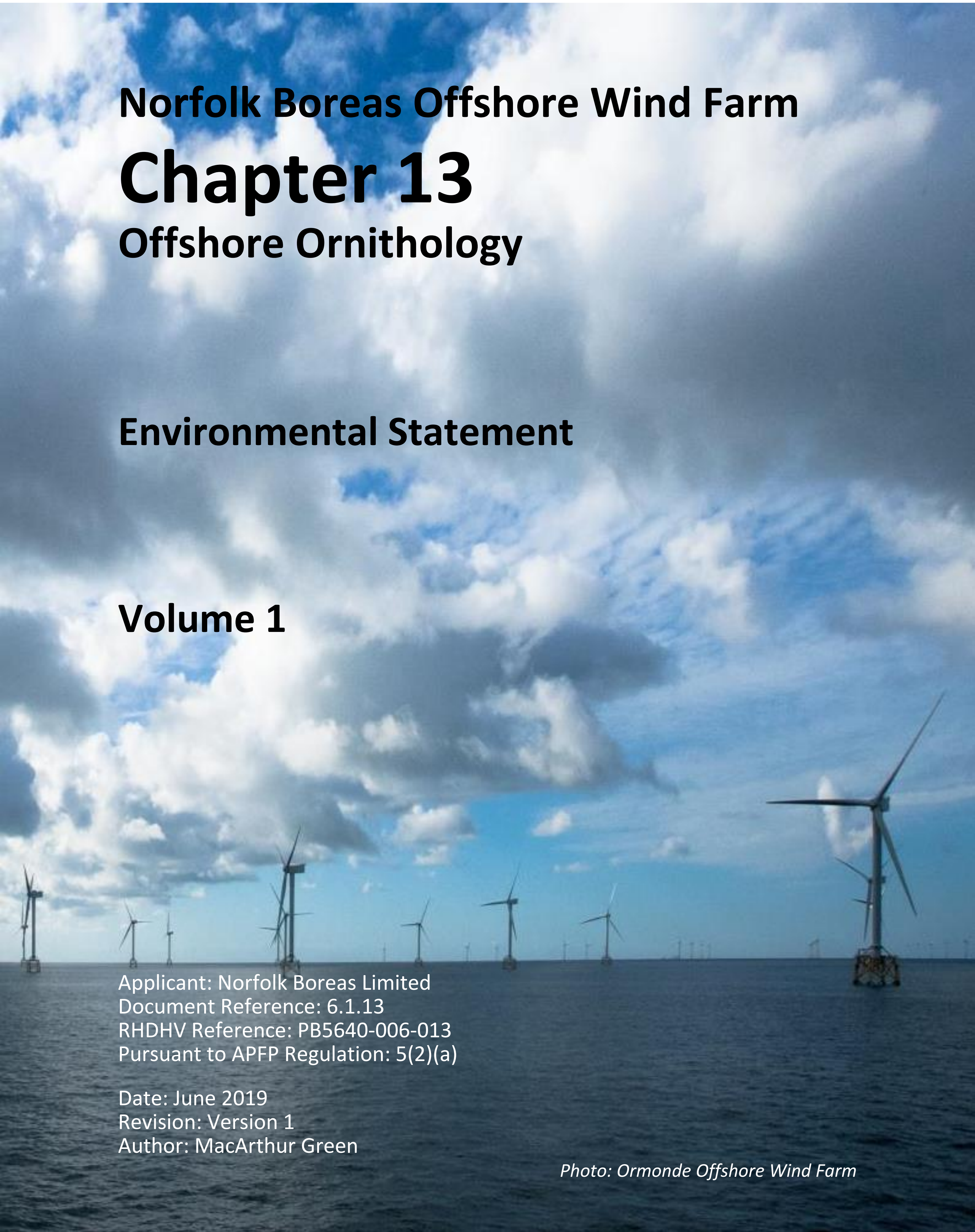

Norfolk Boreas Offshore Wind Farm

Chapter 13

Offshore Ornithology

Environmental Statement

Volume 1

Applicant: Norfolk Boreas Limited
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Glossary of Acronyms

AR	Avoidance Rates
BDMPS	Biologically Defined Minimum Population Scale
BoCC	Birds of Conservation Concern
BTO	British Trust for Ornithology
CIA	Cumulative Impact Assessment
CRM	Collision Risk Modelling
DCO	Development Consent Order
EATL	East Anglia THREE Limited
EIA	Environmental Impact Assessment
EMF	Electro-magnetic Field
EPP	Evidence Plan Process
ES	Environmental Statement
FAME	Future of the Atlantic Marine Environment
GGOWL	Greater Gabbard Offshore Wind Farm Limited
HRA	Habitats Regulations Assessment
IEEM	Institute of Ecology and Environmental Management
JNCC	Joint Nature Conservation Committee
KDE	Kernel Density Estimate
MAGIC	Multi-Agency Geographic Information for the Countryside
MLWS	Mean Low Water Springs
MW	Megawatt
NGO	Non-Governmental Organisation
NPPF	National Planning Policy Framework
NPS	National Policy Statement
OETG	Ornithology Expert Technical Group (part of the Evidence Plan process)
ORJIP	Offshore Renewables Joint Industry Programme
OWEZ	Offshore Wind Farm Egmond aan Zee
OWF	Offshore Wind Farm
PBR	Potential Biological Removal
PCH	Potential Collision Height
PEIR	Preliminary Environmental Information Report
pSPA	Proposed Special Protection Area
PVA	Population Viability Analysis
RSPB	Royal Society for the Protection of Birds
SAC	Special Area of Conservation
SNCB	Statutory Nature Conservation Body
SoS	Strategic Ornithological Support
SOSS	Strategic Ornithological Support Services
SPA	Special Protection Area (note, pSPA indicates a proposed site not yet fully designated)
SSSI	Site of Special Scientific Interest
WWT	Wildfowl and Wetlands Trust

Glossary of Terminology

Array cables	Cables which link the wind turbine to wind turbine and wind turbine to offshore electrical platforms.
Biologically defined minimum population scales	Species-specific non-breeding season seabird populations at biologically defined minimum population scales (BDMPS). Used as reference populations in assessments.
Evidence Plan Process	A voluntary consultation process with specialist stakeholders to agree the approach to the EIA and information to support HRA.
Interconnector cables	Buried offshore cables which link offshore electrical platforms within the Norfolk Bores site
Landfall	Where the offshore cables come ashore at Happisburgh South
Norfolk Boreas site	The Norfolk Boreas wind farm boundary. Located offshore, this will contain all the wind farm array.
Offshore service platform	A platform to house workers offshore and/or provide helicopter refuelling facilities. An accommodation vessel may be used as an alternative for housing workers.
Offshore cable corridor	The corridor of seabed from Norfolk Boreas to the landfall site within which the offshore export cables will be located.
Offshore electrical platform	A fixed structure located within the Norfolk Boreas site, containing electrical equipment to aggregate the power from the wind turbines and convert it into a suitable form for export to shore.
Offshore export cables	The cables which bring electricity from the offshore electrical platform to the landfall.
Offshore project area	The area including the Norfolk Boreas site, project interconnector search area and offshore cable corridor.
Population Viability Analysis	Modelling methods used to explore and understand potential consequences of additional mortality on populations.
Project interconnector cable	Offshore cables which would link an offshore electrical platform in the Norfolk Boreas site with an offshore electrical platform in one of the Norfolk Vanguard sites.
Project interconnector search area	The area within which the project interconnector cable would be installed.
Safety zones	An area around a vessel which should be avoided during offshore construction.
Scour protection	Protective materials to avoid sediment being eroded away from the base of the foundations as a result of the flow of water.
The Applicant	Norfolk Boreas Limited
The Norfolk Vanguard OWF sites	Term used exclusively to refer to the two distinct offshore wind farm areas, Norfolk Vanguard East and Norfolk Vanguard West (also termed NV East and NV West) which will contain the Norfolk Vanguard arrays.
The project	Norfolk Boreas Wind Farm including the onshore and offshore infrastructure.

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13 OFFSHORE ORNITHOLOGY

13.1 Introduction

1. This chapter has been prepared by MacArthur Green using survey data collected by APEM Ltd. and presents the assessment of the potential impacts on ornithological receptors that might arise from construction, operation and decommissioning of the offshore components of the proposed Norfolk Boreas project (hereafter ‘the project’).

2. This chapter describes the offshore components of the project in relation to ornithology; the consultation that has been held with stakeholders; the scope and methodology of the assessment; the avoidance and mitigation measures that have been embedded through project design; the baseline data on birds and important sites and habitats for birds acquired through desk study and surveys; and assesses the potential impacts on birds.

3. Full details of the baseline data acquired through the surveys specifically carried out within the Norfolk Boreas site and a 4km buffer can be found in Technical Appendix 13.1 Norfolk Boreas Offshore Wind Farm Ornithology Technical Appendix.

4. Vattenfall Wind Power Limited (VWPL) (the parent company of Norfolk Boreas Limited) is also developing Norfolk Vanguard, a ‘sister project’ to Norfolk Boreas. Norfolk Vanguard’s development schedule is approximately one year ahead of Norfolk Boreas and as such the Development Consent Order (DCO) application was submitted in June 2018.

5. Norfolk Vanguard may undertake some enabling works for Norfolk Boreas, but these are only relevant to the assessment of impacts onshore. This assessment does however assume a worst case which includes interconnector cables between the Norfolk Boreas and Norfolk Vanguard projects (herein, ’the project interconnector’). If Norfolk Vanguard does not proceed then the project interconnector would not be required.

13.2 Legislation, Guidance and Policy

13.2.1 Guidance

6. The most relevant guidance on Environmental Impact Assessment (EIA) for marine ecology receptors, including birds, is the ‘Guidelines for Ecological Impact Assessment in Britain and Ireland: Marine and Coastal’ published by the Institute of Ecology and Environmental Management (IEEM, 2010). The EIA methodology described in section 13.4.1 and applied in this chapter is based on that IEEM guidance.

7. Additional guidance on the assessment of the potential impacts of renewable energy generation on birds has been produced by a number of statutory bodies, NGOs and consultants including, but not limited to the following:

- Assessment methodologies for offshore wind farms (Maclean et al., 2009);
- Guidance on ornithological cumulative impact assessment for offshore wind developers (King et al., 2009);
- Advice on assessing displacement of birds from offshore wind farms (JNCC et al., 2017);
- Collision Risk Modelling (CRM) to assess bird collision risks for offshore wind farms (Band, 2012);
- Assessing the risk of offshore wind farm development to migratory birds (Wright et al., 2012);
- Vulnerability of seabirds to offshore wind farms (Furness and Wade, 2012; Furness et al., 2013; Wade et al., 2016);
- Mapping seabird sensitivity to Offshore Wind Farms (Bradbury et al., 2014);
- The avoidance rates of collision between birds and offshore turbines (Cook et al., 2014); and
- Joint Response from the Statutory Nature Conservation Bodies to the Marine Scotland Science Avoidance Rate Review (JNCC et al., 2014).

13.2.2 Legislation

8. Table 13.1 identifies the relevant legislation and summarises the important measures derived from it.

Table 13.1 Legislation and relevant measures.

Legislation	Relevant Measures	Section reference
Birds Directive - Council Directive 79/409/EEC on the Conservation of Wild Birds	This Directive provides a framework for the conservation and management of wild birds in Europe. The most relevant provisions of the Directive are the identification and classification of Special Protection Areas (SPAs) for rare or vulnerable species listed in Annex I of the Directive and for all regularly occurring migratory species (required by Article 4). It also establishes a general scheme of protection for all wild birds (required by Article 5). The Directive requires national Governments to establish SPAs and to have in place mechanisms to protect and manage them. The SPA protection procedures originally set out in Article 4 of the Birds Directive have been replaced by the Article 6 provisions of the Habitats Directive.	Designated sites, including SPAs, with potential for connectivity to the wind farm are listed for consideration in section 13.6.1. Assessment of the potential impacts on the features of these SPAs, together with assessment on other Natura sites and features (e.g. Special Areas of Conservation) will be provided in a Habitats Regulations Assessment.
The Conservation of Habitats and Species	The Conservation of Habitats and Species Regulations 2017 ('the Habitats Regulations') consolidates the Conservation of Habitats and Species Regulations 2010	The assessment has been conducted in accordance with the protections afforded by this

Legislation	Relevant Measures	Section reference
Regulations 2017 and	with subsequent amendments. These regulations include the marine environment up to the 12nm territorial limit. The Habitats Regulations transpose the Birds Directive and the Habitats Directive into national law. The Habitats Regulations place an obligation on 'competent authorities' to carry out an appropriate assessment of any proposal likely to affect a Natura 2000 site, to seek advice from Natural England and not to approve an application that would have an adverse effect on a Natura 2000 site except under very tightly constrained conditions that involve decisions by the Secretary of State. The competent authority in the case of the proposed project is the Secretary of State for Business Energy and Industrial Strategy.	legislation. Features of Sites of Special Scientific Interest (SSSI)'s have also been listed in section 13.6.1.
The Conservation of Offshore Marine Habitats and Species Regulations 2017	In November 2017, the Conservation of Habitats and Species Regulations 2010 and the Offshore Marine Conservation (Natural Habitats, &c.) Regulations 2007 were consolidated into the Conservation of Offshore Marine Habitats and Species Regulations 2017 ('the Offshore Habitats Regulations 2017'). These regulations apply to UK waters beyond the 12nm limit within British Fishery Limits and the seabed within the UK Continental Shelf Designated Area.	As above
Wildlife and Countryside Act 1981	The Wildlife and Countryside Act 1981 (as amended) is an important mechanism for the legislative protection of wildlife in Great Britain. It provides protection for all birds by establishing the system of Sites of Special Scientific Interest (SSSI).	As above

13.2.3 Policy

9. Table 13.2 identifies policy and summarises the important measures derived from it that are relevant to offshore ornithology.

Table 13.2 Policy and relevant measures.

Policy	Relevant Measures	Section reference
Overarching National Policy Statement (NPS) for Energy (NPS EN-1) (July 2011)	Paragraph 5.3.3 states that the applicant should ensure that the ES clearly sets out any effects on internationally, nationally and locally designated sites of ecological importance, on protected species and on habitats and other species identified as being of principal importance for the conservation of biodiversity. Paragraph 5.3.4 states that the applicant should also show how the proposed project has taken advantage of opportunities to conserve and enhance biodiversity interests. Paragraph 5.3.18 states that the applicant should include appropriate mitigation measures as an integral part of the proposed project.	Protected sites are listed in 13.6.1. Assessment of the potential effects of the wind farm on the features of these protected sites is provided in 13.7. Further consideration and assessment for designated sites with potential connectivity to

Policy	Relevant Measures	Section reference
		the wind farm will be provided in an Information to Support Habitats Regulations Assessment (HRA) Report (document reference 5.3) which has been submitted as part of the DCO application.
NPS for Renewable Energy Infrastructure (NPS EN-3) (July 2011)	Paragraph 2.6.64 states that the assessment of offshore ecology and biodiversity should be undertaken by the applicant for all stages of the lifespan of the proposed offshore wind farm. Paragraph 2.6.102 states that the scope, effort and methods required for ornithological surveys should have been discussed with the relevant statutory advisor. Paragraph 2.6.104 states that it may be appropriate for the assessment to include CRM for certain bird species.	Potential impacts assessed include during construction (section 13.7.3), operation (section 13.7.4 and decommissioning (section 13.7.5). The survey methods were discussed and agreed with Natural England through the Evidence Plan Process (see Chapter 7 Technical consultation).
National Planning Policy Framework	The National Planning Policy Framework (2018) sets out the Government's planning policies for England and how these are expected to be applied. The document establishes a number of core land-use planning principles that should underpin both plan-making and decision-taking, including contributing to conserving and enhancing the natural environment. Paragraph 170 states that "Planning policies and decisions should contribute to and enhance the natural and local environment", through the adoption of various measures, *inter alia*, "minimising impacts on and providing net gains for biodiversity, including by establishing coherent ecological networks that are more resilient to current and future pressures".	The underlying principles of the NPPF have been adhered to throughout the assessment.
UK Post-2010 Biodiversity Framework	The 'UK Post-2010 Biodiversity Framework' succeeds the UK Biodiversity Action Plan. The Framework demonstrates how the work of the four countries and the UK contributes to achieving the Aichi Biodiversity Targets, and identifies the activities required to complement the country biodiversity strategies in achieving the targets. The following seabirds are identified as a priority for action: common scoter, black-throated diver, Balearic shearwater, Arctic skua, herring gull and roseate tern.	It should be noted that most of the named species have not been recorded on the Norfolk Boreas site. For those which have, potential impacts have been assessed where relevant, e.g. section 13.7.4.3 (Arctic skua collision risk).

13.3 Consultation

10. To inform the offshore ornithology assessment, Norfolk Boreas Limited has undertaken a pre-application consultation process including the following key consultation:

- Scoping Report submitted to the Planning Inspectorate (Royal HaskoningDHV, 2017);
- Scoping Opinion received from the Planning Inspectorate (the Planning Inspectorate, 2017); and
- Consultation with the Ornithology Expert Topic Group (OETG) on a method statement (February and March 2018).
- Production of a Preliminary Environmental Information Report (PEIR) which presented a full assessment using a slightly reduced dataset (as surveys were ongoing at the time). Natural England, the RSPB and Ministry of Infrastructure and Water Management Netherlands provided comments as a formal section 42 response on this document. These comments have been used to revise and update the assessment presented below.
- A conference call was held with the OETG on the 27th February 2019 at which the PEIR was discussed and guidance on updates provided.
- On this call the Marine Scotland Science stochastic Collision Risk Model was discussed. Due to errors encountered whilst attempting to use this model it was agreed that the collision risk assessment would be based on results from the Band (2012) deterministic model.

11. Following submission of the Preliminary Environmental Information Report (PEIR) further consultation was undertaken with key statutory consultees through the Evidence Plan Process (EPP; for further detail on the EPP please refer to Chapter 7 Technical Consultation; minutes from the OETG meetings will be included as a technical appendix to the ES).

12. In addition to stakeholder consultation for Norfolk Boreas, the assessment presented here has also been informed by the information gathered and assessment carried out for Norfolk Vanguard and East Anglia THREE. Norfolk Vanguard was subject to consultation prior to submission of its application for consent in June 2018 and the East Anglia THREE project was consulted on prior to submission of its application in November 2015.

13. Detailed consultation and iteration of the overall approach to the Environmental Impact Assessment (EIA) on ornithological receptors has been discussed and agreed with stakeholders through the Evidence Plan process. The OETG includes representatives of Natural England and the Royal Society for the Protection of Birds

(RSPB). The OETG provided a forum for consultation during preparation of this Environmental Statement (ES) and this will continue during the examination phase.

14. The comments arising from the consultation process (to date comprising scoping and the Evidence Plan Process) and the Applicant's response made to each are summarised in Table 13.3.

15. As Norfolk Boreas and Norfolk Vanguard share an offshore cable corridor, the pre-application consultation undertaken as part of Norfolk Vanguard has been used to inform the approach to the Norfolk Boreas benthic ecology assessment. Furthermore, information submitted as part of the Norfolk Vanguard examination, has also been incorporated. However, in order that the programmed submission of the Norfolk Boreas DCO has not been impacted it has been necessary to use a cut-off point of the 20th March (which coincided with Norfolk Vanguard Examination Deadline 5) after which information provided at the Vanguard examination as well as any wider information has not been included in this assessment unless it could be done without impacting the programme for submission.

Table 13.3 Consultation Responses.

Consultee	Date /Document	Comment	Response / where addressed ES
Natural England	Scoping opinion, June 2017	The potential effects of this development proposal on birds during all phases of development encompassing displacement, indirect effects (through impacts on prey species) and collision mortality – both at a project-level and cumulatively.	These aspects are considered in relevant sections of this ES. Specific points are given additional consideration below.
Secretary of State	Scoping Opinion, June 2017	The SoS recommends that the Applicant seeks agreement with Natural England regarding the suitability of the ornithological data proposed to be utilised for the offshore cable corridor.	Discussion and agreement on this matter have been undertaken through the Evidence Plan Process and agreement reached on the proposed methods.
		The ES should explain how population estimates/densities will be calculated.	This has been included in the ornithology technical appendix (13.1)
		The ES should consider impacts on prey species during construction not only from construction of the array, but also from the offshore cable corridor.	This potential impact has been considered in section 13.7.3.2.
		The Scoping Report proposes to scope out indirect impacts to bird species during the operational phase on the basis that there is growing evidence from existing offshore wind farms that underwater noise, EMF and elevated suspended sediment could cause prey to avoid the operational area and affect their physiology and behaviour. The SoS notes that this approach contradicts proposals within the Fish and Shellfish Ecology Chapter to assess impacts on fish and shellfish (Table 2.16). Accordingly, the SoS does not agree to scope this out.	This proposal has been reviewed and indirect impacts have been considered in the relevant section 13.7.4.2.
		The SoS agrees that indirect impacts on prey species and habitat along the export cable can be scoped out of the operational phase assessment on the basis that maintenance or repair operations would be localised and episodic.	Noted.
		The SoS welcomes that the assessment scope and methodology will be discussed and agreed during the EPP. The Applicant's	Noted.

Consultee	Date /Document	Comment	Response / where addressed ES
		attention is drawn to the comments of Natural England (see Appendix 3 of this Opinion), for example regarding the use of Band (2012) model for collision risk.	
		Paragraph 575 of the Scoping Report refers to matrices in order to assess the potential effects of displacement on sensitive species. This approach is agreed and commented upon by Natural England in its consultation response (see Appendix 3 of this Opinion). The ES should clearly set out the methodology associated with the use of matrices.	The assessment method is described in section 13.4.1
		The potential for cumulative construction impacts should be considered, particularly with Norfolk Vanguard.	This aspect has been considered in section 13.8
Natural England	Scoping opinion, June 2017	Natural England advises that the aerial survey data sets that have been collected so far and are proposed in the Scoping Report to be continued to be collected for the Norfolk Boreas site and 4km buffer, will provide a sufficient baseline for site characterisation; provided the surveys cover the required 24 months.	Noted.
		Scoping Report para. 558: We suggest that the following additional literature and data sources that are not listed in paragraph 558 or referenced in the Scoping Report are considered (noting that this is not an exhaustive list): • Bradbury G., Trinder M., Furness B, Banks A.N., Caldow R.W.G., et al. (2014) Mapping Seabird Sensitivity to Offshore Wind Farms. PLoS ONE 9(9): e106366. doi:10.1371/journal.pone.0106366 • Langston, R. (2010) Offshore wind farms and birds - Round 3 Zones, extensions to Round 1 and 2 sites, and Scottish territorial waters. RSPB Research Report 39. RSPB. • At sea densities of seabirds (ESAS data): https://data.gov.uk/dataset/at-sea-densities-of-allmodelled-seabird-species-combined-for-the-breeding-season https://data.gov.uk/dataset/atsea- densities-of-all-modelled-	Norfolk Boreas Limited have reviewed the suggested additional sources of information and these are listed and referred to in the relevant sections as appropriate.

Consultee	Date /Document	Comment	Response / where addressed ES
		seabird-species-combined-for-the-non-breeding-season • Seabird Monitoring Programme reports and data: http://jncc.defra.gov.uk/smp/counts.aspx and http://jncc.defra.gov.uk/page-1530 • Stroud, D.A., Bainbridge, I.P., Maddock, A., Anthony, S., Baker, H., Buxton, N., Chambers, D., Enlander, I., Hearn, R.D., Jennings, K.R, Mavor, R., Whitehead, S. and Wilson, J.D. - on behalf of the UK SPA & Ramsar Scientific Working Group (eds.) 2016. The status of UK SPAs in the 2000s: the Third Network Review. 108 pp. JNCC, Peterborough. Available online: http://jncc.defra.gov.uk/pdf/UKSPA3_StatusofUKSPAsinthe2000s.pdf The Applicant should also review any relevant papers and guidance documents that are published between this response and the submission of the Environmental Statement.	
		Construction The 'Potential Impacts from Construction' section currently covers disturbance and displacement resulting from the construction of the offshore wind farm and the laying of the offshore cables. It also covers indirect impacts through effects on habitats and prey species via underwater noise and generation of suspended sediments through activities such as piling and seabed preparation for installation of foundations. However, it is unclear whether the indirect impacts on habitats and prey also covers such impacts resulting from cable laying activities. The potential for impact from this aspect of construction should also be considered.	This potential impact has been reviewed and is assessed in section 13.7.3.2.
		Operation The potential operational impacts are listed as disturbance and displacement; indirect impacts include effects on habitats and prey species, collision risk and barrier effect. Consideration could also be given to direct habitat loss from the turbine locations (not in	This potential impact has been reviewed and is assessed in section 13.7.4.2.

Consultee	Date /Document	Comment	Response / where addressed ES
		terms of the whole OWF footprint); although it is acknowledged that this is likely to be small.	
		Decommissioning We agree that decommissioning impacts will be similar to construction.	Noted.
		General comment on potential impacts Additionally, we note that the EIA should consider the environment as a whole, and not as a discrete set of individually sensitive receptors. Any indirect impacts on habitat and prey for all assessment stages (construction, operation, decommissioning) should be linked to the relevant habitat and prey assessment chapters - fish and shellfish ecology, benthic ecology and water and sediment quality assessments. We note that within the Scoping Report there is a section (2.16) on offshore inter-related effects where the Applicant has outlined suggestions regarding the assessment of linkages between receptors, and how impacts on one receptor may influence others. We advise that Table 2.31 should highlight inter-relationships in terms of how offshore ornithology could be affected by benthic and intertidal ecology and marine water and sediment quality as well as fish ecology. We consider that such inter-relationships are likely to be key in interpreting the environmental impacts of the development and welcome the applicant’s intention to integrate these aspects as part of the EIA process.	An assessment of potential interrelated impacts is provided in section 13.10
		Do you agree with the potential impacts that have been scoped out for each topic? If not, please provide details. Table 2.21: This table summarises the impacts relating to offshore ornithology and indicates those impacts scoped in and out for the different phases of the development. We do not agree that	Consideration of indirect operational impacts has been provided in section 13.7.4.2

Consultee	Date /Document	Comment	Response / where addressed ES
		indirect impacts through effects on habitats and prey species should be scoped out for the operation phase with regard to the Norfolk Boreas site itself. This is due to the potential for underwater noise and generation of suspended sediments that may alter behaviour or availability of bird prey species (as highlighted in paragraph 577 of the Scoping Report). However, we would agree that this potential impact for the operational phase could be scoped out with regard to potential impacts along the export cable (for the reasons highlighted in paragraph 578).	
		579: We agree with the scoping in of the collision risk during operation for the Norfolk Boreas site and that the operation of the export cable is scoped out. We note that whilst there is the possibility of bird collision with vessels during construction and decommissioning, this is likely to be minor, with the main impact from collision being with the operational turbines.	Noted.
		580: We agree that the main barrier effect of the project will be whilst it is operational and should therefore be scoped in.	Barrier effects have been considered in section 13.7.4.4
		Have the relevant potential cumulative impacts been identified? If not, please provide details. 583: We agree with the potential cumulative impacts that have been identified by the Applicant, namely: collision risk, barrier effects which impact upon migration routes and indirect impacts on prey species. However, consideration should also be given to cumulative displacement impacts.	Cumulative displacement has been assessed in section 13.8.2.6
		We also note that other offshore wind farms within the former East Anglia Zone could be of relevance in terms of potential for overlap in construction periods (particularly Norfolk Vanguard) and hence advise that cumulative construction impacts are considered.	The potential for cumulative construction impacts has been considered in section 13.8.1

Consultee	Date /Document	Comment	Response / where addressed ES
		Have the relevant potential transboundary impacts been identified? If not, please provide details 586: We agree with the Applicant's approach to assessing potential transboundary impacts and welcome building upon the work undertaken by East Anglia ONE and East Anglia THREE to identify potential receptors and stakeholders	Noted.
		Do you agree with that the proposed approach to assessing each impact is appropriate? If not, please provide details. The information provided on the proposed approach to assessing each impact is very high level/brief and in many cases further detail could be provided regarding the actual approach to the assessments.	Further detail on impact assessment methods has been provided in this ES chapter in section 13.4.1 and the supporting technical appendix.
		579: This paragraph states; 'Collision risk modelling (CRM) will be undertaken using industry standard approaches (Band 2012, Masden 2015) to predict potential mortality levels from this impact.' We note that Masden (2015) is still undergoing testing and we would currently advise that the Band (2012) model is used and that the Applicant presents outputs from the Band model that account for variability in the input parameters – especially densities of birds in flight, flight heights and avoidance rates. We advise the same approach as used in the Hornsea Project 2 assessment using upper and lower confidence intervals for each parameter.	Collision modelling has been undertaken with consideration of uncertainty in the parameters identified by Natural England, as well as in nocturnal activity rates (for gannet, kittiwake and large gulls).
		We welcome the commitment in paragraph 579 that the exact option and version of the collision risk model to be used, avoidance rates, flight height data and parameters for modelling will be based upon the best available evidence and will be agreed through the evidence plan process and clearly defined within the ES and HRA.	Noted.err

Consultee	Date /Document	Comment	Response / where addressed ES
		We agree that the predicted potential effects of displacement on sensitive species will be assessed using matrices to compare varying levels of displacement with varying levels of additional mortality and advise that the approach outlined in the recent (2017) SNCB interim guidance on displacement is followed (available from: http://jncc.defra.gov.uk/pdf/Joint_SNCB_Interim_Displacement_AdviceNote_2017.pdf). Further information could be provided in the section of the Scoping Report on operational disturbance and displacement regarding which sensitive species might assessed and we also recommend the inclusion of an example matrix.	Noted. Further details on the species included and the methods is provided in the relevant sections (13.7.4.1) of this ES.
		576: This paragraph states that; 'For species at risk of displacement during the non-breeding season, consideration will be given to a proposed approach for standardising assessments (i.e. to account for different numbers of nonbreeding seasons between species for which data is available).' We note that in discussions at the first Offshore Ornithology Expert Topic Group meeting (15th Feb 2017) as part of the Evidence Plan Process for Norfolk Boreas this proposed approach was discussed and Natural England advised that summed impacts across all Biologically Defined Minimum Population Scale (BDMPS) seasons for the non-breeding season (and breeding season) should be presented.	The assessment in section13.7.4.1 follows the Natural England recommended approach, whilst noting that this is considered precautionary. It should also be noted that the OETG meeting referred to was for Norfolk Vanguard. However, the Norfolk Vanguard EPP and OETG meetings have informed Norfolk Boreas's assessment.
		Is there any further guidance relating to each topic that we should be aware of? If so, please provide details. Please see the suggested additional literature and data sources listed in our response to question 1.	Noted.
		The scoping report does not provide any detail about how the baseline data will be analysed, e.g. how population estimates/densities will be calculated.	These have been provided in the Offshore Ornithology Technical Appendix 13.1.

Consultee	Date /Document	Comment	Response / where addressed ES
		Table 2.20: Where appropriate the various conservation listings (e.g. BoCC listing, whether a migratory species and/or Annex 1 species, IUCN red listing) should be presented for all species, as for some species some of these listings have not been included.	These listings have been completed for the relevant species in this assessment.
		535: Regarding the Greater Wash pSPA, the Applicant states that the pSPA encompasses the foraging areas of common, Sandwich and little terns from a number of colonies, including The Wash SPA (for little and Sandwich tern). We note that the species in brackets should be the little tern and not Sandwich for the Wash SPA. We advise the Applicant to consider the draft conservation advice package for the Greater Wash pSPA, available at: https://www.gov.uk/government/publications/marine-conservation-advice-for-special-protectionarea-the-wash-uk9008022/the-wash-spa-site-information.	The relevant sections have been updated. Assessment in relation to the Greater Wash SPA will be provided in the Information to support the Habitat Regulations Assessment.
Natural England	PEIR 27th November 2018	**Ornithological assessment – CRM** We request going forward that any ornithological analysis present both the Marine Scotland Science Stochastic Collision Risk Model (April, 2018) and the Band model (or non-stochastic/deterministic version) outputs using the central values for the various variables (bird density, flight heights, avoidance rates, nocturnal activity etc.) in line with other current OWF applications. The use of this model has also been requested for Vanguard.	Collision risk estimates are presented in section 13.7.4.3. However, attempts to use the Marine Scotland Science stochastic CRM were unsuccessful due to the presence of errors in the model code. These were brought to the attention of the model developer who addressed these issues. However there was insufficient time following this for the model to be used for this assessment.
		Natural England has identified a number of concerns that have not been addressed sufficiently and need addressing in the assessment of impacts on offshore ornithology receptors. These can be summarised as follows: • Seasonal definitions; • Seasonal apportioning of impacts for Habitats Regulations Assessments (HRA); • Assessment of displacement impacts (EIA and HRA);	Seasonal definitions are defined in section 13.6.2.1. Where relevant the assignment of months to seasons has been discussed in the text. Impacts in relation to Special Protection Areas (SPAs) are assessed in full in the Information for the Habitats Regulations Assessment, including consideration of appropriate apportioning among populations and seasons.

Consultee	Date /Document	Comment	Response / where addressed ES
		• CRM (EIA and HRA); • Cumulative and in-combination assessments (displacement and CRM); • Population modelling approaches (EIA and HRA). • Implications for EIA and HRA assessments	Displacement is assessed in sections 13.7.3.1, 13.7.4.1 and 13.8.2.6. These assessments have been informed by responses provided for the Norfolk Vanguard project by Natural England and the applicant. Collision risk is assessed in section 13.7.4.3. This assessment has been informed by responses provided for the Norfolk Vanguard project by Natural England and the applicant. No new population modelling has been undertaken for the current assessment as the existing population projections produced for previous applications are considered to remain valid.
RSPB	PEIR 7th December 2018	Impact significance. The RSPB is unable to agree at this stage that no impacts greater than minor adverse significance will occur to ornithological interests as a result of offshore elements of the project. Our concerns relate principally to collision risk to gannet and kittiwake, particularly in relation to the Flamborough and Filey Coast SPA, lesser black-backed gull of the Alde-Ore Estuary SPA and great black-backed gull, and to displacement of red-throated diver (including those of the Greater Wash SPA), razorbill and guillemot.	The RSPB's stated position on impact significance is acknowledged. Collision risk and displacement concerns for all species designated at SPAs which may have connectivity with the Norfolk Boreas wind farm have been considered and discussed in The Information for the Habitats Regulations Assessment (document reference 5.3). The impact assessment follows the methods set out in this ES (see section 13.4.1) and conclusions on impact significance are backed up with evidence in the appropriate sections.
		Methodological issues. The RSPB considers that some methodological procedures used in the assessment are inadequate to ensure a robust assessment and therefore a proper understanding of the likely impacts of the scheme. We have particular concerns regarding the stochastic model used in the assessment of collision risk, the use of median values for bird	The assessment has been updated to address the comments raised by the RSPB (section 13.7.4.3).

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		density within the deterministic collision risk model, the use of revised nocturnal activity factors and the change in approach to the baseline used in cumulative assessments.	
		Habitats Regulation Assessment (HRA). We note that apportioning of offshore impacts (collision risk and displacement) to SPAs both alone and in-combination with other projects has not yet been carried out and that this will need to be addressed to ensure compliance with the Conservation of Habitats and Species Regulations 2017 and the Conservation of Offshore Marine Habitats and Species Regulations 2017 requirements.	The Information for the Habitats Regulations Assessment (document reference 5.3) provides assessment of potential impacts on species designated at SPAs which may have connectivity with the Norfolk Boreas wind farm.
		Table 5.3 indicates that the project design life is around 30 years. Assessments of impacts, including population modelling to assess the effects of potential collision risk, should therefore work to this timescale.	Where necessary, impact consequences have been assessed in relation to population modelling outputs produced for previous wind farm applications. Cumulative collision risk for kittiwake (see section 13.8.2.7.2) makes reference to population predictions covering a 30 year period as requested. .
		We understand that the assessment presented in the PEIR is based on the 18 months of survey data available at the time of production. Our comments on impact significance are therefore subject to change, depending on the findings based on the full 24 months of survey data.	This caveat is noted. The assessment presented in this ES uses baseline data collected over a full 24 month period.
		The PEIR throughout makes the assertion that birds present in the breeding season are unlikely to be breeding birds, yet notes that the site is within mean-maximum foraging range of gannets from the FFC SPA and lesser black-backed gulls from the Alde-Ore Estuary SPA. It is stated that tracking of individuals from these colonies shows limited connectivity. However, no references are	This aspect is discussed and considered in The Information for the Habitats Regulations Assessment (document reference 5.3) for species designated at SPAs which may have connectivity with the Norfolk Boreas wind farm.

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		provided in support of this and it is not therefore possible to assess the numbers of birds studied and whether sufficient evidence to rule out connectivity exists.	
		Benacre-Easton Bavents SPA (designated for breeding little tern and marsh harrier, and breeding and wintering bittern) has been omitted from Table 13.9. This should be included for completeness.	Screening for SPA features is included in The Information for the Habitats Regulations Assessment (document reference 5.3).
		Collision risk: Our concerns are principally around the assessment of impacts on gannet, kittiwake, lesser black-backed gull and great black-backed gull and relate to both the methods used in the assessment and the significance of potential impacts.	Impacts on these species are considered in detail in under appropriate species in section 13.7.4.3 of this ES.
		In order to predict the collision risk mortality of an offshore wind farm in the UK, the Band (2012) model has previously been used in assessment. This model uses a number of input parameters, such as bird size, flight speed and turbine blade dimensions, to calculate the probability of a bird that passes through the swept area of a turbine blade colliding with that blade. For this deterministic model the input parameters were defined as single values with no indication of variability around them. In reality, most of the parameters will exhibit a considerable degree of variability and stochastic CRM has been developed to allow this to be incorporated into the model and thus generate a potential range of output predicted collision mortalities. McGregor et al., (2018), under commission of Marine Scotland Science and overseen by an expert steering panel, produced a revised and fully tested stochastic model to widespread stakeholder acceptance. By contrast, the Applicant has presented an entirely untested new version that does not follow a recognised methodology, with insufficient detail provided as to how it incorporates variability or	The assessment has been updated to use deterministic CRM models to address the comments raised by the RSPB (section 13.7.4.3).

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		how it overcomes the statistical difficulties of non- independence (the degree of interrelation) of some of the variables. The RSPB therefore does not agree that the model presented by the Applicant is fit for purpose and recommend that the Marine Scotland (McGregor et al., 2018) model version is used in preference.	
		The documents present deterministic and stochastic versions of the CRM (see above). For the deterministic version (Band 2012) of the CRM the correct value to use for bird density is the mean monthly value, however, the values used in this assessment are median values, which will result in the model predicting considerably lower collision mortalities.	The assessment has been updated to use deterministic models to address the comments raised by the RSPB (section 13.7.4.3). These use the mean seabird densities as requested.
		We note that, with the exception of lesser black-backed gull, the migration-free breeding season has been used rather than the standard breeding season as it is assumed that there is a very low presence of breeding birds within the project area. We disagree with this assertion, as discussed above. For example for gannet, the migration-free breeding season excludes March and September, which reduces the number of predicted collisions. But gannets start arriving in January and establishing their nest sites in March. Whilst peak fledging is in August, some birds are still fledging in September, hence there is a strong argument for considering these months to be part of the breeding season. For kittiwake, the migration-free breeding season excludes March-April and August, which again significantly reduces the number of collisions. The first kittiwakes arrive at the colony in February, with most birds back by March and remaining until August, hence there is a strong argument for considering March, April and August to be part of the breeding season.	These aspects are discussed and considered in The Information for the Habitats Regulations Assessment (document refernce 5.3) for species designated at SPAs which may have connectivity with the Norfolk Boreas wind farm.

