NORTH FALLS

Offshore Wind Farm

ENVIRONMENTAL STATEMENT

Chapter 13 Offshore Ornithology

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Glossary of Acronyms

AON	Apparently Occupied Nest
AOS	Apparently Occupied Site
BDMPS	Biologically Defined Minimum Population Scales
BEIS	Department for Business, Energy and Industrial Strategy
BM	Baseline annual mortality
BMR	Baseline Mortality Rate
BoCC	Birds of Conservation Concern
BTO	British Trust for Ornithology
CEA	Cumulative Effects Assessment
CIEEM	Chartered Institute of Ecology and Environmental Management
CLs	Confidence Limits
CRM	Collision Risk Modelling
DCO	Development Consent Order
DECC	Department of Energy and Climate Change
DESNZ	Department of Energy Security and Net Zero
EIA	Environmental Impact Assessment
EMF	Electromagnetic Fields
EMR	Effect Mortality Rate
EPP	Evidence Plan Process
ES	Environmental Statement
ESAS	European Seabirds at Sea
ETG	Expert Topic Group
EU	European Union
GGOW	Greater Gabbard Offshore Wind Farm
GPS	Global Positioning System
GWF	Galloper Wind Farm
HAT	Highest Astronomical Tide
HPAI	Highly Pathogenic Avian Influenza
HRA	Habitats Regulations Assessment

INLA	Integrated Nested Laplace Approximations
IPCC	Intergovernmental Panel on Climate Change
JNCC	Joint Nature Conservation Committee
LCL	Lower Confidence Limit
MaRD	Maximum Rotor Diameter
MHWS	Mean High Water Springs
MiRD	Minimum Rotor Diameter
MLWS	Mean Low Water Springs
MMFR	Mean Maximum Foraging Range
MMO	Marine Management Organisation
NAF	Nocturnal Activity Factor
NGO	Non-Governmental Organisation
NPS	National Policy Statements
NSIP	Nationally Significant Infrastructure Project
NVIS	Night Vision Imaging Systems
OCP	Offshore Converter Platform
OCSS	Offshore Coordination Support Scheme
ORJIP	Offshore Renewables Joint Industry Programme
OSP	Offshore Substation Platform
OWEZ	Egmond aan Zee Offshore Wind Farm
OWF	Offshore Wind Farm
P	Reference Population Size
PAWP	Prinses Amalia Windpark
PCH	Potential Collision Height
PEIR	Preliminary Environmental Information Report
pSPA	Potential Special Protection Area
PVA	Population Viability Analysis
R	Size of the reference population
RIAA	Report to Inform Appropriate Assessment
RSPB	Royal Society for the Protection of Birds
sCRM	Stochastic Collision Risk Modelling
SD	Standard Deviation
SeaMAST	The Seabird Mapping and Sensitivity Tool
SEANSE	Strategic Environmental Assessment North Seas Energy
SMP	Seabird Monitoring Programme
SNCB	Statutory Nature Conservation Body
SPA	Special Protection Area
SSSI	Site of Special Scientific Interest
UCI	Upper Confidence Limit
WTG	Wind Turbine Generator

Glossary of Terminology

Array area	The offshore wind farm area, within which the wind turbine generators, array cables, platform interconnector cable, offshore substation platform(s) and / or offshore converter platform will be located
Array cables	Cables which link the wind turbine generators with each other, the offshore substation platform(s) and / or the offshore converter platform.
As-built	A term used for offshore wind farm developments that are operational and where the turbine array 'as built' is different to the worst case scenario in the Environmental Impact Assessment for the development (for example where a wind farm is built out with fewer turbines than the consented design envelope).
Bathymetry	Topography of the seabed
Evidence Plan Process	A voluntary consultation process with specialist stakeholders to agree the approach to the EIA and information to support the HRA through ETG meetings.
Intertidal	Area on a shore that lies between Mean High Water Springs (MHWS) and Mean Low Water Springs (MLWS)
Landfall	The location where the offshore cables come ashore at Kirby Brook.
MaRD	Maximum Rotor Diameter (MaRD) Scenario (larger turbines) – 34 WTGs, 337m rotor diameter
Migration free breeding season	The breeding season for migratory seabird species is defined as a wider breeding season and a narrower window known as the migration free breeding season. In a given species, the timing of breeding will vary depending on the location of the breeding area; with the start of breeding usually later in more northerly locations. Thus, while birds at some colonies are beginning to nest, others may still be migrating to breeding sites. A core or migration free breeding season is defined as the period when all or the majority of breeding adults of a given species are present at breeding colonies.
MiRD	Minimum Rotor Diameter (MiRD) Scenario (smaller turbines) - 57 WTGs, 236m rotor diameter
Offshore cable corridor	The corridor of seabed from the array area to the landfall within which the offshore export cables will be located.
Offshore converter platform	Should an offshore connection to a third party HVDC cable be selected, an offshore converter platform would be required. This is a fixed structure located within the array area, containing HVAC and HVDC electrical equipment to aggregate the power from the wind turbine generators, increase the voltage to a more suitable level for export and convert the HVAC power generated by the wind turbine generators into HVDC power for export to shore via a third party HVDC interconnector cable.
Offshore export cables	The cables which bring electricity from the offshore substation platform(s) to the landfall, as well as auxiliary cables.
Offshore project area	The overall area of the array area and the offshore cable corridor.
Offshore substation platform(s)	Fixed structure(s) located within the array area, containing HVAC electrical equipment to aggregate the power from the wind turbine generators and increase the voltage to a more suitable level for export to shore via offshore export cables.
Platform interconnector cable	Cable connecting the offshore substation platforms (OSP); or the OSP and offshore converter platform (OCP)
Sandwave	Bedforms with wavelengths of 10 to 100m, with amplitudes of 1 to 10m
Scour protection	Protective materials to avoid sediment being eroded away from the base of the wind turbine generator foundations and offshore substation platform (OSP) or / and offshore converter platform (OCP) foundations as a result of the flow of water.
The Applicant	North Falls Offshore Wind Farm Limited (NFOW)

The Project Or ‘North Falls’	North Falls Offshore Wind Farm, including all onshore and offshore infrastructure.
Wind turbine generator	Power generating device that is driven by the kinetic energy of the wind

13 Offshore Ornithology

13.1 Introduction

1. This chapter of the Environmental Statement (ES) considers the likely significant effects of the North Falls offshore wind farm (OWF) (hereafter 'North Falls' or 'the Project') on offshore ornithology receptors. The chapter provides an overview of the existing environment for the proposed offshore project area, followed by an assessment of the likely significant effects for the construction, operation, and decommissioning phases of the Project.