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		If figures for the migration-free breeding season are to be presented, we consider that it would be necessary to attribute birds in the crossover months to breeding and dispersal in order to ensure collision risk to breeding birds is not underestimated. We would therefore prefer to see mortality figures presented for the standard breeding season (alongside the migration-free breeding season, if required), as well as the autumn period, so that the contribution of the different seasons to total annual mortality can be determined and, for the purposes of HRA, impacts on the FFC SPA understood more clearly.	
		We note that an avoidance rate (AR) for gannet of 98.9% is used for all seasons. Whilst the RSPB accepts the SNCB's recommended amendment to the gannet AR (from 98% to 98.9%) for non-breeding birds, we do not agree that this figure should be applied to the breeding season due to the lack of available evidence relating to breeding birds. The reason for the difference between Natural England and the RSPB in their preferred avoidance rates for gannet is that the avoidance rate review carried out by the BTO for gannet was almost entirely based on birds outside the breeding season. It would be expected that breeding gannets would behave differently from non-breeding birds, and work by Cleasby et al. (2015) demonstrated that foraging birds flew higher, and were therefore at greater risk of collision, than commuting birds. In light of this recent evidence, and given that the BTO review was so heavily biased to non- breeding birds, while we accept the rate for the non-breeding season, we prefer a lower, more precautionary rate for the breeding season. We therefore consider that an AR of 98% should be presented for the breeding season.	The RSPB's stated position on gannet collision avoidance rates is acknowledged, however the evidence based rates used in the assessment are those advised by Natural England.

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		We do not agree with the changes in Nocturnal Activity Factor (a parameter used in collision risk modelling) proposed. The value presented for kittiwake is based on unpublished evidence and therefore we are unable to assess the robustness of the study. The current factor is derived from the expert opinion collected by Garthe and Huppop (2004) and this use is endorsed by Band (2012). A review of seabird vulnerability to offshore wind farms (Furness et al., 2013) recommended that no changes be made to the nocturnal activity scores for these species, and an update, including the same authors (Wade et al., 2016) maintained this recommendation. It is also not clear how these revised rates account for the distinction between the definition of daylight as used in the Band model and with the official concept of 'twilight' and 'night'. This is an issue as the Band (2012) model considers the nocturnal period as between sunset to sunrise and so treats flight activity that occurs at twilight as being within the nocturnal flight period. Evidence from tagging shows that an important number of seabirds actively forage at twilight. While we welcome the latest published evidence review for gannet (Furness et al., 2018), we are concerned that the mortalities predicted using revised nocturnal activity rates for gannet (and this is also applicable to kittiwake) are potentially underestimated because they do not account for the potential interaction between survey timing and diurnal behavioural patterns. Peaks in foraging activity at first and last light (see for example Fig. 3 in Furness et al. 2018) will not be accounted for in the assessment if these did not coincide with surveys (the timings	The RSPB's stated position on the use of nocturnal activity rates in collision risk modelling is acknowledged. However, it is considered that the evidence for the revised rates presented in Furness et al. (2018) is robust and the rates identified are appropriate for their intended purpose (i.e. accounting for nocturnal flight activity in assessing gannet collision risk). With respect to comments on the timing of surveys during the day and how these relate to diurnal patterns of behaviour, it is agreed with the RSPB that peaks in activity may be missed by daytime aerial surveys, however, it is in fact more important that these surveys are conducted at a time of day when activity is around an average level, rather than either a peak or a trough in activity, since the latter two will over and under estimate flight activity respectively. Thus, it can be seen in the example cited by the RSPB (Figure 3 in Furness et al. 2018) that surveys conducted during the day (e.g. between 9am and 4pm as is typical for offshore aerial surveys) will record activity in the middle of the range and are thus, contrary to the RSPB's comment, appropriate for estimating average activity levels.

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		of which are currently unknown, but likely to be midday if aerial), and the survey may have been carried out at a time of much lower activity. Thereby the application of the revised nocturnal activity factor recommended by Furness et al., (2018) could result in inaccurate underestimates of collision risk. The Nocturnal Activity Scores presented for gannet in the application documents are also not in accordance with this latest review (Furness et al., 2018). The values used in the assessment, 4.3% and 2.3% respectively, are even lower than the recommendations of the review (8% in the breeding season and 4% in the non-breeding season) and thus reduce predictions of collision risk further. The robustness of this assessment must therefore be questioned.	
		The assessment of collision risk to migrant non-seabirds is taken from work carried out for East Anglia THREE and the population and flight activity data used in that assessment have not been updated. We recommend that this assessment is updated to include more locally relevant species, such as those from the Breydon Water, Broadland and North Norfolk Coast SPAs. These may also require consideration in the HRA.	Updated assessment of collision risk for non-seabird migrants is provided in section 13.7.4.3 and the supporting technical appendix.
		For collision risk modelling of breeding season kittiwake, a biologically defined minimum population size (BDMPS) for ‘breeding season populations of nonbreeding individuals’ is calculated based on the percentage of the spring BDMPS which are subadults. This equates to 47.3% of the spring BDMPS for kittiwake. We do not agree, as stated above, that there is sufficient evidence that all birds present in the breeding season are likely to be non-breeders. We also would not agree that these assumptions	The RSPB’s stated position on kittiwake populations is acknowledged. Additional work has been undertaken on population connectivity and movements (see section 13.7.4.3) and this has informed the relevant sections of this assessment (section 13.7.4.3 and 13.8.2.7.2).

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		could be used to avoid apportioning any impacts to the SPAs in the HRA.	
		The PEIR claims that the longest foraging trips from the RSPB FAME/STAR kittiwake data were largely from colonies where the breeding success was zero or close to zero. This is incorrect. The longest trips were recorded from Flamborough and Filey, where breeding success was comparatively high over the time of tracking.	With respect to comments on kittiwakes for the longest recorded foraging trips, it is agreed with the RSPB that the longest kittiwake trips have been recently recorded from Flamborough and Filey. However, the PEIR stated that longer trips tended to be recorded at colonies with poor breeding success, but this did not preclude long trips being recorded at other colonies, such as Flamborough and Filey Coast SPA.
		We are concerned that the methods used for calculating a reference population for lesser black-backed gulls are inadequately explained, with insufficient reference to current knowledge and lacking precaution. Such a calculation is difficult because of two competing factors. Throughout the UK, the urban population of lesser black-backed gulls is increasing, while those in “natural” colonies is generally decreasing (JNCC, 2018). In simplistic terms this could be argued as reducing any impact on the Alde-Ore Estuary SPA. The calculations of the number of breeding birds within foraging range of the developments includes a number of inland, urban colonies, such as Ipswich and Norwich as likely sources of birds foraging in the development areas. While we acknowledge that there is a need for more research on the foraging behaviour of urban gulls, it is unlikely that such gulls, especially those from non- coastal urban colonies will forage in the offshore marine environment to the same extent as those breeding at coastal “natural” colonies, such as the Alde-Ore Estuary SPA. The inclusion of birds from such sites dilutes the potential significance of impact on the Alde-Ore Estuary SPA. Furthermore in calculating the number of non-SPA birds the	Additional discussion on lesser black-backed gull population sizes is provided in The Information for the Habitats Regulations (document reference 5.3), to which this comment applies.

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		Applicant gives a rounded up figure of 5400 birds, then simply doubles it (and rounds up further) to 11000, with scant justification other than saying 5400 was a likely underestimate, but presenting no supporting evidence. By overstating the non-SPA population in this way, the potential impact on the Alde- Ore Estuary SPA is again significantly understated.	
		Cumulative collision Risk: Our concerns are principally around the assessment of impacts on gannet, kittiwake, lesser black-backed gull and great black-backed gull and relate to both the methods used in the assessment and the significance of potential impacts. We do not agree that cumulative collision risk to these species can be considered to be of minor adverse significance. These impacts should be regarded as of moderate adverse significance.	The cumulative collision risk assessment has been updated (section 13.8.2.7) and is considered to provide a robust, evidence based assessment
		Projects constructed in 2016 or earlier are considered part of the baseline for the purposes of the cumulative collision risk assessment for the reason that these pre-date the Norfolk Boreas ornithological surveys. We note that previous projects have considered that the baseline does not include the effects of older windfarms due to the fact that much of the available seabird population data pre-dates these projects. Given that this represents a change to the previously accepted approach and the justification does not address the original issues raised, we do not consider that sufficient evidence has been presented to accept this change.	This statement by the RSPB appears to be in error: this approach was not used in the assessment of collision risk presented in the PEIR and has also not been used in the collision assessment presented in this ES.
		It is stated that many of the collision estimates for other windfarms are based on higher numbers of turbines than were actually installed – based on a method of updating collision estimates presented by EATL (2016) this is stated to overestimate mortality by 13% for gannets, 14% for kittiwakes, 35% for lesser	It is acknowledged that the legal aspect of the argument made by the RSPB with respect to acceptance of lower collision risks for wind farms constructed with fewer turbines (and invariably using turbines which generate lower per capita collision risks). However, it is still informative to

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		black-backed gull and 30% for great black-backed gull. This is an acceptable point for windfarms where the DCO has been amended and therefore there is legal certainty regarding the reduction, but where windfarms still have their original DCOs, it is not appropriate to do anything less than assess the full extent of those DCOs when considering in-combination/cumulative effects.	consider this aspect as it contributes to the growing degree of precaution in offshore wind farm impact assessments.
		We do not accept the arguments for including compensatory density dependence in the PVAs for kittiwake and great black-backed gull put forward in the PEIR. The reasons for this are outlined in Green et al. (2016) and the BTO review (Cook and Robinson, 2015), and are not that density dependence does not exist, but rather that we do not have the means to accurately quantify the strength and form of it in a biologically meaningful way in order to incorporate it into PVA. Whilst we accept that density dependence is likely to exist in seabird populations, precise species and colony specific knowledge of its size and shape are needed to correctly parameterise the population models. This is important to acknowledge because density dependence is not always compensatory, but can also be depensatory, slowing the rate of population growth at lower population densities. In other words, a population decline arising from an offshore wind farm could have larger consequences on the population than are predicted by the compensatory density dependent or even density independent models. Horswill and Robinson (2015) identified depensation occurring in three gull species (black- legged kittiwake, black-headed gull and herring gull). As such it would be very wrong to simply assume that density independent outputs are "highly precautionary", rather that they are the most sensible to use for assessment.	It is acknowledged that the RSPB's stated position on the inclusion of density dependence in population modelling. Indeed the population modelling to which the RSPB makes reference explicitly considered the uncertainties in these aspects of seabird population dynamics and used density dependent methods suggested by RSPB experts. A range of strengths of density dependent regulation were reviewed alongside available evidence and the most realistic ones used in the modelling. In all cases outputs have been provided for both density dependent and density independent models which are considered to bracket the range of probable population projections. It is also acknowledged that density dependence is not always compensatory (as has been used in the population models) however the examples noted by the RSPB all relate to very small populations of these species, and thus are not relevant to the very large populations currently being considered.
		Displacement:	The assessment of red-throated diver displacement (sections 13.7.3.1.2, 13.7.4.1.1 and 13.8.2.6.1) has been conducted

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		Our concerns are principally around the assessment of impacts on red-throated diver (including those of the Greater Wash SPA during construction) and relate to both the methods used in the assessment and the significance of potential impacts. We do not agree that displacement of this species can be considered to result in impacts of minor adverse significance. These impacts should be regarded as of moderate adverse significance.	using accepted methods and with rate of displacement and mortality derived from a detailed review of available evidence. The magnitude and significance of predicted impacts follows the methods as set out in section 13.4.1.
		For red-throated diver, displacement rates of 80% and mortality of 1-5% have been used in the assessment. As there are few robust studies of displacement, results differ, and we do not know the consequences for mortality or population trajectories, it is appropriate to consider a range of putative displacement and mortality rates. The RSPB therefore considers that mortality of up to 10% represents an appropriate level of precaution and should be used in the assessment. We note that this would result in prediction of potentially significant impacts on this species.	The red-throated diver assessment has been updated following a detailed review of evidence presented in relation to the Norfolk Vanguard assessment (Norfolk Vanguard Appendix 3.1- Red-throated diver displacement: Document reference ExA; WQApp3.1;10.D1.3)).
		The annual increase in baseline mortality for red-throated diver is not given, although it is stated that it is unlikely to be detectable. We are concerned that this impact could be significant and therefore request that the annual increase in baseline mortality is presented, based on an assessment using mortality rates of up to 10%.	The red-throated diver assessment presents quantitative details in full (sections 13.7.3.1.2, 13.7.4.1.1 and 13.8.2.6.1).
		Cumulative displacement: Our concerns are principally around the assessment of impacts on red-throated diver, guillemot and razorbill and relate to both the methods used in the assessment and the significance of potential impacts. We do not agree that displacement of these species can be considered to result in impacts of minor adverse significance. These impacts should be regarded as of moderate adverse significance.	The assessment of red-throated diver displacement (sections 13.7.3.1.2, 13.7.4.1.1 and 13.8.2.6.1) and for guillemot and razorbill (sections 13.7.4.1.3 and 13.8.2.6) have been conducted using accepted methods and with rate of displacement and mortality derived from a detailed review of available evidence. The magnitude and significance of predicted impacts follows the methods as set out in section 13.4.1.

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		The assessment of displacement for guillemot and razorbill only considers mortality of 1%, rather than up to 10% as recommended. This, coupled with a failure to present figures for the increase on background mortality (it is only stated that increases are less than 1%), means that we are unable to agree that impacts are of no greater than minor adverse significance.	The assessment of guillemot and razorbill displacement impacts has been informed by an evidence review presented in relation to the Norfolk Vanguard assessment (NV ref). This provides support for the impact rates used derived from available evidence.
Ministry of Infrastructure and Water Management, Netherlands	Email received 14th January 2019	With regard to ornithology we appreciate you took into consideration our earlier comments to Norfolk Vanguard. We also understand your remarks regarding the operational wind parks. But it does not consider the fact that by 2023 4,5 GW of wind parks in the Netherlands will have been built. These volumes can not be ignored when assessing displacement. We understand that there is no international cumulative approach yet.	It is acknowledged that as yet there is no international cumulative approach. As noted in this response, methods for combining impacts from projects assessed in different countries have not been developed. However, the impact assessments for the planned wind farms in the Netherlands have been discussed in section 13.9.

13.4 Assessment Methodology

13.4.1 Impact Assessment Methodology

16. The impact assessment methodology applied in this chapter is based on that described in Chapter 6 EIA Methodology, adapted to make it applicable to ornithology receptors and aligned with the key guidance document produced on impact assessment on ecological receptors (IEEM, 2010). The impact assessment methodology applied in this chapter has also been consulted on with Natural England and RSPB through the scoping report and OETG consultation (details in the project Method Statement, MacArthur Green 2018a and the agreement log, MacArthur Green 2018b) and builds on the approaches adopted for other recent wind farm applications such as East Anglia THREE and Norfolk Vanguard.

17. The assessment approach uses the conceptual 'source-pathway-receptor' model. The model identifies likely environmental impacts resulting from the proposed construction, operation and decommissioning of the offshore infrastructure. This process provides an easy to follow assessment route between impact sources and potentially sensitive receptors, ensuring a transparent impact assessment. The parameters of this model are defined as follows:

18. Source – the origin of a potential impact (noting that one source may have several pathways and receptors) e.g. an activity such as cable installation and a resultant effect such as re-suspension of sediments.

19. Pathway – the means by which the effect of the activity could impact a receptor e.g. for the example above, re-suspended sediment could settle and smother the seabed.

20. Receptor – the element of the receiving environment that is impacted e.g. for the above example, bird prey species living on or in the seabed are unavailable to foraging individuals.

13.4.1.1 Sensitivity

21. Table 13.4 provides example definitions of the different sensitivity levels for ornithology receptors using as its example the potential impact of disturbance through construction activity.

Table 13.4 Definitions of sensitivity levels for ornithological receptors.

Sensitivity	Definition
High	Bird species has <u>very limited</u> tolerance of sources of disturbance such as noise, light, vessel movements and the sight of people.
Medium	Bird species has <u>limited</u> tolerance of sources of disturbance such as noise, light, vessel movements and the sight of people.

Sensitivity	Definition
Low	Bird species has <u>some</u> tolerance of sources of disturbance such as noise, light, vessel movements and the sight of people.
Negligible	Bird species is <u>generally</u> tolerant of sources of disturbance such as noise, light, vessel movements and the sight of people.

22. It should be noted that although sensitivity is a core component of the assessment, conservation value (defined below) is also taken into account in determining each potential impact's significance. Furthermore, high conservation value (defined below) and high sensitivity are not necessarily linked within a particular impact. A receptor could be categorised as being of high conservation value (e.g. an interest feature of a SPA) but have a low or negligible physical/ecological sensitivity to an effect and vice versa. Determination of potential impact significance takes both of these into consideration. The narrative behind the assessment is important here; the conservation value of an ornithological receptor can be used where relevant as a modifier for the sensitivity (to the effect) already assigned to the receptor.

13.4.1.2 Conservation value

23. The conservation value of ornithological receptors is based on the population from which individuals are predicted to be drawn. This reflects current understanding of the movements of species, with site-based protection (e.g. Special Protection Areas, SPA) generally limited to specific periods of the year (e.g. the breeding season). Therefore, conservation value can vary through the year depending on the relative sizes of the number of individuals predicted to be at risk of impact and the population from which they are estimated to be drawn. Ranking therefore corresponds to the degree of connectivity which is predicted between the wind farm site and protected populations. Using this approach, the conservation importance of a species seen at different times of year may fall into any of the defined categories (Table 13.5).

Table 13.5 Definitions of conservation value levels for ornithological receptors.

Value	Definition
High	A species for which individuals at risk can be clearly connected to a particular SPA.
Medium	A species for which individuals at risk are probably drawn from particular SPA populations, although other colonies (both SPA and non-SPA) may also contribute to individuals observed on the wind farm.
Low	A species for which it is not possible to identify the SPAs from which individuals on the wind farm have been drawn, or for which no SPAs are designated.

13.4.1.3 Magnitude

24. The definitions of the magnitude levels for ornithology receptors are set out in Table 13.6. This set of definitions has been determined on the basis of changes to bird populations.

Table 13.6 Definitions of magnitude levels for ornithological receptors.

Magnitude	Definition
High	A change in the size or extent of distribution of the relevant biogeographic population or the population that is the interest feature of a specific protected site that is predicted to irreversibly alter the population in the short-to-long term and to alter the long-term viability of the population and / or the integrity of the protected site. Recovery from that change predicted to be achieved in the long-term (i.e. more than five years) following cessation of the project activity.
Medium	A change in the size or extent of distribution of the relevant biogeographic population or the population that is the interest feature of a specific protected site that occurs in the short and long-term, but which is not predicted to alter the long-term viability of the population and / or the integrity of the protected site. Recovery from that change predicted to be achieved in the medium-term (i.e. no more than five years) following cessation of the project activity.
Low	A change in the size or extent of distribution of the relevant biogeographic population or the population that is the interest feature of a specific protected site that is sufficiently small-scale or of short duration to cause no long-term harm to the feature / population. Recovery from that change predicted to be achieved in the short-term (i.e. no more than one year) following cessation of the project activity.
Negligible	Very slight change from the size or extent of distribution of the relevant biogeographic population or the population that is the interest feature of a specific protected site. Recovery from that change predicted to be rapid (i.e. no more than circa six months) following cessation of the project related activity.
No change	No loss of, or gain in, size or extent of distribution of the relevant biogeographic population or the population that is the interest features of a specific protected site.

13.4.1.4 Impact significance

25. Following the identification of the receptor value and sensitivity and the determination of the magnitude of the effect, the significance of the impact will be determined. That determination will be guided by the matrix as presented in Table 13.7. Impacts shaded red or orange represent those with the potential to be significant in EIA terms.

Table 13.7 Impact significance matrix.

		Negative Magnitude				Beneficial Magnitude			
		High	Medium	Low	Negligible	Negligible	Low	Medium	High
Sensitivity	High	*Major*	*Major*	*Moderate*	*Minor*	*Minor*	*Moderate*	*Major*	*Major*
	Medium	*Major*	*Moderate*	*Minor*	*Minor*	*Minor*	*Minor*	*Moderate*	*Major*
	Low	*Moderate*	*Minor*	*Minor*	*Negligible*	*Negligible*	*Minor*	*Minor*	*Moderate*
	Negligible	*Minor*	*Negligible*	*Negligible*	*Negligible*	*Negligible*	*Negligible*	*Negligible*	*Minor*

26. It is important that the matrix (and indeed the definitions of sensitivity and magnitude) is seen as a framework to aid understanding of how a judgement has been reached from the narrative of each impact assessment and it is not a prescriptive formulaic method. IEEM (2010) guidance and expert judgement has been applied to the assessment of likelihood and ecological significance of a predicted impact.

27. The impact significance categories are divided as shown in Table 13.8.

Table 13.8 Impact significance definitions.

Impact Significance	Definition
Major	Very large or large changes in receptor condition, can be either adverse or beneficial, which are likely to be important considerations at a regional or district level because they contribute to achieving national, regional or local objectives, or, could result in exceedance of statutory objectives and / or breaches of legislation.
Moderate	Intermediate change in receptor condition, which are likely to be important considerations at a local level.
Minor	Small change in receptor condition, which may be raised as local issues but are unlikely to be important in the decision making process.
Negligible	No discernible change in receptor condition.
No change	No impact, therefore no change in receptor condition.

28. Note that for the purposes of this Chapter, major and moderate impacts are considered to be significant. In addition, whilst minor impacts are not significant in their own right, it is important to distinguish these from other non-significant impacts as they may contribute to significant impacts cumulatively or through interactions.

13.4.2 Cumulative Impact Assessment

29. The cumulative impact assessment methodology applied in this Chapter is based on that described in Chapter 6 EIA Methodology, adapted to make it applicable to ornithology receptors.

30. The methodology has also been aligned with the approach to the assessment of cumulative impacts that has been applied by Ministers when consenting offshore wind farms and confirmed in recent consent decisions. It also follows the approach set out in guidance from the Planning Inspectorate (Planning Inspectorate, 2015) and from the renewables industry (RenewableUK, 2013).

13.4.3 Transboundary Impact Assessment

31. The transboundary impact assessment methodology applied in this Chapter is based on that described in Chapter 6 EIA Methodology, adapted to make it applicable to ornithology receptors.

32. The potential for transboundary impacts is identified by consideration of potential linkages to non-UK protected sites and sites with large concentrations of breeding, migratory or wintering birds (including the use of available information on tagged birds).

13.5 Scope

33. This chapter describes the ornithological interests of the Norfolk Boreas site, project interconnector search area and the offshore cable corridor (Figure 5.1) and evaluates the potential impacts of the project on these ornithological interests.

34. The baseline section describes the distribution and abundance of bird species recorded during surveys of the site and draws on additional data as outlined in section 13.5.2.1.

35. The predicted magnitude of impacts and significance of effects arising due to construction, operation and decommissioning of the wind farm on the ornithological interests of the site are assessed on the basis of the worst case project scenario. Measures to prevent or reduce significance of the possible effects are discussed where appropriate. Cumulative impacts arising from the Norfolk Boreas site and offshore cable corridor and other offshore operations are assessed as appropriate.

13.5.1 Survey Area

36. A survey area was defined that was relevant to the consideration of potential impacts on offshore ornithological receptors. The suitability of the survey area for the purpose of environmental impact assessment was agreed through consultation

with Natural England and the RSPB (Offshore Ornithology Agreement Log, 26/02/2018).

37. This survey area includes the Norfolk Boreas site and a 4km buffer (Figure 13.1). Monthly aerial surveys across the survey area commenced in August 2016 and a full 24 months was completed in July 2018. The full 24 month dataset was used for this ES.

38. The data collected during these surveys have been used to identify the species present and their seasonal abundance.

13.5.2 Data Sources

13.5.2.1 Desk based assessment

39. The desk-based assessment has drawn on a wide variety of published literature, covering both peer reviewed scientific literature and the 'grey literature' such as wind farm project submissions and reports. It includes the published literature on seabird ecology and distribution and on the potential impacts of wind farms (both derived from expert judgement and post-construction monitoring studies). The key topics for which the literature has been examined include:

40. Potential impacts of wind farms (Garthe and Hüppop, 2004; Drewitt and Langston, 2006; Stienen et al., 2007; Speakman et al., 2009; Langston, 2010; Band, 2012; Cook et al., 2012; Furness and Wade, 2012; Wright et al., 2012; Furness et al., 2013; Johnston et al., 2014a,b);

- Bird population estimates (Mitchell et al., 2004; BirdLife International 2004; Holling et al. 2011; Holt et al. 2012; Musgrove et al., 2013; Furness, 2015);
- Bird breeding ecology (Cramp and Simmons, 1977-94; Del Hoyo et al., 1992-2011; Robinson, 2005);
- Bird distribution (Stone et al., 1995; Brown and Grice, 2005; Kober et al., 2010);
- Bird migration and foraging movements (Wernham et al., 2002; Thaxter et al., 2012a); and
- Red-throated diver densities in the Outer Thames Estuary SPA (JNCC, 2013), data from an unpublished report on surveys carried out in 2013 by APEM for Natural England and Natural England and JNCC (2016).

41. Owing to the short-term nature and small spatial scale of potential impacts on offshore ornithological receptors from installation of the export cable, no surveys have been conducted along the offshore cable corridor, therefore the above data sources have also been used to inform the baseline characterisation and impact assessment for cable installation.

42. Information on statutory sites and their interest features has been drawn from the web-based resource Multi-Agency Geographic Information for the Countryside (MAGIC www.magic.defra.gov.uk) and the Natural England and JNCC web sites (www.naturalengland.org.uk; www.jncc.defra.gov.uk).

13.5.2.2 Site specific surveys

43. To assess the temporal and spatial abundance and distribution of birds, digital aerial surveys were conducted by APEM Ltd across the survey area. Further details of how these surveys were carried out, how the images acquired were analysed and the results of the surveys are provided in Technical Appendix 13.1.

13.5.3 Assumptions and Limitations

44. The marine environment is highly variable, both spatially and temporally. The baseline site characterisation for this ES is based on two years of survey data which are considered to be representative of the site for the purpose of impact assessment. Given the project's location (beyond the foraging range of most breeding seabirds) and the results obtained from surveys conducted for other wind farm applications in the former East Anglia Zone (e.g. Norfolk Vanguard, East Anglia THREE, East Anglia ONE, zonal surveys, etc.), the data are considered to be consistent with previous survey results.

13.6 Existing Environment

45. This Section details the baseline ornithological information based on the desk-based assessment and the surveys listed above and detailed in Technical Appendix 13.1.

46. A summary of the ornithological receptors potentially affected by the offshore components is provided at the end of this section in Table 13.10.

13.6.1 Statutory Designated Sites

47. Four classes of statutory designated sites that can have birds included as interest features are considered in this section: SPAs, pSPAs, Ramsar sites and SSSIs (Figures 13.2, 13.3, 13.4).

48. Statutory designated sites have been considered in this assessment on the basis of their potential connectivity to the project. These sites can be broadly separated into those designated for their breeding seabird interests and those for their terrestrial / coastal / marine bird interests (typically for overwintering aggregations).

49. Seabird breeding sites may be connected during the breeding season (e.g. the wind farm lies within foraging range of breeding birds) or during the non-breeding season (e.g. birds pass through during spring and autumn migration or are present overwinter), or during both periods.

50. Terrestrial / coastal sites designated for migrant species outside the breeding season may be connected on the grounds of passage movements through the wind farm.

51. Those sites that have been identified for potential connectivity are listed in Table 13.9 and detailed in Appendix 10.3 In each case their ornithological interest features are listed. The legal process of the designation of SPAs and Ramsar sites in the UK means that, other than marine sites, each SPA and Ramsar site is supported by a complementary SSSI, or sometimes several separate SSSI, that cover the same area (sometimes the SSSI may cover a larger area because of SSSI interest features that are not relevant to the international designation).

52. The assessment of likely significant effects on the interest features of the internationally designated sites (SPAs and Ramsar sites) is carried out through the Habitats Regulations Assessment (HRA) process and this will be reported separately in the Information for the Habitats Regulations Assessment report (document reference 5.3) which has been submitted as part of the DCO application.

Table 13.9 SPAs, Ramsar sites and SSSI with potential for connectivity to Norfolk Boreas. Ornithological Interest Features and minimum distance to Norfolk Boreas, listed in increasing distance.

Site	Designation	Ornithological interest features with potential for connectivity to Norfolk Boreas	Minimum distance to the project (km)
Greater Wash	SPA	Classified for its populations of breeding and non-breeding (wintering and migration) bird populations. But note that the cable route will pass through this SPA.	59
Outer Thames Estuary	SPA / pSPA	A marine SPA classified for its non-breeding populations of seabirds.	40
Winterton-Horsey Dunes	SSSI	Notified for its populations of breeding and non-breeding (wintering and migration) bird populations.	73
Great Yarmouth and North Denes	SPA, SSSI	Classified for its populations of breeding seabirds.	73
Breydon Water	SPA, Ramsar	Classified for its populations of wintering and passage waterbirds.	76
	SSSI	Notified for its populations of breeding and non-breeding (wintering and migration) populations of waders and wildfowl.	
Broadland	SPA	Classified for its populations of wintering and passage waterbirds.	76
Pakefield to Easton Bavents	SSSI	Notified for its populations of breeding and non-breeding (wintering and migration) bird populations.	89
Benacre-Easton Bavents	SPA	Classified for its breeding and non-breeding bird populations	89
Minsmere-Walberswick Heaths and Marshes	SSSI	Notified for its populations of breeding and non-breeding (wintering and migration) populations of waders and wildfowl.	96
Minsmere - Walberswick	SPA, Ramsar	Classified for its populations of breeding, wintering and passage waterbirds.	96

Site	Designation	Ornithological interest features with potential for connectivity to Norfolk Boreas	Minimum distance to the project (km)
Sizewell Marshes	SSSI	Notified for its populations of breeding birds.	105
Waddenzee (Netherlands)	SPA	A coastal SPA classified for breeding and non-breeding seabirds, waterbirds and a raptor species.	105
Alde-Ore Estuary	SPA, Ramsar	Classified for its populations of breeding marsh harrier and breeding and non-breeding waterbirds.	117
	SSSI	Notified for its populations of breeding and non-breeding (wintering and migration) populations of waders and wildfowl.	
Voordelta (Netherlands)	SPA	A marine and coastal SPA classified for non-breeding seabirds and waterbirds.	118
Deben Estuary	SPA, Ramsar	Classified for its populations of non-breeding waterbirds, including population of Brent goose at levels of international importance.	128
	SSSI	Notified for its populations of breeding and overwintering waders and wildfowl.	
Orwell Estuary	SSSI	Notified for its populations of breeding and non-breeding (wintering and migration) populations of waders and wildfowl.	140
Stour & Orwell Estuaries	SPA, Ramsar	Classified for its populations of wintering and passage waterbirds.	140
North Norfolk Coast	SPA	Classified for its populations of wintering and passage waterbirds.	142
	SSSI	Notified for its populations of breeding and non-breeding (wintering and migration) bird populations.	
Stour Estuary	SSSI	Notified for its populations of non-breeding (wintering and migration) birds.	143
Hamford Water	SPA	Classified for its populations of wintering and passage waterbirds.	146
	SSSI	Notified for its populations of breeding and non-breeding (wintering and migration) bird populations.	
The Wash	SPA	Classified for its populations of wintering and passage waterbirds.	150
	SSSI	Notified for its populations of breeding and non-breeding (wintering and migration) bird populations.	
Hunstanton Cliffs	SSSI	Notified for its populations of breeding birds.	151
Cattawade Marshes	SSSI	Notified for its populations of breeding waders and wildfowl.	154
Holland Haven Marshes	SSSI	Notified for its populations of breeding and non-breeding (wintering and migration) bird populations.	156
Gibraltar Point	SPA, Ramsar	Classified for its populations of wintering and passage waterbirds.	161
	SSSI	Notified for its populations of breeding and non-breeding (wintering and migration) bird populations.	
Colne Estuary	SPA, Ramsar	Classified for its populations of wintering and passage waterbirds.	164
	SSSI	Notified for its populations of breeding and non-breeding (wintering and migration) bird populations.	

Site	Designation	Ornithological interest features with potential for connectivity to Norfolk Boreas	Minimum distance to the project (km)
Upper Colne Marshes	SSSI	Notified for its populations of breeding and non-breeding (wintering and migration) populations of waders and wildfowl.	166
Saltfleetby – Theddlethorpe Dunes	SSSI	Notified for its populations of breeding and non-breeding (wintering and migration) populations of wildfowl and waders.	170
Abberton Reservoir	SPA, Ramsar	Classified for its populations of wintering and passage waterbirds.	171
	SSSI	Notified for its populations of breeding and non-breeding (wintering and migration) populations.	
Dengie	SPA, Ramsar	Classified for its populations of wintering and passage waterbirds.	175
	SSSI	Notified for its populations of breeding and non-breeding (wintering and migration) bird populations.	
Banc des Flandres	SPA	Classified for its population of breeding little tern and non-breeding (wintering and migration) bird populations.	177
Blackwater Estuary	SPA, Ramsar	Classified for its populations of wintering and passage waterbirds.	185
	SSSI	Notified for its populations of breeding and non-breeding (wintering and migration) bird populations.	
The Lagoons	SSSI	Notified for its populations of breeding and non-breeding (wintering and migration) bird populations.	185
Foulness	SPA, Ramsar	Classified for its populations of wintering and passage waterbirds.	186
	SSSI	Notified for its populations of breeding and non-breeding (wintering and migration) waders and wildfowl populations.	
Crouch & Roach Estuary	SPA	Classified for its populations of wintering and passage waterbirds.	187
	SSSI	Notified for its populations of breeding and non-breeding (wintering and migration) bird populations.	
Thanet Coast	SSSI	Notified for its populations of breeding and non-breeding (wintering and migration) populations of waders and wildfowl.	187
Thanet Coast and Sandwich Bay	SPA, Ramsar	Classified for its populations of wintering and passage waterbirds.	187
Humber Estuary	SPA, Ramsar, SSSI	Classified for its populations of wintering and passage waterbirds.	190
Benfleet & Southend Marshes	SPA, Ramsar	Classified for its populations of wintering and passage waterbirds.	202
	SSSI	Notified for its populations of non-breeding (wintering and migration) populations of waders and wildfowl.	
The Swale	SPA	Classified for its populations of wintering and passage waterbirds.	205
	SSSI	Notified for its populations of breeding and non-breeding (wintering and migration) populations of waders and wildfowl.	
Thames Estuary and Marshes	SPA, Ramsar	Classified for its populations of wintering and passage waterbirds.	210

Site	Designation	Ornithological interest features with potential for connectivity to Norfolk Boreas	Minimum distance to the project (km)
Medway Estuary & Marshes	SPA	Classified for its populations of wintering and passage waterbirds.	210
	SSSI	Notified for its populations of breeding and non-breeding (wintering and migration) populations of waders and wildfowl.	
South Thames Estuary and Marshes	SSSI	Notified for its populations of breeding and non-breeding (wintering and migration) bird populations.	211
Pitsea Marsh	SSSI	Notified for its populations of breeding and non-breeding (wintering and migration) bird populations.	211
Vange and Fobbing Marshes	SSSI	Notified for its population of breeding and non-breeding (wintering and migration) bird population.	212
Holehaven Creek	SSSI	Notified for its populations of non-breeding (wintering) birds.	212
Hornsea Mere	SPA	Classified for its populations of wintering and passage waterbirds.	215
	SSSI	Notified for its populations of breeding and non-breeding (wintering and migration) bird populations.	
Flamborough and Filey Coast	SPA	Classified for its populations of breeding seabirds.	216
Mucking Flats and Marshes	SSSI	Notified for its populations of non-breeding (wintering and migration) and passage bird populations.	218
Borkum-Riffgrund (Germany)	SPA	A marine SPA classified for its non-breeding populations of seabirds.	218
Flamborough Head	SSSI	Notified for its populations of breeding birds.	219
Caps Gris Nez	SPA	Classified for its population of non-breeding (wintering and migration) bird populations.	225
Filey Brigg	SSSI	Notified for its population of non-breeding (wintering and migration) birds	235
Sylter Außenriff (Germany)	SPA	A marine SPA classified for its non-breeding seabirds.	286
Teesmouth and Cleveland Coast	SPA, Ramsar	Classified for its populations of wintering and passage waterbirds.	301
Northumbria Coast	SPA, Ramsar	Classified for its populations of wintering and passage waterbirds.	319
Östliche Deutsche Bucht (Germany)	SPA	A marine SPA classified for its populations of non-breeding seabirds.	328
Littoral Seino-Marin (France)	SPA	A marine, coastal and terrestrial SPA classified for its breeding seabirds and a raptor and non-breeding seabirds, waterbirds and a raptor.	329
Seevogelschutz gebiet Helgoland (Germany)	SPA	A marine and island SPA classified for its populations of breeding and non-breeding seabirds.	329

Site	Designation	Ornithological interest features with potential for connectivity to Norfolk Boreas	Minimum distance to the project (km)
Chichester & Langstone Harbour	SPA	Classified for its populations of migratory waterbirds.	340
Portsmouth Harbour	SPA	Classified for its populations of migratory waterbirds.	347
Solent & Southampton Water	SPA	Classified for its populations of migratory waterbirds.	351
Ramsar-Gebiet S-H Wattenmeer und angrenzende Küstengebiete (Germany)	SPA	A coastal SPA classified for its breeding, wintering and passage waterbirds, other migrant species and Annex 1 species (82 species listed).	355
Coquet Island	SPA	Classified for its populations of breeding seabirds.	373
Farne Islands	SPA	Classified for its populations of breeding seabirds.	397
Lindisfarne	SPA, Ramsar	Classified for its populations of wintering and passage waterbirds.	403
Chesil Beach & The Fleet SPA	SPA	Classified for its populations of migratory waterbirds.	441
St Abbs Head to Fast Castle	SPA	Classified for its populations of breeding seabirds.	441
Baie de Seine Occidentale (France)	SPA	A coastal SPA classified for its populations of breeding and non-breeding seabirds and waterbirds.	447
Falaise du Bessin Occidental (France)	SPA	A marine, coastal and terrestrial SPA classified for its breeding populations of seabirds and a passerine and non-breeding populations of seabirds and raptors.	463
Firth of Forth	SPA	Classified for its populations of wintering and passage waterbirds.	468
Forth Islands (Fife/East Lothian)	SPA	Classified for its populations of breeding seabirds.	476
Exe Estuary	SPA	Classified for its populations of migratory waterbirds.	491
Imperial Dock Lock, Leith	SPA	Classified for its populations of breeding seabirds.	498
Firth of Tay & Eden Estuary	SPA	Classified for its populations of wintering and passage waterbirds.	506
Montrose Basin	SPA	Classified for its populations of wintering and passage waterbirds.	520
Fowlsheugh	SPA	Classified for its populations of breeding seabirds.	524
Ythan Estuary, Sands of Forvie and Meikle Loch	SPA	Classified for its populations of wintering and passage waterbirds.	553
Buchan Ness to Colleston Coast	SPA	Classified for its populations of breeding seabirds.	553

Site	Designation	Ornithological interest features with potential for connectivity to Norfolk Boreas	Minimum distance to the project (km)
Loch of Strathbeg	SPA	Classified for its populations of wintering and passage waterbirds.	576
Troup, Pennan and Lion`s Heads	SPA	Classified for its populations of breeding seabirds.	593
Moray and Nairn Coast	SPA	Classified for its populations of wintering and passage waterbirds.	622
Inner Moray Firth	SPA	Classified for its populations of wintering and passage waterbirds.	652
Cromarty Firth	SPA	Classified for its populations of wintering and passage waterbirds.	664
Dornoch Firth and Loch Fleet	SPA	Classified for its populations of wintering and passage waterbirds.	668
East Caithness Cliffs	SPA	Classified for its populations of breeding seabirds.	682
North Caithness Cliffs	SPA	Classified for its populations of breeding seabirds.	703
Pentland Firth Islands	SPA	Classified for its populations of breeding seabirds.	710
Copinsay (Orkney)	SPA	Classified for its populations of breeding seabirds.	718
Hoy (Orkney)	SPA	Classified for its populations of breeding seabirds.	728
Calf of Eday (Orkney)	SPA	Classified for its populations of breeding seabirds.	753
Fair Isle (Shetland)	SPA	Classified for its populations of breeding seabirds.	750
Rousay (Orkney)	SPA	Classified for its populations of breeding seabirds.	756
Marwick Head (Orkney)	SPA	Classified for its populations of breeding seabirds.	761
West Westray (Orkney)	SPA	Classified for its populations of breeding seabirds.	766
Papa Westray (North Hill and Holm) (Orkney)	SPA	Classified for its populations of breeding seabirds.	770
Sumburgh Head (Shetland)	SPA	Classified for its populations of breeding seabirds.	778
Mousa (Shetland)	SPA	Classified for its populations of breeding seabirds.	793
Noss (Shetland)	SPA	Classified for its populations of breeding seabirds.	802
Foula (Shetland)	SPA	Classified for its populations of breeding seabirds.	822
Papa Stour (Shetland)	SPA	Classified for its populations of breeding seabirds.	839
Fetlar (Shetland)	SPA	Classified for its populations of breeding seabirds.	844
Ronas Hill - North Roe and Tingon (Shetland)	SPA	Classified for its populations of breeding seabirds.	852

Site	Designation	Ornithological interest features with potential for connectivity to Norfolk Boreas	Minimum distance to the project (km)
Hermaness, Saxa Vord and Valla Field (Shetland)	SPA	Classified for its populations of breeding seabirds.	866

13.6.2 Baseline Environment and Assessment of Nature Conservation Value for Each Bird Species

13.6.2.1 Seabirds

53. The bird abundance estimates and how they were derived are presented in detail in the Ornithology Technical Appendix (13.1). Detail from this report has not been repeated in this chapter to minimise unnecessary repetition. Bird abundances and assemblages have been estimated from the site-specific surveys of Norfolk Boreas.

54. Species assessed for impacts are those which were recorded during surveys and which are considered to be at potential risk either due to their abundance, potential sensitivity to wind farm impacts or due to biological characteristics which make them potentially susceptible (e.g. commonly fly at rotor heights). The conservation status of these species is provided in Table 13.10. The locations of all species observed are plotted on figures in Technical Appendix 13.1 Annex 8.

Table 13.10 Summary of nature conservation value of species considered at risk of impacts.

Species	Conservation status
Red-throated diver	BoCC Green listed, Birds Directive Migratory Species, Birds Directive Annex 1
Black-throated diver	BoCC Amber listed, Birds Directive Migratory Species, Birds Directive Annex 1
Great northern diver	BoCC Amber listed, Birds Directive Migratory Species, Birds Directive Annex 1
Fulmar	BoCC Amber listed, Birds Directive Migratory Species
Gannet	BoCC Amber listed, Birds Directive Migratory Species
Arctic skua	BoCC Red listed, Birds Directive Migratory Species
Great skua	BoCC Amber listed, Birds Directive Migratory Species
Puffin	BoCC Red listed, Birds Directive Migratory Species
Razorbill	BoCC Amber listed, Birds Directive Migratory Species
Common guillemot	BoCC Amber listed, Birds Directive Migratory Species
Common tern	BoCC Amber listed, Birds Directive Migratory Species, Birds Directive Annex 1
Arctic tern	BoCC Amber listed, Birds Directive Migratory Species, Birds Directive Annex 1
Kittiwake	BoCC Red listed, Birds Directive Migratory Species
Little gull	BoCC Green listed, Birds Directive Migratory Species

Species	Conservation status
Lesser black-backed gull	BoCC Amber listed, Birds Directive Migratory Species
Herring gull	BoCC Red listed, Birds Directive Migratory Species
Great black-backed gull	BoCC Amber listed, Birds Directive Migratory Species

55. Impacts have been assessed in relation to relevant biological seasons, as defined by Furness (2015). For the non-breeding period, the seasons and relevant population sizes for Biologically Defined Minimum Population Scales (BDMPS) were taken from Furness (2015) (Table 13.11). For the breeding period, the potential for connectivity to known breeding populations has been considered. However, it should be noted that bird abundance was low for all species during the breeding season, with many species absent in one or more of the summer months. This indicated that very few breeding birds utilise the Norfolk Boreas site.

56. The seasonal definitions in Furness (2015) include overlapping months in some instances due to variation in the timing of migration for birds which breed at different latitudes (i.e. individuals from breeding sites in the north of the species' range may still be on spring migration when individuals farther south have already commenced breeding). Due to the very low presence of breeding birds it was considered appropriate to define breeding as the migration-free breeding period (see Table 13.11), sometimes also referred to as the core breeding period. This ensured that any late or early migration movements which were observed were assessed in relation to the appropriate reference populations. One exception to this was lesser black-backed gull, for which there is potential that breeding adults from the Alde Ore Estuary SPA population may forage on the Norfolk Boreas site. Hence for this species the full breeding season was applied in the attribution of potential impacts to relevant populations.

Table 13.11 Species specific seasonal definitions and biologically defined minimum population sizes (in brackets) have been taken from Furness (2015). Shaded cells indicate the appropriate non-breeding season periods used in the assessment for each species.

Species	Breeding	Migration-free breeding	Migration - autumn	Winter	Migration - spring	Non-breeding
Red-throated diver	Mar-Aug	May-Aug	Sep-Nov (13,277)	Dec-Jan (10,177)	Feb-Apr (13,277)	
Black-throated diver*	Apr-Aug	May-Aug				Aug-Apr
Great northern diver	-	-	Sep-Nov	Dec-Feb	Mar-May	Sep-May (200)
Fulmar	Jan-Aug	Apr-Aug	Sep-Oct (957,502)	Nov (568,736	Dec-Mar (957,502)	-
Gannet	Mar-Sep	Apr-Aug	Sep-Nov (456,298)	-	Dec-Mar (248,385)	-
Cormorant	Apr-Aug	May-Jul	-	-	-	Sep-Mar

Species	Breeding	Migration-free breeding	Migration - autumn	Winter	Migration - spring	Non-breeding
						(10,460)
Shag	Feb-Aug	Mar-July	-	-	-	Sep-Jan (4,346)
Arctic skua	May-Jul	Jun-Jul	Aug-Oct (6,427)	-	Apr-May (1,227)	-
Great skua	May-Aug	May-Jul	Aug-Oct (19,556)	Nov-Feb (143)	Mar-Apr (8,485)	-
Puffin	Apr-Aug	May-Jun	Jul-Aug	Sep-Feb	Mar-Apr	Mid-Aug-Mar (231,957)
Razorbill	Apr-Jul	Apr-Jun	Aug-Oct (591,874)	Nov-Dec (218,622)	Jan-Mar (591,874)	-
Guillemot	Mar-Jul	Mar-Jun	Jul-Oct	Nov	Dec-Feb	Aug-Feb (1,617,306)
Sandwich tern	Apr-Aug	Jun	Jul-Sep (38,051)	Oct-Feb	Mar-May (38,051)	Sep-Mar
Commic tern**	May-Aug	Jun	Jul-Sep (308,841)	-	Apr-May (308,841)	-
Kittiwake	Mar-Aug	May-Jul	Aug-Dec (829,937)	-	Jan-Apr (627,816)	-
Little gull (Not included in Furness 2015)	Apr-Jul	May-Jul	-	-	-	Aug-Apr
Lesser black-backed gull	Apr-Aug	May-Jul	Aug-Oct (209,007)	Nov-Feb (39,314)	Mar-Apr (197,483)	-
Herring gull	Mar-Aug	May-Jul	Aug-Nov	Dec	Jan-Apr	Sep-Feb (466,511)
Great black-backed gull	Mar-Aug	May-Jul	Aug-Nov	Dec	Jan-Apr	Sep-Mar (91,399)

* Not included in Furness (2015). Natural England (2012) states: Breeding black-throated divers migrate to saltwater habitats from August, returning to their breeding sites from April. Birds are also seen in small numbers on eastward passage through the English Channel in April and May.

** Commic tern' is used to include common terns and Arctic terns, as these species are not readily identified to species level from the survey data

57. In addition to BDMPS populations, the biogeographic populations have also been considered in the assessment where appropriate. These are provided in Table 13.12.

Table 13.12 Biogeographic population sizes taken from Furness (2015).

Species	Biogeographic population with connectivity to UK waters (adults and immatures)
Red-throated diver	27,000
Black-throated diver (not included in Furness 2015)	56,460*
Great northern diver	430,000
Fulmar	8,055,000
Gannet	1,180,000
Cormorant	324,000
Shag	106,000
Arctic skua	229,000
Great skua	73,000

Species	Biogeographic population with connectivity to UK waters (adults and immatures)
Puffin	11,840,000
Razorbill	1,707,000
Guillemot	4,125,000
Commic tern**	628,000 (Arctic: 480,000; Common: 248,000)
Kittiwake	5,100,000
Great black-backed gull	235,000
Herring gull	1,098,000
Lesser black-backed gull	864,000
Little gull (not included in Furness 2015)	75,000 #

* JNCC (http://jncc.defra.gov.uk/pdf/UKSPA/UKSPA-A6-2.pdf). Note this figure has been calculated as 19,196 breeding pairs multiplied by 2 and divided by the estimated proportion of adults in the population (0.68).

\# Estimated passage population (Steinen et al., 2007)

** 'Commic tern' is used to include common terns and Arctic terns, as these species are not readily identified to species level from the survey data

58. The impact of additional mortality due to wind farm effects is assessed in terms of the change in the baseline mortality rate which could result. It has been assumed that all age classes are equally at risk of effects, with each age class affected in proportion to its presence in the population. Therefore, a weighted average baseline mortality rate has been calculated which is appropriate for all age classes for use in assessments, calculated for those species screened in for assessment (see section 13.7). These were calculated using the different rates for each age class and their relative proportions in the population.

59. Demographic rates for each species were taken from Horswill and Robinson (2015) and entered into a matrix population model. This was used to calculate the expected stable proportions in each age class (note, to obtain robust stable age class distributions for less well studied species such as divers it was necessary to adjust the rates in order to obtain a stable population size). Each age class survival rate was multiplied by its stable age proportion and the total for all ages summed to give the weighted average survival rate for all ages. Taking this value from 1 gives the average mortality rate. The demographic rates and the age class proportions and average mortality rates calculated from them are presented in Table 13.13.

Table 13.13 Average mortality across all age classes. Average mortality calculated using age specific demographic rates and age class proportions.

Species	Parameter	Survival (age class)						Productivity	Average mortality
		0-1	1-2	2-3	3-4	5-6	Adult		
Red-throated diver	Demographic rate	0.6	0.62	-	-	-	0.84	0.571	0.228
	Population age ratio	0.179	0.145	-	-	-	0.676	-	
Gannet	Demographic rate	0.424	0.829	0.891	0.895	-	0.912	0.7	0.191

Species	Parameter	Survival (age class)						Productivity	Average mortality
		0-1	1-2	2-3	3-4	5-6	Adult		
	Population age ratio	0.191	0.081	0.067	0.06	-	0.6	-	
Guillemot	Demographic rate	0.56	0.792	0.917	0.939	0.939	0.939	0.672	0.14
	Population age ratio	0.168	0.091	0.069	0.062	0.056	0.552	-	
Razorbill[1]	Demographic rate	0.63	0.63	0.895	0.895	-	0.895	0.57	0.174
	Population age ratio	0.159	0.102	0.065	0.059	-	0.613	-	
Puffin[2]	Demographic rate	0.709	0.709	0.76	0.805	-	0.906	0.617	0.167
	Population age ratio	0.162	0.115	0.082	0.063	-	0.577	-	
Common tern[3]	Demographic rate	0.441	0.441	0.85	-	-	0.883	0.764	0.263
	Population age ratio	0.223	0.103	0.048	-	-	0.626	-	
Kittiwake	Demographic rate	0.79	0.854	0.854	0.854		0.854	0.69	0.156
	Population age ratio	0.155	0.123	0.105	0.089		0.53	-	
Lesser black-backed gull	Demographic rate	0.82	0.885	0.885	0.885		0.885	0.53	0.124
	Population age ratio	0.134	0.109	0.095	0.083		0.579	-	
Herring gull	Demographic rate	0.798	0.834	0.834	0.834		0.834	0.92	0.172
	Population age ratio	0.178	0.141	0.117	0.097		0.467		
Great black-backed gull	Demographic rate	0.815	0.815	0.815	0.815		0.885	0.53	0.144
	Population age ratio	0.137	0.112	0.093	0.076		0.581	-	

1 – Razorbill have a combined survival rate from 0 – 2 of 0.63, giving an annual rate of 0.79.
2 – Puffin have a combined survival rate from 0 – 3 of 0.709, giving an annual rate of 0.89
3 – Common tern have a combined survival rate from 0 – 2 of 0.441, giving an annual rate of 0.66. Note that the rates for common tern have been used for the commic tern assessment where necessary.

60. The seasonal peak abundance within species specific seasons (as defined in Table 13.11) recorded individually within the Norfolk Boreas site are provided in Table 13.14 (note these abundances do not include birds observed in the 4km buffer around the site boundaries).

52. The method to calculate the seasonal peaks for Norfolk Boreas was as follows:

- The population density and abundance for each survey was calculated using design-based estimation methods, with 95% confidence intervals calculated using non-parametric bootstrapping (see Technical Appendix 13.1 for further details).

- The abundance for each calendar month was calculated as the mean of estimates for each month (i.e. the mean of two survey values per month).

The seasonal peak was taken as the highest from the months falling within each season. In some cases the peak was recorded in a month which is included in overlapping seasons and therefore the same value has been identified in both seasons. These have been identified in italics in Table 13.14.

Table 13.14 Seasonal peak population and 95% confidence intervals within the Norfolk Boreas site (not including buffer). The population size in each calendar month was calculated as the mean of the individual surveys conducted in that month and the values shown in the table are the highest from all months in each season. Figures in italics identify occasions when the same peak was recorded in different seasons due to overlapping months.

Species	Breeding		Migration-free breeding		Migration - autumn		Winter		Migration - spring		Non-breeding	
	Seasonal peak	95% c.i.	Seasonal peak	95% c.i.	Seasonal peak	95% c.i.	Seasonal peak	95% c.i.	Seasonal peak	95% c.i.	Seasonal peak	95% c.i.
Red-throated diver	450.9	0-1052.3	34.7	0-115.8	11.6	0-57.9	81.4	23.1-152.1	*450.9*	*0-1052.3*	-	-
Great northern diver	-	-	-	-	0	0-0	5.9	0-35.1	0	0-0	5.9	0-35.1
Fulmar	*814.8*	*46.8-1743.8*	306.4	196.3-427.3	972.6	289.5-1759.9	46.5	0-104.8	*814.8*	*46.8-1743.8*	-	-
Gannet	*1172.2*	*0-2621.5*	*1172.2*	*0-2621.5*	1201.0	803.3-1644.1	-	-	395.4	269.2-531.9	-	-
Cormorant	0	0-0	0	0-0	*34.7*	*0-127.4*	5.9	0-35.1	0	0-0	*34.7*	*0-127.4*
Shag	0	0-0	0	0-0	*34.7*	*0-115.8*	0	0-0	0	0-0	*34.7*	*0-115.8*
Arctic skua	0	0-0	0	0-0	17.4	0-69.5	-	-	0	0-0	-	-
Great skua	0	0-0	0	0-0	57.9	11.6-127.4	5.9	0-35.1	0	0-0	-	-
Puffin	0	0-0	0	0-0	0	0-0	0	0-0	*23.1*	*0-80.9*	*23.1*	*0-80.9*
Razorbill	*472.9*	*222.6-777.2*	*374.9*	*13.9-864.1*	250.9	83.5-460.3	687.7	222.9-1283.0	290.4	55.7-585.9	-	-
Guillemot	*6292.3*	*1095.5-11864.9*	*1782.3*	*956.9-2966.9*	6292.3	1095.5-11864.9	3442.6	1005.5-6188.5	*10480.2*	*5538.2-15613.2*	*10480.2*	*5538.2-15613.2*
Sandwich tern	*11.5*	*0-46.1*	5.8	0-34.6	*11.5*	*0-46.1*	-	-	11.6	0-34.7	-	-
Commic tern	*347.3*	*23.2-752.6*	0	0-0	213.1	0-541.4	-	-	*347.3*	*23.2-752.6*	-	-
Kittiwake	*499.3*	*277.87-753.1*	*499.3*	*277.9-753.1*	1822.6	1132.3-2586.2	-	-	764.1	198.9-1455.0	-	-
Black-headed gull	-	-	28.3	0-85.2	-	-	-	-	-	-	271.6	0-655.1
Little gull	*201.3*	*0-513.5*	*201.3*	*0-513.5*	-	-	-	-	-	-	65.0	0-186.0
Common gull	-	-	19.3	0-57.8	-	-	-	-	-	-	81.4	23.4-151.1

Species	Breeding		Migration-free breeding		Migration - autumn		Winter		Migration - spring		Non-breeding	
	Seasonal peak	95% c.i.	Seasonal peak	95% c.i.	Seasonal peak	95% c.i.	Seasonal peak	95% c.i.	Season al peak	95% c.i.	Seasonal peak	95% c.i.
Lesser black-backed gull	*1679.8*	*57.8-3560.1*	*1679.8*	*57.8-3560.1*	541.0	57.7-1137.2	87.0	11.7-186.8	17.3	0-69.4	-	-
Herring gull	*124.2*	*11.6-296.2*	*124.2*	*11.6-296.2*	194.0	11.3-465.0	*484.1*	*222.3-797.9*	70.0	0-176.4	*194.0*	11.3-465.0
Great black-backed gull	*98.2*	*0-242.5*	*98.2*	*0-242.5*	*1239.0*	*527.6-2081.0*	554.0	210.6-977.2	593.8	11.7-1327.0	*1239.0*	*527.6-2081.0*

* Combined population presented due to difficulty of separating common and Arctic tern species in survey data.

61. The following sections provide a summary of the observations for each species with reference to the offshore wind farm site, and offshore cable corridor (where relevant). The population estimates provided are those estimated from data within the site boundary, not including the buffer.

62. Note that some species, such as skuas, terns and little gull are likely to be poorly represented in the survey data (e.g. due to infrequent passage movements) and therefore the impact assessments for these species draw on additional sources of information with regards their anticipated movements and utilise methods developed for migratory species (e.g. WWT & MacArthur Green, 2013).

13.6.2.1.1 *Red-throated diver*

63. Red-throated divers were recorded on the Norfolk Boreas site between November and May. The peak abundance was estimated in March (451), coinciding with the period of spring migration to breeding sites. The species was absent between June and October.

64. The offshore cable corridor will pass through the proposed Greater Wash SPA. This marine SPA includes nonbreeding red-throated diver as a feature. Aerial surveys of the SPA have recorded moderate numbers of red-throated divers in the vicinity of the offshore cable corridor with densities of around one to two birds per km^2 (Natural England and JNCC, 2016).

13.6.2.1.2 *Great Northern diver*

65. Great Northern divers were recorded in January on the Norfolk Boreas site coinciding with the winter season with a population of 6.

13.6.2.1.3 *Fulmar*

66. Fulmars were recorded in all months on the Norfolk Boreas site. There was no clear pattern in abundance across the year, with a peak estimated population of 973 in September.

13.6.2.1.4 *Gannet*

67. Gannets were recorded in all months on the Norfolk Boreas site. Numbers were generally low in most months except the peaks in August (1,172) and November (1,201).

13.6.2.1.5 *Cormorant*

68. Cormorants were recorded in October and December on the Norfolk Boreas site. The peak estimated population was recorded in October (35).

13.6.2.1.6 *Shag*

69. Shags were recorded in September on the Norfolk Boreas site, with a population of 35.

13.6.2.1.7 *Arctic skua*

70. Arctic skuas were recorded on the Norfolk Boreas site in August and September, with a peak estimated population of 17 individuals in August. This pattern is consistent with post-breeding migration through the region.

13.6.2.1.8 *Great skua*

71. Great skuas were recorded on the Norfolk Boreas site in September, October and December with a peak estimated population of 58 individuals in September. This pattern is consistent with occasional autumn migrants passing through the region.

13.6.2.1.9 *Puffin*

72. Puffins were recorded in March on the Norfolk Boreas site. The estimated peak population was 23.

13.6.2.1.10 *Razorbill*

73. Razorbills were recorded on the Norfolk Boreas site in all months. The estimated peak population was 688 in December (accounting for availability bias).

13.6.2.1.11 *Guillemot*

74. Guillemots were recorded in all months on the Norfolk Boreas site, with an estimated peak population of 10,480 in December (accounting for availability bias).

13.6.2.1.12 *Sandwich tern*

75. Sandwich terns were recorded on the Norfolk Boreas site in March, May, June, July and September. The estimated peak population was 12 individuals, recorded in both March and July.

13.6.2.1.13 *Commic tern*

76. Common and/or Arctic terns were recorded on Norfolk Boreas site in May, July and August. The estimated peak population was 347 in May.

13.6.2.1.14 *Kittiwake*

77. Kittiwakes were recorded on the Norfolk Boreas site in all months, with higher numbers between November and January. The estimated peak population was 1,822 in December (including an allocated proportion of unidentified small gulls).

13.6.2.1.15 *Black-headed gull*

78. Black-headed gulls were recorded on the Norfolk Boreas site in January, March, July, September and December. The peak population was 272 individuals in March (including an allocated proportion of unidentified small gulls).

13.6.2.1.16 *Little gull*

79. Little gulls were recorded on the Norfolk Boreas site March, April, May, October and November. The estimated peak population was 201 individuals in May (including an allocated proportion of unidentified small gulls).

13.6.2.1.17 *Common gull*

80. Common gulls were recorded on the Norfolk Boreas site in all months except May and June. The estimated peak population was 81 individuals in December (including an allocated proportion of unidentified small gulls).

13.6.2.1.18 *Lesser black-backed gull*

81. Lesser black-backed gulls were recorded on the Norfolk Boreas site in all months. Highest numbers occurred post-breeding, with a peak in July of 1,680 (including an allocated proportion of unidentified black-backed and large gulls).

13.6.2.1.19 *Herring gull*

82. Herring gulls were recorded on the Norfolk Boreas site in all months. The estimated population was low in most months, but with higher numbers in mid-winter. The peak estimated population was 484 in December (including an allocated proportion of unidentified large gulls).

13.6.2.1.20 *Great black-backed gull*

83. Great black-backed gulls were recorded on the Norfolk Boreas site in all months. Low numbers were recorded during the breeding season, with a peak in September of 1,239 (including an allocated proportion of unidentified black-backed and large gulls).

13.6.2.2 Non-seabird migrants

84. Migrant terrestrial bird species are typically not well recorded by offshore surveys as they rapidly traverse marine areas, often at altitudes which make them difficult to see or identify and often during the night. Consequently, and in recognition of this, previous wind farm assessments have included estimates of the potential risk of collisions on the basis of knowledge of migration flight paths and migratory population sizes (e.g. for East Anglia THREE: EATL, 2015).

85. The EATL (2015) assessment comprised a screening exercise which identified 23 species as being at potential collision risk at the East Anglia THREE site on migration. The proportion of each flyway population predicted to pass through the East Anglia THREE site was estimated using the approach described in the Strategic Ornithological Support Services (SOSS) 05 Project (Wright et al., 2012). Collisions were estimated using the Band collision risk model Option 1 using the Migrant sheet to calculate the number of potential collisions in each migration season (with a 98% avoidance rate).

86. The results from this modelling indicated that none of the species were at risk of significant collisions whilst on migration. Indeed, the impacts were of such small magnitude (for most species between zero and one collision was predicted per year) that the potential for the proposed East Anglia THREE project to contribute to cumulative impacts was ruled out (EATL, 2015) and no cumulative assessment was therefore necessary (there were only five species with annual collisions greater than one: dark-bellied Brent goose (six), wigeon (two), oystercatcher (two), lapwing (three) and dunlin (ten)).

87. The approach taken used generic data (e.g. Wright et al., 2012) and considers broad migration fronts and the degree to which these overlap with offshore wind farms.

88. Although the conclusions for the East Anglia THREE assessment are expected to apply equally to Norfolk Boreas, an assessment of collision risk for non-seabird migrants has been undertaken and the results are summarised in the relevant section of this ES (section 13.7.4.3.1). Technical Appendix 13.1 provides further details.

13.6.3 Anticipated Trends in Baseline Conditions

89. Key drivers of seabird population size in western Europe are climate change (Sandvik et al., 2012; Frederiksen et al., 2004, 2013; Burthe et al., 2014; Macdonald et al., 2015; Furness, 2016; JNCC, 2016), and fisheries (Tasker et al., 2000; Frederiksen et al., 2004; Ratcliffe, 2004; Carroll et al., 2017; Sydeman et al., 2017). Pollutants (including oil, persistent organic pollutants, plastics), alien mammal predators at colonies, disease, and loss of nesting habitat also impact on seabird populations but are generally much less important and often more localised in their effect (Ratcliffe, 2004; Votier et al., 2005, 2008; JNCC, 2016).

90. Trends in seabird numbers in breeding populations are better known, and better understood, than trends in numbers at sea within particular areas. Breeding numbers are regularly monitored at many colonies (JNCC, 2016), and in the British Isles there have been three comprehensive censuses of breeding seabirds in 1969-70, 1985-88 and 1998-2002 (Mitchell et al., 2004) as well as single-species surveys

(such as the decadal counts of breeding gannet numbers, Murray et al., 2015). In contrast, the European Seabirds at Sea (ESAS) database is incomplete, and few data have been added since 2000, so that current trends in numbers at sea in areas of the North Sea are not so easy to assess.

91. Breeding numbers of many seabird species in the British Isles are declining, especially in the northern North Sea (Foster and Marrs, 2012; Macdonald et al., 2015; JNCC, 2016). The most striking exception is gannet, which continues to increase (Murray et al., 2015), although the rate of increase has been slowing (Murray et al., 2015). These trends seem likely to continue in the short to medium term future.

92. Climate change is likely to be the strongest influence on seabird populations in coming years, with anticipated deterioration in conditions for breeding and survival for most species of seabirds (Burthe et al., 2014; Macdonald et al., 2015; Capuzzo et al., 2018) and therefore further declines in numbers of most of our seabird populations are anticipated. It is therefore highly likely that breeding numbers of most of our seabird species will continue to decline under a scenario with continuing climate change due to increasing levels of greenhouse gases. Fisheries management is also likely to influence future numbers in seabird populations. The Common Fisheries Policy (CFP) Landings Obligation ('discard ban') will likely reduce an unnaturally high level of available food as a result from discard from fishing practices that has been a food supply for scavenging seabirds such as great black-backed gulls, lesser black-backed gulls, herring gulls, fulmars, kittiwakes and gannets (Votier et al., 2004; Bicknell et al., 2013; Votier et al., 2013; Foster et al., 2017).Recent changes in fisheries management that aid recovery of predatory fish stock biomass are likely to further reduce food supply for seabirds that feed primarily on small fish such as sandeels, as those small fish are major prey of large predatory fish. Therefore, anticipated future increases in predatory fish abundance resulting from improved management to constrain fishing mortality on those commercially important species at more sustainable levels than in the past are likely to cause further declines in stocks of small pelagic seabird 'food-fish' such as sandeels (Frederiksen et al., 2007; Macdonald et al., 2015).

93. Future decreases in kittiwake breeding numbers are likely to be particularly pronounced, as kittiwakes are very sensitive to climate change (Frederiksen et al., 2013; Carroll et al., 2015) and to fishery impacts on sandeel stocks near breeding colonies (Frederiksen et al., 2004; Carroll et al., 2017), and additionally, the species may not be able to feed on a readily available food supply from fishery discards as the Landings Obligation comes into effect. Gannet numbers may continue to increase for some years, but evidence suggests that this increase is already slowing, and numbers may peak not too far into the future. While the Landings Obligation will

reduce discard availability to gannets in European waters, in recent years increasing proportions of adult gannets have wintered in west African waters rather than in UK waters (Kubetzki et al., 2009), probably because there are large amounts of fish discarded by west African trawl fisheries and decreasing amounts available in the North Sea (Kubetzki et al., 2009; Garthe et al., 2012). It appears that the flexible behaviour and diet of gannets makes this species comparatively robust to changes in fishery practices or to climate change impacts on fish communities (Garthe et al., 2012).

94. Fulmars, terns, common guillemot, razorbill and puffin appear to be highly vulnerable to climate change, so numbers may decline over the next few decades (Burthe et al., 2014). Strong declines in shag numbers are likely to continue as they are adversely affected by climate change, by low abundance of sandeels and especially by stormy and wet weather conditions in winter (Burthe et al., 2014; Frederiksen et al., 2008). Most of the red-throated divers and common scoters wintering in the southern North Sea originate from breeding areas at high latitudes in Scandinavia and Russia. Numbers of red-throated divers and common scoters wintering in the southern North Sea may possibly decrease in future if warming conditions make the Baltic Sea more favourable as a wintering area for those species so that they do not need to migrate as far as UK waters. There has been a trend of increasing numbers of sea ducks remaining in the Baltic Sea overwinter (Mendel et al., 2008; Fox et al., 2016; Ost et al., 2016) and decreasing numbers coming to the UK (Austin and Rehfisch, 2005; Pearce-Higgins and Holt, 2013), and that trend is likely to continue, although to an uncertain extent.

95. ESAS data indicate that there has already been a long-term decrease in numbers of great black-backed gulls wintering in the southern North Sea (S. Garthe et al., in prep.), and the Landings Obligation which may reduce unnaturally high levels of available food will probably result in further decreases in numbers of north Norwegian great black-backed gulls and herring gulls coming to the southern North Sea in winter. It is likely that further redistribution of breeding herring gulls and lesser black-backed gulls will occur into urban environments (Rock and Vaughan, 2013), although it is unclear how the balance between terrestrial and marine feeding by these gulls may alter over coming years; that may depend greatly on the consequences of Brexit for UK fisheries and farming. Some of the human impacts on seabirds are amenable to effective mitigation (Ratcliffe et al., 2009; Brooke et al., 2018), but the scale of efforts to reduce these impacts on seabird populations has been small by comparison with the major influences of climate change and fisheries. This is likely to continue to be the case in future, and the conclusion must be that with the probable exception of gannet, numbers of almost all other seabird species in the UK North Sea region will most likely be on a downward trend over the next few decades, due to population declines, redistributions or a combination of both.

13.7 Potential Impacts

96. The impacts that could potentially arise during the construction, operation and decommissioning of the proposed project and that require assessment are:

- In the construction phase:
    - Impact 1: Disturbance / displacement; and
    - Impact 2: Indirect impacts through effects on habitats and prey species.
- In the operational phase:
    - Impact 3: Disturbance / displacement;
    - Impact 4: Indirect impacts through effects on habitats and prey species;
    - Impact 5: Collision risk; and
    - Impact 6: Barrier effect.
- In the decommissioning phase:
    - Impact 7: Disturbance / displacement; and
    - Impact 8: Indirect impacts through effects on habitats and prey species.

13.7.1 Embedded Mitigation

97. Norfolk Boreas Limited has committed to a number of techniques and engineering designs/modifications inherent as part of the project, during the pre-application phase, in order to avoid a number of impacts or reduce impacts as far as possible. Embedding mitigation into the project design is a type of primary mitigation and is an inherent aspect of the EIA process.

98. A range of different information sources has been considered as part of embedding mitigation into the design of the project (for further details see Chapter 5 Project Description, Chapter 4 Site Selection and Assessment of Alternatives) including engineering requirements, ongoing discussions with stakeholders and regulators, commercial considerations and environmental best practice.

99. Mitigation measures which are embedded into the proposed project design and are relevant to offshore ornithology receptors are listed in Table 13.15.

Table 13.15 Embedded mitigation relating to offshore ornithology.

Parameter	Mitigation measures embedded in the proposed project design
Site Selection	The Norfolk Boreas site was identified through the Zonal Appraisal and Planning process and avoids European protected sites for birds (e.g. the distance between the Norfolk Boreas site and Flamborough and Filey Coast SPA is more than 218km and from the Alde-Ore Estuary SPA is over 112km). This means the site is beyond the foraging range of almost all seabird species, the exceptions being gannet and lesser black-backed gull for which mean maximum ranges of up to 229km and 141km have been estimated respectively (Thaxter et al., 2012a). However, tracking of individuals from the colonies

Parameter	Mitigation measures embedded in the proposed project design
	within potential foraging range (Flamborough Head and Alde Ore) have revealed a very low degree of connectivity.
Turbine model	Norfolk Boreas Limited has committed to a smallest turbine model of 10MW which would result in a maximum of 180 turbines and is investigating larger models which could result in as few as 90 turbines. Collision risks are typically reduced when fewer larger turbines are used to achieve the same overall maximum export capacity (1800MW). This is also likely to reduce displacement effects.

13.7.2 Worst Case

100. The detailed design of Norfolk Boreas (including numbers of wind turbines, layout configuration etc.) will not be finalised until after the DCO has been determined. Therefore, realistic worst case scenarios in relation to impacts/effects on ornithology are adopted.

101. The worst case assumptions with regards to offshore ornithology are presented by impact in Table 13.16.

Table 13.16 Worst Case Assumptions.

Impact	Parameter	Notes
Construction		
Impact 1: Disturbance and displacement from increased vessel activity	Up to 57 vessels on site at any one time. Total estimated movements; up to 1,180 for single or two phase construction.	Maximum estimated number of vessel movements would cause greatest displacement to birds from wind farm site, cable corridor and project interconnector area. This assumes a maximum construction schedule of 24 hours a day, 7 days a week for a maximum construction period of 24 months within an overall period of up 4 years. Note, however, that there will be periods of downtime.
Impact 2: Indirect effects as a result of displacement of prey species due to increased noise and disturbance to seabed	Spatial worst case impact (maximum area of impact at one time and maximum anticipated pile energy) Monopiles: 2 concurrent piling events, 90 x 15m diameter wind turbine foundations, 2 offshore electrical platforms, a service platform and 2 met masts. 5,000kJ hammer (max. for monopiles). Temporal worst case impact (greatest duration of pile driving	See Chapter 11 Fish and Shellfish Ecology

Impact	Parameter	Notes
	based on the greatest number of piles) Jackets: 2 concurrent piling, 180 wind turbine foundations (with 4 piles each), 2 offshore electrical platforms, 2 service platform and 2 met masts. 2,700kJ hammer.	
	Disturbance/displacement from increased suspended sediment concentration.	Total sediment release over the maximum 4 year build period is listed in Chapter 8 Physical Processes, Table 8.16. However, the release on a daily basis would be temporary and localised with sediment settling out quickly.
	The maximum area of disturbance to benthic habitats during construction would be approximately 23.3km^2 across the Norfolk Boreas offshore project area.	Breakdown is given in Chapter 10 Benthic Ecology, Table 10.9.
Operation		
Impact 3: Disturbance and displacement from offshore infrastructure and due to increased vessel and helicopter activity	An area of 725km^2 plus a 4km buffer with a maximum of 180 wind turbines, with a minimum spacing of 720 x 720m between turbines. Maximum 2 offshore electrical platforms, an offshore service platform, 2 met masts, 2 LiDAR platforms and 2 wave buoys. Support vessels making approximately 445 two-way vessel movements per annum for supporting wind farm operations (average of 1-2 per day). Maximum of 14 two-way helicopter movements per week for scheduled and unscheduled maintenance (728 per year).	Represents maximum density of turbines and structures across the offshore project area, which maximises the potential for avoidance and displacement. Other options represent a smaller total area occupied and reduced density of turbines (e.g. increased spacing). Assessment assumes varying displacement from site and a buffer, where appropriate. See Chapter 5 Project Description.
Impact 4: Indirect effects due to habitat loss / change for key prey species	The maximum possible above seabed footprint of the project including scour protection plus any cable protection. The overall total footprint is approximately 17.67km^2.	Breakdown is given in Chapter 10 Benthic ecology, Table 10.9.
Impact 5: Collision risk	Maximum of 180 x 10MW turbines.	CRM shows that 180 x 10MW turbines have largest collision impact risk.

Impact	Parameter	Notes
		Other options (e.g. 15 MW turbines) have a reduced number of turbines (e.g. 120) and lower collision risks (Technical Appendix 13.1).
Impact 6: Barrier effects	Maximum offshore project area 725km^2 with a maximum of 180 wind turbines, with a minimum spacing of 760 x 760m between turbines. Maximum 2 offshore electrical platforms, offshore service platform, 2 met masts, 2 LiDAR platforms and 2 wave buoys.	Maximum density of turbines and structures across the offshore project area, which maximises the potential barrier to foraging grounds and migration routes for bird species. Other options result in reduced number and density of turbines.
Decommissioning		
Impact 7: Disturbance and displacement from decommissioning activities	Disturbance is anticipated to be similar in nature but of lower magnitude than during construction, but specific details are not currently known.	Maximum estimated number of vessel movements would cause greatest displacement to birds on site.
Impact 8: Indirect effects due to habitat loss / change for key prey species	As above for construction, there would be habitat disturbance effects around sites of activity across the site and offshore cable corridor. There would be limited noise disturbance to prey (as no piling and no use of explosives).	Breakdown is given in Chapter 10 Benthic Ecology, Table 10.2.
Cumulative		
Cumulative impacts are assessed as for the above project alone impacts. The worst case cumulative impacts are defined in the relevant sections and reflect the current knowledge of other projects which could contribute to cumulative effects.		

13.7.3 Potential Impacts during Construction

13.7.3.1 Impact 1: Disturbance and displacement from increased vessel activity

102. The construction phase of the proposed project has the potential to affect bird populations in the marine environment through disturbance due to construction activity leading to displacement of birds from construction sites. This would effectively result in temporary habitat loss through reduction in the area available for feeding, loafing and moulting. The worst case scenario, outlined in Table 13.16, describes the elements of the proposed project considered within this assessment.

103. The maximum duration of offshore construction for the proposed project would be 24 months which would overlap with a maximum of two breeding seasons, two winter periods and up to four migration periods.

104. The construction phase would require the mobilisation of vessels, helicopters and equipment and the installation of foundations, export cables, interconnector cables and other infrastructure. These activities have the potential to disturb and displace birds from within and around the offshore elements of the proposed project, including the wind farm and the subsea cables. The level of disturbance at each work location would differ dependent on the activities taking place, but there could be vessel movements at any time of day or night over the worst case 24 month construction period.

105. Any impacts resulting from disturbance and displacement from construction activities are considered likely to be short-term, temporary and reversible in nature, lasting only for the duration of construction activity, with birds expected to return to the area once construction activities have ceased. Construction related disturbance and displacement is most likely to affect foraging birds.

106. Some species are more susceptible to disturbance than others. Gulls are not considered susceptible to disturbance, as they are often associated with fishing boats (e.g. Camphuysen, 1995; Hüppop and Wurm, 2000) and have been noted in association with construction vessels at the Greater Gabbard offshore wind farm (GGOWL 2011) and close to active foundation piling activity at the Egmond aan Zee (OWEZ) wind farm, where they showed no noticeable reactions to the works (Leopold and Camphuysen, 2007). However, species such as divers and scoters have been noted to avoid shipping by several kilometres (Mitschke et al*.,* 2001 from Exo et al*.,* 2003; Garthe and Hüppop, 2004; Schwemmer et al*.* 2011).

107. There are a number of different measures used to assess bird disturbance and displacement from areas of sea in response to activities associated with an offshore wind farm. Garthe and Hüppop (2004) developed a scoring system for such disturbance factors, which is used widely in offshore wind farm EIAs. Furness and Wade (2012) developed disturbance ratings for particular species, alongside scores for habitat flexibility and conservation importance. These factors were used to define an index value that highlights the sensitivity of a species to disturbance and displacement. As many of these references relate to disturbance from helicopter and vessel activities, these are considered relevant to this assessment. Although, all else being equal, a helicopter may constitute a more pronounced source of disturbance than a vessel, the combination of higher speed (and hence briefer presence) and greater distance to the sea surface means that helicopter disturbance is considered to be the same or lower than that resulting from vessel movements. Thus, the following assessment is based on disturbance due to vessels and it has been assumed that this also encompasses disturbance due to helicopters.