2. This chapter has been written by Royal HaskoningDHV, with the assessment undertaken with specific reference to the relevant legislation and guidance, of which the primary sources are the National Policy Statements (NPS). Details of these and the methodology used for the Environmental Impact Assessment (EIA) and Cumulative Effects Assessment (CEA) are presented in Section 13.4.

3. The assessment should be read in conjunction with following linked chapters:

    - Chapter 10 Benthic and Intertidal Ecology (Document Reference: 3.1.12); and
    - Chapter 11 Fish and Shellfish Ecology (Document Reference: 3.1.13).

4. Additional information to support the offshore ornithology assessment includes:

    - Appendix 13.1 Offshore Ornithology Consultation (Document Reference: 3.3.12);
    - Appendix 13.2 Offshore Ornithology Technical Report (Document Reference: 3.3.13); and
    - Appendix 13.3 Supplementary Information for the Offshore Ornithology Cumulative Effects Assessment (Document Reference: 3.3.14).

5. The Report to Inform Appropriate Assessment (RIAA) includes shadow appropriate assessments for offshore ornithology features of Special Protection Areas (SPAs) that have been screened in for North Falls.

13.2 Consultation

6. Consultation with regard to offshore ornithology has been undertaken in line with the general process described in ES Chapter 6 EIA Methodology (Document Reference: 3.1.8). This has included the Scoping Opinion, consultation on the Preliminary Environmental Report (PEIR) (and accompanying draft RIAA and without prejudice compensation documents) and the ongoing technical consultation via the Offshore Ornithology Expert Topic Group (ETG). The feedback received has been considered in preparing this ES chapter. Full details of the issues raised by consultees and how these have been addressed are provided in ES Appendix 13.1 (Document Reference: 3.3.12).

7. This chapter has been updated following the consultation on the PEIR in order to produce the final assessment. Full details of the consultation process are also presented in the Consultation Report as part of the Development Consent Order (DCO) application.

13.3 Scope

13.3.1 Study area and survey area

8. The study area for the ES for offshore ornithology is the North Falls array area, a 4km buffer around it, and (where it extends outside the 4km buffer from the array area) the offshore cable corridor to mean low water springs (MLWS) at the landfall[1]. This is shown in Figure 13.1 (Document Reference: 3.2.9) and is the area considered in the assessment in this chapter, which has been defined based on the maximum zone of influence of the effects to be considered by the assessment.

9. Baseline surveys were carried out over the array area and a 4km buffer, referred to as the survey area.

10. For the RIAA Part 4 Offshore Ornithology (Document Reference: 7.1.4), the survey area was extended to the west to include a 12km buffer from the array area during the final two months of surveys (January and February 2021). The 12km extension was included in anticipation of requirements to consider displacement of red-throated divers beyond 4km from the North Fall array boundary in the shadow appropriate assessment for the Outer Thames Estuary SPA. A 12km buffer was based on the findings of post-construction monitoring at the London Array OWF (APEM 2021), that displacement effects on red-throated divers were detected out to 11.5km from the OWF array.

11. The area surveyed for the North Falls offshore ornithology assessment is consistent with the Natural England (2022a) best practice advice for baseline surveys for OWFs, although the survey programme was completed before this advice was issued.

13.3.2 Realistic worst case scenario

12. The final design of North Falls will be confirmed through detailed engineering design studies that will be undertaken post-consent. In order to provide a precautionary but robust impact assessment at this stage of the development process, realistic worst case scenarios have been defined in terms of the potential effects that may arise. This approach to EIA, referred to as the Rochdale Envelope, is common practice for developments of this nature, as set out in Planning Inspectorate Advice Note Nine (2018). The Rochdale Envelope for a project outlines the realistic worst case scenario for each individual impact, so that it can be safely assumed that all other scenarios within the design envelope will have less impact. Further details are provided in ES Chapter 6 EIA Methodology (Document Reference: 3.1.8).

13. One area of optionality is in relation to the National Grid connection point (discussed further in ES Chapter 5 Project Description (Document Reference:

[1] Onshore ornithology above MLWS is discussed in ES Chapter 24 Onshore Ornithology (Document Reference: 3.1.26).

3.1.7)). The following grid connection options are included in the Project design envelope:

- Option 1: Onshore electrical connection at a national grid connection point within the Tendring peninsula of Essex, with a project alone onshore cable route and onshore substation infrastructure;
- Option 2: Onshore electrical connection at a national grid connection point within the Tendring peninsula of Essex, sharing an onshore cable route and onshore duct installation (but with separate onshore export cables) and co-locating separate project onshore substation infrastructure with Five Estuaries; or
- Option 3: Offshore electrical connection, supplied by a third party.

14. For the offshore project area, Options 1 and 2 would be the same. Within the array area, under Options 1 and 2 there would be up to two offshore substation platforms (OSP); whereas for Option 3 there would be one offshore converter platform (OCP) and up to one OSP, i.e. under all scenarios there would be a maximum of two platforms. For Option 3, there would be no project export cables to shore.

15. The realistic worst case scenarios for the likely significant effects scoped into the EIA for the offshore ornithology assessment relate to Options 1 and 2 and these are summarised in Table 13.1. These are based on North Falls parameters described in ES Chapter 5 Project Description (Document Reference: 3.1.7), which provides further details regarding specific activities and their durations.