108. Birds recorded during the species-specific spring and autumn migration periods are assumed to be moving through the area between breeding and wintering areas. As these individuals will be present in the site for a short time only and the potential

zone of construction displacement will be small (that located around up to three construction vessels) it is likely that the assessment presented below for the migration periods will over-estimate population impacts.

109. In order to focus the assessment of disturbance and displacement, a screening exercise was undertaken to identify those species most likely to be at risk (Table 13.17). Any species with a low sensitivity to displacement or recorded only in very small numbers within the Study Area (including the offshore cable corridor) was screened out of further assessment.

Table 13.17 Disturbance and displacement screening.

Receptor	Sensitivity to disturbance and displacement	Screening result (IN/OUT)
Common scoter	High	**Screened IN** for export cable installation through near shore areas (i.e. the Greater Wash SPA) only.
Red-throated diver	High	**Screened IN** for the Norfolk Boreas site and export cable installation through near shore areas (i.e. the Greater Wash SPA).
Great northern diver	High	**Screened OUT** as species recorded in very low numbers and therefore additional displacement would be negligible.
Fulmar	Very Low	**Screened OUT** as the species has a Very Low sensitivity and is not known to avoid vessels.
Gannet	Low	**Screened OUT** as has a Low sensitivity to disturbance and displacement.
Puffin	Low to Medium	**Screened OUT** as present in low numbers in very few months and due to low sensitivity to disturbance and displacement.
Razorbill	Medium	**Screened IN** for the Norfolk Boreas site only due to numbers recorded and classified as Medium sensitivity to disturbance and displacement.
Guillemot	Medium	**Screened IN** for the Norfolk Boreas site only due to numbers recorded and classified as Medium sensitivity to disturbance and displacement.
Sandwich tern	Low to Medium	**Screened OUT** for the Norfolk Boreas site as classified of Low to Medium sensitivity to disturbance and displacement, and very low numbers recorded in the Norfolk Boreas site. **Screened OUT** for export cable installation as route does not overlap areas identified in Natural England and JNCC (2016).
Commic tern	Low to Medium	**Screened IN** for the Norfolk Boreas site due to moderate peak population, although classified as Low to Medium sensitivity to disturbance and displacement. **Screened OUT** for export cable installation as route does not overlap foraging areas identified in Natural England and JNCC (2016).
Kittiwake	Low	**Screened OUT** as has a Low sensitivity to disturbance and displacement.

Receptor	Sensitivity to disturbance and displacement	Screening result (IN/OUT)
Great black-backed gull	Low	**Screened OUT** as has a Low sensitivity to disturbance and displacement.
Herring gull	Low	**Screened OUT** as has a Low sensitivity to disturbance and displacement.
Lesser black-backed gull	Low	**Screened OUT** as has a Low sensitivity to disturbance and displacement.
Little gull	Low	**Screened OUT** as has a Low sensitivity to disturbance and displacement.

13.7.3.1.1 *Common scoter*

Export cable installation

110. Common scoter over-winter on inshore waters around the British coast with notable concentrations in the Greater Wash area, Carmarthen Bay and the Irish Sea. This species has been identified as being particularly sensitive to human activities in marine areas including through the disturbance effects of ship and helicopter traffic (Garthe and Hüppop, 2004; Schwemmer et al*.,* 2011; Furness and Wade, 2012; Bradbury et al*.,* 2014).

111. Common scoter is not considered at risk of construction impacts on the Norfolk Boreas site since it was not recorded during surveys. This is to be expected given their habitat preferences (less than 20m sea depth). However, there is potential for disturbance and displacement of non-breeding common scoters resulting from the presence of construction vessels installing the offshore cables through the Greater Wash SPA, for which this species is a nonbreeding feature.

112. Cable laying vessels are static for large periods of time and move only short distances as cable installation takes place, and offshore cable installation activity is a relatively low noise emitting operation. Therefore, the potential magnitude of disturbance is very small. Furthermore, Natural England and JNCC (2016) indicate that no birds were recorded within 10km of the export cable route, and the main concentrations of this species were located along the north Norfolk coast, towards the Wash.

113. On this basis, the potential risks to common scoter resulting from disturbance due to offshore cable laying are considered to be temporary and localised in nature and the magnitude of effect has been determined as negligible or no change. As the species is of high sensitivity to disturbance, the impact significance is at worst **minor adverse**.

13.7.3.1.2 *Red-throated diver*

Export cable installation

114. Red-throated diver has been identified as being particularly sensitive to human activities in marine areas (Dierschke et al., 2016), including through the disturbance effects of ship and helicopter traffic (Garthe and Hüppop, 2004; Schwemmer et al*.,* 2011; Furness and Wade, 2012; Bradbury et al*.,* 2014; Dierschke et al*.,* 2017).

115. There is potential for disturbance and displacement of non-breeding red-throated divers resulting from the presence of construction vessels installing the offshore cables, including when they are laid through the Greater Wash SPA. However, cable laying vessels are static for large periods of time and move only short distances as cable installation takes place. Offshore cable installation activity is also a relatively low noise emitting operation.

116. 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 2km 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 1km.

117. In order to calculate the number of red-throated divers that would potentially be at risk of displacement from the offshore cable corridor (including the project interconnector search area) 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.

118. The worst case area from which birds could be displaced was defined as a circle with a 2km radius around each cable laying vessel, which is $25.2km^2$ ($2 \times 12.6km^2$). 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 400m per hour if surface laying, 300m per hour for ploughing and 80m per hour if trenching (Chapter 5 Project Description). This represents a maximum speed of 7m per minute. For context, a modest tidal flow rate for the region would be in the region of 1m per second (60m 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 has been assumed that the estimated number displaced at any one time represents the total number displaced over the course of a single winter (i.e. rather than many individuals for a short duration each, the same individuals for the duration of a single winter).

119. Definitive mortality rates associated with displacement for red-throated divers, or for any other seabird species, are not known and precautionary estimates have to be used. There is no evidence that birds displaced from wind farms suffer any mortality as a consequence of displacement; any mortality due to displacement would be most likely a result of increased density in areas outside the affected area, resulting in increased competition for food where density was elevated (Dierschke et al*.,* 2017). Such impacts are most likely to be negligible, and below levels that could be quantified, as the available evidence suggests that red-throated divers are unlikely to be affected by density-dependent competition for resources during the non-breeding period (Dierschke et al*.,* 2017). Impacts of displacement are also likely to be context-dependent. In years when food supply has been severely depleted, as for example by unsustainably high fishing mortality of sandeel stocks as has occurred several times in recent decades (ICES, 2013), displacement of sandeel-dependent seabirds from optimal habitat may increase mortality. In years when food supply is good, displacement is unlikely to have any negative effect on seabird populations. Red-throated divers may feed on sandeels, but take a wide diversity of small fish prey, so would be buffered to an extent from fluctuations in abundance of individual fish species. It is not possible for the proposed project to predict future fishing effort.

120. For recent wind farm assessments Natural England have advised that an unconfirmed 10% mortality rate should be used for birds displaced by cable laying vessels. This magnitude of impact is not supported in the literature and given that this would equate to more than half the natural adult annual mortality (16%) from a single occasion of disturbance (as described above), it is highly improbable that such an effect would occur. To put this in context it is worth considering that disturbance from vessels in the southern North Sea has been ongoing for decades and the Norfolk Boreas site is bordered by the Deep Water Route, a designated shipping lane which accommodates regular, high frequency of marine traffic (See Chapter 15 Shipping and Navigation) . With this in mind, additional mortality of 10% of the population due to single instances of vessel disturbance during the course of the winter, as proposed by Natural England, would reduce a population of 1,500 (i.e. the Greater Wash SPA population) to fewer than 100 within 10 years (alternatively the SPA population would need to have been 16 times larger 10 years prior to the SPA designation surveys in order to have been reduced to 1,500). Neither of these scenarios is supported by the evidence.

121. A review of available evidence for red-throated diver displacement was submitted for the Norfolk Vanguard assessment (MacArthur Green 2019a) and this concluded that there would be little or no effect of displacement on diver survival. Consequently, a maximum, and hence precautionary, displacement caused mortality rate of 1% was identified as appropriate for this assessment.

122. At this level of additional mortality, only a maximum of 1 individual would be expected to die across the entire winter period (September to April) as a result of any potential displacement effects from the offshore cable installation activities, which would be restricted to a single season, and only if cable laying takes place during these months. 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.

123. 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**.

Offshore wind farm

124. Red-throated divers were recorded in Norfolk Boreas in low numbers between November and May (and in the buffer in September and October), with numbers peaking in March (mean density 0.62/km^2) with none present between June and August. Although March and April were identified as breeding months in Furness (2015) this species does not breed in the southern North Sea and individuals recorded at this time are considered to be part of the spring migration population (February – April; Furness, 2015).

125. 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 Norfolk Boreas site.

126. For this precautionary assessment it has been assumed that between 1% (evidence based precautionary rate, MacArthur Green 2019a) and 10% (Natural England's preferred precautionary value) of displaced individuals could die as a result of displacement by construction vessels (see section 13.7.4.1.1 for further details).

127. During autumn migration, with a seasonal peak density on the wind farm site of 0.02/km^2 and a precautionary 2km radius of disturbance around each construction vessel, less than 1 individual (0.02 x 12.56 x 2) could be at risk of displacement with

between 0.005 (1%) and 0.05 (10%) individuals at risk of mortality in a maximum of two autumn periods.

128. 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**.

129. During winter, with a seasonal peak density of 0.11/km^2 and a precautionary 2km radius of disturbance around each construction vessel, 2.8 individuals (0.11 x 12.56 x 2) could be at risk of displacement with between 0.03 (1%) and 0.3 (10%) individuals at risk of mortality during a maximum of two winter periods.

130. At the average baseline mortality rate for red-throated diver of 0.228 (Table 13.13) 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 0.3 to this increases the mortality rate by 0.013%. 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**.

131. During spring, with a seasonal peak density of 0.62/km^2 and a precautionary 2km radius of disturbance around each construction vessel, 16 individuals (0.62 x 12.56 x 2) could be at risk of displacement with between 0.15 (1%) and 1.5 (10%) individuals at risk of mortality during a maximum of two spring periods.

132. At the average baseline mortality rate for red-throated diver of 0.228 (Table 13.13) 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 1.5 to this increases the mortality rate by 0.05%. 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**.

133. The combined nonbreeding impact of construction, with between 0.2 (1%) and 2 (10%) 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 a maximum of 0.09%). 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**.

13.7.3.1.3 *Razorbill*

Offshore wind farm

134. Razorbills were recorded in the Norfolk Boreas site year round, with numbers peaking in December (mean density $0.95/km^2$) and at their lowest in June (mean density $0.09/km^2$). Razorbills are considered to have a medium general sensitivity to disturbance and displacement, based on their sensitivity to ship and helicopter traffic in Garthe and Hüppop (2004) and Furness and Wade (2012). Dierschke et al. (2016) categorized razorbill as ‘weakly avoiding offshore wind farms’ based on a review of numbers inside and outside of operational offshore wind farms; their behavioural response to construction is likely to be similar and probably slightly stronger than during operation.

135. There is potential for disturbance and displacement of razorbills 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 Norfolk Boreas site.

136. For recent wind farm assessments, Natural England has advised that a unconfirmed 10% mortality rate should be used for auks displaced from wind farms. This magnitude of impact is not supported in the literature and given that this would equate to a doubling of natural adult annual mortality (10.5%), it is highly improbable that such an effect would occur.

137. A review of available evidence for auks displacement was submitted for the Norfolk Vanguard assessment (MacArthur Green 2019b) and this concluded that precautionary rates of displacement and mortality from operational wind farms would be 50% and 1% respectively. These figures are also considered suitably precautionary for the potential displacement around construction vessels. Thus the assessment presents estimates using 1% mortality (evidence based) and 10% (Natural England unconfirmed rate).

138. During the autumn migration season, at a seasonal peak density of $0.35/km^2$ and with a highly precautionary 2km radius of disturbance around each construction vessel, 9 individuals (0.35 x 12.56 x 2) could be at risk of displacement with between 0.1 (1%) and 1 (10%) individuals at risk of mortality. The autumn migration BDMPS for razorbill is 591,874 (Furness, 2015). At the average baseline mortality rate for razorbill of 0.174 (Table 13.13) the number of individuals expected to die in the autumn migration BDMPS is 102,986 (591,874 x 0.174). The addition of 1 individual to this would increase the mortality rate by an undetectable amount (<0.001%).

Therefore, during the autumn period, the magnitude of effect is assessed as negligible even on the basis of this highly precautionary approach. The construction works are temporary and localised in nature and the magnitude of effect has been determined as negligible. As the species is of medium sensitivity to disturbance, the impact significance is **minor adverse**.

139. During the winter, at a seasonal peak density of 0.95/km^2 and with a highly precautionary 2km radius of disturbance around each construction vessel, 24 individuals (0.95 x 12.56 x 2) could be at risk of displacement with between 0.2 (1%) and 2.4 (10%) individuals at risk of mortality. The winter (nonbreeding season) BDMPS for razorbill is 218,622 (Furness, 2015). At the average baseline mortality rate for razorbill of 0.174 (Table 13.13) the number of individuals expected to die in the winter BDMPS is 38,040 (218,622 x 0.174). The addition of 2.4 individuals to this would increase the mortality rate by 0.006% which 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.

140. The construction works are temporary and localised in nature and the magnitude of effect has been determined as negligible. As the species is of medium sensitivity to disturbance, the impact significance is **minor adverse**.

141. During the spring migration season, at a peak mean density of 0.40/km^2 and with a highly precautionary 2km radius of disturbance around each construction vessel, 10 individuals (0.40 x 12.56 x 2) could be at risk of displacement with between 0.1 (1%) and 1 (10%) individuals at risk of mortality. The spring migration BDMPS for razorbill is 591,874 (Furness, 2015). At the average baseline mortality rate for razorbill of 0.174 (Table 13.13) the number of individuals expected to die in the spring migration BDMPS is 102,986 (591,874 x 0.174). The addition of 1 individual to this would increase the mortality rate by an undetectable amount (0.001%). Therefore, during the spring migration period, the magnitude of effect is assessed as negligible even on the basis of this highly precautionary approach.

142. The construction works are temporary and localised in nature and the magnitude of effect has been determined as negligible. As the species is of medium sensitivity to disturbance, the impact significance is **minor adverse**.

143. During the breeding season the seasonal peak density of razorbills on the site was 0.65/km^2 (July) which suggests that 16 individuals (0.65 x 12.56 x 2) could be at risk of displacement and between 0.2 (1%) and 1.6 (10%) individuals at risk of mortality.

144. The mean maximum foraging range for breeding razorbill is 48.5km (Thaxter et al*.,* 2012a) which places the Norfolk Boreas site considerably beyond the range of any razorbill breeding colonies. It should be noted that some recent tagging studies have recorded larger apparent foraging ranges (one razorbill was recorded travelling 312km from Fair Isle) which would indicate the possibility of connectivity to breeding

colonies. However, further consideration of this apparent potential for connectivity indicates how exceptional this result is. A razorbill flies at about 16m per second (Pennycuick, 1987) so would take almost 11 hours to complete this round trip even if it spent no time resting on the water or diving for food. This is incompatible with bringing enough food back to keep a chick alive as razorbill chicks receive about three feeds per day (Harris and Wanless, 1989). Yet chicks are normally attended and protected by one adult at the nest site while the partner is foraging (Wanless and Harris, 1986), so there are simply not enough hours in the day to allow successfully breeding razorbills to make such long trips to provision a chick. At 16m per second the Norfolk Boreas site is 3.8 hours direct flight time away from the nearest razorbill breeding colony (Flamborough Head which is 220k from the Norfolk Boreas Site). A return trip would take 7.6 hours, not allowing for foraging. As for the Fair Isle example, travelling such distances is incompatible with successful breeding. On the basis of three feeds per day, the furthest away a bird could fly per trip to achieve this in 24 hours is 115km (i.e. a round trip of 230km), with no allowance for foraging time. Even if the bird spends a maximum of only 30 minutes foraging, this reduces the farthest distance to 108km (i.e. approximately half the distance to Norfolk Boreas).

145. On the basis of the above evidence, it can be stated with confidence that there are no breeding colonies for razorbill within foraging range of the Norfolk Boreas site, therefore it is reasonable to assume that individuals seen during the breeding season are nonbreeding (e.g. immature birds). Since immature seabirds are known often to remain in wintering areas, the number of immature birds in the relevant population during the breeding season may be estimated as 43% of the total wintering BDMPS population (Furness, 2015). This gives a breeding season population of 94,007 (BDMPS for the UK North Sea and Channel, 218,622 x 43%). At the average baseline mortality rate for razorbill of 0.174 (Table 13.13) the number of individuals expected to die in the breeding season is 16,357 (94,007 x 0.174). The addition of a maximum of 1.6 individual to this would increase the mortality rate by less than 0.01% which would be undetectable. Therefore, during the breeding season, the magnitude of effect is assessed as negligible even on the basis of this highly precautionary approach. As the species is of medium sensitivity to disturbance, the impact significance is **minor adverse**.

146. The construction works are temporary and localised in nature and the magnitude of effect has been determined as negligible. As the species is of medium sensitivity to disturbance, even when the individual season impacts are combined (between 0.6 and 6 additional mortalities in total, across the year as a whole) the increase in mortality would be no more than 0.03% (using the smaller BDMPS) therefore the impact significance is **minor adverse**.

13.7.3.1.4 *Guillemot*

Offshore wind farm

147. Guillemots have been recorded in the Norfolk Boreas site year round, with numbers peaking in December (mean density 14.45/km^2) and at their lowest in June (mean density 0.35/km^2). Guillemots are considered to have a medium general sensitivity to disturbance and displacement, based on their sensitivity to ship and helicopter traffic in Garthe and Hüppop (2004), Furness and Wade (2012), Furness et al. (2013) and Bradbury et al. (2014). Dierschke et al. (2016) categorized guillemot as 'weakly avoiding offshore wind farms' based on a review of numbers inside and outside of operational offshore wind farms; their behavioural response to construction is likely to be similar and probably slightly stronger than during operation.

148. There is potential for disturbance and displacement of guillemots 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 no more than two foundations expected to be installed at any time within the Norfolk Boreas site. Consequently, the effects will occur only in the areas where vessels are operating at any given point and not the entire site.

149. For recent wind farm assessments Natural England have advised that an unconfirmed10% mortality rate should be used for auks displaced from wind farms. This magnitude of impact is not supported in the literature and given that this would equate to more than double natural adult annual mortality (6%), it is highly improbable that such an effect would occur.

150. A review of available evidence for auks displacement was submitted for the Norfolk Vanguard assessment (MacArthur Green 2019b) and this concluded that precautionary rates of displacement and mortality from operational wind farms would be 50% and 1% respectively. These figures are also considered suitably precautionary for the potential displacement around construction vessels. Thus the assessment presents estimates using 1% mortality (evidence based) and 10% (Natural England unconfirmed rate).During the nonbreeding season, at a seasonal peak density of 14.45/km^2 and with a highly precautionary 2km radius of disturbance around each construction vessel, a maximum of 363 individuals (14.4 x 12.56 x 2) could be at risk of displacement with between 3.6 (1%) and 36 (10%) at risk of mortality. The nonbreeding season BDMPS for common guillemot is 1.6 million birds (Furness, 2015). At the average baseline mortality rate for guillemot of 0.14 (Table 13.13) the number of individuals expected to die in the nonbreeding BDMPS is 226,423 (1,617,306 x 0.14). The addition of a maximum of 36 individuals to this would increase the mortality rate by 0.016% which would be undetectable.

Therefore, during the nonbreeding period, the magnitude of effect is assessed as negligible even on the basis of this highly precautionary approach.

151. The construction works are temporary and localised in nature and the magnitude of effect has been determined as negligible. As the species is of medium sensitivity to disturbance, the impact significance is **minor adverse**.

152. During the breeding season the seasonal peak density on the Norfolk Boreas site was 8.68/km^2 (July) which suggests that 218 individuals (8.68 x 12.56 x 2) could be at risk of displacement with between 2.2 (1%) and 22 (10%) individuals at risk of mortality.

153. The mean maximum foraging range for breeding guillemot is 84.2km (Thaxter et al., 2012a) which places the Norfolk Boreas site considerably beyond the range of any guillemot breeding colonies. It should be noted that some recent tagging studies have recorded larger apparent distances than this (one guillemot was recorded travelling 340km from Fair Isle) which would indicate connectivity to breeding colonies. However, further consideration of this apparent potential for connectivity indicates how exceptional this result is. The 340km figure is derived from an individual guillemot on Fair Isle in a year when the local sandeel stock collapsed and breeding success was close to zero (this bird's chick died). A common guillemot flies at about 19m per second (Pennycuick, 1987) so would take almost ten hours to complete this round trip even if it spent no time on the water or diving for food. This is incompatible with bringing enough food back to keep a chick alive. The species carries only one fish at a time and common guillemot chicks need about five feeds per day. Yet chicks are normally attended and protected by one adult at the nest site while the partner is foraging (Uttley et al., 1994), so there are simply not enough hours in the day to allow successfully breeding guillemots to make such long trips to provision a chick. At 19m per second the Norfolk Boreas site is 3.2 hours direct flight time away from the nearest guillemot breeding colony (Flamborough Head, 218km from Norfolk Boreas). A return trip would take 6.4 hours, not allowing for foraging. As is the case for the Fair Isle example, travelling such distances is incompatible with successful breeding. On the basis of five feeds per day, the furthest away a bird could fly per trip to achieve this in 24 hours is 164km (i.e. a round trip of 328km), with no allowance for foraging time. Even if the bird spends a maximum of only 30 minutes foraging, this reduces the farthest distance to 147km.

154. On the basis of the above evidence, it can be stated with confidence that there are no breeding colonies for guillemot within foraging range of the Norfolk Boreas site, therefore it is reasonable to assume that individuals seen during the breeding season are nonbreeding (e.g. immature birds). Since immature seabirds are known often to remain in wintering areas, the number of immature birds in the relevant population during the breeding season may be estimated as 43% (the proportion of the population that is of immature status) of the total wintering BDMPS population (Furness, 2015). This gives a breeding season population of nonbreeding immature

birds of 695,441 (BDMPS for the UK North Sea and Channel, 1,617,306 x 43%). At the average baseline mortality rate for guillemot of 0.14 (Table 13.13) the number of individuals expected to die in the breeding season is 97,362 (695,441 x 0.14). The addition of a maximum of 22 individuals to this would increase the mortality rate by 0.02% which would be undetectable. Therefore, during the breeding season, the magnitude of effect is assessed as negligible even on the basis of this highly precautionary approach. Therefore, an impact on 22 (likely immature) individuals during the breeding season will be negligible.

155. The construction works are temporary and localised in nature and the magnitude of effect has been determined as negligible. As the species is of medium sensitivity to disturbance, the impact significance is **minor adverse**.

156. Guillemot are of medium sensitivity to disturbance, thus even when the individual season impacts are combined (between 5.8 and 58 additional mortalities in total across the year as a whole) the increase in mortality would be no more than 0.43% (using the smaller BDMPS) therefore the impact significance is **minor adverse**.

13.7.3.1.5 *Commic tern*

Offshore wind farm

157. Commic terns were recorded in the Norfolk Boreas site in May, July and August, with a peak in May (mean density 0.48/km^2). Commic terns are considered to have a low to medium general sensitivity to disturbance and displacement, based on their sensitivity to ship and helicopter traffic in Garthe and Hüppop (2004), Furness and Wade (2012), Furness et al. (2013) and Bradbury et al. (2014).

158. There is potential for disturbance and displacement of commic tern 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 no more than two foundations expected to be installed at any time within Norfolk Boreas. Consequently, the effects will occur only in the areas where vessels are operating at any given point and not the entire site.

159. For this precautionary assessment it has been assumed that 10% of displaced individuals could die as a result of displacement by construction vessels.

160. The Norfolk Boreas site is a minimum of 73km to the coast, which is more than three times the mean maximum foraging range for Arctic tern and almost five times that for common tern. Therefore, since both months when birds were recorded fall within migration periods it is appropriate to assess impacts against the relevant migratory populations.

161. During the autumn migration period, at a seasonal peak density of 0.29/km^2 and with a highly precautionary 2km radius of disturbance around each construction vessel, a maximum of 7 individuals (0.29 x 12.56 x 2) could be at risk of displacement and 0.7 at risk of mortality. The autumn migration period BDMPS for commic tern is 308,841 (Furness, 2015). The average baseline mortality has been calculated using the demographic rates for common tern as there are more data available for this species (Horswill and Robinson, 2015). At the average baseline mortality rate of 0.263 (Table 13.13) the number of individuals expected to die in the autumn migration BDMPS is 81,225 (308,841 x 0.263). The addition of less than 1 individual to this would increase the mortality rate by <0.001% which would be undetectable. Therefore, during the nonbreeding period, the magnitude of effect is assessed as negligible even on the basis of this highly precautionary approach.

162. The construction works are temporary and localised in nature and the magnitude of effect has been determined as negligible. As the species is of medium sensitivity to disturbance, the impact significance is no higher than **minor adverse**.

163. During the spring migration period, at a seasonal peak density of 0.48/km^2 and with a highly precautionary 2km radius of disturbance around each construction vessel, a maximum of 12 individuals (0.48 x 12.56 x 2) could be at risk of displacement and 1 at risk of mortality. The spring migration period BDMPS for commic tern is 308,841 (Furness, 2015). The average baseline mortality has been calculated using the demographic rates for common tern as there are more data available for this species (Horswill and Robinson, 2015). At the average baseline mortality rate of 0.263 (Table 13.13) the number of individuals expected to die in the spring migration BDMPS is 81,225 (308,841 x 0.263). The addition of 1 individual to this would increase the mortality rate by an undetectable amount (0.002%). Therefore, during the nonbreeding period, the magnitude of effect is assessed as negligible even on the basis of this highly precautionary approach.

164. The construction works are temporary and localised in nature and the magnitude of effect has been determined as negligible. As the species is of medium sensitivity to disturbance, the impact significance is no higher than **minor adverse**.

165. Commic terns are of low to medium sensitivity to disturbance, thus even when the individual season impacts are combined (up to 2 additional mortalities in total) the increase in mortality would be 0.002%, therefore the impact significance is no higher than **minor adverse**.

13.7.3.2 Impact 2: Indirect effects as a result of displacement of prey species due to increased noise and disturbance to seabed

166. Indirect disturbance and displacement of birds may occur during the construction phase if there are impacts on prey species and the habitats of prey species. These indirect effects include those resulting from the production of underwater noise (e.g.

during piling) and the generation of suspended sediments (e.g. during preparation of the seabed for foundations) that may alter the behaviour or availability of species which are prey for birds. Underwater noise may cause fish and mobile invertebrates to avoid the construction area and also affect their physiology and behaviour. Suspended sediments may cause fish and mobile invertebrates to avoid the construction area and may smother and hide immobile benthic prey. These mechanisms result in less prey being available within the construction area to foraging seabirds. Such potential effects on benthic invertebrates and fish have been assessed in Chapter 10 Benthic Ecology and Chapter 11 Fish and Shellfish Ecology and the conclusions of those assessments inform this assessment of indirect effects on ornithological receptors.

167. With regard to noise impacts on fish, Chapter 11 Fish and Shellfish Ecology discusses the potential impacts upon fish as prey species relevant to birds. With regard to physical injury or behavioural changes, underwater noise impacts on fish during construction of the proposed project are considered to be minor or negligible (see Chapter 11 Fish and Shellfish Ecology) for species such as herring, sprat and sandeel which are main prey items of seabirds such as gannet and auks. Given that Norfolk Boreas is situated in a region of lower importance for foraging seabirds (i.e. beyond foraging range of breeding colonies), a minor or negligible adverse impact on fish that are bird prey species will give rise to impacts on seabirds occurring in or around the proposed project during the construction phase of a **negligible to minor adverse** significance.

168. With regard to changes to the seabed and to suspended sediment levels, Chapter 8 Marine geology, oceanography and physical process discusses the nature of any change and impact. Such changes are considered to be temporary, small scale and highly localised. The consequent indirect impact on benthic and fish species through habitat loss is considered to be minor or negligible (see Chapters 10 Benthic Ecology and 11 Fish and Shellfish Ecology) for species such as herring, sprat and sandeel which are main prey items of seabirds such as gannet and auks. With a minor or negligible impact on fish that are bird prey species, it is concluded that the indirect impact significance on seabirds occurring in or around the project during the construction phase is similarly **negligible to minor adverse**.

13.7.4 Potential Impacts during Operation

13.7.4.1 Impact 3: Disturbance and displacement from offshore infrastructure

169. The presence of wind turbines has the potential to directly disturb and displace birds from within and around the offshore project area. This is assessed as an indirect habitat loss, as it has the potential to reduce the area available to birds for feeding, loafing and moulting. Vessel activity and the lighting of wind turbines and associated

ancillary structures could also attract (or repel) certain species of birds and affect migratory behaviour on a local scale.

170. Seabird species vary in their reactions to the presence of operational infrastructure (e.g. wind turbines, offshore project substations and met masts) and to the maintenance activities that are associated with them (particularly ship and helicopter traffic), with Garthe and Hüppop (2004) presenting a scoring system for such disturbance factors, which is used widely in offshore wind farm EIAs. As offshore wind farms are a new feature in the marine environment, there is limited evidence as to the disturbance and displacement effects of the operational infrastructure in the long term. However, Dierschke et al. (2016) reviewed all available evidence from operational offshore wind farms on the extent of displacement or attraction of seabirds in relation to these structures. They found strong avoidance of operational offshore wind farms by great crested grebe, red-throated diver, black-throated diver and gannet. They found weak avoidance by long-tailed duck, common scoter, fulmar, Manx shearwater, razorbill, guillemot, little gull and Sandwich tern. They found no evidence of any consistent response by eider, kittiwake, common tern and Arctic tern, and evidence of weak attraction to operating offshore wind farms for common gull, black-headed gull, great black-backed gull, herring gull, lesser black-backed gull and red-breasted merganser, and strong attraction for shags and cormorants. Dierschke et al. (2016) suggested that strong avoidance would lead to some habitat loss for those species, while attracted birds appear to benefit from increases in food abundance within operational offshore wind farms.

171. The Statutory Nature Conservation Bodies (SNCBs) issued a joint Interim Displacement Guidance Note (JNCC, 2017), which provides recommendations for presenting information to enable the assessment of displacement effects in relation to offshore wind farm developments. This guidance note has been used in the assessment provided below.

172. There are a number of different measures used to determine bird displacement from areas of sea in response to activities associated with an offshore wind farm. Furness et al. (2013), for example, use disturbance ratings for particular species, alongside scores for habitat flexibility and conservation importance to define an index value that highlights the sensitivity to disturbance and displacement. These authors also recognise that displacement may contribute to individual birds experiencing fitness consequences, which at an extreme level could lead to the mortality of individuals.

173. Both the presence of the infrastructure and the operational activities associated with the proposed project have the potential to directly disturb birds. These activities could potentially displace birds from important areas for feeding, moulting and loafing. Reduced access to some areas could result, at the extreme, in changes to feeding and other behavioural activities resulting in a loss of fitness and a reduction

in survival chances. This would be unlikely for seabirds that have large areas of alternative habitat available but would be more likely to affect seabirds with highly specialised habitat requirements that are limited in availability (Furness et al., 2013; Bradbury et al., 2014).

174. The methodology presented in JNCC (2017) recommends a matrix is presented for each key species showing bird losses at differing rates of displacement and mortality. This assessment uses the range of predicted losses, in association with the scientific evidence available from post-construction monitoring studies, to quantify the level of displacement and the potential losses as a consequence of the proposed project. These losses are then placed in the context of the relevant population (e.g. SPA, BDMPS or biogeographic) to determine the magnitude of effect.

175. The population estimate used for each species to assess the displacement effects was the relevant seasonal peak (i.e. the highest value for the months within each season). The seasonal peaks were calculated as follows; first the density for each calendar month was calculated (as the average of the density in each survey undertaken in that month), then the highest value from the months within each season extracted. As per JNCC (2017), for divers, the assessment used all data recorded within the 4km buffer, for all other species the assessment used all data recorded within the 2km buffer (although it should be noted that the evidence reviews in MacArthur Green 2019a and 2019b indicate that these buffer distances are highly precautionary for both divers and auks).

176. Birds are considered to be most at risk from operational disturbance and displacement effects when they are resident (e.g. during the breeding season or wintering season). The small risk of impact to migrating birds is better considered in terms of barrier effects. However, JNCC (2017) suggests that migration periods should also be assessed using the matrix approach and this has been undertaken where appropriate.

177. Following installation of the offshore cable, the required operational and maintenance activities (in relation to the cable) may have short-term and localised disturbance and displacement impacts on birds using the Norfolk Boreas site. However, disturbance from operational cable activities (e.g. maintenance) would be temporary and localised, and is unlikely to result in detectable effects at either the local or regional population level. Therefore, no impact due to cable operation and maintenance is predicted. The focus of this section is therefore on the disturbance and displacement of birds due to the presence and operation of wind turbines, other offshore infrastructure and any maintenance operations associated with these structures.

178. In order to focus the assessment of disturbance and displacement, a screening exercise was undertaken to identify those species most likely to be at risk (Table

13.18), focussing on the main species described in the Ornithology Technical Report (Technical Appendix 13.1). The species identified as at risk were then assessed within the biological seasons within which effects were potentially likely to occur. Any species with a low sensitivity to displacement or recorded only in very small numbers within the Norfolk Boreas site during the breeding and wintering seasons, were screened out of further assessment.

179. This process screened out great northern diver as only a single individual of this species was recorded in one survey.

180. Operational disturbance and displacement screening (Table 13.18) presents the general sensitivity to disturbance and displacement for each species.

Table 13.18 Operational disturbance and displacement screening.

Receptor	Sensitivity to Disturbance and Displacement (Garthe and Hüppop, 2004; Furness and Wade, 2012, Wade *et al.*, 2016, Dierschke *et al.*, 2016)	Biological Season(s) with peak numbers	Screening Result (IN or OUT)
Red-throated diver	Very High	Spring migration	**Screened IN** for potential effects during autumn migration, midwinter and spring migration.
Fulmar	Considered Low in some studies, but possibly high according to Dierschke et al. (2016)	Breeding & migration periods	**Screened OUT** as the species has a maximum habitat flexibility score of 1 in Furness & Wade (2012), suggesting species utilises a wide range of habitats over a large area.
Gannet	Considered Low in some studies, but possibly high according to Dierschke et al. (2016)	Autumn migration	**Screened IN** for autumn and spring migration seasons, as has a high macro avoidance rate.
Guillemot	Medium	Migration periods	**Screened IN** as present in moderate numbers in nonbreeding season and due to medium sensitivity to disturbance and displacement.
Razorbill	Medium	Nonbreeding season	**Screened IN** as present in moderate numbers in nonbreeding season and due to medium sensitivity to disturbance and displacement.
Puffin	Low	Spring migration season	**Screened OUT** as present in low numbers in very few months and due to low sensitivity to disturbance and displacement.
Kittiwake	Low	Migration periods	**Screened OUT** as migration numbers low relative to BDMPS and not known to avoid wind turbines (low macro avoidance rate)
Lesser black-backed gull	Low	No clear seasonal peak	**Screened OUT** as present in low numbers in all seasons and not known to avoid wind turbines (low macro avoidance rate)
Herring gull	Low	Breeding	**Screened OUT** as present in low numbers in all seasons and not known to avoid wind turbines (low macro avoidance rate)
Great black-backed gull	Low	Breeding & Wintering	**Screened OUT** as present in low numbers in all season and not known to avoid wind turbines (low macro avoidance rate)

181. The impact of mortality caused by displacement on the population is assessed in terms of the change in the baseline mortality rate which could result. It has been assumed that all age classes are equally at risk of displacement (i.e. in proportion to their presence in the population), therefore the average mortality rate calculated above (Table 13.13) has been used.

182. For assessment a worst case assumption has been made that birds will be at risk of displacement from the complete extent of the wind farm site plus species specific buffers. This will over-estimate impacts since it is highly unlikely that the entire area within the site will contain turbines, and even if it did then the inter-turbine separation distance would be such that birds would be very likely to use areas between turbines. Therefore, either a smaller area will be developed, or the magnitude of displacement will be lower than the level assumed in the assessment. There is evidence to suggest that the density of turbines influences the magnitude of displacement (Leopold et al., 2011). Indeed, since the cause of operational displacement is bird responses to the turbines, it is logical to infer that a wind farm with a lower turbine density will cause lower displacement levels than one with a higher density of turbines.

183. Natural England guidance is that displacement effects estimated in different seasons should be combined to provide an annual effect for assessment which should then be assessed in relation to the largest of the component BDMPS populations, and also the biogeographic population. Natural England have acknowledged that summing impacts in this manner almost certainly over-estimates the number of individuals at risk through double counting (i.e. some individuals may potentially be present in more than one season) and assessing against the BDMPS almost certainly under-estimates the population from which they are drawn (which must be at least this size and is likely to be considerably larger as a consequence of turnover of individuals). However, at the present time there is no agreed alternative method for undertaking assessment of annual displacement and therefore the above approach is presented, albeit with the caveat that the results are anticipated to be highly precautionary.

13.7.4.1.1 *Red-throated diver*

184. Red-throated divers are considered to have a very high general sensitivity to disturbance and displacement and they are notoriously shy and prone to avoiding disturbed areas (Garthe & Hüppop, 2004; Petersen et al., 2006; Furness and Wade, 2012; Percival, 2014; Dierschke et al., 2016; Dierschke et al., 2017). Monitoring studies of red-throated divers at the Kentish Flats offshore wind farm found an observable shift of birds away from the turbines, particularly within 500m of the site (Percival, 2010). This is consistent with a study of pre-construction and post-construction abundance and distribution of birds conducted at Horns Rev, Denmark. This study found that red-throated divers avoided areas of sea that were apparently

suitable (favoured habitat, suitable depth and abundant food sources) following the construction of an offshore wind farm, and that this effect remained for a period of three years (Petersen et al., 2006). Further pre-construction and post-construction abundance and distribution studies published more recently on red-throated divers at the Kentish Flats site (Percival, 2014) have provided displacement values for both the site footprint and within distance bands away from the site boundary and indicate how displacement has changed over the periods following construction.

185. Natural England's preferred method assumes that displacement will occur at a constant level to a distance of 4km and that within this area, 100% of birds will be displaced and mortality of displaced birds will be 10%. This is considered to be over-precautionary since it combines high values for three aspects of the assessment: the distance over which birds will be affected (4km), the rate of displacement within this distance (100%) and the mortality rate of displaced individuals (10%). Further consideration of these is provided below.

186. Studies at Kentish Flats and Thanet have provided evidence that red-throated divers are displaced to a decreasing extent with increasing distance from wind turbines (Percival, 2013, 2014). Percival (2014) reported that at Kentish Flats, while displacement within the wind farm boundary was around 80% (compared to pre-construction), this declined to 10% at 1km from the wind farm and was 0% from 2km. A similar within wind farm reduction in density was reported at Thanet, but there was no detectable displacement beyond the wind farm boundary (Percival, 2013). Displacement rates of 60% to 80% were reported for OWEZ (Leopold et al., 2011) and the review by Dierschke et al. (2016) also suggested a figure in this range. The 4km exclusion distance advised by Natural England is greater than the evidence suggests is required for this species, and it is therefore considered over-precautionary to combine this with a displacement rate as high as 100%.

187. A review of evidence undertaken by a panel of experts brought together by JNCC concluded that mortality associated with displacement of red-throated divers may well be zero (Dierschke et al., 2017) and is certainly very unlikely to be as high as the 10% recommended by Natural England. This conclusion is also supported by modelling of individual energy budgets (Topping and Petersen, 2011).

188. A comprehensive literature review investigating displacement impacts on red-throated divers was conducted for the Norfolk Vanguard assessment (MacArthur Green 2019a). This review advocated an evidence-based displacement rate of 90% extending 2km from the wind farm boundary with a consequent maximum mortality rate of 1%.

189. Therefore, this assessment presents displacement across a range from 90% displaced and 1% mortality (evidence based) and 100% displaced and 10% mortality (Natural

England unconfirmed rates). Note that the evidence based rates retain precaution, not only in the interpretation of the evidence to derive the rates, but also in the use of a 4km buffer around the wind farm, despite the evidence that 2km would be a sufficiently precautionary distance.

190. The displacement matrices in Table 13.19 to Table 13.26 have been populated with data for red-throated diver during the autumn migration, nonbreeding and spring migration periods within the site and those calculated within a 4km 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.

191. Using the seasonal peak autumn migration abundance on the Norfolk Boreas site (and 4km buffer) of 23, 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 and 2 individuals (Table 13.19).

192. 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 (Table 13.13) the number of individuals expected to die is 3,027 (13,277 x 0.228). The addition of a maximum of two individuals to this would increase 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 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**.

Table 13.19 Displacement matrix presenting the number of red-throated divers in the Norfolk Boreas site (and 4km buffer) during the autumn migration season that may be subject to mortality (highlighted) on the assumption of complete development of the site.

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	0
3	0	0	0	0	0	0	0	1	1	1
4	0	0	0	0	0	1	1	1	1	1
5	0	0	0	0	1	1	1	1	1	1
6	0	0	0	1	1	1	1	1	1	1
7	0	0	0	1	1	1	1	1	1	2
8	0	0	1	1	1	1	1	1	2	2
9	0	0	1	1	1	1	1	2	2	2
10	0	0	1	1	1	1	2	2	2	2
20	0	1	1	2	2	3	3	4	4	5
30	1	1	2	3	3	4	5	6	6	7

Mortality (%)	Displacement (%) 10	20	30	40	50	60	70	80	90	100
50	1	2	3	5	6	7	8	9	10	12
75	2	3	5	7	9	10	12	14	16	17
100	2	5	7	9	12	14	16	18	21	23

193. Using the seasonal peak winter abundance on the Norfolk Boreas site (and 4km buffer) of 156, the maximum number of individual red-throated divers which could potentially suffer mortality as a consequence of displacement has been estimated to be between 1 and 16 individuals (Table 13.20).

194. 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 (Table 13.23) the number of individuals expected to die is 2,320 (10,177 x 0.228). The addition of a maximum of 16 individuals to this would increase the mortality rate by 0.7%. 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. As the species is of high sensitivity to disturbance, the impact significance is **minor adverse**.

Table 13.20 Displacement matrix presenting the number of red-throated divers in the Norfolk Boreas site (and 4km buffer) during the winter period that may be subject to mortality (highlighted) on the assumption of complete development of the site.

Mortality (%)	Displacement (%) 10	20	30	40	50	60	70	80	90	100
1	0	0	0	1	1	1	1	1	1	2
2	0	1	1	1	2	2	2	2	3	3
3	0	1	1	2	2	3	3	4	4	5
4	1	1	2	2	3	4	4	5	6	6
5	1	2	2	3	4	5	5	6	7	8
6	1	2	3	4	5	6	7	7	8	9
7	1	2	3	4	5	7	8	9	10	11
8	1	2	4	5	6	7	9	10	11	12
9	1	3	4	6	7	8	10	11	13	14
10	2	3	5	6	8	9	11	12	14	16
20	3	6	9	12	16	19	22	25	28	31
30	5	9	14	19	23	28	33	37	42	47
50	8	16	23	31	39	47	55	62	70	78
75	12	23	35	47	59	70	82	94	105	117
100	16	31	47	62	78	94	109	125	140	156

195. Using the seasonal peak spring migration abundance on the Norfolk Boreas site (and 4km buffer) of 620 the maximum number of individual red-throated divers which could potentially suffer mortality as a consequence of displacement has been estimated to be between 6 and 62 individuals (Table 13.21). However, this abundance estimate is very strongly influenced by the March 2017 survey, conducted on the 29th and 30th March, which generated an estimated abundance on Norfolk Boreas and the 4km buffer of 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 the spring migration, rather than indicating a resident population (Wernham et al. 2002). Consequently, individuals recorded at this time will only be present for a short span of time and the assumption of 10% mortality is likely to over-estimate the magnitude of impact at this time. For these reasons, the evidence based 1% mortality rate is considered to be more appropriate for birds recorded in late March.

196. At an average mortality rate of 0.228 (Table 13.13), the number of individuals expected to die is 3,027 (13,277 x 0.228). The addition of a maximum of 62 individuals to this would increase the mortality rate by 2%, while this would be 0.2% using the evidence based 1% mortality rate. Thus during the spring 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**.

Table 13.21 Displacement matrix presenting the number of red-throated divers in the Norfolk Boreas site (and 4km buffer) during the spring migration season that may be subject to mortality (highlighted) on the assumption of complete development of the site.

Mortality (%)	Displacement (%)									
	10	20	30	40	50	60	70	80	90	100
1	1	1	2	2	3	4	4	5	6	6
2	1	2	4	5	6	7	9	10	11	12
3	2	4	6	7	9	11	13	15	17	19
4	2	5	7	10	12	15	17	20	22	25
5	3	6	9	12	16	19	22	25	28	31
6	4	7	11	15	19	22	26	30	33	37
7	4	9	13	17	22	26	30	35	39	43
8	5	10	15	20	25	30	35	40	45	50
9	6	11	17	22	28	33	39	45	50	56
10	6	12	19	25	31	37	43	50	56	62
20	12	25	37	50	62	74	87	99	112	124
30	19	37	56	74	93	112	130	149	167	186
50	31	62	93	124	155	186	217	248	279	310
75	47	93	140	186	233	279	326	372	419	465
100	62	124	186	248	310	372	434	496	558	620

197. The summed Norfolk Boreas site (and 4km buffer) displacement mortality for autumn, winter and spring is estimated to be between 7 (1%) and 80 (10%) individuals, although this figure also includes 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.

198. The total nonbreeding season mortality is therefore very unlikely to exceed the level at which it would increase the background mortality rate by a detectable amount. Therefore, the magnitude of effect is assessed as negligible even on the basis of the highly precautionary assessment approach. As the species is of high sensitivity to disturbance, the impact significance is **minor adverse**.

13.7.4.1.2 *Gannet*

199. Gannets show a low level of sensitivity to ship and helicopter traffic (Garthe and Hüppop, 2004; Furness and Wade, 2012); however, a detailed study (Krijgsveld et al., 2011) using radar and visual observations to monitor the post-construction effects of the Windpark Egmond aan Zee OWEZ established that 64% of gannets avoided entering the wind farm (macro-avoidance) and a similar result (80% macro avoidance) was also observed during a study at the Thanet wind farm (Skov et al., 2018). Leopold et al. (2013) reported that most gannets avoided Dutch offshore wind farms and did not forage within these. Dierschke et al. (2016) concluded that gannets show high avoidance of offshore wind farms despite showing little avoidance of ships.

200. The displacement matrices have been populated with data for gannets during the autumn and spring migration periods within the Norfolk Boreas site and those calculated within a 2km buffer, in line with guidance (JNCC, 2017). It should be noted that the inclusion of birds within the 2km buffer to determine the total number of birds subject to displacement is precautionary since in reality the avoidance rate is likely to fall with distance from the site, as demonstrated in a study of gannet distribution in relation to the Greater Gabbard wind farm (APEM, 2014).

201. For the purpose of this assessment, percentage displacement rates between 10 and 100% at 10% increments have been combined with mortality between 1 and 100% at varying increments. The highlighted cells in the matrices indicate displacement rates of 60% to 80% (as the OWEZ and Thanet data suggest the actual rate lies between these two figures based on macro-avoidance; Leopold et al., 2013; Skov et al., 2018) and the most likely mortality rate during the nonbreeding seasons, which is assumed to be no more than 1% (as they score highly for habitat flexibility; Furness and Wade,

2012). A high score in habitat flexibility is given to species that use a wide range of habitats over a large area, and usually with a relatively wide range of foods (Furness and Wade, 2012).

202. Within the range of 60-80% displacement and 1% mortality, the maximum number of individual gannets which could potentially suffer mortality as a consequence of displacement from the Norfolk Boreas site (and 2km buffer) during the autumn migration period has been estimated as 14 individuals (Table 13.22).

Table 13.22 Displacement matrix presenting the number of gannets in the Norfolk Boreas site (and 2km buffer) during the autumn migration season that may be subject to mortality (highlighted).

Mortality (%)	Displacement (%)									
	10	20	30	40	50	60	70	80	90	100
1	2	3	5	7	9	10	12	14	16	17
2	3	7	10	14	17	21	24	28	31	34
3	5	10	16	21	26	31	36	41	47	52
4	7	14	21	28	34	41	48	55	62	69
5	9	17	26	34	43	52	60	69	78	86
6	10	21	31	41	52	62	72	83	93	103
7	12	24	36	48	60	72	84	96	109	121
8	14	28	41	55	69	83	96	110	124	138
9	16	31	47	62	78	93	109	124	140	155
10	17	34	52	69	86	103	121	138	155	172
20	34	69	103	138	172	207	241	276	310	345
30	52	103	155	207	258	310	362	414	465	517
50	86	172	258	345	431	517	603	689	775	862
75	129	258	388	517	646	775	905	1034	1163	1292
100	172	345	517	689	862	1034	1206	1378	1551	1723

203. The BDMPS for gannet in autumn is 456,298 (Furness, 2015). At the average baseline mortality rate for gannet of 0.191 (Table 13.13), the number of individuals expected to die is 87,153 (456,298 x 0.191). The addition of a maximum of 14 to this increases the mortality rate by 0.016%. 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. Although gannets are considered to show high macro-avoidance of wind farms, which would suggest a high sensitivity score, this has been accounted for in the assessment in the application of a precautionary level of displacement (60-80%). Therefore, since this species has low sensitivity to other sources of disturbance such as vessels, a medium to low sensitivity has been assumed for displacement, with impact significance assessed as **negligible** to **minor adverse**.

204. Within the range of 60-80% displacement and 1% mortality, the maximum number of individual gannets which could potentially suffer mortality as a consequence of displacement from the Norfolk Boreas site (and 2km buffer) during the spring migration period has been estimated as four individuals (Table 13.23).

Table 13.23 Displacement matrix presenting the number of gannets in the Norfolk Boreas site (and 2km buffer) during the spring migration season that may be subject to mortality (highlighted).

Mortality (%)	Displacement (%)									
	10	20	30	40	50	60	70	80	90	100
1	1	1	2	2	3	3	4	4	5	5
2	1	2	3	4	5	6	7	8	9	11
3	2	3	5	6	8	9	11	13	14	16
4	2	4	6	8	11	13	15	17	19	21
5	3	5	8	11	13	16	18	21	24	26
6	3	6	9	13	16	19	22	25	28	32
7	4	7	11	15	18	22	26	29	33	37
8	4	8	13	17	21	25	29	34	38	42
9	5	9	14	19	24	28	33	38	43	47
10	5	11	16	21	26	32	37	42	47	53
20	11	21	32	42	53	63	74	84	95	105
30	16	32	47	63	79	95	110	126	142	158
50	26	53	79	105	132	158	184	210	237	263
75	39	79	118	158	197	237	276	316	355	395
100	53	105	158	210	263	316	368	421	473	526

205. The BDMPS for gannet in spring is 248,385 (Furness, 2015). At the average baseline mortality rate for gannet of 0.191 (Table 13.13), the number of individuals expected to die is 47,442 (248,385 x 0.191). The addition of a maximum of four 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 spring migration period, the magnitude of effect is assessed as negligible. Although gannets are considered to show high macro-avoidance of wind farms, which would suggest a high sensitivity score, this has been accounted for in the assessment in the application of a precautionary level of displacement (60-80%). Therefore, since this species has low sensitivity to other sources of disturbance such as vessels, a medium to low sensitivity has been assumed for displacement, with impact significance assessed as **negligible to minor adverse**.

206. Within the range of 60-80% displacement and 1% mortality, the maximum number of individual gannets which could potentially suffer mortality as a consequence of

displacement from the Norfolk Boreas site (and 2km buffer) during the breeding season has been estimated as 10 individuals (Table 13.24).

Table 13.24 Displacement matrix presenting the number of gannets in the Norfolk Boreas site (and 2km buffer) during the breeding season that may be subject to mortality (highlighted).

Mortality (%)	Displacement (%)									
	10	20	30	40	50	60	70	80	90	100
1	1	2	4	5	6	7	9	10	11	12
2	2	5	7	10	12	15	17	20	22	25
3	4	7	11	15	18	22	26	29	33	37
4	5	10	15	20	25	29	34	39	44	49
5	6	12	18	25	31	37	43	49	55	61
6	7	15	22	29	37	44	52	59	66	74
7	9	17	26	34	43	52	60	69	77	86
8	10	20	29	39	49	59	69	79	88	98
9	11	22	33	44	55	66	77	88	100	111
10	12	25	37	49	61	74	86	98	111	123
20	25	49	74	98	123	147	172	197	221	246
30	37	74	111	147	184	221	258	295	332	369
50	61	123	184	246	307	369	430	492	553	615
75	92	184	277	369	461	553	645	737	830	922
100	123	246	369	492	615	737	860	983	1106	1229

207. Although the Norfolk Boreas site, at 220km from Flamborough Head, is within the mean maximum gannet foraging range (229km) from the colony at Bempton Cliffs, the degree of connectivity indicated from tagging studies is considered to be low (e.g. Langston et al., 2013). However, as a precautionary assessment the breeding season impact has been assessed against this population. The population was estimated at 11,061 pairs in 2012 (Furness, 2015) but had risen to 13,391 pairs in 2017 (RSPB, unpubl colony report). At the average baseline mortality rate for gannet of 0.191 (Table 13.13), the number of individual adults predicted to die would be between 4,225 and 5,115 (22,122 to 26,782 x 0.191). The addition of 10 individuals to these would increase the mortality rate by 0.2% (note that this has been calculated for the adult breeding population only, which would be expected to comprise around 60% of the total population, thus adding further precaution to this assessment). This magnitude of increase in mortality would not materially alter the background mortality of the population and would be undetectable. Therefore, during the breeding season, and assessing the impact against a small adult population, the magnitude of effect is assessed as negligible. Although gannets are considered to show high macro-avoidance of wind farms, which would suggest a high sensitivity score, this has been accounted for in the assessment in the application of a precautionary level of displacement (60-80%). Therefore, since this

species has low sensitivity to other sources of disturbance such as vessels, a low to medium sensitivity has been assumed for displacement, with impact significance assessed as **negligible** to **minor adverse**.

208. Within the range of 60-80% displacement and 1% mortality, the maximum number of individual gannets which could potentially suffer mortality as a consequence of displacement from the Norfolk Boreas site (and 2km buffer) during the breeding, autumn migration and spring migration periods combined has been estimated as 28 individuals. The biogeographic gannet population is 1,180,000 (Furness, 2015).

209. At the average baseline mortality rate for gannet of 0.191 (Table 13.13) the number of individuals expected to die during the annual period is 225,380 (1,180,000 x 0.191). The addition of a maximum of 28 to this increases the mortality rate by 0.01%. This magnitude of increase in mortality would not materially alter the background mortality of the population and would be undetectable. Therefore, during the whole year, the magnitude of effect is assessed as negligible. Although gannets are considered to show high macro-avoidance of wind farms, which would suggest a high sensitivity score, this has been accounted for in the assessment in the application of a precautionary level of displacement (60-80%). Therefore, since this species has low sensitivity to other sources of disturbance such as vessels, a low to medium sensitivity has been assumed for displacement, with impact significance assessed as **negligible** to **minor adverse**.

13.7.4.1.3 *Auks (Guillemot and Razorbill)*

210. Razorbill and guillemot are considered to have a medium sensitivity to disturbance and displacement, based on their sensitivity to ship and helicopter traffic in Garthe and Hüppop (2004), Langston (2010), an interpretation of the Furness and Wade (2012) species concern index value in the context of disturbance and/or displacement from a habitat, and the meta-analysis of avoidance and attraction responses of seabirds to offshore wind farms by Dierschke et al. (2016).

211. Displacement of foraging seabirds due to the presence of wind turbines cannot readily be assessed from observing birds in flight as only a very small proportion of flying seabirds land in any particular location. There is not yet very much empirical data on displacement of foraging seabirds from offshore wind farms with the consequence that assessment of the amount of displacement arising from developments is somewhat speculative. Available pre- and post-construction data have yielded variable results but indicate that auks may be displaced to some extent by some wind farms, but this is partial, and apparently negligible in some sites (Dierschke et al., 2016).

212. Common guillemots were displaced at Blighbank (Vanermen et al., 2012), were displaced only in a minority of surveys at two Dutch wind farms (OWEZ and PAWP; Leopold et al., 2011; Krijgsveld et al., 2011), but were not significantly displaced at Horns Rev (although the data suggest that slight displacement was probably occurring; Petersen et al., 2006) or Thornton Bank (Vanermen et al., 2012). Razorbills were displaced in one out of six surveys at two Dutch wind farms (OWEZ and PAWP; Leopold et al., 2011, Krijgsveld et al., 2011), but not at Horns Rev (Petersen et al., 2006), Thornton Bank or Blighbank (Vanermen et al., 2012).

213. For recent wind farm assessments Natural England has advised that an unconfirmed 10% mortality rate should be used for auks displaced from wind farms. This magnitude of impact is not supported in the literature. For, example this would equate to a doubling of natural adult annual mortality for razorbill (10.5%; Horswill and Robinson 2015) and more than double that for guillemot (6%; Horswill and Robinson 2015). Therefore it is considered that this mortality rate is highly conservative and improbable in reality.

214. A review of available evidence for auk displacement was submitted for the Norfolk Vanguard assessment (MacArthur Green 2019b) and this concluded that precautionary rates of displacement and mortality from operational wind farms would be 50% and 1% respectively. These figures are also considered suitably precautionary for the potential displacement around construction vessels. Thus, the assessment presents estimates using 1% mortality (evidence based) and 10% (Natural England unconfirmed rate).

215. Following statutory guidance (Joint SNCB Note, 2017), the abundance estimates for the most relevant biological periods have each been placed into individual displacement matrices. Each displacement matrix contains the abundance of each auk species within the Norfolk Boreas site and the 2km buffer.

216. Each matrix displays displacement rates and mortality rates for each species. For the purpose of this assessment a displacement rate range of 50 to 70% and a precautionary mortality rate range of 1 to 10% are highlighted in each matrix , with the 50% / 1% derived from a review of evidence (Macarthur Green 2019b) and the 70% / 10% combination representing a highly precautionary worst case scenario as advised by Natural England. Mortality due to displacement might arise if displacement increased competition for resources in the remaining areas of auk habitat outside the wind farm. However, it should be recognised that the mortality rate due to displacement may well be 0% since the increase in density of auks outside the wind farm area will be negligible (because the rest of the available habitat is vast), and is very unlikely to be as high as these precautionary values.

217. 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 695,441 guillemots (BDMPS for the UK North Sea and Channel, 1,617,306 x 43%) and 94,007 razorbills (BDMPS for the UK North Sea and Channel, 218622 x 43%). For guillemot there is only one defined nonbreeding season (August - February), while for razorbill there are three (August - October, November - December and January - March; Table 13.11). The number of birds which could potentially be displaced has been estimated for each species-specific relevant season.

Razorbill

218. The estimated number of razorbills subject to mortality during the breeding period due to displacement from the Norfolk Boreas site (and 2k buffer; Table 13.25) is between 3 and 44 individuals (from 50%/1% to 70%/10%).

Table 13.25 Displacement matrix presenting the number of razorbills in the Norfolk Boreas site (and 2km buffer) during the breeding season that may be subject to mortality (highlighted).

Mortality (%)	Displacement (%)									
	10	20	30	40	50	60	70	80	90	100
1	1	1	2	3	3	4	4	5	6	6
2	1	3	4	5	6	8	9	10	11	13
3	2	4	6	8	9	11	13	15	17	19
4	3	5	8	10	13	15	18	20	23	25
5	3	6	9	13	16	19	22	25	28	32
6	4	8	11	15	19	23	26	30	34	38
7	4	9	13	18	22	26	31	35	40	44
8	5	10	15	20	25	30	35	40	45	50
9	6	11	17	23	28	34	40	45	51	57
10	6	13	19	25	32	38	44	50	57	63
20	13	25	38	50	63	76	88	101	113	126
30	19	38	57	76	95	113	132	151	170	189
50	32	63	95	126	158	189	221	252	284	315
75	47	95	142	189	236	284	331	378	425	473
100	63	126	189	252	315	378	441	504	567	630

219. At the average baseline mortality rate for razorbill of 0.174 (Table 13.13) the number of individuals expected to die in the breeding season is 16,357 (94,007 x 0.174). The addition of a maximum of 44 to this increases the mortality rate by 0.27%. This

magnitude of 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**.

220. The estimated number of razorbills subject to mortality during the autumn migration period due to displacement from the Norfolk Boreas site (and 2km buffer; Table 13.26) is between 1 and 18 individuals (from 50%/1% to 70%/10%).

Table 13.26 Displacement matrix presenting the number of razorbills in the Norfolk Boreas site (and 2km buffer) during autumn migration that may be subject to mortality (highlighted).

Mortality (%)	Displacement (%)									
	10	20	30	40	50	60	70	80	90	100
1	0	1	1	1	1	2	2	2	2	3
2	1	1	2	2	3	3	4	4	5	5
3	1	2	2	3	4	5	6	6	7	8
4	1	2	3	4	5	6	7	8	9	11
5	1	3	4	5	7	8	9	11	12	13
6	2	3	5	6	8	9	11	13	14	16
7	2	4	6	7	9	11	13	15	17	18
8	2	4	6	8	11	13	15	17	19	21
9	2	5	7	9	12	14	17	19	21	24
10	3	5	8	11	13	16	18	21	24	26
20	5	11	16	21	26	32	37	42	47	53
30	8	16	24	32	39	47	55	63	71	79
50	13	26	39	53	66	79	92	105	118	132
75	20	39	59	79	99	118	138	158	178	197
100	26	53	79	105	132	158	184	210	237	263

221. At the average baseline mortality rate for razorbill of 0.174 (Table 13.13) the number of individuals expected to die in the autumn migration period is 102,986 (591,874 x 0.174). The addition of a maximum of 18 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 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**.

222. The estimated number of razorbills subject to mortality during the winter period due to displacement from the Norfolk Boreas site (and 2km buffer; Table 13.27) is between 5 and 75 individuals (from 50%/1% to 70%/10%).

Table 13.27 Displacement matrix presenting the number of razorbills in the Norfolk Boreas site (and 2km buffer) during the winter period that may be subject to mortality (highlighted).

Mortality (%)	Displacement (%)									
	10	20	30	40	50	60	70	80	90	100
1	1	2	3	4	5	6	7	9	10	11
2	2	4	6	9	11	13	15	17	19	21
3	3	6	10	13	16	19	22	26	29	32
4	4	9	13	17	21	26	30	34	38	43
5	5	11	16	21	27	32	37	43	48	53
6	6	13	19	26	32	38	45	51	58	64
7	7	15	22	30	37	45	52	60	67	75
8	9	17	26	34	43	51	60	68	77	85
9	10	19	29	38	48	58	67	77	86	96
10	11	21	32	43	53	64	75	85	96	107
20	21	43	64	85	107	128	149	170	192	213
30	32	64	96	128	160	192	224	256	288	320
50	53	107	160	213	266	320	373	426	479	533
75	80	160	240	320	399	479	559	639	719	799
100	107	213	320	426	533	639	746	852	959	1065

223. At the average baseline mortality rate for razorbill of 0.174 (Table 13.13) the number of individuals expected to die in the winter is 38,040 (218,622 x 0.174). The addition of a maximum of 75 to this increases the mortality rate by 0.20%. This magnitude of increase in mortality would not materially alter the background mortality of the population and would be undetectable. Therefore, during the winter, the magnitude of effect is assessed as negligible. As the species is of medium sensitivity to disturbance, the impact significance is **minor adverse**.

224. The estimated number of razorbills subject to mortality during the spring migration period due to displacement from the Norfolk Boreas site (and 2km buffer; Table 13.28) is between 2 and 24 individuals (from 50%/1% to 70%/10%).

Table 13.28 Displacement matrix presenting the number of razorbills in the Norfolk Boreas site (and 2km buffer) during spring migration that may be subject to mortality (highlighted).

Mortality (%)	Displacement (%)									
	10	20	30	40	50	60	70	80	90	100
1	0	1	1	1	2	2	2	3	3	3
2	1	1	2	3	3	4	5	6	6	7
3	1	2	3	4	5	6	7	8	9	10
4	1	3	4	6	7	8	10	11	12	14
5	2	3	5	7	9	10	12	14	16	17
6	2	4	6	8	10	12	14	17	19	21
7	2	5	7	10	12	14	17	19	22	24

Mortality (%)	Displacement (%)									
	10	20	30	40	50	60	70	80	90	100
8	3	6	8	11	14	17	19	22	25	28
9	3	6	9	12	16	19	22	25	28	31
10	3	7	10	14	17	21	24	28	31	35
20	7	14	21	28	35	41	48	55	62	69
30	10	21	31	41	52	62	72	83	93	104
50	17	35	52	69	86	104	121	138	155	173
75	26	52	78	104	129	155	181	207	233	259
100	35	69	104	138	173	207	242	276	311	345

225. At the average baseline mortality rate for razorbill of 0.174 (Table 13.13) the number of individuals expected to die in the spring migration season is 102,986 (591,874 x 0.174). The addition of a maximum of 24 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 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**.

226. The estimated number of razorbills subject to mortality combined across all seasons due to displacement from the Norfolk Boreas site (and 2km buffer; Table 13.29) is between 12 and 161 individuals (from 50%/1% to 70%/10%).

Table 13.29 Displacement matrix presenting the number of razorbills in the Norfolk Boreas site (and 2km buffer) combined across the breeding, autumn migration, winter and spring migration periods that may be subject to mortality (highlighted).

Mortality (%)	Displacement (%)									
	10	20	30	40	50	60	70	80	90	100
1	2	5	7	9	12	14	16	18	21	23
2	5	9	14	18	23	28	32	37	41	46
3	7	14	21	28	35	41	48	55	62	69
4	9	18	28	37	46	55	64	74	83	92
5	12	23	35	46	58	69	81	92	104	115
6	14	28	41	55	69	83	97	111	124	138
7	16	32	48	64	81	97	113	129	145	161
8	18	37	55	74	92	111	129	147	166	184
9	21	41	62	83	104	124	145	166	187	207
10	23	46	69	92	115	138	161	184	207	230
20	46	92	138	184	230	276	322	368	415	461
30	69	138	207	276	345	415	484	553	622	691
50	115	230	345	461	576	691	806	921	1036	1152

Mortality (%)	Displacement (%)									
	10	20	30	40	50	60	70	80	90	100
75	173	345	518	691	864	1036	1209	1382	1555	1727
100	230	461	691	921	1152	1382	1612	1842	2073	2303

227. At the average baseline mortality rate for razorbill of 0.174 (Table 13.13) the number of individuals from the largest BDMPS population expected to die across all seasons is 102,986 (591,874 x 0.174). The addition of a maximum of 161 to this increases the mortality rate by 0.16%. The number of individuals from the biogeographic population expected to die across all seasons is 297,018 (1,707,000 x 0.174). The addition of a maximum of 161 to this increases the mortality rate by 0.05%. Thus, the increase in background mortality is between 0.05% and 0.16%.

228. These magnitudes of 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**.

Guillemot

229. The estimated number of guillemots subject to mortality during the breeding period due to displacement from the Norfolk Boreas site (and 2km buffer; Table 13.30) is between 39 and 544 individuals (within the range of displacement/mortality of 50%/1% to 70%/10%).

Table 13.30 Displacement matrix presenting the number of guillemots in the Norfolk Boreas site (and 2km buffer) in the breeding season that may be subject to mortality (highlighted).

Mortality (%)	Displacement (%)									
	10	20	30	40	50	60	70	80	90	100
1	8	16	23	31	39	47	54	62	70	78
2	16	31	47	62	78	93	109	124	140	155
3	23	47	70	93	117	140	163	186	210	233
4	31	62	93	124	155	186	217	249	280	311
5	39	78	117	155	194	233	272	311	350	388
6	47	93	140	186	233	280	326	373	419	466
7	54	109	163	217	272	326	381	435	489	544
8	62	124	186	249	311	373	435	497	559	621
9	70	140	210	280	350	419	489	559	629	699
10	78	155	233	311	388	466	544	621	699	777
20	155	311	466	621	777	932	1087	1243	1398	1553
30	233	466	699	932	1165	1398	1631	1864	2097	2330
50	388	777	1165	1553	1942	2330	2718	3107	3495	3884
75	583	1165	1748	2330	2913	3495	4078	4660	5243	5825

Mortality (%)	Displacement (%)									
	10	20	30	40	50	60	70	80	90	100
100	777	1553	2330	3107	3884	4660	5437	6214	6990	7767

230. At the average baseline mortality rate for guillemot of 0.140 (Table 13.13) the number of individuals expected to die in the breeding season is 97,362 (695,441 x 0.140). The addition of a maximum of 544 to this increases the mortality rate by 0.6%. This magnitude of 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**.

231. The estimated number of guillemots subject to mortality during the nonbreeding period due to displacement from the Norfolk Boreas site (and 2km buffer; Table 13.31) is between 69 and 964 individuals (from 50%/1% to 70%/10%).

Table 13.31 Displacement matrix presenting the number of guillemots in the Norfolk Boreas site (and 2km buffer) in the nonbreeding season that may be subject to mortality (highlighted).

Mortality (%)	Displacement (%)									
	10	20	30	40	50	60	70	80	90	100
1	14	28	41	55	69	83	96	110	124	138
2	28	55	83	110	138	165	193	220	248	276
3	41	83	124	165	207	248	289	331	372	413
4	55	110	165	220	276	331	386	441	496	551
5	69	138	207	276	344	413	482	551	620	689
6	83	165	248	331	413	496	579	661	744	827
7	96	193	289	386	482	579	675	772	868	964
8	110	220	331	441	551	661	772	882	992	1102
9	124	248	372	496	620	744	868	992	1116	1240
10	138	276	413	551	689	827	964	1102	1240	1378
20	276	551	827	1102	1378	1653	1929	2204	2480	2755
30	413	827	1240	1653	2067	2480	2893	3306	3720	4133
50	689	1378	2067	2755	3444	4133	4822	5511	6200	6889
75	1033	2067	3100	4133	5166	6200	7233	8266	9299	10333
100	1378	2755	4133	5511	6889	8266	9644	11022	12399	13777

232. At the average baseline mortality rate for guillemot of 0.140 (Table 13.13) the number of individuals expected to die in the nonbreeding season is 226,423 (1,617,306 x 0.140). The addition of a maximum of 964 to this increases the mortality rate by 0.4%. This magnitude of increase in mortality would not materially

alter the background mortality of the population and would be undetectable. Therefore, during the nonbreeding period, the magnitude of effect is assessed as negligible. As the species is of medium sensitivity to disturbance, the impact significance is **minor adverse**.

233. The estimated number of guillemots subject to mortality combined across all seasons due to displacement from the Norfolk Boreas site (and 2km buffer; Table 13.32) is between 108 and 1508 individuals (from 50%/1% to 70%/10%).

Table 13.32 Displacement matrix presenting the number of guillemots in the Norfolk Boreas site (and 2km buffer) combined across the breeding and nonbreeding seasons that may be subject to mortality (highlighted).

Mortality (%)	Displacement (%)									
	10	20	30	40	50	60	70	80	90	100
1	22	43	65	86	108	129	151	172	194	215
2	43	86	129	172	215	259	302	345	388	431
3	65	129	194	259	323	388	452	517	582	646
4	86	172	259	345	431	517	603	689	776	862
5	108	215	323	431	539	646	754	862	969	1077
6	129	259	388	517	646	776	905	1034	1163	1293
7	151	302	452	603	754	905	1056	1206	1357	1508
8	172	345	517	689	862	1034	1206	1379	1551	1724
9	194	388	582	776	969	1163	1357	1551	1745	1939
10	215	431	646	862	1077	1293	1508	1724	1939	2154
20	431	862	1293	1724	2154	2585	3016	3447	3878	4309
30	646	1293	1939	2585	3232	3878	4524	5171	5817	6463
50	1077	2154	3232	4309	5386	6463	7540	8618	9695	10772
75	1616	3232	4847	6463	8079	9695	11311	12926	14542	16158
100	2154	4309	6463	8618	10772	12926	15081	17235	19390	21544

234. At the average baseline mortality rate for guillemot of 0.140 (Table 13.13) the number of individuals from the largest BDMPS population expected to die across all seasons is 226,423 (1,617,306 x 0.140). The addition of a maximum of 1508 to this increases the mortality rate by 0.67%. The number of individuals from the biogeographic population expected to die across all seasons is 577,500 (4,125,000 x 0.140). The addition of a maximum of 1508 to this increases the mortality rate by 0.26%. Thus, the maximum estimate of increase in background mortality is between 0.67% and 0.26%.

235. These magnitudes of 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**.

13.7.4.2 Impact 4: Indirect impacts through effects on habitats and prey species

236. Indirect disturbance and displacement of birds may occur during the operational phase if there are impacts on prey species and the habitats of prey species. These indirect effects include those resulting from the production of underwater noise (e.g. the turning of the wind turbines), loss of habitat, electro-magnetic fields (EMF) and the generation of suspended sediments (e.g. due to scour or maintenance activities) that may alter the behaviour or availability of bird prey species. Underwater noise and EMF may cause fish and mobile invertebrates to avoid the operational area and also affect their physiology and behaviour. Suspended sediments may cause fish and mobile invertebrates to avoid the operational area and may smother and hide immobile benthic prey. These mechanisms could result in less prey being available within the operational area to foraging seabirds. Changes in fish and invertebrate communities due to changes in presence of hard substrate (resulting in colonisation by epifauna and provision of novel habitat providing shelter for fish and invertebrates) may also occur, and changes in fishing activity could influence the communities present.

237. With regard to noise impacts on fish, Chapter 11 Fish and Shellfish Ecology discusses the potential impacts upon fish relevant to ornithology as prey species. With regard to behavioural changes related to underwater noise impacts on fish during the operation of the proposed project, Chapter 11 reports that the sensitivity of fish and shellfish species to operational noise is considered to be low and the magnitude of effect negligible. It concludes a negligible impact on fish. With a negligible impact on fish that are bird prey species, it is reasonable to conclude that the indirect impact on seabirds occurring in or around the Norfolk Boreas site during the operational phase is similarly a negligible adverse impact.

238. With regard to changes to the seabed and to suspended sediment levels, Chapter 8 Marine geology, Oceanography and Physical Processes discusses the nature of any change and impact. It identifies that the small quantities of sediment released due to scour processes would rapidly settle within a few hundred metres of each wind turbine or cable protection structure. Therefore, the magnitude of the impact on benthic species is likely to be negligible to low (see Chapter 10 Benthic Ecology) and that smothering due to increased suspended sediment during operation of the project would result in an impact of minor adverse significance. With a minor impact on benthic habitats and species, it is reasonable to conclude that the indirect impact on seabirds occurring in or around the Norfolk Boreas site during the operational phase is similarly a minor adverse impact.

239. With regard to EMF effects these are identified as highly localised with the majority of cables being buried to up to 3m depth, further reducing the effect of EMF (see Chapter 10).

240. Very little is known about potential long-term changes in invertebrate and fish communities due to colonisation of hard substrate and changes in fishing pressures at the Norfolk Boreas site. Whilst the impact of the colonisation of introduced hard substrate is seen as a minor adverse impact in terms of benthic ecology (as it is a change from the baseline conditions), the consequences for seabirds may be positive or negative locally but are unlikely to be significant at a wider scale. Dierschke et al. (2016) concluded that cormorants (both great cormorant and European shag) tend to be attracted to offshore wind farms because structures provide an opportunity for cormorants to roost and to dry their wings so extend their potential foraging habitat further offshore. Several gull species and red-breasted mergansers were found to tend to increase in abundance at offshore wind farms, which Dierschke et al. (2016) interpreted as most likely to be responses to increased foraging opportunities resulting from higher abundance of fish and invertebrates associated with offshore wind farm structures and possibly the reduction in fishing activity.

241. Overall the magnitude of impact is considered negligible on benthic invertebrates and low on fish. With a minor or negligible impact on invertebrates and fish, it is reasonable to conclude that the indirect impact on seabirds occurring in or around the Norfolk Boreas site during the operational phase is similarly a **negligible** or **minor adverse impact**.

13.7.4.3 Impact 5: Collision risk

242. There is a potential risk of collision with the wind turbine rotors and associated infrastructure resulting in injury or fatality to birds which fly through the Norfolk Boreas site whilst foraging for food or commuting between breeding sites and foraging areas.

243. Initial screening for species to include in the collision risk assessment is presented in Table 13.33. Species where risk of collision is assessed as very low were screened out. Species where risk of collision is assessed as low were screened out if their abundance in flight was very low or low. To be precautionary, all species where risk of collision is assessed as medium or high were screened in, even if their abundance in flight was very low.

Table 13.33 Collision risk screening. Species were screened in on the basis of columns two and three.

Receptor	Risk of collisions (Garthe and Hüppop, 2004; Furness and Wade, 2012, Wade *et al.*, 2016)	Estimated density of birds in flight at Norfolk Boreas	Screening Result (IN or OUT)
Red-throated diver	Low	Medium	**IN**
Great northern diver	Low	Very low	**OUT**
Fulmar	Low	High	**IN**
Gannet	Medium	Medium	**IN**

Receptor	Risk of collisions (Garthe and Hüppop, 2004; Furness and Wade, 2012, Wade *et al.*, 2016)	Estimated density of birds in flight at Norfolk Boreas	Screening Result (IN or OUT)
Arctic skua	Medium	Very low	**IN**
Great skua	Medium	Very low	**IN**
Puffin	Very low	Very low	**OUT**
Razorbill	Very low	High	**OUT**
Common guillemot	Very low	High	**OUT**
Common tern	Low	Low	**OUT**
Arctic tern	Low	Low	**OUT**
Sandwich tern	Low	Low	**OUT**
Kittiwake	Medium	High	**IN**
Black-headed gull	Medium	Low	**IN**
Little gull	Medium	Low	**IN**
Common gull	Medium	Low	**IN**
Lesser black-backed gull	High	Medium	**IN**
Herring gull	High	Low	**IN**
Great black-backed gull	High	Low	**IN**

244. CRM has been used in this assessment to estimate the collision risk mortality to birds associated with the proposed project. Deterministic CRM, using the Band model (Band, 2012) Options 1 and 2 has been used to produce predictions of mortality for particular species across set time periods (biological seasons). The approach to CRM is summarised here and further details are provided in Technical Appendix 13.1.

245. The difference between Options 1 and 2 is the source of flight height data used to estimate the proportion of time each species will spend at potential collision height (PCH). Option 1 uses site and species-specific data collected during site characterisation surveys. Option 2 uses generic estimates of flight height for each species (Johnston et al., 2014 a,b) to estimate PCH. Natural England advice is to present the results from both options, but to base assessment on option 1 if sufficient height data records are available. The minimum threshold for use of Option 1 for a particular species which has typically been applied is 100 flight height observations.

246. However, following a review of their data collection and analysis methods, APEM advised Vattenfall Wind Power Limited that the flight height estimates supplied as part of the survey data were not sufficiently reliable for use in CRM. Furthermore, the parameters required to correct for the methodological errors had not been

recorded during the surveys and therefore it was not possible to re-estimate the heights.

247. Consequently, and in agreement with Natural England, the collision mortalities used for impact assessment for all species are those calculated using Option 2 of the Band model (although the erroneous flight height estimates and Option 1 results have also been provided in Technical Appendix 13.1).

248. Natural England requested that the CRM results should incorporate uncertainty in seabird density, collision avoidance rates, flight heights and also to provide consideration of a range of nocturnal activity rates. These requests reflect the fact that many of the CRM input parameters include sources of both natural variation (e.g. seabird densities) and measurement error, as well as precautionary estimates of parameters.