Table 13.1 Realistic worst case scenarios

Impact	Parameter	Notes
Construction		
Disturbance and displacement from construction activities	Length of offshore construction period: two years Daily timing of works offshore: 24 hours Maximum no of vessels of all types operating simultaneously in the offshore project area: 35 Construction vessel trips to port: 2,532 over two year offshore construction period (average of three movements per day) Construction port: To be determined, could be any North Sea port (UK and / or European Union (EU)). Maximum no. of foundation installation activities occurring at any one time: three (including maximum of two simultaneous pile driving operations) Maximum number of helicopter round trips per annum: c. 100 (1 – 2 per week) Installation period for offshore export cables: six months Number of cable laying vessels operating simultaneously: two Speed of cable-laying vessels: 150 – 400m/h	The worst case scenario is based on the longest construction period and the maximum numbers of plant on site and operational at a given time.
Indirect effect as a result of displacement of prey species due to increased noise and disturbance to seabed	Underwater noise effects on fish (ES Chapter 11 Fish and Shellfish Ecology, Document Reference: 3.1.13): Spatial WCS: • 57 Wind Turbine Generators (WTGs) on monopile foundations; • Two OSPs / OCP on monopile foundations; • Maximum pile diameter for WTG and OSP monopiles: 17m; • 6,000Kj hammer energy, 7.5 hours piling duration per monopile including a 10 minute soft start at 15% hammer energy and 120 minute (2 hour) ramp up to full energy (where required); • Maximum number of monopiles to be installed per 24 hour period: three; • Total WTG active piling duration: 427.5 hours (equivalent to 17.8 days);	N/A.

Impact	Parameter	Notes
	• Total OSPs / OCP active piling duration: 15 hours (less than one day); • Duration of foundation installation: 12 months; and • Maximum no. of foundation installation activities occurring at any one time: three (including maximum of two simultaneous pile driving operations). Temporal WCS: • 57 WTGs on pin-piled jacket foundations, with up to four legs per jacket and two piles per leg (i.e. eight piles per jacket; 456 total); • Two OSPs / OCP on pin-piled jacket foundations, with up to six legs per jacket and two piles per leg (i.e. 12 piles per jacket; 24 total); • Maximum pile diameter for WTG pin piles: 6m; • Maximum pile diameter for OSP / OCP pin piles: 3.5m; • WTGs: 4,400kJ hammer energy, 4.5 hours piling duration including a 10 minute soft start at 15% hammer energy, and 80 minute ramp up to full energy (where required); • OSP / OCPs: 3,000kJ hammer energy; • Maximum number of pin piles to be installed per 24 hour period: six; • Total WTG active piling duration: 2,052 hours (equivalent to 85.5 days); • Total OSP active piling duration: 108 hours (equivalent to 4.5 days); • Duration of foundation installation: 12 months; and • Maximum no. of foundation installation activities occurring at any one time: three (including maximum of two simultaneous pile driving operations).	
	Habitat disturbance effects on fish (ES Chapter 11 Fish and Shellfish Ecology, Document Reference: 3.1.13): The maximum worst case area of temporary disturbance to benthic habitats during construction would be 5.88km^2 within the	Based on es Chapter 10 Benthic and Intertidal Ecology (Document Reference: 3.1.12).

Impact	Parameter	Notes
	array area (equivalent to 6.2% of the maximum offshore development footprint (95km^2)) and 3.31km^2 within the offshore cable corridor.	
	Increased suspended sediment concentration effects on fish (ES Chapter 11 Fish and Shellfish Ecology (Document Reference: 3.1.13)): • Seabed preparation for foundation installation = 1,14Mm^3; • Array and interconnector platform cables installation = 28.96Mm^3; and • Offshore export cables installation = 1.7Mm^3.	Based on ES Chapter 10 Benthic and Intertidal Ecology (Document Reference: 3.1.12).
Operation and maintenance		
Displacement / barrier effect from offshore infrastructure and associated operational activity	A wind farm area of 95km^2 plus 4km buffer with maximum of 57 WTGs at a minimum spacing of: • 5 x the rotor diameter (i.e. 1180m for the smallest turbines with 236m rotor diameter or 1,685m for the largest turbines with 337m rotor diameter) in the downwind direction; and • 4 x the rotor diameter (i.e. 944m for the smallest turbines with 236m rotor diameter or 1,348m for the largest turbines with 337m rotor diameter) in the cross wind direction.	For most offshore ornithology receptors the assessment of displacement considers the array area plus a 2km buffer, the maximum 4km buffer is considered for red-throated diver only. For all species the assessment of operational displacement covers all different WTGs scenarios described below (unlike collision risk where a separate assessment is presented for each scenario).
	Maximum of 1,222 vessel round trips per annum to support wind farm operations. Maximum of 100 helicopter round trips per annum (c.1 – 2 per week) for scheduled and unscheduled maintenance. Lighting requirements Aviation light: Only on specific structures, usually the perimeter, mounted on the top of the nacelles. Off during the day. Red, up to 2,000 Candela (Cd) light displayed at night only	N/A.

Impact	Parameter	Notes
	Dimmable to 200Cd when visibility is greater than 5km at night Synchronised flashing Morse "W" A reduced intensity at and below the horizontal. 360° visibility Compatible with Night Vision Imaging Systems (NVIS) UPS: eight hours required to maintain all aviation warning lights Helihoist light: Low intensity green 200Cd light. Off, unless the WTG is being prepared for helicopter approach	
Collision risk	Two design scenarios: Minimum Rotor Diameter (MiRD) Scenario (smaller turbines) – 57 WTGs, 236m rotor diameter, (air gap 26.6m above highest astronomical tide (HAT), 27m above mean high water spring (MHWS)); Maximum Rotor Diameter (MaRD) Scenario (larger turbines) – 34 WTGs, 337m rotor diameter, (air gap also 26.6m above HAT, 27m above MHWS).	Collision Risk Modelling (CRM) has been carried out for both WTG scenarios based on the WTG specifications (Table 13.36, ES Appendix 13.2 (Document Reference: 3.3.13)). For each bird species, the WTG scenario which produces the highest collision risk has been used in the assessment (see Section 13.6.2.2 below).
Indirect effects due to habitat loss / change for key prey species	Maximum permanent habitat loss 5.37km^2 from 57 WTGs, two OSPs / OCP, scour protection and array / inter-platform cable protection within the array area; 5.7% of total array area. Offshore export cable: Up to 12.5km of cable protection may be required in the unlikely event that offshore export cables cannot be buried (based on 10% of the length) x 6m cable protection width = 75,240m^2	Based on ES Chapter 10 Benthic and Intertidal Ecology (Document Reference: 3.1.12).
Decommissioning		
Disturbance and displacement from decommissioning activities	<u>Array area:</u> Cutting of piles below the seabed surface: • 480 pin-piles of 6m diameter; • 57 wind turbines x 8 piles; and • 2 OSPs / OCP x 12 piles.	Assumed similar to construction and therefore a worst case would be as above.

Impact	Parameter	Notes
Indirect effects as a result of displacement of prey species due to increased noise and disturbance to seabed	Or • 59 monopiles of 17m diameter (57 wind turbines + 2 OSPs / OCP) Or Removal of largest foundations (GBS): • 57 WTG x 65m diameter; and • 2 OSPs / OCP x 65m diameter. Or A mixture of the above foundation types. The foundation types could also include suction caissons, however these do not represent a worst case scenario for decommissioning. <u>Offshore export cables:</u> Up to 125.4km of export cable (removal to be determined in consultation with key stakeholders as part of the decommissioning plan) <u>Array cables:</u> Up to 170km of array cable (removal to be determined in consultation with key stakeholders as part of the decommissioning plan) <u>Platform interconnector cables:</u> Up to 20km of array cable (removal to be determined in consultation with key stakeholders as part of the decommissioning plan) The following infrastructure is likely to be decommissioned in situ depending on available information at the time of decommissioning, however where it represents the worst case scenario (e.g. for disturbance), removal is assessed: • Scour protection;	Any area affected would be less than or at worst equal to the areas of disturbance during construction. There would be limited noise disturbance to prey (as no piling and no use of explosives).

Impact	Parameter	Notes
	• Offshore cables may be removed or left in situ; and • Crossings and cable protection. The detail and scope of the decommissioning works will be determined by the relevant legislation and guidance at the time of decommissioning and will be agreed with the regulator.	

13.3.3 Summary of mitigation embedded in the design

16. This section outlines the embedded mitigation relevant to the offshore ornithology assessment, which has been incorporated into the design of North Falls (Table 13.2).

Table 13.2 Embedded mitigation measures

Parameter	Mitigation measures embedded into North Falls design
Array area	Following PEIR, the array area has been reduced from 149.5km^2 down to 95km^2. This has involved the complete removal of the former northern array and refinement of the former southern array (now the array area), increasing the distance from the Outer Thames Estuary SPA.
Reduced turbine numbers	Following PEIR, the maximum number of turbines (assuming the smallest turbine model) has been reduced from 72 to 57 and the number of the largest turbine model has been reduced from 40 to 34.
Offshore cable corridor	Offshore cable corridor site selection reduced overlap with the Outer Thames Estuary SPA. Site selection was undertaken in consultation with Natural England (see ES Chapter 4 Site Selection and Assessment of Alternatives (Document Reference: 3.1.6)).
WTG air gap	A minimum air gap (the distance between the lower rotor tip of a WTG and the sea surface) of 27m above MHWS (26.6m above HAT). This is an increase of 5m above the minimum of 22m MHWS required for navigation purposes to reduce collision risk for birds (as most seabirds tend to fly low to the sea surface).
Protocol for reducing disturbance to red-throated divers	The protocol is designed to minimise disturbance to non-breeding red-throated diver, and would apply during the core winter period between 1 November and 1 March inclusive. Details of the protocol are set out in the Outline Project Environmental Management Plan, Appendix B.

13.4 Assessment methodology

13.4.1 Legislation, guidance and policy

13.4.1.1 National Policy Statements

17. The assessment of likely significant effects upon offshore ornithology receptors has been made with specific reference to the relevant NPS. These are the principal decision making documents for Nationally Significant Infrastructure Projects (NSIPs). Those relevant to the Project and offshore ornithology are:

- Overarching NPS for Energy (EN-1) (Department of Energy Security and Net Zero) (DESNZ, 2023a); and
- NPS for Renewable Energy Infrastructure (EN-3) (DESNZ, 2023b).

18. The specific assessment requirements for offshore ornithology, as detailed in the NPS, are summarised in Table 13.3 together with an indication of the section of the ES chapter where each is addressed.

Table 13.3 NPS assessment requirements

NPS requirement	NPS reference	ES reference
Overarching NPS for Energy (EN-1)		
The applicant should show how the project has taken advantage of opportunities to conserve and enhance biodiversity and geological conservation interests.	Paragraph 5.4.19	Embedded mitigation measures have been outlined in Section 13.3.3.

NPS requirement	NPS reference	ES reference
NPS for Renewable Energy Infrastructure (EN-3)		
Applicants should assess the potential of their proposed development to have net positive effects on marine ecology and biodiversity as well as negative effects.	Paragraph 2.8.103	This has been discussed throughout the assessment, Section 13.6 and 13.7.
Any relevant data that has been collected as part of post-construction ecological monitoring from existing, operational OWF should be referred to where appropriate	Paragraph 2.8.106	Evidence from operational OWFs is referred to throughout the assessment.
Currently, cumulative impact assessments for ornithology are based on the consented Rochdale Envelope parameters of projects, rather than the 'as-built' parameters, which may pose a lower risk to birds. The applicant must ensure any draft consents include provisions to define the final 'as built' parameters (which may not then be exceeded). These parameters must be used in future cumulative assessments.	Paragraph 2.8.137 – 2.8.138	Provisions to define and confirm the 'as built' parameters so that these can be used in CEAs for future developments has been considered in the preparation of the draft DCO.
Applicants should discuss the scope, effort and methods required for ornithological surveys with the relevant statutory advisor, taking into consideration baseline and monitoring data from operational wind farms.	Paragraph 2.8.143	Natural England were appraised of the survey programme prior to the commencement of the Evidence Plan Process (EPP).
Applicants must undertake collision risk modelling, as well as displacement and population viability analysis for certain species of birds. Applicants are expected to seek advice from SNCBs.	Paragraph 2.8.144	Displacement assessments have been undertaken based on guidance from UK Statutory Nature Conservation Bodies (SNCBs) (2017) and specific advice from Natural England during the EPP. For the ES, where appropriate, reference has been made to existing population viability assessments (PVAs) for species scoped in for assessment and project specific PVAs have been undertaken where required for SPA populations (for the RIAA).
Turbine parameters should also be developed to reduce collision risk where assessment shows there is a significant risk of collision (e.g. altering rotor height)	Paragraph 2.8.241	The project designs of North Falls include an air gap of 27m MHWS. This includes a 5m increase on the standard air gap of 22m MHWS required for navigational purposes. This commitment was made in response to consultation with Natural England and the Royal Society for the Protection of Birds (RSPB) through EPP.