249. To incorporate variation in the model input parameters, CRM was run using the mean values for each of the above list of parameters as well as using the upper and lower 95% confidence interval values. The input parameters for CRM are provided in Technical Appendix 13.1 Annex 3 and the outputs are presented in full in Technical Appendix 13.1 Annex 4.

250. For the collision risk assessments, the monthly mean density values calculated from the survey data were used (the monthly values were derived as the mean of two months of survey density data) and the upper and lower 95% confidence intervals surrounding the mean densities were derived from 1,000 nonparametric bootstrap simulations (see Technical Appendix 13.1 for details).

251. Collision avoidance rates used were those recommended by the SNCBs (JNCC et al., 2014) following the review conducted by the British Trust for Ornithology (BTO) on behalf of Marine Scotland (Cook et al., 2014). These are 98.9% for gannet and kittiwake, 99.5% for lesser black-backed gull, herring gull and great black-backed gull, 99.2% for little gull, common gull and black-headed gull and 98% for all other species..

252. It should be noted that estimation of avoidance rates at offshore wind farms is an area of ongoing research. For example, a study on gannet behaviour in relation to offshore wind farms (APEM, 2014) gathered evidence which suggests this species may exhibit a higher avoidance rate than the current recommended rate of 98.9%. This work, conducted during the autumn migration period, indicated an overall wind turbine avoidance of 100%, although a precautionary rate of 99.5% was proposed (for the autumn period at least). Although this rate has not been applied to the estimates presented in this assessment, it indicates that gannet collision mortality estimated at 98.9% is likely to overestimate the risk for this species, potentially by at least 50% and possibly higher. Indeed, as noted in Cook et al. (2014), all the

recommended avoidance rates remain precautionary and thus the results presented in this assessment are worst case estimates.

253. A bird flight behaviour study has been conducted for the Offshore Renewables Joint Industry Programme (ORJIP). The final report for this study provides further evidence relating to the precautionary nature of current avoidance rates and other parameters used in wind farm assessment (Skov et al., 2018).

254. A further analysis of the data collected by Skov et al. (2018) was conducted by Bowgen and Cook (2018) and recommended avoidance rates for use with the deterministic Band model (options 1 and 2) 99.5% for gannet and large gulls and 99.0% for kittiwake. Bowgen and Cook (2018) also recommend avoidance rates for stochastic versions of the basic model (Options 1 and 2) for kittiwake of 99.4% (95% c.i. of 97.6% to 99.8%) and for large gulls of 99.7% (95% c.i. of 99.2% to 99.9%). However, as a precautionary approach has been taken in this ES, these higher avoidance rates have not been incorporated into the CRM analysis.

255. The nocturnal activity parameter used in the CRM defines the level of nocturnal activity of each seabird species, expressed in relation to daytime activity levels. For example, a value of 50% for the nocturnal activity factor is appropriate for a species which is half as active at night as during the day ('activity' in the current context refers to flight activity). This factor is used to enable estimation of nocturnal collision risk from survey data collected during daylight, with the total collision risk the sum of those for day (sunrise to sunset) and night (sunset to sunrise). The values typically used for each species were derived from reviews of seabird activity reported in Garthe and Hüppop (2004). This review ranked species from 1 to 5 (1 low, 5 high) for relative nocturnal activity, and these were subsequently modified for the purposes of CRM into 1 = 0% to 5 = 100%. This approach was not anticipated by Garthe and Hüppop (2004), who considered that their 1 to 5 scores were simply categorical and were not intended to represent a scale of 0 to 100% of daytime activity (not least because the lowest score given was 1 and not 0). This is clear from their descriptions of the scores: for example, for score 1 '*hardly any flight activity at night'*.

256. Recently however, a number of studies have deployed loggers on seabirds, and data from those studies provide empirical evidence of the actual flight activity level throughout the daily cycle. These studies indicate that the rates derived from Garthe and Hüppop (2004) overestimate the levels of nocturnal activity in the species studied. For example, across four studies of gannet, nocturnal activity relative to daytime was reported as between 0% and 2%, across four studies of kittiwake nocturnal activity relative to daytime was reported as between 0% and 12% and in one study of lesser black-backed gull nocturnal activity relative to daytime was reported as 25%. These compare to the much higher values

recommended by SNCBs for use in CRM of 25%, 50% and 50% for gannet, kittiwake and lesser black-backed gull respectively.

257. As the relative proportion of daytime to night-time varies considerably during the year at UK latitudes, the effect of changes in the nocturnal activity factor for CRM outputs depends on the relative abundance of birds throughout the year (i.e. a reduction in this parameter will have a much greater effect on collision estimates in mid-winter than mid-summer). The extent of mortality reduction obtained by reducing the categorical score for five species (gannet, kittiwake, lesser black-backed gull, herring gull and great black-backed gull) by 1 (i.e. from 3 to 2 for kittiwake) has been investigated previously (EATL, 2015). That work revealed annual mortality estimate reductions of between 14.5% (lesser black-backed gull) and 27.7% (gannet).

258. In the light of the growing evidence of the over-precaution in nocturnal activity scores, Natural England has advised that CRM should use upper and lower nocturnal activity rates of 0% and 25% for gannet and 25% and 50% for kittiwake, lesser black-backed gull, great black-backed gull and herring gull, rather than just the higher value as used previously.

259. However, it is obviously preferable to obtain evidence-based estimates derived from empirical data. Consequently, reviews of evidence from gannet and kittiwake tracking studies have been undertaken (Furness et al., 2018 and Furness et al., in prep.).

260. Furness et al. (2018) recommended precautionary nocturnal activity rates for gannet in the breeding and nonbreeding seasons of 8% and 4% respectively. Evidenced based nocturnal factors were additionally presented for gannet in the current CRM assessment in addition to the Natural England recommended rates noted above. The evidence-based nocturnal activity rate estimated in Furness et al. (in prep) for kittiwake in the d nonbreeding seasons is 20% (SE 5%). Further work is underway to extend the kittiwake analysis to the breeding season, however results from this are not yet available.

261. Preliminary modelling was conducted for eight turbine models from 180 x 10MW to 90 x 20MW. This identified the 10MW turbine as the worst case for collisions and only the results from this turbine have been presented. The full results for the 10MW turbine are provided in Technical Appendix 13.1 Annex 4.

262. A number of the seabird species which were only recorded in small numbers during aerial surveys of the survey programme were identified as potential migrants through the Norfolk Boreas site (e.g. great skua, Arctic skua). These species were included in the CRM but were also assessed using the methods described in WWT and MacArthur Green (2013).

263. The risk of collisions for non-seabird migrants was considered using the migrant collision Band (2012) model in conjunction with population estimates and migration routes presented in Wright et al. (2012). The full analysis is presented in Technical Appendix 13.1 Annex 7 with a summary of the results provided below.

264. Seasonal mortality predictions have been compared to the relevant BDMPS populations and the predicted increases in background mortality which could result have been estimated.

13.7.4.3.1 *Assessment of CRM results*

Seabirds

265. The following sections provide a summary of the outputs for assessment, using the seasons defined in Table 13.11. Annual collision risk estimates for all species assessed are presented in Table 13.34. For each species, annual totals are presented.

266. Several species had very low predicted annual collision risks (Table 13.34). These were red-throated diver, fulmar, Arctic skua, great skua, black-headed gull, little gull and common gull. As the magnitudes of predicted impact were so small, even for the worst case 10MW turbine, no further assessment is considered necessary for these species.

267. The seasonal collision estimates for species at higher risk of collision (gannet, kittiwake, herring gull, lesser black-backed gull and great black-backed gull) are presented in more detail below (Table 13.35).

268. Impacts during the non-breeding periods have been assessed in relation to the relevant BDMPS (Furness, 2015). Impacts during the breeding season have been assessed in relation to reference populations calculated as described in the following sections.

Table 13.34 Annual collision risk for the Norfolk Boreas site using the worst case 10MW turbine option and Band Option 2. Estimates are mean and 95% confidence intervals.

Species	Deterministic CRM
Red-throated diver	1.85 (0-6.64)
Fulmar	6.97 (0.54-16.42)
Gannet	117.63 (32.45-239.62)
Arctic skua	0.35 (0-1.57)
Great skua	1.86 (0-5.43)
Kittiwake	202.80 (86.16-354.67)
Black-headed gull	14.91 (0-40.34)
Little gull	3.88 (0-13.87)
Common gull	14.02 (0.91-47.92)
Lesser black-backed gull	39.78 (3.96-108.27)
Herring gull	18.43 (0-56.23)
Great black-backed gull	93.11 (14.37-201.62)

Table 13.35 Seasonal and annual collision risks (mean and 95% c.i.) for gannet, kittiwake, lesser black-backed gull, herring gull and great black-backed gull at the Norfolk Boreas site for the worst case 10MW turbine, Band Option 2 deterministic CRM. Outputs for the full and migration free breeding seasons are provided with other seasons adjusted to avoid overlap between months (months as per Furness 2015).

Species	Breeding season	Breeding	Autumn migration	Mid-winter / nonbreeding	Spring migration	Annual
Gannet	Migration free	45.49 (0.97-112.89)	55.07 (24.36-94.69)	-	17.06 (7.11-32.03)	117.63 (32.45-239.62)
	Full	54.13 (2.61-132.47)	48.5 (22.72-80.75)	-	14.99 (7.11-26.39)	
Kittiwake	Migration free	29.92 (7.76-60.02)	116.59 (59.95-191)	-	56.29 (18.45-103.66)	202.8 (86.16-354.67)
	Full	46.9 (12.19-96.63)	113.74 (59.95-182.54)	-	42.16 (14.02-75.51)	
Lesser black-backed gull	Migration free	8.02 (1.02-22.33)	25.71 (2.94-61.4)	4.14 (0-15.34)	1.91 (0-9.22)	39.78 (3.96-108.27)
	Full	17.3 (3.96-42.6)	17.88 (0-47.64)	4.14 (0-15.34)	0.46 (0-2.71)	
Herring gull	Migration free	1.18 (0-4.71)	7.03 (0-20.42)	5.57 (0-15.75)	4.66 (0-15.36)	18.43 (0-56.23)
	Full	3.93 (0-14.66)	4.8 (0-13.61)	5.57 (0-15.75)	4.14 (0-12.22)	
Great black-backed gull	Migration free	7.75 (0-18.22)	40.45 (5.17-91.8)	15.43 (8.21-23.89)	29.47 (0.99-67.7)	93.11 (14.37-201.62)
	Full	17.94 (0-42.31)	36.03 (5.17-81.76)	15.43 (8.21-23.89)	23.7 (0.99-53.65)	

Breeding season reference populations for collision assessment

Gannet

269. While the Norfolk Boreas site is within the foraging range of gannets (mean max. 229km; Thaxter et al., 2012a) from Flamborough and Filey Coast SPA (220km at its closest), tracking studies of adult gannets breeding at Flamborough and Filey Coast SPA have found that very few foraging trips extend as far as the wind farm (e.g. Langston et al., 2013). Nevertheless, the potential for connectivity exists for this population, therefore assessment has been conducted against this breeding population. To estimate the total population for these colonies (i.e. accounting for sub-adult ages classes) the number of breeding pairs has been multiplied by 2 (to obtain the number of adults) and divided by the adult proportion in Table 13.13. For gannet, the most recent census for Flamborough and Filey Coast SPA was in 2017 which recorded 13,391 pairs. This gives a breeding season reference population of 44,637 ((13391 x 2)/0.6).

Kittiwake

270. The Norfolk Boreas site is generally beyond the range of kittiwake from any large breeding colonies (Flamborough and Filey Coast SPA is the nearest SPA population, at 220km, although more recent tracking work has found some instances of birds foraging over these distances (Wischnewski et al. 2018). It is therefore very unlikely that birds present during the breeding season are breeding. While RSPB's Future of the Atlantic Marine Environments (FAME) studies have shown some extremely long foraging trips for this species, those extreme values tend to occur at colonies where food supply is extremely poor and breeding success is low (for example Orkney and Shetland). Daunt et al. (2002) point out that seabirds, as central place foragers, have an upper limit to their potential foraging range from the colony, set by time constraints. For example, they assess this limit to be 73km for kittiwake based on foraging flight speed and time required to catch food, based on observations of birds from the Isle of May. This means that kittiwakes would be unable to consistently travel more than 73km from the colony and provide enough food to keep chicks alive. Hamer et al. (1993) recorded kittiwake foraging ranges exceeding 40km in 1990 when sandeel stock biomass was very low and breeding success at the study colony in Shetland was 0.0 chicks per nest, but <5km in 98% of trips in 1991 when sandeel abundance was higher and breeding success was 0.98 chicks per nest. Kotzerka et al. (2010) reported a maximum foraging range of 59km, with a mean range of around 25km for a kittiwake colony in Alaska. Consequently, the breeding season impact on kittiwake has been assessed against a reference population estimated using the same approach as that for the displacement assessment (section 13.7.4.1). This is based on the observation that immature birds tend to remain in wintering areas. Thus, the number of immature birds in the relevant populations

during the breeding season may be estimated as the proportion of the relevant BDMPS (the one immediately preceding the breeding season) which are sub-adults. Thus, the breeding season reference population can be calculated as 47.3% of the spring BDMPS populations of kittiwake (see Table 13.13). This yields a breeding season population of nonbreeding kittiwake of 296,956 (Spring BDMPS for the UK North Sea and Channel, 627,816 x 47.3%).

<u>Lesser black-backed gulls</u>

271. Lesser black-backed gulls breed at the Alde-Ore Estuary SPA which, at 117km from the nearest point to the Norfolk Boreas site, is within the species' 141km mean maximum foraging range (Thaxter et al., 2012a). Thus, there is potential for connectivity with the Norfolk Boreas site during the breeding season.

272. 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 SCM database). That means that 68% of the breeding population was within the Alde-Ore Estuary SPA in 2001. 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 SCM database for Orfordness since 2012 and that limits understanding of any changes that have occurred since.

273. 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 2012). 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), and the 2012 census of urban breeding gulls in Suffolk was carried out after adverse conditions resulted in considerable breeding failure of many gulls (Piotrowski 2012) 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. In addition, breeding numbers have increased at Felixstowe (1,401 pairs in 2013) and Ipswich (99 pairs in 2001, 262 pairs in 2012), which are also urban colonies, and at Outer Trial Bank (1,704 pairs in 2006, 1,457 pairs in 2009 and 1,294 pairs in 2018, although this latter site is located beyond the species' reported foraging range) (JNCC SCM database) (Table 13.36).

Table 13.36 Lesser black-backed gull colonies in Norfolk And Suffolk.

Colony / Town	Minimum distance to Norfolk Boreas	Approximate number of pairs in 2008-2015
Great Yarmouth	73	750
Southtown	77	450
Lowestoft	78	2000
Alde-Ore Estuary SPA	117	2000
Felixstowe	135	700
Total within foraging range (141km)		5900
Outer Trial Bank	169	1300

274. 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 34% in 2012-2016 (2000/5900). 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 between 2012-2016.

275. There is also potential for connectivity between the project and colonies of lesser black-backed gulls in the Netherlands which are within 181km. However, extensive colour ringing and tracking of breeding lesser black-backed gulls from multiple colonies in the Netherlands has shown that there is very little or no connectivity during the breeding season between birds breeding in the Dutch colonies and the UK, and indeed that there is remarkably little migration of birds from the colonies in the Netherlands through UK waters even after the breeding season in autumn, winter or spring (Camphuysen, 2013). Not only do breeding adult lesser black-backed gulls from colonies in the Netherlands normally remain on the continental side of the North Sea while breeding, but 95% of their foraging trips in the 1990s and 2000s were less than 135km from those colonies (Camphuysen, 1995, 2013), and between 2008 and 2011 95% of foraging trips were within 60.5km of the colony (Camphuysen et al., 2015). Based on these foraging ranges, breeding adult lesser black-backed gulls from colonies in the Netherlands would be very unlikely to reach the Norfolk Boreas site. Therefore, during the breeding season, it is likely that adult lesser black-backed gulls at the Norfolk Boreas site will originate from the Alde-Ore Estuary SPA and other non-SPA colonies in East Anglia. However, these birds may be mixed with non-breeding birds from a variety of sources, so that any impact on lesser black-backed gulls due to Norfolk Boreas will be on a mixture of breeding birds from Alde-Ore Estuary, breeding birds from non-SPA colonies and immatures/nonbreeders from many different sources.

276. Thaxter et al., (2012b, 2015) tracked breeding adult lesser black-backed gulls from the Alde-Ore Estuary SPA and showed that birds differed in feeding habitat and area use both within and between seasons, as well as individually. Marine foraging

occurred more during chick-rearing, suggesting that connectivity with the Norfolk Boreas site would be most likely during the chick-rearing part of the breeding season, whereas early and late in the breeding season these birds foraged more in terrestrial and coastal habitats. This work has found that while the core areas, defined as the 50% and 75% kernel density estimates (KDE) respectively, remained fairly consistent across years, at the larger scale (defined as the 95% KDE) spatial distributions showed more variation. However, from the perspective of Norfolk Boreas, there was virtually no overlap between the foraging areas and the wind farm. It is therefore likely that few of the birds recorded during the breeding season on the Norfolk Boreas sites are breeding adults from this colony (see Norfolk Boreas ES Technical Appendix 13.1 Annex 7 for further details).

277. As discussed above, the non-SPA adult lesser black-backed gull population with potential for connectivity to the Norfolk Boreas site is likely to be at least 12,300 (6,150 pairs x 2) and could easily be larger when allowance is made for population increases since surveys were last conducted. This estimate is also derived from partial coverage of urban locations at which gulls may breed (e.g. Norfolk appears to have had very limited coverage). This, together with the fact that there are over 230km of coastline within foraging range of the Norfolk Boreas site, also suggests the actual non-SPA lesser black-backed gull population within range of the Norfolk Boreas site is likely to be much larger. On the basis that adults comprise approximately 58% of the population (Furness 2015), the total population in the region is likely to be in excess of 21,200.

278. The Alde-Ore SPA lesser black-backed gull breeding population has been around 2,000 pairs between 2007 and 2014 (minimum 1,580 pairs in 2011, maximum 2,769 pairs in 2008). This estimate for the breeding population size is considered robust since it takes into account observed inter-annual variations over a span of representative years for which data are available. This suggests that the total population (all age classes) associated with the SPA is around 6,700 individuals.

279. Incorporating all of the above evidence, a worst case (precautionary) assumption has been made that the breeding season reference population is 21,200 individuals, 32% of which potentially originate from the Alde-Ore SPA population (tracking data suggest a much lower value than this, but do not permit a robust quantification).

Herring gull

280. Norfolk Boreas is 117 km from the nearest breeding colony for herring gull at Alde-Ore Estuary. This species has a mean maximum foraging range of 61 km, and a maximum recorded foraging range of 92 km. Therefore the likelihood that herring gulls breeding at Alde-Ore Estuary would reach the Norfolk Boreas site is extremely small. Consequently, the breeding season impact on herring gull has been assessed against a reference population estimated using the same approach as that used in

the ES for other species for which breeding adults were considered unlikely to be present. This is based on the observation that immature birds tend to remain in wintering areas. Thus, the number of immature birds in the relevant populations during the breeding season may be estimated as the proportion of the relevant biologically defined minimum population scale (BDMPS) season (the one immediately preceding the breeding season) which are sub-adults. Thus, the breeding season reference population can be calculated as 66.4% (the proportion of sub-adults in the population, (see Table 13.13) of the nonbreeding BDMPS populations of herring gull. This yields a breeding season population of nonbreeding herring gull of 309,763 (nonbreeding BDMPS for the UK North Sea and Channel, 466,511 x 66.4%). The nonbreeding season reference population was 466,511 (Furness 2015).

Great black-backed gull

281. There are no breeding colonies for this species within foraging range of the Norfolk Boreas site (the closest SPA populations are the Isles of Scilly and East Caithness Cliffs, both over 650 km away). Consequently, the breeding season impact on great black-backed gull has been assessed against a reference population estimated using the same approach as that for the displacement assessment (section 13.7.4.1). This is based on the observation that immature birds tend to remain in wintering areas. Thus, the number of immature birds in the relevant populations during the breeding season may be estimated as the proportion of the relevant BDMPS (the one immediately preceding the breeding season) which are sub-adults. Thus, the breeding season reference population can be calculated as 57.8% of the nonbreeding BDMPS populations of great black-backed gull (see Table 13.13). This yields a breeding season population of nonbreeding great black-backed gull of 52,829 (nonbreeding BDMPS for the UK North Sea and Channel, 91,399 x 57.8%).

Nonbreeding season reference populations for collision assessment

282. As advised by Natural England, the nonbreeding season reference populations were taken from Furness (2015).

Collision impacts

283. The impacts of mortality caused by collisions on the populations are assessed in terms of the change in the baseline mortality rate which could result. It has been assumed that all age classes are equally at risk of collisions (i.e. in proportion to their presence in the population), therefore it is necessary to calculate an average baseline mortality rate for all age classes for each species assessed. These were calculated using the different survival rates for each age class and their relative proportions in the population (Table 13.13).

284. Table 13.37 provides the baseline survival rates, the relevant breeding season and nonbreeding season BDMPS and the percentage increase in mortality for each seabird species due to collisions.

285. The mean collision predictions for all species in all seasons and also summed across the year resulted in increases in background mortality below 1%. Therefore, the magnitude of effects due to collision mortality for gannet, kittiwake, lesser black-backed gull, herring gull and great black-backed gull are considered to be negligible, resulting in impact significances of **negligible** to **minor adverse**.

286. In two cases the upper 95% confidence interval of the seasonal collision estimate corresponded to an increase in the background mortality above 1% in the breeding season; gannet (1.55%) and lesser black-backed gull in the breeding season (1.62%). However, these results reflect a combination of worst cases in project design (the 10MW turbine) and were only obtained at the top end of the range of combined upper values for seabird density, flight height, avoidance rate and nocturnal activity and therefore this does not alter the assessed impact significance.

287. Natural England have advised that they consider gannet may potentially be at risk of both operational displacement and collision risk (although it is important to note that combining the estimated mortality for these effects will include an unknown degree of double counting). The addition of the maximum annual displacement total estimate of 28 (13.7.4.1.2) to the predicted annual collision mortality of 118 would not materially alter the above conclusion of at worst a minor adverse effect.

Table 13.37. Percentage increases in the background mortality rate of seasonal and annual populations due to predicted collisions calculated using Band Option 2 deterministic CRM. Note that the annual mortalities have been assessed against both the biogeographic populations and the largest BDMPS (as advised by Natural England) in order to bracket likely effects.

Species		Gannet			Kittiwake			Lesser black-backed gull			Herring gull			Great black-backed gull		
		Mean	Lower c.i.	Upper c.i	Mean	Lower c.i.	Upper c.i	Mean	Lower c.i.	Upper c.i	Mean	Lower c.i.	Upper c.i	Mean	Lower c.i.	Upper c.i
Baseline average mortality		0.191			0.156			0.124			0.172			0.144		
Breeding season	Reference population	44,637			296,956			21,200			309,763			52,829		
	Seasonal mortality	54.13	2.61	132.47	29.92	7.76	60.02	17.30	3.96	42.60	3.93	0.00	14.66	7.75	0.00	18.22
	Increase in background mortality (%)	0.635	0.031	1.554	0.065	0.017	0.130	0.658	0.151	1.621	0.007	0.000	0.028	0.102	0.000	0.240
Autumn	Reference population	456298			829937			209007								
	Seasonal mortality	48.50	22.72	80.75	116.59	59.95	191.00	17.88	0.00	47.64	-	-	-	-	-	-
	Increase in background mortality (%)	0.056	0.026	0.093	0.090	0.046	0.148	0.069	0.000	0.184	-	-	-	-	-	-
Wintering	Reference population							39,316			466,511			91,399		
	Seasonal mortality	-	-	-	-	-	-	4.14	0.00	15.34	5.57	0.00	15.75	15.43	8.21	23.89
	Increase in background mortality (%)	-	-	-	-	-	-	0.085	0.000	0.315	0.007	0.000	0.020	0.117	0.062	0.182
Spring	Reference population	248,385			627,816			197,483								
	Seasonal mortality	14.99	7.11	26.39	56.29	18.45	103.66	0.46	0.00	2.71	-	-	-	-	-	-

Species		Gannet			Kittiwake			Lesser black-backed gull			Herring gull			Great black-backed gull		
		Mean	Lower c.i.	Upper c.i	Mean	Lower c.i.	Upper c.i	Mean	Lower c.i.	Upper c.i	Mean	Lower c.i.	Upper c.i	Mean	Lower c.i.	Upper c.i
	Increase in background mortality (%)	0.032	0.015	0.056	0.057	0.019	0.106	0.002	0.000	0.011	-	-	-	-	-	-
Annual – largest BDMPS	Reference population	456,298			829,937			209,007			466,511			91,399		
	Annual mortality	117.63	32.45	239.62	202.80	86.16	354.67	39.78	3.96	108.27	18.43	0.00	56.23	93.11	14.37	201.62
	Increase in background mortality (%)	0.135	0.037	0.275	0.157	0.067	0.274	0.153	0.015	0.418	0.023	0.000	0.070	0.707	0.109	1.532
Annual - biogeographic	Reference population	1,180,000			5,100,000			864,000			1,098,000			235,000		
	Annual mortality	117.63	32.45	239.62	202.80	86.16	354.67	39.78	3.96	108.27	18.43	0.00	56.23	93.11	14.37	201.62
	Increase in background mortality (%)	0.052	0.014	0.106	0.025	0.011	0.045	0.037	0.004	0.101	0.010	0.000	0.030	0.275	0.042	0.596

Migrant seabirds

288. Some migratory seabirds may not have been accounted for from the standard survey methods as they may migrate across seas in large numbers, but over a short time period. These movements are also often at night and sometimes in bad weather (Cook et al., 2012). Most of the seabirds migrating through the Norfolk Boreas site were frequently detected on surveys, but five species (great skua, Arctic skua, common tern, Arctic tern and little gull) have been identified from previous studies as potentially traversing the region during migration seasons in large numbers (Wright et al., 2012).

289. Collision risk for these migrant seabirds was estimated following the approach in WWT & MacArthur Green (2013) and using population estimates in Furness (2015). These migrating seabirds tend to move parallel to the coast, in broad bands, their preferred distance from coasts dependent on species' ecology (Wernham et al., 2002; WWT & MacArthur Green, 2013). For example, great skuas tend to concentrate in the zone from one to thirty kilometres from the coast, and are much less frequently seen more than 40 km from coasts (Furness, 1987). The key parameters to be considered for these species are therefore the width of the coastal migration corridors (i.e. the routes followed on passage through the North Sea) and the percentage at collision height (Table 13.38).

Table 13.38 Key parameters for predicting collision risk for migrating seabirds.

Species	Main migration corridor (WWT & MacArthur Green 2013)	Percentage at rotor height calculated as >22m (Johnston et al. 2014a,b)
Arctic skua	0 – 20km	1.8
Great skua	0 – 40km	4.4
Arctic tern	0 – 20km	2.9
Common tern	0 – 10km	5.7
Little gull	0 – 20km	12.5

290. The Norfolk Boreas site is located a minimum of 73km from the coast at its nearest point. This is considerably further offshore than any of the corridor widths for the migrant seabird species in Table 13.38. While a few individuals will travel beyond the outer edges of these corridors, given the low percentages at collision height the overall collision risk will be very small. Consequently, any effects from Norfolk Boreas will be negligible and cause no material difference to current baseline mortality rates. The magnitude of effects is considered to be negligible for all species. Therefore, **no impacts** would be expected to result from collisions for any of these migrant seabird species. This conclusion is also consistent with the aerial survey data indicating very low numbers of these species in the survey area even during the migration seasons.

Migrant non-seabirds

291. This assessment included modelling to estimate the occurrence of other (terrestrial) migrant birds, including waders and wildfowl, in order to estimate potential collision risks (see Technical Appendix 13.1 Annex 7 for further details). Following a screening exercise, 25 non-seabird species with the potential to migrate through the Norfolk Boreas site were assessed.

- Bewick's swan (*Cygnus columbianus bewickii*);
- Dark-bellied brent goose (*Branta bernicla bernicla*);
- Wigeon (*Anas Penelope*);
- Gadwall (*Anas Strepera*);
- Teal (*Anas crecca*);
- Pintail (*Anas acuta*);
- Shoveler (*Anas clypeata*);
- Pochard (*Aythya farina*);
- Tufted duck (*Aythya fuligula*);
- Common scoter (*Melanitta nigra*);
- Goldeneye (*Bucephala clangula*);
- Marsh harrier (*Circus aeruginosus*);
- Oystercatcher (*Haematopus ostralegus*);
- Avocet (nonbreeding) (*Recurvirostra avosetta*);
- Ringed plover (*Charadrius hiaticula*);
- Golden plover (*Pluvialis apricaria*);
- Grey plover (*Pluvialis squatarola*);
- Lapwing (*Vanellus vanellus*);
- Knot (*Calidris canutus*);
- Sanderling (*Calidris alba*);
- Dunlin (*Calidris alpine*);
- Bar-tailed godwit (*Limosa lapponica*);
- Curlew (*Numenius arquata*);
- Redshank (*Tringa totanus* (including each sub-species));
- Turnstone (*Arenaria interpres*).

292. Relevant population sizes and migration routes were obtained from the Strategic Ornithological Support Services (SOSS) Migration Assessment Tool (hereafter referred to as SOSSMAT; Wright et al. 2012). The SOSSMAT Geographical Information System tool enables estimation of the proportion of migrating populations which could encounter offshore wind farms. The species-specific migration routes were derived by Wright et al. (2012) from a review of literature, and the tool enables identification of those routes which cross user-defined wind farm footprints.

293. Natural England requested that the non-seabird migrant collision assessment should consider potential impacts on the wider populations of each species as well as the populations at the Breydon Water SPA, Broadland SPA and North Norfolk Coast SPA. Migrant collision modelling was undertaken using the migrant option in the Band (2012) to estimate the proportion of the total collisions which could affect each population. It was assumed that the SPA populations would be affected in proportion to the size of the SPA population relative to the total population.

294. Ten species were estimated to be at risk of 1 or fewer collisions per year: avocet, Bewick’s swan, common scoter, dark-bellied brent goose, gadwall, goldeneye, marsh harrier, pintail, sanderling and shoveler.

295. Ten species were estimated to be at risk of between 1 and 10 collisions per year: bar-tailed godwit, curlew, grey plover, knot, pochard, redshank (summed across all races), ringed plover, teal, tufted duck and turnstone.

296. The remaining five species with predicted annual collisions of 10 or more were dunlin (23), golden plover (25), lapwing (18), oystercatcher (12) and wigeon (11).

297. There are no species for which the total annual collisions exceeded 0.01% of the migratory population. With respect to the potential increases in background mortality as a result of these collisions, the background mortality rate would only be increased by more than 1% (the threshold below which additional mortality is considered to have an undetectable effect) for any of these species if the natural mortality rate was extremely low at less than 1% (i.e. the annual survival rate would need to be at least 99%). This is much lower than the natural mortality rates for any of the species assessed, most have natural mortality of at least 10% per year.

298. Consequently, the collision risk predictions for all the migrant non-seabird species included in the assessment generate negligible magnitude impacts which are therefore of negligible or minor significance.

299. Due to the very low numbers of collisions apportioned to the relevant SPA populations, no likely significant effects are predicted for the Breydon Water SPA, Broadland SPA and North Norfolk Coast SPA due to migrant collisions at the Norfolk Boreas Wind Farm.

13.7.4.4 Impact 6: Barrier effects

300. The presence of the proposed project could potentially create a barrier to bird migration and foraging routes, and as a consequence, the proposed project has the potential to result in long-term changes to bird movements. It has been shown that some species (such as divers and scoters) avoid wind farms by making detours around wind turbine arrays which potentially increases their energy expenditure (Petersen et al., 2006; Petersen and Fox, 2007; Masden et al., 2010, 2012), which

under some circumstances could potentially decrease survival chances. Such effects may have a greater impact on birds that regularly commute around a wind farm (e.g. birds heading to / from foraging grounds and roosting / nesting sites) than on migrants that would only have to negotiate around a wind farm once per migratory period, or twice per annum, if flying the same return route (Speakman et al., 2009; Masden et al., 2012).

301. During the spring and autumn migration periods, the route taken by migrating individuals may change due to the barrier effect created by the wind turbines. Although migrating birds may have to increase their energy expenditure to circumvent the Norfolk Boreas site at a time when their energy budgets are typically restricted, this effect is likely to be small for one-off avoidances. Masden et al. (2010, 2012) and Speakman et al. (2009) calculated that the costs of one-off avoidances during migration were small, accounting for less than 2% of available fat reserves. Therefore, the impacts on birds that only migrate (including seabirds, waders and waterbirds on passage) through the site can be considered negligible and these species have been scoped out of detailed assessment.

302. Several species of seabirds could be susceptible to a barrier effect, outside of passage movements, if the presence of wind turbines prevented access to foraging grounds or made the journey to or from foraging grounds more energetically expensive, particularly during the breeding season. The Norfolk Boreas site is located beyond the foraging range of the majority of species during the breeding season, with the exception of fulmar, gannet and lesser black-backed gull. However, even for these species, the Norfolk Boreas site is towards the periphery of their mean maximum foraging ranges (Thaxter et al., 2012a) so it is highly unlikely that anything other than a negligible magnitude barrier effect would be created. In addition, all of these species are considered to have a low sensitivity to barrier effects (Maclean et al., 2009). Assessment of barrier effects of offshore wind farms in the Forth and Tay area for gannets breeding in the Forth Islands SPA concluded that even in this situation where wind farms were planned in close proximity to the Bass Rock gannet colony, the barrier effect for that population would be negligible (Searle et al., 2014; Searle et al., 2017). The impact significance of the barrier effect for all of these species is assessed as **negligible**.

13.7.5 Potential Impacts during Decommissioning

303. There are two potential impacts that may affect bird populations during the decommissioning phase of the proposed project that have been screened in. These are:

- Disturbance / displacement; and
- Indirect impacts through effects on habitats and prey species.

304. Any effects generated during the decommissioning phase of Norfolk Boreas are expected to be similar, or of reduced magnitude, to those generated during the construction phase, as certain activities such as piling would not be required. This is because it would generally involve a reverse of the construction phase through the removal of structures and materials installed, including some or all of the array cables, interconnector cables, project interconnector cables and offshore export cables, although It is anticipated that scour and cable protection would remain in-situ.

305. Potential impacts predicted during the decommissioning phase include those associated with disturbance and displacement and indirect effects on birds through effects on habitats and prey species. Disturbance and displacement is likely to occur due to the presence of working vessels and crews and the movement and noise associated with these. Indirect effects would occur as structures are removed.

306. As no offshore wind farms in UK waters have yet been decommissioned, it is anticipated that any future activities would be programmed in close consultation with the relevant statutory marine and nature conservation bodies, to allow any future guidance and best practice to be incorporated to minimise any potential impacts.

13.7.5.1 Impact 7: Direct disturbance and displacement

307. Disturbance and displacement is likely to occur due to the presence of working vessels and crews and the movement and noise associated with these. Such activities have already been assessed for relevant bird species in the construction section above and have been found to be of negligible to minor negative magnitude.

308. Any impacts generated during the decommissioning phase of Norfolk Boreas are expected to be similar, but likely of reduced magnitude compared to those generated during the construction phase; therefore the magnitude of effect is predicted to be negligible. This magnitude of impact on a range of species of low to high sensitivity to disturbance is of **negligible** to **minor adverse** significance.

13.7.5.2 Impact 8: Indirect impacts through effects on habitats and prey species

309. Indirect effects such as displacement of seabird prey species are likely to occur as structures are removed. Such activities have already been assessed for relevant bird species in the construction section above and have been found to be of negligible magnitude.

310. Any impacts generated during the decommissioning phase of the proposed project are expected to be similar, but likely of reduced magnitude compared to those generated during the construction phase; therefore the magnitude of effect is predicted to be negligible. This magnitude of impact on a range of species of low to high sensitivity to disturbance is of **negligible to minor adverse** significance.

13.8 Cumulative Impacts

13.8.1 Screening for cumulative impacts

311. The screened in potential effects arising from Norfolk Boreas alone that have been identified above are presented in Table 13.39, within which they are assessed for their potential to create a cumulative impact.

Table 13.39 Potential cumulative impacts.

Impact	Potential for cumulative impact	Data confidence	Rationale
Construction			
1. Disturbance and displacement	No	High	There is a possibility that construction would overlap temporally with construction of East Anglia THREE and Norfolk Vanguard. However, the impact assessments for all three wind farms have identified very small magnitudes of impact, and even if these occurred at the same time this would not constitute a significant effect. This also applies to the installation of the export cable, as it is very unlikely that this would coincide both spatially and temporally with installation for other wind farms.
2. Indirect impacts through effects on habitats and prey species	No	High	There is a possibility that construction would overlap temporally with construction of East Anglia THREE and Norfolk Vanguard. However, the impact assessments for all three wind farms have identified very small magnitudes of impact, and even if these occurred at the same time this would not constitute a significant effect.
Operation			
3. Disturbance and displacement	Yes	Medium-Low	There is a sufficient likelihood of a cumulative impact to justify a detailed, quantitative cumulative impact assessment. Note that data confidence is lower for older wind farms due to variations in the level of detail reported. There is greater confidence in assessments for more recent wind farms which have typically followed a standard approach to assessment and reporting.
4. Indirect impacts through effects on habitats and prey species	No	High	The likelihood that there would be a cumulative impact is low because the contribution from the proposed project is small.
5. Collision risk	Yes	Medium	There is a sufficient likelihood of a cumulative impact to justify a detailed, quantitative cumulative impact assessment.

Impact	Potential for cumulative impact	Data confidence	Rationale
6. Barrier effect	No	High	The likelihood that there would be a cumulative impact is low for the following reasons; the region has very low presence of breeding seabirds (only lesser black-backed gulls breed within foraging range, but there is no evidence for barrier effects in this species) so no risk of daily barrier to movement. Diversion around wind farms by migrating seabirds has negligible costs, and non-seabird migrants will primarily fly over the wind farm and therefore will not face a barrier to movement.
Decommissioning			
7. Disturbance and displacement	No	High	The likelihood that there would be a cumulative impact is low because the contribution from the proposed project is small and it is dependent on a temporal and spatial co-incidence of disturbance / displacement from other plans or proposed projects.
8. Indirect impacts through effects on habitats and prey species	No	High	The likelihood that there would be a cumulative impact is low because the contribution from the proposed project is small and it is dependent on a temporal and spatial co-incidence of disturbance / displacement from other plans or projects.

312. The classes of projects that could potentially be considered for the cumulative assessment of offshore ornithological receptors include:

- Offshore wind farms;
- Marine aggregate extraction;
- Oil and gas exploration and extraction;
- Subsea cables and pipelines; and
- Commercial shipping.

313. The identification of plans and projects to include in the cumulative assessment of offshore ornithological receptors has been based on:

- Approved plans;
- Constructed projects;
- Approved but as yet unconstructed projects; and
- Projects for which an application has been made, are currently under consideration and may be consented before Norfolk Boreas.

314. 'Foreseeable' projects, that is those for which an application has not been made but they have been the subject of consultation by the developer, or they are listed in plans that have clear delivery mechanisms, have been included for consideration,

but the absence of firm or any relevant data could preclude a quantitative cumulative assessment being carried out.

13.8.2 Screened in sources of effect for the Cumulative Impact Assessment (CIA)

315. Potential plans and projects have been considered for how they might act cumulatively with the proposed project and a screening process carried out. Any new information which has become available after the cut-off point of 20th March has not been included in the CIA (see Table 13.13).

13.8.2.1 Activities which may affect benthic habitats and fish

316. This includes marine aggregate extraction, oil and gas exploration and extraction, and installation and maintenance of subsea cables and pipelines. Effects on benthic habitats could affect seabird prey species, including fish, thereby constituting an indirect source of impact.