13.4.1.2 Other

19. In addition to the NPS, there are a number of pieces of policy and guidance applicable to the assessment of offshore ornithology.

20. England currently has nine marine plans; those relevant to North Falls are the East Inshore, The East Offshore Marine Plans and the South East Marine Plan (HM Government, 2014, HM Government, 2021).

21. The East Inshore and Offshore Marine Plans contain the following objectives which are of relevance to offshore ornithology:

- Objective 6: 'To have a healthy, resilient and adaptable marine ecosystem in the East Marine Plan areas'; and
- Objective 7: 'To protect, conserve and, where appropriate, recover biodiversity that is in or dependent upon the East marine plan areas'.

22. The South East Marine Plan contains the following objectives which are of relevance to offshore ornithology:

- Objective 11: 'Biodiversity is protected, conserved and, where appropriate, recovered, and loss has been halted.'; and
- Objective 13: 'Our oceans support viable populations of representative, rare, vulnerable, and valued species.'

23. Further information on the Applicant's compliance with the Marine Plans is provided in the Marine Plan Assessment (Document Reference: 7.5).

24. Guidance of relevance to offshore ornithology includes:

- The most relevant EIA guidance for offshore ornithology receptors is Chartered Institute of Ecology and Environmental Management (CIEEM) (2018). The EIA methodology applied in this chapter is based on this guidance;
- Guidance documents for the assessment of OWF impacts on offshore ornithology receptors produced by Natural England (Natural England, 2022a, 2022b, 2022c, 2023);
- Headroom in Cumulative Offshore Wind Farm Impacts for Seabirds: Legal Issues and Possible Solutions (The Crown Estate and Womble Bond Dickinson, 2021); and
- A wide range of additional guidance has been referred to throughout the assessment as required.

13.4.2 Data sources

13.4.2.1 Site specific

25. To provide site specific and up to date information on which to base the impact assessment, baseline surveys for offshore ornithology receptors were carried out between March 2019 and February 2021.

26. The area for baseline surveys was based on a 4km buffer of the former array areas (comprising a northern and southern array area). Following PEIR consultation feedback, the former array areas were refined from 149.5km^2 down to 95km^2. This has involved the removal of the northern array and a reduction in the size of the southern array (now referred to as the 'array area').

27. The surveys comprised 24 monthly digital aerial surveys flown along strip transects (oriented roughly north-west to south-east and at a spacing of 2.5km). Details of the survey methodology are included in ES Appendix 13.2 (Document Reference: 3.3.13); HiDef 2020, 2021).

28. Baseline survey data were used to derive abundance and density estimates for offshore ornithology receptors for the array area alone, and the array area and 2km and 4km buffers, for use in the assessment Section 13.3.1, Figure 13.1 (Document Reference: 3.2.9)) (2km buffer estimates were used for assessment of displacement effects for all species except red-throated diver, for which a 4km buffer was used, see Section 13.6.2.1 below). As noted above, the baseline surveys were based on a 4km buffer of the former array areas, however data analysis has been revised based on the refined array area to provide updated densities and population abundance estimates for offshore ornithology receptors, for the array area and 2km and 4km buffers.

29. Data was processed to give 15% coverage in all surveys.

30. The baseline survey methodology is consistent with the Natural England (2022a) best practice advice for baseline surveys for OWFs.

31. In January and February 2021, the baseline survey area was extended to 12km from the former southern array area in the west, to include additional areas within and close to the Outer Thames Estuary SPA for red-throated diver (noting that red-throated diver abundance within this SPA peaks in January and February). As stated above (paragraph 10), the 12km extension was based on the findings of post-construction monitoring at the London Array OWF (APEM 2021), where displacement effects on red-throated divers were detected out to 11.5km from the OWF array. Data for the extended survey area in January and February 2021 has been used for the RIAA but is not applicable to the EIA.

13.4.2.2 Other data sets

32. Other data sources used in the offshore ornithology assessment are listed in Table 13.4.

Table 13.4 Additional data sources used for offshore ornithology

Data Set	Spatial Coverage	Year	Notes
Seabird Mapping and Sensitivity Tool	English Offshore Waters	1979 – 2012	Bradbury *et al.* (2014). Used for cumulative assessment of red-throated diver displacement
Red-throated diver survey of Outer Thames Estuary SPA	SPA boundary	2018	Irwin *et al.,* 2019. Used as source of red-throated diver densities in the Offshore Cable Corridor where this overlaps with the SPA, and for RIAA

13.4.2.3 Desk-based assessment

33. 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 published literature on seabird ecology and distribution and on the potential impacts of wind farms (both as derived from expert judgement and post-construction monitoring studies), as well as guidance from Regulators and SNCBs on offshore ornithology assessment for OWFs. Key sources are as follows:

- Potential impacts of OWFs on seabirds (Garthe and Hüppop, 2004; Drewitt and Langston, 2006; Stienen *et al.*, 2007; Speakman *et al.*, 2009; Langston, 2010; Band, 2012; Cook *et al.*, 2014; Furness and Wade, 2012; Wright *et al.*, 2012; Furness *et al.*, 2013);
- Methodology for assessment (Band, 2012; Johnston *et al.*, 2014a and b; SNCBs, 2017, 2022; McGregor *et al.*, 2018; Natural England, 2022c and 2023);
- Bird population estimates (SPA citations / departmental briefs / conservation advice from the websites of SNCBs; Joint Nature Conservation Committee (JNCC) Seabird Monitoring Programme (SMP) database[2]; Mitchell *et al.*, 2004; Furness, 2015; Frost *et al.*, 2019; Woodward *et al.*, 2020);
- Bird distribution (Stone *et al.*, 1995; Kober *et al.*, 2010; Balmer *et al.*, 2013; Wakefield *et al.*, 2013, 2017; Waggit *et al.*, 2019; Cleasby *et al.*, 2018, 2020);
- Bird migration and foraging movements (Wernham *et al.*, 2002; Thaxter *et al.*, 2012; Wright *et al.*, 2012; Furness, 2015; Woodward *et al.*, 2019);
- Red-throated diver densities in the Outer Thames Estuary SPA (O'Brien *et al.*, 2012; APEM, 2013; Irwin *et al.*, 2019);
- Relevant documents from marine licence applications for other OWFs in the North Sea and Channel; and
- Relevant ecological studies for species included in EIA and Habitats Regulations Assessment (HRA) (published scientific papers and 'grey' literature).

34. Interim updated advice on demographic rates, EIA scale mortality rates and reference populations for use in offshore wind impact assessments was received from Natural England in March 2024. As the North Falls assessment was at a late stage of drafting, this chapter does not reflect this new advice.. The advice includes small changes to the average annual mortality rate used in the assessment (see Section 13.5.4 below), and updated guidance on the reference populations used during the breeding season (see Section 13.5.2 below). A review of the implications indicates that for all species and seasons scoped in for assessment, there would be no change to the magnitude of effect or the outcome of the assessment. For all species scoped in for assessment for displacement and collision, the new breeding season reference populations are larger than those which have been used in this chapter, meaning that predicted mortality from displacement and / or collision, expressed as a percentage increase in baseline mortality of a species population, would be smaller than presented (notes to this effect are included in the assessments throughout). The exception to this is great black-backed gull, where the new breeding season reference population is smaller than that used, but, for this species, scoped in for collision risk only, there is no predicted mortality during the breeding season (Section 13.6.2.2.2).

[2] https://app.bto.org/seabirds/public/index.jsp

13.4.3 Impact assessment methodology

35. ES Chapter 6 EIA Methodology (Document Reference: 3.1.8) explains the general impact assessment methodology applied to North Falls. The impact assessment methods applied in this chapter are adapted for offshore ornithology receptors and aligned with the key guidance document produced on impact assessment on ecological receptors (CIEEM, 2018).

36. The methodology applied in this chapter has also been the subject of extensive consultation with Natural England and the RSPB through the EPP for the proposed North Falls project; and been informed by recent DCO examinations for other OWFs in the southern North Sea.

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

- 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.
- 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.
- 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 birds.

38. The terms impact and effect are defined as follows (after CIEEM, 2018):

- Impact – a change resulting from an activity associated with the Project, e.g. increased suspended sediments or increased noise, defined in terms of magnitude.
- Effect – the consequence of an impact combining with a receptor, defined in terms of significance (significance being dependant on magnitude of impact and the sensitivity / value / importance of receptor).

13.4.3.1 Sensitivity

39. For each potential impact, the assessment identifies receptors within the study area which are sensitive to that impact. Definitions of sensitivity for ornithology receptors are included in Table 13.5 using the example of disturbance from construction activity.

Table 13.5 Definition of sensitivity for offshore ornithology receptors (illustrated for potential effects of disturbance from construction activities)

Sensitivity	Definition
High	Ornithology receptor (bird species) has very limited tolerance of a potential impact, e.g. strongly displaced by sources of disturbance such as noise, light, vessel movements and the sight of people

Sensitivity	Definition
Medium	Ornithology receptor (bird species) has limited tolerance of a potential impact, e.g. moderately displaced by sources of disturbance such as noise, light, vessel movements and the sight of people
Low	Ornithology receptor (bird species) has some tolerance of a potential impact, e.g. partially displaced by sources of disturbance such as noise, light, vessel movements and the sight of people.
Negligible	Ornithology receptor (bird species) is generally tolerant of a potential impact e.g. not displaced by sources of disturbance such as noise, light, vessel movements and the sight of people.

13.4.3.2 Value

40. The conservation value of ornithological receptors is based on the population from which individuals are predicted to be drawn, based on current understanding of the movements of bird species. Therefore, conservation value for a species can vary through the year depending on changes in 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 in part to the degree of connectivity which is predicted between the array area and protected populations.

41. Example definitions of the value levels for ornithology receptors are given in Table 13.6. These are related to connectivity with populations that are protected as qualifying species of SPAs. SPAs are internationally designated sites which carry strong protection for populations of qualifying bird species. These SPA qualifying species are a key consideration for the ornithology assessment.

Table 13.6 Definition of conservation value for offshore ornithology receptors

Magnitude	Definition
High	A species for which individuals at risk can be clearly connected to a particular SPA or potential SPA (pSPA).
Medium	A species for which individuals at risk are probably drawn from particular SPA or pSPA populations, although other populations (both SPA and non-SPA) may also contribute to individuals at risk
Low	A species for which individuals at risk have no known connectivity to SPAs, or for which no SPAs are designated.

13.4.3.3 Magnitude

42. The definitions of the impact magnitude levels for ornithology receptors are set out in Table 13.7. This set of definitions has been determined based on changes to bird populations.

Table 13.7 Definition of magnitude for offshore ornithology receptors

Magnitude	Definition
High	A change that is predicted to irreversibly alter the receptor population in the short to long term, and to alter the long-term viability of the receptor population and / or the integrity of a protected site.
Medium	A change that occurs in the short to long-term, but which is not predicted to alter the long-term viability of the receptor population and / or the integrity of a protected site.

Magnitude	Definition
Low	A change that is sufficiently small-scale or of short duration to cause no long-term harm to the receptor population and / or the integrity of a protected site.
Negligible	A very slight change that is sufficiently small scale or of such short duration that it may be undetectable in the context of natural variation.
No change	No positive or negative change is predicted.

13.4.3.4 Significance of effect

43. The assessment of the significance of an effect is a function of the sensitivity of the receptor and the magnitude of the impact (see ES Chapter 6 EIA Methodology (Document Reference: 3.1.8) for further details). The determination of significance is guided using a ‘significance of effect’ matrix, as shown in Table 13.8.

44. 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. It is not a prescriptive formulaic method. Expert judgement has been applied to the assessment of likelihood and ecological significance of a predicted impact.

45. In particular, it should be noted that conservation value *per se* is not included in the matrix but is taken into account in the narrative of the assessment. Conservation value and high sensitivity are not necessarily linked for a particular impact. A receptor could be 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. Likely effect significance will not be inflated simply because a feature is ‘valued’. Similarly, potentially highly significant impacts will not be deflated simply because a feature is not of high value.

46. Where possible, assessment is based upon quantitative and accepted criteria (for example, industry standard guidance on collision risk modelling (Band, 2012; McGregor *et al.*, 2018), and displacement (SNCB, 2017; 2022), and / or predicted changes in demographic parameters determined through population modelling), together with the use of professional judgement and expert interpretation to establish to what extent an impact is ecologically significant. The assessment refers to and includes embedded mitigation (Section 13.3.3).