317. The potential for cumulative indirect impacts acting through adverse effects on benthic habitats and consequently on bird prey species was considered as part of Chapter 10 Benthic Ecology, section 10.7. This identified that the potential cumulative impacts to the benthos caused by interactions of Norfolk Boreas and other activities are:

- Physical disturbance and habitat loss;
- Increased suspended sediment concentrations;
- Re-mobilisation of contaminated sediments;
- Underwater noise and vibration; and
- Colonisation of foundations and cable protection.

318. The cumulative assessment identified that these impacts would mostly be temporary, small scale and localised. Given the distances to other activities in the region (e.g. other offshore wind farms and aggregate extraction), it concluded that there is no pathway for interaction between impacts cumulatively. Whilst it is recognised that across the former East Anglia Zone and wider southern North Sea there would be additive impacts, the combined magnitude of these would be negligible relative to the scale of the habitats affected. Accordingly, the cumulative impacts on birds through these effects could be no more than negligible and these are screened out from further assessment.

13.8.2.2 Shipping and navigation

319. Wide ranging species such as gannet and fulmar have low sensitivity to human activity disturbance and are relatively flexible in their habitat choice (Garthe & Hüppop, 2004). These species are therefore unlikely to be subject to cumulative effects of disturbance from Norfolk Boreas and existing ship traffic.

320. Gulls are undisturbed by the close proximity of boats, and therefore no potential adverse cumulative effects are expected for kittiwake, common gull, lesser black-backed gull, herring gull or great black-backed gull.

321. Divers, particularly red-throated divers, are known to be sensitive to disturbance from shipping. Consequently, they usually occur in areas with light sea traffic (Mitschke et al., 2001). It has been noted from aerial survey data that while red-throated divers avoid shipping lanes (tending to prefer areas 1km or more away), they do not display complete absence, and vessel activity in these shipping lanes is considerably higher than any proposed wind farm service boat activity (DTI, 2006). The high shipping activity in the Thames Strategic Area due to bulk carriers, tankers and passenger ferries, does not seem to affect the overwintering population of red-throated divers inside and outside of the Outer Thames SPA. Auks also tend to move away from vessels, although their responses are less marked than for divers. While it can be expected that red-throated divers, guillemots and razorbills will be displaced from shipping lanes, it is reasonable to assume that such effects are accounted for in the baseline data which underpin this assessment.

322. In conclusion, it is likely that the seabirds present in the vicinity of Norfolk Boreas have already adapted to shipping operations in the area. The increase in shipping activities associated with construction of Norfolk Boreas would be short-term and temporary. Therefore, no significant cumulative disturbance and displacement effects are predicted for any seabird species and shipping and navigation is screened out of further cumulative assessment.

13.8.2.3 Wind farms

323. UK offshore wind farms that are operational, under construction, consented but not constructed, subject to current applications, subject to consultation or notified to the Planning Inspectorate have been screened in. Consideration is given below (section 13.9) to non-UK offshore wind farms. This list of wind farms with their status is provided in Table 13.40. Although some of the wind farms included in this list have been operational for over 10 years, in most cases the seabird population data pre-date the installations (e.g. Seabird 2000, Mitchell et al., 2004) and therefore the baseline cannot be assumed to include the effects of these wind farms.

324. The wind farms have been assigned to Tiers following the approach proposed by Natural England and JNCC (Natural England, 2013a) as follows:

1. Built and operational projects;
2. Projects under construction;
3. Consented;
4. Application submitted and not yet determined;
5. In planning (scoped), application not yet submitted; and,
6. Identified in Planning Inspectorate list of projects.

Table 13.40 Summary of projects considered for the CIA in relation to offshore ornithology.

Project	Tier	Status	Development period	[1]Distance from Norfolk Boreas site (km)	Project data status	Included in CIA	Rationale
Greater Gabbard	1	Built and operational	Fully commissioned Aug 2013	111	Complete for the ornithology receptors being assessed	Yes	Included as an operational project that does not yet form part of the baseline.
Gunfleet Sands	1	Built and operational	Fully commissioned Jun 2010	160	Complete for the ornithology receptors being assessed	Yes	Included as an operational project that does not yet form part of the baseline.
Kentish Flats	1	Built and operational	Fully commissioned Dec 2005	191	Complete but limited quantitative species assessment	Yes	Operational for a sufficiently long time that its effects will have been incorporated in surveys but not yet in population responses
Lincs	1	Built and operational	Fully commissioned Sep 2013	150	Complete but limited quantitative species assessment	Yes	Included as an operational project that does not yet form part of the baseline.
London Array (Phase 1)	1	Built and operational	Fully commissioned Apr 2013	128	Complete but limited quantitative species assessment	Yes	Included as an operational project that does not yet form part of the baseline.
Lynn and Inner Dowsing	1	Built and operational	Fully commissioned Mar 2009	151	Complete but limited quantitative species assessment	Yes	Included as an operational project that does not yet form part of the baseline.
Scroby Sands	1	Built and operational	Fully commissioned Dec 2004	68	Complete but limited quantitative species assessment	Yes	Operational for a sufficiently long time that its effects will have been incorporated in surveys but not yet in population responses
Sheringham Shoal	1	Built and operational	Fully commissioned Sep 2012	104	Complete but limited quantitative species assessment	Yes	Included as an operational project that does not yet form part of the baseline.

[1] Shortest distance between the considered project and the Norfolk Boreas site – unless specified otherwise.

Project	Tier	Status	Development period	[1]Distance from Norfolk Boreas site (km)	Project data status	Included in CIA	Rationale
Beatrice (demonstrator)	1	Built and operational	Fully commissioned Sep 2007	677	Complete but limited quantitative species assessment	Yes	Included as an operational project that does not yet form part of the baseline.
Thanet	1	Built and operational	Fully commissioned Sep 2010	175	Complete for the ornithology receptors being assessed	Yes	Included as an operational project that does not yet form part of the baseline.
Teesside	1	Built and operational	Fully commissioned Aug 2013	307	Complete but limited quantitative species assessment	Yes	Included as an operational project that does not yet form part of the baseline.
Westermost Rough	1	Built and operational	Fully commissioned May 2015	188	Complete for the ornithology receptors being assessed	Yes	Included as a consented project that does not yet form part of the baseline.
Humber Gateway	1	Built and operational	Fully commissioned May 2015	174	Complete but limited quantitative species assessment	Yes	Included as a consented project that does not yet form part of the baseline.
Hywind	1	Built and operational	Fully commissioned	551	Complete but limited quantitative species assessment	Yes	Included as a consented project that does not yet form part of the baseline.
EOWDC (Aberdeen OWF)	3	Operational	Consent August 2014, offshore construction commenced April 2018	545	Complete for the ornithology receptors being assessed	Yes	Included as a consented project that does not yet form part of the baseline.
Kincardine	2	Under construction	Consent	533	Complete but limited quantitative species assessment	Yes	Included as a consented project that does not yet form part of the baseline.
Galloper	2	Under construction	Consent March 2018	107	Complete for the ornithology receptors being assessed	Yes	Included as a consented project that does not yet form part of the baseline.

Project	Tier	Status	Development period	[1]Distance from Norfolk Boreas site (km)	Project data status	Included in CIA	Rationale
Dudgeon	2	Under construction	Consent November 2017	90	Complete but limited quantitative species assessment	Yes	Included as a consented project that does not yet form part of the baseline.
Race Bank	2	Under construction	Consent February 2018	124	Complete but limited quantitative species assessment	Yes	Included as a consented project that does not yet form part of the baseline.
Beatrice	2	Under construction	Consent Mar 2014. Construction commenced Jan 2017	677	Complete for the ornithology receptors being assessed	Yes	Included as a consented project that does not yet form part of the baseline.
Hornsea Project 1	2	Under construction	Consent Dec 2014, no construction start date	89	Complete for the ornithology receptors being assessed	Yes	Included as a consented project that does not yet form part of the baseline.
Rampion	2	Under construction	Consent Aug 2014. Construction commenced Apr 2017 (expected to be commissioned 2018)	325	Complete for the ornithology receptors being assessed	Yes	Included as a consented project that does not yet form part of the baseline.
East Anglia ONE	2	Under construction	Consent Jun 2014, offshore construction due to commence August 2018	61	Complete for the ornithology receptors being assessed	Yes	Included as a consented project that does not yet form part of the baseline.
Blyth (NaREC Demonstration)	3	Consented	Consent Nov 2013, no construction start date	346	Complete but limited quantitative species assessment	Yes	Included as a consented project that does not yet form part of the baseline.
Dogger Bank Creyke Beck A & B	3	Consented	Consent Feb 2015, no construction start date	173	Complete for the ornithology receptors being assessed	Yes	Included as a consented project that does not yet form part of the baseline.

Project	Tier	Status	Development period	[1]Distance from Norfolk Boreas site (km)	Project data status	Included in CIA	Rationale
Inch Cape	3	Consented	Consent Sep 2014, no construction start date	483	Complete for the ornithology receptors being assessed	Yes	Included as a consented project that does not yet form part of the baseline.
Neart ne Goithe	3	Consented	Consent Oct 2014, no construction start date	468	Complete for the ornithology receptors being assessed	Yes	Included as a consented project that does not yet form part of the baseline.
Firth of Forth Alpha and Bravo	3	Consented	Consent Oct 2014, no construction start date	470	Complete for the ornithology receptors being assessed	Yes	Included as a consented project that does not yet form part of the baseline.
Moray Firth (EDA)	3	Consented	Consent Mar 2014, no construction start date	651	Complete for the ornithology receptors being assessed	Yes	Included as a consented project that does not yet form part of the baseline.
Dogger Bank Teesside A & B	3	Consented	Consent Aug 2015, no construction start date	191	Complete for the ornithology receptors being assessed	Yes	Included as a consented project that does not yet form part of the baseline.
Hornsea Project 2	3	Consented	Consent Aug 2016, no construction start date	109	Complete for the ornithology receptors being assessed	Yes	Included as a consented project that does not yet form part of the baseline.
Triton Knoll	3	Consented	Consent Jul 2013, no construction start date	123	Complete for the ornithology receptors being assessed	Yes	Included as a consented project that does not yet form part of the baseline.
East Anglia THREE	3	Consented	Consent Aug 2017. No construction start date	13	Complete for the ornithology receptors being assessed	Yes	Included as a foreseeable project.
Hornsea Project 3	5	In planning	ES submitted May 2018	61	Complete for the ornithology receptors being assessed	Yes	Outputs from the ES have been included.

Project	Tier	Status	Development period	[1]Distance from Norfolk Boreas site (km)	Project data status	Included in CIA	Rationale
Thanet Extension	5	In planning	ES submitted June 2018	174	Complete for the ornithology receptors being assessed	Yes	Outputs from the ES have been included.
Norfolk Vanguard	5	In planning	ES submitted June 2018	1	Complete for the ornithology receptors being assessed	Yes	Outputs from the ES have been included.
Moray West	6	In planning	ES submitted July 2018	655	Complete for the ornithology receptors being assessed	Yes	Outputs from the ES have been included.
East Anglia ONE North	6	Pre-planning application	PEIR submitted November 2018	50	Complete for the ornithology receptors being assessed	Yes	Outputs from the PIER have been included in the ES.
East Anglia TWO	6	Pre-planning application	PEIR submitted November 2018	72	Complete for the ornithology receptors being assessed	Yes	Outputs from the PIER have been included in the ES.
Hornsea Project 4	6	Pre-planning application	Scoping report submitted October 2018	140	Not yet available	No	In the absence of data, the inclusion of this project is only on a qualitative basis.

325. The level of data available and the ease with which impacts can be combined across the wind farms in Table 13.40 is quite variable, reflecting the availability of relevant data for older projects and the approach to assessment taken. Wherever possible the cumulative assessment is quantitative (i.e. where data in an appropriate format have been obtained). Where this has not been possible (e.g. for older projects) a qualitative assessment has been undertaken.

13.8.2.4 Bird species included in the cumulative assessment of operational disturbance and displacement

326. The species assessed for project alone operational displacement impacts (and the relevant seasons) were red-throated diver (autumn, winter, spring), gannet (breeding, autumn, spring), guillemot (breeding, nonbreeding) and razorbill (breeding, autumn, winter, spring).

13.8.2.5 Bird species included in the cumulative assessment of collision risk

327. The species assessed for project alone collision impacts (and the relevant seasons) were gannet, kittiwake, lesser black-backed gull, herring gull and great black-backed gull. As these were the only species where an impact may be possible at the project level, only these species were assessed for cumulative risks.

13.8.2.6 Cumulative assessment of disturbance and displacement during operation

13.8.2.6.1 *Red-throated diver*

328. There is potential for wind farms in the southern North contribute to cumulative red-throated diver displacement. Table 13.41 summarises the results of a review of wider southern North Sea project environmental statements. This review of 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).

Table 13.41 Summary of red-throated diver assessments for wind farms in southern North Sea (excluding former East Anglia Zone wind farms) with potential to contribute to a cumulative operational disturbance and displacement impact.

Wind farm	Red-throated diver assessment method	Estimated no. of red-throated diver mortalities due to displacement	Conclusion for Norfolk Boreas cumulative assessment
Scroby Sands	None	No number presented	Part of baseline
Kentish Flats	Qualitative	No number presented	Part of baseline
Lynn & Inner Dowsing	Qualitative	No number presented	Part of baseline
Gunfleet Sands	Qualitative	very small'	Part of baseline
Thanet	Quantitative	<1-2	Part of baseline

Wind farm	Red-throated diver assessment method	Estimated no. of red-throated diver mortalities due to displacement	Conclusion for Norfolk Boreas cumulative assessment
Sheringham Shoal	None	No number presented	Part of baseline
Greater Gabbard	Quantitative	4-40	Part of baseline
London Array	Qualitative	No number presented	Part of baseline
Lincs	Qualitative	No number presented	Part of baseline
Kentish Flats Extension	Qualitative	No number presented	Assumed very small
Galloper	Quantitative	1-14	Very small impact
Dudgeon	Not assessed	No number presented	Assumed very small
Race Bank	Not assessed	No number presented	Assumed very small
Triton Knoll	Not assessed	No number presented	Assumed very small
Thanet Extension	Quantitative	1-9	Very small impact
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 Project 1	Not assessed	No number presented	NA
Hornsea Project 2	Not assessed	No number presented	NA
Hornsea Project 3	Not assessed	No number presented	NA

329. Cumulative red-throated diver displacement mortality has also been calculated for wind farms in the former East Anglia Zone which have a higher potential to contribute to a cumulative effect. This has been conducted using the same precautionary magnitudes of displacement (90-100%) and mortality (1 to 10%) applied to all birds within the 4km wind farm buffer, as defined in section 13.7.4.1.1.

330. The red-throated diver displacement mortality across wind farms in the East Anglia Zone is presented in Table 13.42. Displacement from these wind farms is considered to be the most likely source of cumulative impact in combination with Norfolk Boreas.

Table 13.42 Red-throated diver cumulative displacement mortality calculated on the basis of a precautionary assumption of 90-100% displacement within 4km of the wind farm and 1% to 10% mortality of displaced individuals.

Wind farm	Autumn	Midwinter	Spring	Annual
Wider region projects (see Table 13.41)	N/A	N/A	N/A	6 - 56
Thanet Extension	0	4 - 43	2 - 26	6 - 69
East Anglia ONE	0.4 - 5	1 - 10	1.4 - 15	2.8 -30
East Anglia THREE	0.4 - 5	0.2 – 2	2 - 20	2.6 - 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

Wind farm	Autumn	Midwinter	Spring	Annual
Norfolk Boreas	0 - 1	1 - 15	5 - 62	6 - 78
Total	1.2 - 19	9.4 - 109	13.4 - 155	30 - 339

331. The cumulative red-throated diver displacement mortality total combines several sources of precaution:

- Natural England guidance is to assume that all birds within 4km 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 falling to zero within 1km (see Kentish Flats monitoring reports);
- It is very likely that the final wind farm will comprise more widely spaced turbines than that assumed for the worst case scenario and this will reduce the magnitude of impact (and this applies equally to the other former East Anglia Zone wind farms);
- It includes an unknown degree of double counting across seasons since some individuals will be present within more than one season;
- The Norfolk Boreas, Norfolk Vanguard and East Anglia THREE 4km buffers overlap with one or both of the other two wind farms, therefore including the buffer for all three sites double counts birds in the overlapping area (by approx. 15%); and
- Half of the total was predicted to occur during the spring migration period when the potential consequences of displacement are expected to be much lower due to the brief duration that birds spend in the area at this time.

332. 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 (Table 13.13) the number of individuals expected to die is 3,027 (13,277 x 0.228). The addition of between 30 and 339 to this would increase the mortality rate between 0.99% and 11.2%. 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 (Table 13.13) the number of individuals expected to die is 6,156 (27,000 x 0.228). The addition of between 30 and 339 to this would increase the mortality rate between 0.49% and 5.5%. Thus, at the more realistic end of the predicted range of cumulative impacts (90% displaced and 1% mortality), mortality would increase below the threshold of detectability, while at the highly precautionary worst case upper end of the cumulative range (100% displaced and 10% mortality), the predicted impacts would exceed the 1% threshold below which impacts are considered to be undetectable.

333. If it is assumed that the abundance at the other wind farms within 2 km is 25% lower than within 4 km (as is the case at Norfolk Boreas), then application of the evidence-based finding that displacement extends no further than 1.5 km (Annex 1), these totals would decline to between 23 and 254.

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

- 11.2% (4 km buffer, 100% displacement, 10% mortality);
- 8.4% (2 km buffer, 100% displacement, 10% mortality);
- 1.0% (4 km buffer, 90% displacement, 1% mortality);
- 0.7% (2 km buffer, 90% displacement, 1% mortality);

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

- 5.5% (4 km buffer, 100% displacement, 10% mortality);
- 4.1% (2 km buffer, 100% displacement, 10% mortality);
- 0.49% (4 km buffer, 90% displacement, 1% mortality);
- 0.37% (2 km buffer, 90% displacement, 1% mortality).

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

- The evidence review (MacArthur Green 2019a) found that 90% displacement and 1% mortality are more appropriate (and still precautionary) 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 Boreas, Norfolk Vanguard East and East Anglia THREE 4 km buffers overlap with each other so including the buffer for all three sites leads to double counting birds in the overlapping areas;
- The inclusion of total displacement within the 4km buffers from both Norfolk Vanguard East and Norfolk West is precautionary since this would only result from turbines being installed across the entirety of both sites and this will not in fact occur; and
- Half 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.

337. 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 (particularly the more recent projects such as those in the former East Anglia zone). 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.

338. To inform consideration of the combinations of displacement and mortality which result in increases in background mortality of <1% and between 1% and 2%, displacement matrices with highlighted cells have been produced for the BDMPS population (Table 13.43) and the biogeographic population (

339. Table 13.44). These tables indicate that, with respect to the BDMPS population, cumulative displacement of 90% combined with 2% mortality would generate an increase in background mortality of less than 2%. Furthermore, any level of displacement combined with 1% mortality generates an increase in mortality of less than 1%. With respect to the biogeographic population, displacement of 90% and 2% mortality would reduce the estimated mortality increase to below 1%.

Table 13.43 Red-throated diver cumulative displacement matrix. Levels of mortality which would increase the baseline mortality of the BDMPS population by percentage thresholds indicated by shading: green <1%; orange >1% and <2%; clear >2%.

Mortality (%)	Displacement (%)									
	10	20	30	40	50	60	70	80	90	100
1	3	6	9	11	14	17	20	23	26	29
2	6	11	17	23	29	34	40	46	52	57
3	9	17	26	34	43	52	60	69	77	86
4	11	23	34	46	57	69	80	92	103	115
5	14	29	43	57	72	86	100	115	129	143
6	17	34	52	69	86	103	120	137	155	172
7	20	40	60	80	100	120	140	160	180	200
8	23	46	69	92	115	137	160	183	206	229
9	26	52	77	103	129	155	180	206	232	258
10	29	57	86	115	143	172	200	229	258	339

Table 13.44 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%; clear >2%.

Mortality (%)	Displacement (%)									
	10	20	30	40	50	60	70	80	90	100
1	3	6	9	11	14	17	20	23	26	29
2	6	11	17	23	29	34	40	46	52	57
3	9	17	26	34	43	52	60	69	77	86
4	11	23	34	46	57	69	80	92	103	115
5	14	29	43	57	72	86	100	115	129	143
6	17	34	52	69	86	103	120	137	155	172
7	20	40	60	80	100	120	140	160	180	200
8	23	46	69	92	115	137	160	183	206	229
9	26	52	77	103	129	155	180	206	232	258
10	29	57	86	115	143	172	200	229	258	339

340. 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**.

341. However, on the basis of the evidence review (MacArthur Green 2019a) 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 the basis of this more representative assessment, 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.**

13.8.2.6.2 *Gannet*

342. There is evidence that gannets avoid flying through wind farms (Krijgsveld et al., 2011; Skov et al., 2018). If this prevents them accessing important foraging areas this could have an impact on affected individuals. However, for the reasons set out below, the potential for the proposed project to contribute to a cumulative effect such as this is considered to be very low. The period when gannet displacement is of potential concern is during autumn migration. At this time very large numbers of gannets migrate from breeding colonies in Northern Europe to wintering areas farther south (off southern Europe and off the coast of West Africa). Thus, displacement due to wind farms in the North Sea is trivial when compared with the range over which individuals of this species travel (Garthe et al., 2012, see also Masden et al., 2010, 2012). Furthermore, gannets are considered to be highly

flexible in their foraging requirements (capable of catching a wide range of prey species), and exclusion from wind farms in the southern North Sea during the migration period, when combined with the low overall numbers of birds present, is very unlikely to represent a loss of any importance. Consequently, the potential that even the worst case precautionary prediction of 14 displacement mortalities at Norfolk Boreas could contribute to a significant cumulative displacement effect on gannets during migration is considered to be very small and the impact significance of cumulative displacement is **negligible**.

13.8.2.6.3 *Auks*

343. Post-construction monitoring of nonbreeding season auks has found evidence of wind farm avoidance behaviour, with indications that wind turbine density may affect the magnitude of avoidance (Leopold et al., 2011; Krijgsveld et al., 2011; Dierschke et al., 2016). The only auk species present in sufficient numbers in those studies to permit robust estimation of wind farm avoidance was guillemot, for which an avoidance rate of around 68% was calculated, although it should be noted that this was based on observations of flying birds and this value may not be appropriate for swimming birds. Furthermore, those studies were conducted at sites with relatively closely spaced wind turbines (e.g. 550m), while the minimum spacing at Norfolk Boreas will be 720m (within rows) and 720m (between rows), which equates to a minimum turbine density reduction of almost 25%.

344. The pressures on nonbreeding birds in terms of energy requirements are lower outside the breeding season when they only need to obtain sufficient food to maintain their own survival. In addition, species such as auks remain at sea for extended periods and thus flight costs are minimised. Recoveries of ringed auks have revealed wide winter distributions, with birds spread throughout the North Sea (Furness, 2015). This pattern has received further support from recent studies using geolocator tags, which have revealed that birds from Scottish colonies spread out through much of the North Sea (S. Wanless, pers. comm.). These studies have also found quite marked levels of variation between years, which suggests that birds are relatively flexible in terms of where they spend the winter and are not dependent on particular foraging locations. Hence, the consequence of winter displacement from wind farms in terms of increased mortality is likely to be minimal. Given that, even when fish stocks have collapsed, seabird adult survival rates have shown declines of no more than 6 - 7% (e.g. kittiwake, Frederiksen et al., 2004) an increase in mortality due to displacement from wind farm sites seems likely to be at the low end of the proposed 1 - 10% range, and a value of 1% when combined with the precautionary 70% displacement rate is considered appropriate for wintering auks. Thresholds for additional mortality for each species are provided in Table 13.45.

Table 13.45. Auk populations in UK North Sea waters (see Natural England 2015) used in the displacement assessment, the baseline mortality averaged across age classes (Table 13.13) 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

13.8.2.6.4 *Razorbill*

345. Norfolk Boreas is 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 Boreas site (combined across the breeding season and all the nonbreeding seasons) was a maximum of 2,303 individuals. The totals at risk on other North Sea wind farms are presented in Table 13.46.

Table 13.46. Cumulative razorbill numbers on wind farms in the North Sea.

Project	Breeding season	Post-breeding season	Non-breeding season	Pre-breeding season
Aberdeen	161	64	7	26
Beatrice	873	833	555	833
Blyth Demonstration	121	91	61	91
Dogger Bank Creyke Beck A	1250	1576	1728	4149
Dogger Bank Creyke Beck B	1538	2097	2143	5119
Dogger Bank Teesside A	834	310	959	1919
Dogger Bank Teesside B	1153	592	1426	2953
Dudgeon	256	346	745	346
East Anglia ONE	16	26	155	336
East Anglia THREE	1807	1122	1499	1524
East Anglia TWO	288	55.0	148.0	263.0
East Anglia ONE North	403	85.0	54.0	207.0
Galloper	44	43	106	394
Greater Gabbard	0	0	387	84
Hornsea Project One	1109	4812	1518	1803
Hornsea Project Two	2511	4221	720	1668
Hornsea Project Three	630	2020	3694	1236
Humber Gateway	27	20	13	20
Hywind	30.0	719.0	10.0	
Inch Cape	1436	2870	651	
Kincardine	22.0			
Lincs and LID6	45	34	22	34
London Array I & II	14	20	14	20
Moray East	2423	1103	30	168
Moray West	2808	3544	184	3585
Neart na Gaoithe	331	5492	508	
Norfolk Vanguard East	599	491	279	752
Norfolk Vanguard West	280	375	348	172

Project	Breeding season	Post-breeding season	Non-breeding season	Pre-breeding season
Race Bank	28	42	28	42
Seagreen A	3208	N/A	N/A	N/A
Seagreen B	886	N/A	N/A	N/A
Sheringham Shoal	106	1343	211	30
Teesside	16	61	2	20
Thanet	3	0	14	21
Thanet Extension			34	50
Triton Knoll	40	254	855	117
Westermost Rough	91	121	152	91
Norfolk Boreas	630	263.0	1065.0	345.0
Seasonal total	26,017	35,045	20,325	28,418
Annual total				**109,805**

346. 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 assessment, with an emphasis on displacement between 30% and 70% and mortality between 1% and 10%. Within this range, upper levels of displacement would result in increases in baseline mortality above 1%. However, evidence in support of the use of a precautionary displacement rate of 50% with a 1% mortality rate for razorbill has been presented in MacArthur Green (2019b). For the current cumulative assessment presented in Table 13.47, 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 13.47).

347. 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 13.47. 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%.

		Mortality (%)																			
		1	2	3	4	5	6	7	8	9	10	20	25	30	40	50	60	70	80	90	
Displacement (%)	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	2	22	44	66	88	110	132	154	176	198	220	439	549	659	878	1098	1318	1537	1757	1976	2196
	4	44	88	132	176	220	264	307	351	395	439	878	1098	1318	1757	2196	2635	3075	3514	3953	4392
	6	66	132	198	264	329	395	461	527	593	659	1318	1647	1976	2635	3294	3953	4612	5271	5929	6588
	8	88	176	264	351	439	527	615	703	791	878	1757	2196	2635	3514	4392	5271	6149	7028	7906	8784
	10	110	220	329	439	549	659	769	878	988	1098	2196	2745	3294	4392	5490	6588	7686	8784	9882	10981
	12	132	264	395	527	659	791	922	1054	1186	1318	2635	3294	3953	5271	6588	7906	9224	10541	11859	13177
	14	154	307	461	615	769	922	1076	1230	1384	1537	3075	3843	4612	6149	7686	9224	10761	12298	13835	15373
	16	176	351	527	703	878	1054	1230	1406	1581	1757	3514	4392	5271	7028	8784	10541	12298	14055	15812	17569
	18	198	395	593	791	988	1186	1384	1581	1779	1976	3953	4941	5929	7906	9882	11859	13835	15812	17788	19765
	20	220	439	659	878	1098	1318	1537	1757	1976	2196	4392	5490	6588	8784	10981	13177	15373	17569	19765	21961
	25	275	549	824	1098	1373	1647	1922	2196	2471	2745	5490	6863	8235	10981	13726	16471	19216	21961	24706	27451
	30	329	659	988	1318	1647	1976	2306	2635	2965	3294	6588	8235	9882	13177	16471	19765	23059	26353	29647	32942
	40	439	878	1318	1757	2196	2635	3075	3514	3953	4392	8784	10981	13177	17569	21961	26353	30745	35138	39530	43922
	50	549	1098	1647	2196	2745	3294	3843	4392	4941	5490	10981	13726	16471	21961	27451	32942	38432	43922	49412	54903
	60	659	1318	1976	2635	3294	3953	4612	5271	5929	6588	13177	16471	19765	26353	32942	39530	46118	52706	59295	65883
	70	769	1537	2306	3075	3843	4612	5380	6149	6918	7686	15373	19216	23059	30745	38432	46118	53804	61491	69177	76864
	80	878	1757	2635	3514	4392	5271	6149	7028	7906	8784	17569	21961	26353	35138	43922	52706	61491	70275	79060	87844
	90	988	1976	2965	3953	4941	5929	6918	7906	8894	9882	19765	24706	29647	39530	49412	59295	69177	79060	88942	98825
	100	1098	2196	3294	4392	5490	6588	7686	8784	9882	10981	21961	27451	32942	43922	54903	65883	76864	87844	98825	109805

13.8.2.6.5 *Guillemot*

348. Norfolk Boreas is 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 Boreas site with a mean maximum of 13,777 individuals (Table 13.48).

Table 13.48. 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
East Anglia TWO	305.0	593.0
East Anglia ONE North	345.0	548.0
Galloper	9836.0	8097.0
Greater Gabbard	7735.0	13164.0
Hornsea Project One	13374.0	17772.0
Hornsea Project Two	2126.0	2020.0
Hornsea Project Three	4183.0	1847.0
Humber Gateway	99.0	138.0
Hywind	249.0	2136.0
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 East	9820.0	547.0
Moray West	24426.0	38174.0
Neart na Gaoithe	1755.0	3761.0
Norfolk Vanguard East	2931	2197
Norfolk Vanguard West	1389	2579
Race Bank	361.0	708.0
Seagreen A	13606.0	4688.0
Seagreen B	11118.0	4112.0
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
Norfolk Boreas	7767.0	13777.0
Seasonal total	159807.0	156109.0
Annual total		**315916.0**

Table 13.49 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%.

	Mortality (%)																			
Displacement (%)	1	2	3	4	5	6	7	8	9	10	20	25	30	40	50	60	70	80	90	
0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
2	63	126	190	253	316	379	442	505	569	632	1264	1580	1895	2527	3159	3791	4423	5055	5686	6318
4	126	253	379	505	632	758	885	1011	1137	1264	2527	3159	3791	5055	6318	7582	8846	10109	11373	12637
6	190	379	569	758	948	1137	1327	1516	1706	1895	3791	4739	5686	7582	9477	11373	13268	15164	17059	18955
8	253	505	758	1011	1264	1516	1769	2022	2275	2527	5055	6318	7582	10109	12637	15164	17691	20219	22746	25273
10	316	632	948	1264	1580	1895	2211	2527	2843	3159	6318	7898	9477	12637	15796	18955	22114	25273	28432	31592
12	379	758	1137	1516	1895	2275	2654	3033	3412	3791	7582	9477	11373	15164	18955	22746	26537	30328	34119	37910
14	442	885	1327	1769	2211	2654	3096	3538	3981	4423	8846	11057	13268	17691	22114	26537	30960	35383	39805	44228
16	505	1011	1516	2022	2527	3033	3538	4044	4549	5055	10109	12637	15164	20219	25273	30328	35383	40437	45492	50547
18	569	1137	1706	2275	2843	3412	3981	4549	5118	5686	11373	14216	17059	22746	28432	34119	39805	45492	51178	56865
20	632	1264	1895	2527	3159	3791	4423	5055	5686	6318	12637	15796	18955	25273	31592	37910	44228	50547	56865	63183
25	790	1580	2369	3159	3949	4739	5529	6318	7108	7898	15796	19745	23694	31592	39490	47387	55285	63183	71081	78979
30	948	1895	2843	3791	4739	5686	6634	7582	8530	9477	18955	23694	28432	37910	47387	56865	66342	75820	85297	94775
40	1264	2527	3791	5055	6318	7582	8846	10109	11373	12637	25273	31592	37910	50547	63183	75820	88456	101093	113730	126366
50	1580	3159	4739	6318	7898	9477	11057	12637	14216	15796	31592	39490	47387	63183	78979	94775	110571	126366	142162	157958
60	1895	3791	5686	7582	9477	11373	13268	15164	17059	18955	37910	47387	56865	75820	94775	113730	132685	151640	170595	189550
70	2211	4423	6634	8846	11057	13268	15480	17691	19903	22114	44228	55285	66342	88456	110571	132685	154799	176913	199027	221141
80	2527	5055	7582	10109	12637	15164	17691	20219	22746	25273	50547	63183	75820	101093	126366	151640	176913	202186	227460	252733
90	2843	5686	8530	11373	14216	17059	19903	22746	25589	28432	56865	71081	85297	113730	142162	170595	199027	227460	255892	284324
100	3159	6318	9477	12637	15796	18955	22114	25273	28432	31592	63183	78979	94775	126366	157958	189550	221141	252733	284324	315916

349. 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 assessment, with an emphasis on displacement between 30% and 70% and mortality between 1% and 10%. Within this range, upper levels of displacement would result in increases in baseline mortality above 1%. However, evidence in support of the use of a precautionary displacement rate of 50% with a 1% mortality rate for guillemot has been presented here. For the current cumulative assessment presented in Table 13.48, 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 13.49).

350. Consequently, the potential cumulative annual displacement mortality for guillemot 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**.

13.8.2.7 Cumulative assessment of collision risk

13.8.2.7.1 *Gannet*

351. The cumulative gannet collision risk prediction is set out in the form of a 'tiered approach' in Table 13.50. This collates collision predictions from other wind farms which may contribute to the cumulative total.

352. The cumulative totals of collision mortality in each season, and summed across seasons, are presented in Table 13.50. Assessments at other wind farms have been conducted using a range of avoidance rates and alternative collision model Options. In order to simplify interpretation of the data across sites and also to bring these assessments up to date with the current Natural England Advice, the values in Table 13.50 are those estimated using the Band model Option 1 (or 2, if that was the one presented) standardised at an avoidance rate of 98.9%. The worst case scenario for Norfolk Boreas has been included along with the revised cumulative total.

Table 13.50 Cumulative Collision Risk Assessment for gannet.

Tier	Wind farm	Breeding		Autumn		Spring		Annual	
		CRM	Total	CRM	Total	CRM	Total	CRM	Total
1	Beatrice Demonstrator	0.6	0.6	0.9	0.9	0.7	0.7	2.2	2.2
1	Greater Gabbard	14.0	14.5	8.8	9.7	4.8	5.5	27.5	29.7
1	Gunfleet Sands	0.0	14.5	0.0	9.7	0.0	5.5	0.0	29.7
1	Kentish Flats	1.4	15.9	0.8	10.5	1.1	6.6	3.3	33.0
1	Lincs	2.1	18.0	1.3	11.8	1.7	8.3	5.0	38.0
1	London Array	2.3	20.3	1.4	13.2	1.8	10.1	5.5	43.5
1	Lynn and Inner Dowsing	0.2	20.5	0.1	13.3	0.2	10.3	0.5	44.1
1	Scroby Sands	0.0	20.5	0.0	13.3	0.0	10.3	0.0	44.1
1	Sheringham Shoal	14.1	34.6	3.5	16.8	0.0	10.3	17.6	61.7

Tier	Wind farm	Breeding		Autumn		Spring		Annual	
		CRM	Total	CRM	Total	CRM	Total	CRM	Total
1	Teesside	4.9	39.5	1.7	18.5	0.0	10.3	6.7	68.3
1	Thanet	1.1	40.6	0.0	18.5	0.0	10.3	1.1	69.4
1	Humber Gateway	1.9	42.5	1.1	19.7	1.5	11.8	4.5	73.9
1	Westermost Rough	0.2	42.7	0.1	19.8	0.2	12.0	0.5	74.4
1	Hywind	5.6	48.3	0.8	20.6	0.8	12.8	7.2	81.6
2	Kincardine	3.0	51.3	0.0	20.6	0.0	12.8	3.0	84.6
2	Beatrice	37.4	88.7	48.8	69.4	9.5	22.3	95.7	180.3
2	Dudgeon	22.3	111.0	38.9	108.3	19.1	41.3	80.3	260.6
2	Galloper	18.1	129.1	30.9	139.2	12.6	54.0	61.6	322.2
2	Race Bank	33.7	162.8	11.7	150.9	4.1	58.0	49.5	371.7
2	Rampion	36.2	198.9	63.5	214.4	2.1	60.1	101.8	473.5
2	Hornsea Project One	11.5	210.4	32.0	246.4	22.5	82.6	66.0	539.5
3	Blyth Demonstration Project	3.5	214.0	2.1	248.5	2.8	85.4	8.4	548.0
3	Dogger Bank Creyke Beck Projects A and B	5.6	219.5	6.6	255.1	4.3	89.8	16.5	564.5
3	East Anglia ONE	3.4	222.9	131.0	386.1	6.3	96.1	140.7	705.2
3	European Offshore Wind Deployment Centre	4.2	227.1	5.1	391.3	0.1	96.1	9.3	714.5
3	Firth of Forth Alpha and Bravo	800.8	1027.9	49.3	440.6	65.8	161.9	915.9	1630.4
3	Inch Cape	336.9	1364.8	29.2	469.8	5.2	167.1	371.3	2001.7
3	Moray Firth (EDA)	80.6	1445.4	35.4	505.2	8.9	176.0	124.9	2126.6
3	Neart na Gaoithe	143.0	1588.4	47.0	552.2	23.0	199.0	213.0	2339.6
3	Dogger Bank Teesside Projects A and B	14.8	1603.1	10.1	562.3	10.8	209.9	35.7	2375.3
3	Triton Knoll	26.8	1629.9	64.1	626.4	30.1	239.9	121.0	2496.3
3	Hornsea Project Two	7.0	1636.9	14.0	640.4	6.0	245.9	27.0	2523.3
4	East Anglia THREE	6.1	1643.0	33.3	673.7	9.6	255.6	49.0	2572.2
5	Hornsea Project Three	18.0	1661.0	12.0	685.7	8.0	263.6	38.0	2610.2
5	Thanet Extension	0.0	1661.0	4.4	690.1	9.1	272.7	13.5	2623.7
5	Norfolk Vanguard	21.6	1682.6	71.6	761.7	49.3	322.0	142.5	2766.2
6	Moray West	8.8	1691.4	8.6	770.3	1.2	323.1	18.6	2784.8
6	East Anglia TWO (PEIR)	8.8	1700.2	5.5	775.8	1.3	324.4	15.6	2800.4
6	East Anglia ONE North (PEIR)	10.0	1710.2	2.0	777.8	1.0	325.4	13.0	2813.4
6	Norfolk Boreas	54.13	1764.33	48.50	826.3	14.99	339.8	117.63	2931.03
	Total		**1764.33**		**826.3**		**340.39**		**2931.03**

353. On the basis of the worst case Norfolk Boreas collision estimates the annual cumulative total for UK North Sea wind farms is 2,931. Note, however that many of the collision estimates for other wind farms were calculated on the basis of 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. A method for updating

collision estimates for changes in wind farm design such as this was presented in EATL (2016). Updating the collision estimates for the Beatrice wind farm using this approach reduces the predicted annual mortality from 96 to 64 (a 33% reduction in mortality). Applying the same method to the other wind farms in Table 13.50 results in a reduction in the cumulative annual mortality of approximately 400. Therefore, the values presented in Table 13.50, as well as being based on precautionary calculation methods, can be seen to further overestimate the total risk by around 13% due to the reduced collision risks for projects which undergo design revisions post-consent.

354. 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.

355. A recent review of avoidance behaviour at an operational wind farm (Bowgen and Cook 2018) has also recommended a higher avoidance rate for gannet of 99.5%. Use of this rate would more than halve the number of collisions (i.e. the cumulative total at this rate would be 1,325).