47. As a rule of thumb, assessment outcomes of major or moderate are regarded within this chapter as potentially ecologically significant.

48. CIEEM (2018) guidance states that “*significance is a concept related to the weight that should be attached to effects when decisions are made… so that the decision maker is adequately informed of the environment consequences of permitting a project… A significant effect does not necessarily equate to an effect so severe that consent for the project should be refused … For example, many projects with significant negative ecological effects have been lawfully permitted following EIA procedures”*. Ecological significance is defined as follows: “*In broad terms, significant effects encompass impacts on the structure and function of defined sites, habitats or ecosystems and the conservation status of habitats and species (including extent, abundance, and distribution). Significant effects should be qualified with reference to an appropriate*

geographic scale, for example a significant effect on a Site of Special Scientific Interest (SSSI) … is likely to be of national significance."

49. Potential ecological significance may include situations where an effect would result in a failure to meet legally binding objectives such as conservation objectives for designated sites; or a breach of environmental legislation.

50. Should major or moderate effects be identified within the assessment, these would be regarded within this chapter as significant. Should the assessment indicate any likely significant effect, mitigation measures would be identified, where possible, in consultation with the regulatory authorities and relevant stakeholders. The aim of mitigation measures is to avoid or reduce the overall significance of effect to determine a residual effect upon a given receptor

Table 13.8 Significance of effect matrix

		Adverse 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*	*Negligible*	*Negligible*	*Negligible*	*Minor*	*Minor*	*Moderate*
	Negligible	*Minor*	*Negligible*	*Negligible*	*Negligible*	*Negligible*	*Negligible*	*Negligible*	*Minor*

13.4.4 Cumulative effects assessment methodology

51. The CEA considers other plans, projects and activities that may result in cumulation on offshore ornithology receptors with North Falls. ES Chapter 6 EIA Methodology (Document Reference: 3.1.8) provides further details of the general framework and approach to the CEA.

52. The methodology has also been aligned with the approach to the assessment of cumulative impacts that has been applied by Ministers when consenting OWFs and confirmed in recent consent decisions. It also follows the approach set out in guidance from the Planning Inspectorate (Planning Inspectorate, 2019) and from the renewables industry (RenewableUK, 2013).

13.4.5 Transboundary impact assessment methodology

53. As discussed in the North Falls Scoping Report, due to the level of development in the southern North Sea by EU Member States (i.e. Belgium, the Netherlands, Germany and Denmark), and given that birds are highly mobile and migratory, there is potential for transboundary impacts especially with regard to displacement / barrier effects and collision risk. Transboundary impacts are assessed in Section 13.9.

54. The potential for transboundary impacts in relation to 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) is assessed in the RIAA.

13.4.6 Assumptions and limitations

55. The offshore ornithology assessment contains a wide range of sources of uncertainty. These include the process of estimating seabird density and abundance estimates from baseline survey data, estimated values and seabird characteristics used to predict displacement (e.g. displacement and mortality rates), collisions (e.g. flight height distributions, avoidance rates, bird size, flight speeds, bird behaviour, and the parameters of the WTGs), and demographic rates (e.g. environmental and demographic variations in survival and productivity). This is not an exhaustive list. Where specific limitations apply, for example in relation to the use of baseline data for the quantification of potential effects, or where assumptions have been made in the assessment of particular potential effects, these are included in the description of the assessment.

56. The marine environment is inherently highly variable, and the analytical methods used make allowance for the associated uncertainties through the estimate of variance around central point estimates. It is important that these uncertainties are given consideration in impact assessment (MacArthur Green, 2019a).

57. Uncertainties within the assessment process for OWFs can arise from environmental or natural variation in the baseline data (for example year-to-year changes in the numbers and distribution of offshore birds within a project study area related to environmental factors such as prey availability), and uncertainty in models used to predict the effects of OWFs on birds (for example species specific parameters such as flight height distribution used in CRM). The key general distinction between environmental variation and other sources of uncertainty is that environmental variation is an inherent feature of the system (e.g. arising from seabird biology), and so cannot be reduced through additional data collection, whereas uncertainty is a feature of the state of knowledge, and so can, at least in principle, be reduced through additional data collection and improved understanding, thereby enhancing validity of models (Searle *et al.*, 2021).

58. The precautionary principle employed in the process of impact assessment (CIEEM, 2018) means that there can be a tendency to add precaution, or make precautionary assumptions, at each stage of an assessment by focusing attention on the upper limits of each component variable. The end result is that worst-case scenarios compound to over-estimate the magnitude of the impacts. This is then further compounded when individual project level effects are added together in cumulative and in-combination assessments (MacArthur Green, 2019a).

59. Sources of uncertainty and precaution relating to the quantification and assessment of the effects of OWFs on ornithology receptors are described in the individual species assessments in Sections 13.6.1, 13.6.2, 13.8.3 and 13.8 below.

13.5 Existing environment

60. The characterisation of the existing or baseline environment is undertaken based on the baseline surveys (outlined in Section 13.4.2.1 above and as

detailed in ES Appendix 13.2 (Document Reference: 3.3.13)), the desk study (Section 13.4.2.3), and other relevant literature.

13.5.1 Key bird species

61. Birds present in offshore waters and potentially affected by the construction, operation, maintenance and decommissioning of North Falls will be predominantly seabirds (defined for this report as auks, gulls, terns, gannets, skuas, shearwaters, petrels, divers and sea duck). These species have the potential to be present during the breeding season and non-breeding season (including spring / autumn migration / passage periods). Other bird species that may be affected by the Project include waterfowl (e.g. swans, geese, ducks, and waders) and other bird species which may fly through the Project areas during spring and / or autumn migration / passage periods.

62. The key bird species recorded during site-specific digital (video) aerial bird surveys (surveys described in Section 13.4.2 and ES Appendix 13.2 (Document Reference: 3.3.13)) are listed in Table 13.9 along with details of their conservation status. Plots of the distribution of all species recorded in the baseline surveys are included in ES Appendix 13.2 (Document Reference: 3.3.13), Annex 3.