356. A review of nocturnal activity in gannets (Furness et al., 2018) has found that the value previously used for this parameter (25%) to estimate flight activity at night is a considerable overestimate and has identified evidence based rates of 8% during the breeding season and 4% during the nonbreeding season. These rates were used in the Norfolk Boreas collision modelling (and also the Norfolk Vanguard assessment), however they will also apply to the estimates for other wind farms calculated using the old rate of 25%.

357. It is straightforward to adjust existing 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.

358. 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%.

359. The WWT 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 13.50. 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).

360. 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.

361. 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.

362. In conclusion, the cumulative impact on the gannet population due to collisions both year round and within individual seasons is considered to be of low magnitude, and the relative contribution of Norfolk Boreas to this cumulative total is small. Furthermore, there are many additive sources of precaution in the collision assessment which mean that the total mortality is almost certainly considerably lower than that based on the precautionary approaches used here. Gannet are considered to be of low to medium sensitivity to collision mortality and the impact significance is therefore **minor adverse**.

13.8.2.7.2 *Kittiwake*

363. The cumulative kittiwake collision risk prediction is set out in the form of a ‘tiered approach’ in Table 13.51. This collates collision predictions from other wind farms which may contribute to the cumulative total.

364. The cumulative totals of collision mortality in each season, and summed across seasons, are presented in Table 13.51. Assessments at other wind farms have been conducted using a range of avoidance rates and alternative collision model Options. In order to simplify interpretation of the data across sites and also to bring these assessments up to date with the current Natural England Advice, the values in Table 13.51 are those estimated using the Band model Option 1 (or 2, if that was the one presented) standardised at an avoidance rate of 98.9%. The worst case scenario for Norfolk Boreas has been included along with the revised cumulative total.

Table 13.51 Cumulative Collision Risk Assessment for kittiwake.

Tier	Wind farm	Breeding		Autumn		Spring		Annual	
		CRM	Total	CRM	Total	CRM	Total	CRM	Total
1	Beatrice Demonstrator	0.0	0.0	2.1	2.1	1.7	1.7	3.8	3.8
1	Greater Gabbard	1.1	1.1	15.0	17.1	11.4	13.1	27.5	31.3
1	Gunfleet Sands	0.0	1.1	0.0	17.1	0.0	13.1	0.0	31.3
1	Kentish Flats	0.0	1.1	0.9	18.0	0.7	13.8	1.6	32.9
1	Lincs	0.7	1.8	1.2	19.2	0.7	14.5	2.6	35.5
1	London Array	1.4	3.2	2.3	21.5	1.8	16.3	5.5	41.0
1	Lynn and Inner Dowsing	0.0	3.2	0.0	21.5	0.0	16.3	0.0	41.0
1	Scroby Sands	0.0	3.2	0.0	21.5	0.0	16.3	0.0	41.0
1	Sheringham Shoal	0.0	3.2	0.0	21.5	0.0	16.3	0.0	41.0
1	Teesside	38.4	41.6	24.0	45.5	2.5	18.8	64.9	105.9
1	Thanet	0.3	41.9	0.5	46.0	0.4	19.2	1.2	107.1
1	Humber Gateway	1.9	43.8	3.2	49.2	1.9	21.1	7.0	114.0
1	Westermost Rough	0.1	43.9	0.2	49.4	0.1	21.2	0.5	114.5
1	Hywind	16.6	60.5	0.9	50.2	0.9	22.1	18.3	132.8
2	Kincardine	22.0	82.5	9.0	59.2	1.0	23.1	32.0	164.8
2	Beatrice	94.7	177.2	10.7	69.9	39.8	62.9	145.2	310.0
2	Dudgeon	0.0	177.2	0.0	69.9	0.0	62.9	0.0	310.0
2	Galloper	6.3	183.5	27.8	97.7	31.8	94.7	65.9	375.9
2	Race Bank	1.9	185.4	23.9	121.6	5.6	100.3	31.4	407.3
2	Rampion	54.4	239.8	37.4	159.0	29.7	130.0	121.5	528.8
2	Hornsea Project One	44.0	283.8	55.9	214.9	20.9	150.9	120.8	649.6
3	Blyth Demonstration Project	1.4	285.2	2.3	217.2	1.4	152.3	5.1	654.7
3	Dogger Bank Creyke Beck Projects A and B	288.0	573.2	135.0	352.2	295.0	447.3	718.0	1372.7
3	East Anglia ONE	1.8	575.0	160.4	512.6	46.8	494.1	209.0	1581.7
3	European Offshore Wind Deployment Centre	11.8	586.8	5.8	518.4	1.1	495.2	18.7	1600.4

Tier	Wind farm	Breeding		Autumn		Spring		Annual	
		CRM	Total	CRM	Total	CRM	Total	CRM	Total
3	Firth of Forth Alpha and Bravo	153.1	739.9	313.1	831.5	247.6	742.8	713.8	2314.2
3	Inch Cape	13.1	753.0	224.8	1056.3	63.5	806.3	301.4	2615.6
3	Moray Firth (EDA)	43.6	796.6	2.0	1058.3	19.3	825.6	64.9	2680.5
3	Neart na Gaoithe	32.9	829.5	56.1	1114.4	4.4	830.0	93.4	2773.9
3	Dogger Bank Teesside Projects A and B	136.9	966.4	90.7	1205.1	216.9	1046.9	444.5	3218.4
3	Triton Knoll	24.6	991.0	139.0	1344.1	45.4	1092.3	209.0	3427.4
3	Hornsea Project Two	16.0	1007.0	9.0	1353.1	3.0	1095.3	28.0	3455.4
4	East Anglia THREE	6.1	1013.1	69.0	1422.1	37.6	1132.9	112.7	3568.1
5	Hornsea Project Three	121.0	1134.1	76.0	1498.1	40.0	1172.9	237.0	3805.1
5	Thanet Extension	1.5	1135.6	3.4	1501.5	9.8	1182.7	14.7	3819.8
5	Norfolk Vanguard	31.3	1166.9	134.1	1635.6	150.5	1333.2	315.9	4135.7
6	Moray West	79.0	1245.9	24.0	1659.6	7.0	1340.2	110.0	4245.7
6	East Anglia TWO (PEIR)	13.6	1259.5	2.9	1662.5	9.3	1349.5	25.8	4271.5
6	East Anglia ONE North (PEIR)	6.0	1265.5	4.3	1666.8	17.4	1366.9	27.7	4299.2
6	Norfolk Boreas	29.92	1295.42	116.59	1783.39	56.29	1423.19	202.80	4460.2
	Total		**1295.42**		**1783.39**		**1423.19**		**4502**

365. On the basis of the worst case Norfolk Boreas collision estimates the annual cumulative total is 4,502. Note, however that many of the collision estimates for other wind farms were calculated on the basis of 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. A method for updating collision estimates for changes in wind farm design was presented in EATL (2016). Updating the collision estimates for the Beatrice wind farm using this approach reduces the predicted annual mortality from 145 to 97 (a 33% reduction in mortality). Applying the same method to the other wind farms in Table 13.51 can achieve a reduction in the estimated cumulative annual mortality of around 550. Therefore, the values presented in Table 13.51, as well as being based on precautionary calculation methods, can be seen to further overestimate the total risk by around 14% due to the reduced collision risks for projects which undergo design revisions post consent.

366. A recent review of avoidance behaviour at an operational wind farm (Bowgen and Cook 2018) has also recommended a higher avoidance rate for kittiwake of 99%. Use of this rate would reduce the total by 10% (i.e. the cumulative total at this rate would be 4,055).

367. A review of nocturnal activity in kittiwakes (Furness et al., 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. These rates were used in the Norfolk Boreas collision modelling (and also for the Norfolk Vanguard assessment), however they will also apply to the estimates for other wind farms calculated using the old rate of 50%.

368. 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.

369. 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. For annual mortality of 4,000, the density dependent model predicted the population after 25 years would be 3.6% to 4.4% smaller than that predicted in the absence of additional mortality, while the more precautionary density independent model predicted equivalent declines of 10.3% to 10.9%. 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 3% and 10% across a longer (25 year) period against a background of natural changes an order of magnitude larger would almost certainly be undetectable.

370. 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*' (Natural England, 2017).

371. 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 the results were the worst case). 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 (including 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.

372. 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. Furthermore, there are many additive sources of precaution in the collision assessment which mean that the total mortality is almost certainly considerably lower than that based on the precautionary approaches used here. 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 in to account (precautionary avoidance rate estimates, reduction in wind farm sizes and number of turbines, over-estimated nocturnal activity) the cumulative collision risk impact magnitude is almost certainly smaller still.

13.8.2.7.3 *Lesser black-backed gull*

373. The cumulative lesser black-backed gull collision risk prediction is set out in the form of a ‘tiered approach’ in Table 13.52. This collates collision predictions from other wind farms which may contribute to the cumulative total.

374. The collision values presented in Table 13.52 include totals for breeding, nonbreeding and annual periods. However, not all projects provide a seasonal breakdown of collisions, 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). Therefore, for those sites where a seasonal split was not presented, the annual numbers in Table 13.52 have been multiplied by 0.8 to estimate the nonbreeding component and 0.2 to estimate the breeding component.

375. Assessments for other wind farms have been conducted using a range of avoidance rates and alternative collision model options. In order to simplify interpretation of the data across sites and also to bring these assessments up to date with the current Natural England advice, the values in Table 13.52 are those estimated using the Band model Option 1 (or 2, if that was the one presented) at an avoidance rate of 99.5%. (Note that estimates for the Dogger Bank projects have only been presented using

Band model Option 3. Therefore, these values in Table 13.52 have been converted to the Natural England advised rate for this model of 98.9%).

Table 13.52 Cumulative Collision Risk Assessment for lesser black-backed gull.

Tier	Wind farm	Breeding		Non-breeding		Annual	
		CRM	Total	CRM	Total	CRM	Total
1	Beatrice Demonstrator	0.0	0.0	0.0	0.0	0.0	0.0
1	Greater Gabbard	12.4	12.4	49.6	49.6	62.0	62.0
1	Gunfleet Sands	1.0	13.4	0.0	49.6	1.0	63.0
1	Kentish Flats	0.3	13.7	1.3	50.9	1.6	64.6
1	Lincs	1.7	15.4	6.8	57.7	8.5	73.1
1	London Array	0.0	15.4	0.0	57.7	0.0	73.1
1	Lynn and Inner Dowsing	0.0	15.4	0.0	57.7	0.0	73.1
1	Scroby Sands	0.0	15.4	0.0	57.7	0.0	73.1
1	Sheringham Shoal	1.7	17.1	6.6	64.3	8.3	81.3
1	Teesside	0.0	17.1	0.0	64.3	0.0	81.3
1	Thanet	3.2	20.3	12.8	77.1	16.0	97.3
1	Humber Gateway	0.3	20.5	1.1	78.2	1.3	98.7
1	Westermost Rough	0.1	20.6	0.3	78.4	0.3	99.0
1	Hywind	0.0	20.6	0.0	78.4	0.0	99.0
2	Kincardine	0.0	20.6	0.0	78.4	0.0	99.0
2	Beatrice	0.0	20.6	0.0	78.4	0.0	99.0
2	Dudgeon	7.7	28.2	30.6	109.0	38.3	137.2
2	Galloper	27.8	56.0	111.0	220.0	138.8	276.0
2	Race Bank	43.2	99.2	10.8	230.8	54.0	330.0
2	Rampion	1.6	100.8	6.3	237.1	7.9	337.8
2	Hornsea Project One	4.4	105.1	17.4	254.5	21.8	359.6
3	Blyth Demonstration Project	0.0	105.1	0.0	254.5	0.0	359.6
3	Dogger Bank Creyke Beck Projects A and B	2.6	107.7	10.4	264.9	13.0	372.6
3	East Anglia ONE	5.9	113.6	33.8	298.7	39.7	412.3
3	European Offshore Wind Deployment Centre	0.0	113.6	0.0	298.7	0.0	412.3
3	Firth of Forth Alpha and Bravo	2.1	115.7	8.4	307.1	10.5	422.8
3	Inch Cape	0.0	115.7	0.0	307.1	0.0	422.8
3	Moray Firth (EDA)	0.0	115.7	0.0	307.1	0.0	422.8
3	Neart na Gaoithe	0.3	116.0	1.2	308.3	1.5	424.3
3	Dogger Bank Teesside Projects A and B	2.4	118.4	9.6	317.9	12.0	436.3
3	Triton Knoll	7.4	125.8	29.6	347.5	37.0	473.3
3	Hornsea Project Two	2.0	127.8	2.0	349.5	4.0	477.3
4	East Anglia THREE	1.8	129.6	8.2	357.7	10.0	487.3
5	Hornsea Project Three	15.0	144.6	3.0	360.7	18.0	505.3
5	Thanet Extension	1.5	146.1	0.8	361.5	2.3	507.6
5	Norfolk Vanguard	32.2	178.3	7.8	369.3	40.0	547.6
6	Moray West	0.0	178.3	0.0	369.3	0.0	547.6
6	East Anglia TWO (PEIR)	0.5	178.8	0.0	369.3	0.5	548.1
6	East Anglia ONE North (PEIR)	0.6	179.4	0.0	369.3	0.6	548.7
6	Norfolk Boreas	17.30	196.7	22.48	391.78	39.78	588.48
	Total		**196.7**		**391.78**		**588.48**

376. On the basis of the worst case Norfolk Boreas collision estimates the annual cumulative total is 588. Note, however that many of the collision estimates for other wind farms were calculated on the basis of 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 EATL (2016). 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 13.52 can achieve a reduction in the cumulative annual mortality of around 200. Therefore, the values presented in Table 13.52, as well as being based on precautionary calculation methods, can be seen to further overestimate the total risk by around 35% due to the reduced collision risks for projects which undergo design revisions post consent.

377. 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 an evidence-based avoidance rate of 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 13.52. 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 583, 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, 1.4 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.

378. 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 lesser black-backed gull is almost certainly an overestimate, possibly by as much as a factor of two (i.e. empirical data from logger deployments suggest that 25% is more appropriate). Reducing the nocturnal activity factor to 25% reduced collision estimates by around 15%. Natural England have recognised this aspect of precaution and advised recent projects to undertake collision modelling with nocturnal activity set to both 25% and 50%. This was included in the Norfolk Boreas collision modelling (and also the Norfolk Vanguard assessment) by setting the nocturnal factor in simulated model runs to be randomly selected as one of these

two values. However, this adjustment to nocturnal activity is also applicable to the collision estimates for other wind farms. Applying the same approach would reduce the cumulative collision estimate 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.

379. In conclusion, the current cumulative total is considerably lower than previously consented cumulative totals (between 1.4 and 2.8 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**.

13.8.2.7.4 *Herring gull*

380. The cumulative herring gull collision risk prediction is set out in the form of a 'tiered approach' in Table 13.53. This collates collision predictions from other wind farms which may contribute to the cumulative total.

381. Assessments at other wind farms have been conducted using a range of avoidance rates and alternative collision model Options. In order to simplify interpretation of the data across sites and also to bring these assessments up to date with the current Natural England Advice, the values in Table 13.53 are those estimated using the Band model Option 1 (or 2, if that was the one presented) standardised at an avoidance rate of 98.9%. The worst case scenario for Norfolk Boreas has been included along with the revised cumulative total.

Table 13.53 Cumulative Collision Risk Assessment for herring gull.

Tier	Wind farm	Breeding		Non-breeding		Annual	
		CRM	Total	CRM	Total	CRM	Total
1	Beatrice Demonstrator	0.0	0.0		0.0	0.0	0.0
1	Greater Gabbard	0.0	0.0		0.0	0.0	0.0
1	Gunfleet Sands	0.0	0.0		0.0	0.0	0.0
1	Kentish Flats	0.5	0.5	1.7	1.7	2.2	2.2
1	Lincs	0.0	0.5		1.7	0.0	2.2
1	London Array	0.0	0.5		1.7	0.0	2.2
1	Lynn and Inner Dowsing	0.0	0.5		1.7	0.0	2.2
1	Scroby Sands	0.0	0.5		1.7	0.0	2.2
1	Sheringham Shoal	0.0	0.5		1.7	0.0	2.2
1	Teesside	8.7	9.1	34.5	36.2	43.2	45.3
1	Thanet	4.9	14.0	19.6	55.8	24.5	69.8
1	Humber Gateway	0.4	14.4	1.1	56.9	1.5	71.3
1	Westermost Rough	0.1	14.5	0.0	56.9	0.1	71.4
1	Hywind	0.6	15.1	7.8	64.7	8.4	79.8

Tier	Wind farm	Breeding		Non-breeding		Annual	
		CRM	Total	CRM	Total	CRM	Total
2	Kincardine	1.0	16.1	0.0	64.7	1.0	80.8
2	Beatrice	49.4	65.5	197.4	262.1	246.8	327.6
2	Dudgeon	0.0	251.1		275.9	0.0	527.0
2	Galloper	27.2	265.9		314.9	27.2	580.8
2	Race Bank	0.0	92.7		262.1	0.0	354.8
2	Rampion	155.0	322.9		340.9	155.0	663.8
2	Hornsea Project One	2.9	265.9	11.6	314.9	14.5	580.8
3	Blyth Demonstration Project	0.5	251.1	2.2	275.9	2.7	527.0
3	Dogger Bank Creyke Beck Projects A and B	0.0	251.1		275.9	0.0	527.0
3	East Anglia ONE	0.0	251.1	28.0	303.9	28.0	555.0
3	European Offshore Wind Deployment Centre	4.8	255.9		303.9	4.8	559.8
3	Firth of Forth Alpha and Bravo	10.0	265.9	21.0	324.9	31.0	590.8
3	Inch Cape	0.0	265.9	13.5	338.4	13.5	604.3
3	Moray Firth (EDA)	52.0	317.9		338.4	52.0	656.3
3	Neart na Gaoithe	5.0	322.9	12.5	350.9	17.5	673.8
3	Dogger Bank Teesside Projects A and B	0.0	322.9		350.9	0.0	673.8
3	Triton Knoll	0.0	322.9		350.9	0.0	673.8
3	Hornsea Project Two	23.8	346.6		350.9	23.8	697.5
4	East Anglia THREE	0.0	346.6	23.0	373.9	23.0	720.5
5	Hornsea Project Three	1.0	347.6	7.0	380.9	8.0	728.5
5	Thanet Extension	10.0	357.6	4.0	384.9	14.0	742.5
5	Norfolk Vanguard	0.0	357.6	17.1	402.0	17.1	759.6
6	Moray West	12.0	369.6	1.0	403.0	13.0	772.6
6	East Anglia TWO (PEIR)	0.0	369.6	0.0	403.0	0.0	772.6
6	East Anglia ONE North (PEIR)	0.0	369.6	0.0	403.0	0.0	772.6
6	Norfolk Boreas	3.93	373.5	14.51	417.5	18.43	791.0
	Total		**373.5**		**417.5**		**791.0**

382. On the basis of the worst case Norfolk Boreas collision estimates the annual cumulative total is 791 of which the proposed Norfolk Boreas project contributes 18. Note, however that many of the collision estimates for other wind farms were calculated on the basis of 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. A method for updating collision estimates for changes in wind farm design was presented in EATL (2016). Updating the collision estimates for the Beatrice wind farm using this approach reduces the predicted annual mortality from 247 to 88 (a 65% reduction in mortality), and all the more notable as this project contributes a third of the total mortality for this species. It is clear that the application of a similar reductions to the other wind farms in Table 13.53 will achieve a very large reduction in the estimated cumulative annual mortality. Therefore, the values presented in

Table 13.53, as well as being based on precautionary calculation methods, can be seen to be considerable overestimates of the real risk presented by built projects compared with consented ones.

383. Previous herring gull collision assessments 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 13.48. The only projects consented after November 2014 were Hornsea Project 1 (14 annual collisions at 99.5%), Dogger Bank Creyke Beck A&B (0 annual collisions at 98.9% Option 3) and Dogger Bank Teesside A&B (0 annual collisions at 98.9% Option 3). Therefore, the previous cumulative collision total (at 98%) excluding these three projects would have been 2,652 (677 - (14) x 4; note this includes the Triton Knoll estimate as the windfarm was consented in July 2013). The current cumulative total of 787, including all consented and still to be consented projects, is therefore much lower than the previously accepted cumulative total.

384. 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 herring gull is almost certainly an overestimate, possibly by as much as a factor of two (i.e. empirical data from logger deployments suggest that 25% is more appropriate). Reducing the nocturnal activity factor to 25% reduced collision estimates by around 15%. Natural England have recognised this aspect of precaution and advised recent projects to undertake collision modelling with nocturnal activity set to both 25% and 50%. This was included in the Norfolk Boreas collision modelling (and also the Norfolk Vanguard assessment) by setting the nocturnal factor in simulated model runs to be randomly selected as one of these two values. However, this adjustment to nocturnal activity is also applicable to the collision estimates for other wind farms. Applying the same approach would reduce the cumulative collision estimate 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.

385. In conclusion, the current cumulative total is considerably lower than previously consented cumulative total (around 3.3 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 herring 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**.

13.8.2.7.5 *Great black-backed gull*

386. The cumulative great black-backed gull collision risk prediction is set out in the form of a 'tiered approach' in Table 13.54. This collates collision predictions from other wind farms which may contribute to the cumulative total.

387. The collision values presented in Table 13. include breeding, nonbreeding and annual collision totals. However, not all projects provide a seasonal breakdown of collisions, 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). This ratio is considered to also be appropriate for great black-backed gull, therefore for those sites where a seasonal split was not presented the annual numbers in Table 13.54 have been multiplied by 0.8 to estimate the nonbreeding component and 0.2 to estimate the breeding component.

388. Assessments for other wind farms have been conducted using a range of avoidance rates and alternative collision model Options. In order to simplify interpretation of the data across sites and also to bring these assessments up to date with the current Natural England advice, the values in Table 13.54 are those estimated using the Band model Option 1 (or 2, if that was the one presented) at an avoidance rate of 99.5%. (Note that estimates for the Dogger Bank projects have only been presented using Band model Option 3. Therefore, these values in Table 13.54 have been converted to the Natural England advised rate for this model of 98.9%).

Table 13.54 Cumulative Collision Risk Assessment for great black-backed gull.

Tier	Wind farm	Breeding		Non-breeding		Annual	
		CRM	Total	CRM	Total	CRM	Total
1	Beatrice Demonstrator	0.0	0.0	0.0	0.0	0.0	0.0
1	Greater Gabbard	15.0	15.0	60.0	60.0	75.0	75.0
1	Gunfleet Sands	0.0	15.0	0.0	60.0	0.0	75.0
1	Kentish Flats	0.1	15.1	0.2	60.2	0.3	75.3
1	Lincs	0.0	15.1	0.0	60.2	0.0	75.3
1	London Array	0.0	15.1	0.0	60.2	0.0	75.3
1	Lynn and Inner Dowsing	0.0	15.1	0.0	60.2	0.0	75.3
1	Scroby Sands	0.0	15.1	0.0	60.2	0.0	75.3
1	Sheringham Shoal	0.0	15.1	0.0	60.2	0.0	75.3
1	Teesside	8.7	23.8	34.8	95.1	43.6	118.8
1	Thanet	0.1	23.9	0.4	95.5	0.5	119.3
1	Humber Gateway	1.3	25.1	5.1	100.5	6.3	125.7
1	Westermost Rough	0.0	25.1	0.0	100.6	0.1	125.7
1	Hywind	0.3	25.4	4.5	105.1	4.8	130.5
2	Kincardine	0.0	25.4	0.0	105.1	0.0	130.5
2	Beatrice	30.2	55.6	120.8	225.9	151.0	281.5
2	Dudgeon	0.0	89.6	0.0	361.7	0.0	451.3
2	Galloper	4.5	103.5	18.0	449.5	22.5	553.0

Tier	Wind farm	Breeding		Non-breeding		Annual	
		CRM	Total	CRM	Total	CRM	Total
2	Race Bank	0.0	60.1	0.0	243.9	0.0	304.0
2	Rampion	5.2	113.9	20.8	515.3	26.0	629.3
2	Hornsea Project One	17.2	103.5	68.6	449.5	85.8	553.0
3	Blyth Demonstration Project	1.3	83.8	5.1	338.3	6.3	422.1
3	Dogger Bank Creyke Beck Projects A and B	5.8	89.6	23.3	361.7	29.1	451.3
3	East Anglia ONE	0.0	89.6	46.0	407.7	46.0	497.3
3	European Offshore Wind Deployment Centre	0.6	90.2	2.4	410.1	3.0	500.3
3	Firth of Forth Alpha and Bravo	13.4	103.5	53.4	463.5	66.8	567.0
3	Inch Cape	0.0	103.5	36.8	500.2	36.8	603.8
3	Moray Firth (EDA)	9.5	113.0	25.5	525.7	35.0	638.8
3	Neart na Gaoithe	0.9	113.9	3.6	529.3	4.5	643.3
3	Dogger Bank Teesside Projects A and B	6.4	120.3	25.5	554.8	31.9	675.2
3	Triton Knoll	24.4	144.7	97.6	652.4	122.0	797.2
3	Hornsea Project Two	3.0	147.7	20.0	672.4	23.0	820.2
4	East Anglia THREE	4.6	152.4	34.4	706.8	39.0	859.2
5	Hornsea Project Three	16.0	168.4	50.0	756.8	66.0	925.2
5	Thanet Extension	1.3	169.7	20.8	777.6	22.1	947.3
5	Norfolk Vanguard	0.0	169.7	65.1	842.7	65.1	1012.4
6	Moray West	4.0	173.7	5.0	847.7	9.0	1021.4
6	East Anglia TWO (PEIR)	2.2	175.9	0.5	848.2	2.7	1024.1
6	East Anglia ONE North (PEIR)	0.0	175.9	0.5	848.7	0.5	1024.6
6	Norfolk Boreas	7.8	183.7	85.4	934.1	93.1	1117.8
	Total		**183.7**		**934.1**		**1117.8**

389. On the basis of the worst case Norfolk Boreas collision estimates the annual cumulative total is 1,118. Note, however that many of the collision estimates for other wind farms were calculated on the basis of 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 EATL (2016). 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 13.54 can achieve a reduction in the estimated cumulative annual mortality of around 260. Therefore, the values presented in Table 13.54, as well as being based on precautionary calculations, can be seen to further overestimate the total risk by around 30% due to the reduced collision risks for projects which undergo design revisions post consent.

390. 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 an evidence-based 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 13.54. Accounting for projects up to Triton Knoll consented after November 2014 (Hornsea Project 1, 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 1,118, 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 around 1.2 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.

391. 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%. Natural England have recognised this aspect of precaution and advised recent projects to undertake collision modelling with nocturnal activity set to both 25% and 50%. This was included in the Norfolk Boreas collision modelling (and also the Norfolk Vanguard assessment) by setting the nocturnal factor in simulated model runs to be randomly selected as one of these two values. However, this adjustment to nocturnal activity is also applicable to the collision estimates for other wind farms. Applying the same approach would reduce the cumulative collision estimate 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.

392. 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 Natural England no longer consider PBR to be an appropriate tool for assessing wind farm impacts).

393. The current cumulative mortality of 1,118 (Table 13.54) is much lower than either of the cumulative totals reported for Rampion (1,803 and 3,025). The increase in the estimate of 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).

394. 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.

395. Using the density dependent model, application of an additional annual mortality of 1,000 to the great black-backed gull BDMPS resulted in impacted populations after 25 years which were 6.8% to 8.9% smaller than in the absence of impact. The equivalent density independent predictions generated population reductions of 22.6% to 23%. The component of these total declines attributable to Norfolk Boreas is less than 1% (density dependent predictions) and around 2% (density independent predictions).

396. 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 since the great black-backed gull is considered to be of low to medium sensitivity, the impact significance is **minor adverse**.

13.9 Transboundary Impacts

397. A summary of consultations conducted with other EU Member States (MS) surrounding the North Sea basin is provided in Table 13.3. The only MS which provided a response to the PEIR was Rijkswaterstaat (RWS) in the Netherlands, who noted that consideration should be given to proposed wind farm developments in the Netherlands with respect to displacement impacts. The response also noted that this would require an international cumulative approach, which has not been

developed to date. Owing to the different approaches to impact assessment adopted by each MS it is not currently clear how this could be undertaken quantitatively.

398. Protected sites in countries beyond the UK that may have connectivity with Norfolk Boreas are listed in Table 13.9.

399. To inform this assessment, consideration has been given to the consultation response received for Norfolk Boreas which raised a potential concern over transboundary impacts on ornithology receptors. This was provided by Rijkswaterstaat (RWS) in the Netherlands and noted that non-UK wind farms in the southern North Sea had not been included in the cumulative assessment of displacement.

400. With regards to the potential for transboundary cumulative impacts, there is clearly potential for collisions and displacement at wind farms outside UK territorial waters. However, the operational offshore wind farms in Belgium, the Netherlands and Germany are comparatively small (in combination these projects are of a similar size to no more than one to two of the more recent UK wind farms, such as East Anglia ONE). Since the spatial scale and hence seabird population sizes for a transboundary assessment would be much larger (e.g. assessments would necessarily be with reference to the biogeographic populations which are much larger: for example, for guillemot the biogeographic population is approximately three times the BDMPS, for gannet the difference is between two and four times, and for kittiwake it is more than five times), it is apparent that the comparative scale of wind farm development relative to the seabird populations would be much smaller (perhaps 1.5 times as much in the way of wind farm development). Therefore, the inclusion of non-UK wind farms would be very unlikely to alter the conclusions of the existing cumulative assessment, and any if there were any change it is likely that estimated impacts at population levels would be reduced if calculated at larger spatial scales.

13.10 Inter-relationships

401. The construction, operation and decommissioning phases of the proposed Norfolk Boreas wind farm would cause a range of effects on offshore ornithological interests. The magnitude of these effects has been assessed individually above in section 13.7 using expert judgement, drawing from a wide science base that includes project-specific surveys and previously acquired knowledge of the bird ecology of the North Sea.

402. These effects have the potential to form an inter-relationship and directly impact the terrestrial and seabird receptors and have the potential to manifest as sources for impacts upon receptors other than those considered within the context of offshore ornithology.

403. As none of the offshore impacts to birds were assessed individually to have any greater than a minor adverse impact it is considered unlikely that they would inter-relate to form an overall significant impact on Offshore Ornithology.

404. In terms of how impacts to offshore ornithological interests may form inter-relationships with other receptor groups, assessments of significance are provided in the chapters listed in the second column of Table 13.55. In addition, the table shows where other chapters have been used to inform the offshore ornithology inter-relationships assessment.

Table 13.55 Chapter topic inter-relationships.

Topic and description	Related Chapter	Where addressed in this Chapter
Indirect impacts through effects on habitats and prey during construction	10 – Benthic Ecology 11 – Fish and Shellfish Ecology	Section 13.7.3.2
Indirect impacts through effects on habitats and prey during operation	10 – Benthic Ecology 11 – Fish and Shellfish Ecology	Section 13.7.4.2
Indirect impacts through effects on habitats and prey during decommissioning	10 – Benthic Ecology 11 – Fish and Shellfish Ecology	Section 13.7.5.2

13.11 Interactions

405. The impacts identified and assessed in this chapter have the potential to interact with each other, which could give rise to synergistic impacts as a result of that interaction. The worst case impacts assessed within the chapter take these interactions into account and for the impact assessments are considered conservative and robust. For clarity the areas of interaction between impacts are presented in Table 13.56, along with an indication as to whether the interaction may give rise to synergistic impacts.

Table 13.56 Interaction between impacts.

Potential interaction between impacts Construction		
	1 Disturbance and displacement from increased vessel activity	2 Indirect effects as a result of displacement of prey species due to increased noise and disturbance to seabed
1 Disturbance and displacement from increased vessel activity	-	Yes, but small (possible longer term effects on birds, but spatial magnitude very small)
2 Indirect effects as a result of displacement of prey species due to increased	Yes, but small (possible longer term effects on birds, but spatial magnitude very small)	-

Potential interaction between impacts				
noise and disturbance to seabed				
Operation				
	1 Disturbance and displacement from offshore infrastructure	2 Indirect impacts through effects on habitats and prey species	3 Collision risk	4 Barrier effects
1 Disturbance and displacement from offshore infrastructure	-	No (direct displacement of birds overrides prey effects)	No (mutually exclusive)	No (similar response)
2 Indirect impacts through effects on habitats and prey species	No (direct displacement of birds overrides prey effects)	-	No	No
3 Collision risk	No (mutually exclusive)	No	-	No (mutually exclusive)
4 Barrier effects	No (similar response)	No	No (mutually exclusive)	-
Decommissioning				
It is anticipated that the decommissioning impacts will be no worse than those of construction.				

13.12 Summary

406. This chapter describes the offshore components of the proposed project; the consultation that has been held with stakeholders (to date); the scope and methodology of the assessment; the avoidance and mitigation measures that have been embedded through project design; the baseline data on birds and important sites and habitats for birds acquired through desk study and survey (Technical Appendix 13.1) and assesses the potential impacts on birds.

407. Detailed consultation and iteration of the overall approach to the impact assessment on ornithology receptors is ongoing through the Evidence Plan process for Norfolk Boreas. The assessment has also been informed by the closely related discussions for the adjacent Norfolk Vanguard project. An Ornithology Expert Technical Group has been convened which involves Natural England and the Royal Society for the Protection of Birds (RSPB) for the offshore ornithology discussions. A Schedule of Agreement and Non-agreement will be produced following the final Ornithology Expert Technical Group meeting and will be submitted as part of the final DCO submission.

408. A standard survey area, covering the Norfolk Boreas site and a 4km buffer was surveyed; the final survey was conducted in August 2018. A complete 24 months of data collection was available to inform this ES. The results of these surveys have been used to estimate the abundance and assemblage of birds using or passing across the area.

409. Birds were screened in for assessment taking into account their abundance on the wind farm site and their potential sensitivity to wind farm projects.

410. The impacts that could potentially arise during the construction, operation and decommissioning of Norfolk Boreas were presented in the project Method Statement which was reviewed by Natural England and the RSPB. Following comments received, and also discussions on Norfolk Vanguard, it was agreed that the potential impacts that required detailed assessment were:

411. In the Construction Phase

- Impact 1: Disturbance / displacement; and
- Impact 2: Indirect impacts through effects on habitats and prey species.

412. In the Operational Phase

- Impact 3: Disturbance / displacement;
- Impact 4: Indirect impacts through effects on habitats and prey species;
- Impact 5: Collision risk; and
- Impact 6: Barrier effect.

413. In the Decommissioning Phase

- Impact 7: Disturbance / displacement; and
- Impact 8: Indirect impacts through effects on habitats and prey species.

414. On the basis of the survey data collected, the following conclusions on impact significance were reached.

415. During the construction phase of the proposed project no impacts have been assessed to be greater than of minor adverse significance for any bird species. Similarly, no species is subject to an impact of greater than minor adverse significance from the potential effects of the proposed project during operational lifetime.

416. Displacement effects on red-throated divers, gannets, razorbills and guillemots would not create impacts of more than minor adverse significance during any biological season.

417. The risk to birds from collisions with wind turbines from Norfolk Boreas alone is assessed as no greater than minor adverse significance for all species when considered for all biological seasons against the most appropriate population scale.

418. Potential plans and projects have been considered for how they might act cumulatively with the proposed project and a screening process carried out.

419. The cumulative assessment identified that most impacts would be temporary, small scale and localised. Given the distances to other activities in the region (e.g. other offshore wind farms and aggregate extraction) and the highly localised nature of the impacts above it concluded that there is no pathway for interaction between most impacts cumulatively, which were screened out.

420. In the offshore environment only other wind farms that were operational, under construction, consented but not constructed, subject to current applications or subject to consultation were screened in. This list of wind farms with their status is provided in Table 13.40.

421. The cumulative collision risk impact and displacement impact assessment follows the tiered approach in its presentation of mortality predictions for the identified projects. The risk to birds from cumulative collisions with wind turbines across all wind farms considered is assessed as no greater than minor adverse significance for all species.

422. The identified potential impacts are summarised in Table 13.57.

Table 13.57 Potential Impacts Identified for offshore ornithology.

Potential Impact	Receptor	Value/ Sensitivity	Magnitude	Significance	Mitigation	Residual Impact
Construction						
Disturbance and displacement from increased vessel traffic	Common scoter	High	Negligible / no change	**Negligible to minor adverse**	NA	**Negligible to minor adverse**
	Red-throated diver	High	Negligible	**Minor adverse**	NA	**Minor adverse**
	Razorbill	Medium	Negligible	**Minor adverse**	NA	**Minor adverse**
	Guillemot	Medium	Negligible	**Minor adverse**	NA	**Minor adverse**
	Commic tern	Medium	Negligible	**Minor adverse**	NA	**Minor adverse**
Indirect effects due to prey species displacement	All species	Low to high	Negligible	**Negligible to minor adverse**	NA	**Negligible to minor adverse**
Operation						
Disturbance and displacement	Red-throated diver	High	Negligible	**Minor adverse**	NA	**Minor adverse**
	Gannet	Low	Negligible	**Negligible to minor adverse**	NA	**Negligible**
	Razorbill	Medium	Negligible	**Minor adverse**	NA	**Minor adverse**
	Guillemot	Medium	Negligible	**Minor adverse**	NA	**Minor adverse**
Indirect effects due to impacts on habitats and prey species displacement	All species	Low to high	Negligible	**Negligible to minor adverse**	NA	**Negligible to minor adverse**
Collision Risk - seabirds	Gannet	Low to medium	Negligible	**Negligible to minor adverse**	NA	**Negligible to minor adverse**

Potential Impact	Receptor	Value/ Sensitivity	Magnitude	Significance	Mitigation	Residual Impact
	Kittiwake	Low to medium	Negligible	**Negligible to minor adverse**	NA	**Negligible to minor adverse**
	Lesser black-backed gull	Low to medium	Negligible	**Negligible to minor adverse**	NA	**Negligible to minor adverse**
	Herring gull	Low to medium	Negligible	**Negligible to minor adverse**	NA	**Negligible to minor adverse**
	Great black-backed gull	Low to medium	Negligible	**Negligible to minor adverse**	NA	**Negligible to minor adverse**
Collision risk – migrant seabirds	Arctic skua	Low to medium	Negligible	**Negligible to minor**	NA	**Negligible to minor**
	Great skua	Low to medium	Negligible	**Negligible to minor**	NA	**Negligible to minor**
	Arctic tern	Low to medium	Negligible	**Negligible to minor**	NA	**Negligible to minor**
	Common tern	Low to medium	Negligible	**Negligible to minor**	NA	**Negligible to minor**
	Little gull	Low to medium	Negligible	**Negligible to minor**	NA	**Negligible to minor**
Collision risk – nonseabird migrants	All species	Low to high	Negligible	**Negligible**	NA	**Negligible**
Barrier effects	All species	Low to high	Negligible	**Negligible**	NA	**Negligible**
Decommissioning						
Direct disturbance and displacement	All species	Low to high	Negligible	**Negligible to minor**	NA	**Negligible to minor**
Indirect impacts through effects on habitats and prey	All species	Low to high	Negligible	**Negligible to minor**	NA	**Negligible to minor**
Cumulative						

Potential Impact	Receptor	Value/ Sensitivity	Magnitude	Significance	Mitigation	Residual Impact
Operational disturbance and displacement	Red-throated diver	High	Negligible	**Minor adverse**	NA	**Minor adverse**
	Gannet	Low	Negligible	**Negligible**	NA	**Negligible**
	Razorbill	Medium	Negligible	**Minor adverse**	NA	**Minor adverse**
	Guillemot	Medium	Negligible	**Minor adverse**	NA	**Minor adverse**
Collision Risk - seabirds	Gannet	Low to medium	Low	**Minor adverse**	NA	**Minor adverse**
	Kittiwake	Low to medium	Low	**Minor adverse**	NA	**Minor adverse**
	Herring gull	Low to medium	Low	**Minor adverse**	NA	**Minor adverse**
	Lesser black-backed gull	Low to medium	Low	**Minor adverse**	NA	**Minor adverse**
	Great black-backed gull	Low to medium	Low	**Minor adverse**	NA	**Minor adverse**

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