Table 13.9 Seabird species recorded by baseline surveys between March 2019 and February 2021 within North Falls Array Area and 4km buffer, and their conservation status

Species[1]	Scientific name	Conservation status[2]
Arctic skua	*Stercorarius parasiticus*	Birds of Conservation Concern (BoCC) Red, Birds Directive Migratory Species IUCN ‘Least Concern’
Black-headed gull	*Chroicocephalus ridibundus*	BoCC Amber, Birds Directive Migratory Species IUCN ‘Least Concern’
Common gull	*Larus canus*	BoCC Amber, Birds Directive Migratory Species IUCN ‘Least Concern’
Common tern	*Sterna hirundo*	BoCC Amber, Birds Directive Annex 1 IUCN ‘Least Concern’
Cormorant	*Phalacrocorax carbo*	BoCC Green, Birds Directive Migratory Species IUCN ‘Least Concern’
Fulmar	*Fulmarus glacialis*	BoCC Amber, Birds Directive Migratory Species IUCN ‘Least Concern’
Gannet	*Morus bassanus*	BoCC Amber, Birds Directive Migratory Species IUCN ‘Least Concern’
Great black-backed gull	*Larus marinus*	BoCC Amber, Birds Directive Migratory Species IUCN ‘Least Concern’
Great skua	*Stercorarius skua*	BoCC Amber, Birds Directive Migratory Species IUCN ‘Least Concern’
Guillemot	*Uria aalge*	BoCC Amber, Birds Directive Migratory Species IUCN ‘Least Concern’
Herring gull	*Larus argentatus*	BoCC Red, Birds Directive Migratory Species IUCN ‘Near Threatened’
Kittiwake	*Rissa tridactyla*	BoCC Red, Birds Directive Migratory Species

Species[1]	Scientific name	Conservation status[2]
		IUCN 'Vulnerable'
Lesser black-backed gull	*Larus fuscus*	BoCC Amber, Birds Directive Migratory Species IUCN 'Least Concern'
Little gull	*Hydrocoloeus minutus*	BoCC Green, Birds Directive Migratory Species IUCN 'Near Threatened'
Puffin	*Fratercula arctica*	BoCC Red, Birds Directive Migratory Species
Razorbill	*Alca torda*	BoCC Amber, Birds Directive Migratory Species IUCN 'Near Threatened'
Red-throated diver	*Gavia stellata*	BoCC Green, Birds Directive Annex 1 IUCN 'Least Concern'
Sandwich tern	*Thalasseus sandvicensis*	BoCC Amber, Birds Directive Annex 1 IUCN 'Least Concern'
1. Vernacular British names as defined by the British Ornithologists Union (https://bou.org.uk/british-list/bird-names/) are used (rather than international English bird names) 2. BOCC = in the UK, Stanbury *et al.* (2021), IUCN – International Union for Conservation of Nature's global red list of threatened species (https://www.iucnredlist.org/)		

63. In addition to the seabird species listed in Table 13.9, additional bird species were recorded irregularly including migratory waterfowl (Brent goose *Branta bernicla*, Shelduck *Tadorna tadorna*, Whimbrel *Numenius phaeopus* and Wigeon *Anas penelope*), raptors (Peregrine *Falco peregrinus*, Osprey *Pandion haliaetus* and Sparrowhawk *Accipiter nisus*), passerines (Carrion crow *Corvus corone*, Chaffinch *Fringilla coelebs*, Fieldfare *Turdus pilaris* and Starling *Sternus vulgaris*) and feral pigeon *Columba livia.* Further details are provided in ES Appendix 13.2 (Document Reference: 3.3.13).

64. No site-specific ornithology surveys were carried out for the offshore cable corridor. This is normal practice for OWF baseline surveys (e.g. Natural England (2022a) guidance on baseline data collection for OWFs advises that the survey area covers the whole of an area within which a planned array may be built plus a buffer (usually 4km) around this area). In relation to offshore ornithology receptors, the effects of works within the offshore cable corridor are very small and short-term (limited to disturbance over small areas during cable laying and maintenance works). Therefore, data collection on the numbers and distribution of offshore ornithology receptors within the offshore cable corridor is not merited. For North Falls, where the offshore cable corridor passes through the Outer Thames Estuary SPA, designated for red-throated diver in the non-breeding season, the assessment for this component of the development has been carried out with reference to a report on aerial surveys of the Outer Thames Estuary SPA in 2018 commissioned by Natural England (Irwin *et al.*, 2019)

65. Species scoped into the impact assessment are those which were recorded during baseline surveys, and which are considered to be at potential risk either due to their abundance, conservation importance and / or potential sensitivity to the effects of wind farms (for example due to biological characteristics such as tendency to fly at rotor heights, which make them potentially susceptible to collision). The exception to this is potential effects on migratory species flying through the study area during passage periods. As the baseline digital aerial

surveys are not designed to capture the numbers of such species, effects (notably collision risk) were assessed based on desk study data on the numbers of birds estimated to be transiting through the North Falls study area (Wright *et al.*, 2012).

13.5.2 Biologically Relevant Seasons and Population Scales

66. Effects have been assessed in relation to relevant biological seasons, and Biologically Defined Minimum Population Scales (BDMPS) as defined by Furness (2015). These are given in Table 13.10.

67. 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 a species' range may still be on spring migration when individuals farther south have already commenced breeding). For seabird species for which one or more breeding colonies are considered to have connectivity with the North Falls offshore project, the full breeding period has been applied in the attribution of potential impacts to relevant populations (for example lesser black-backed gull and gannet respectively from the Alde Ore Estuary SPA and Flamborough and Filey Coast SPA). The potential for breeding season connectivity is determined on the basis of colonies being within the mean maximum plus 1 standard deviation (SD) of the breeding season foraging range (Woodward *et al.*, 2019). For the species with breeding season connectivity, the non-breeding periods that have been applied are treated in such a way as to be mutually exclusive of the full breeding period (e.g., with reference to Table 13.10, for gannet the baseline survey data from September are assigned to the breeding period but not to the autumn passage period). Where there are no breeding colonies within foraging range, and a species was absent or present in very low numbers in the breeding season, it was considered appropriate to define breeding as the migration-free breeding period (see Table 13.10), 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.

68. Furness (2015) gives regional BDMPS populations for the non-breeding season only. Recent interim guidance from Natural England and Natural Resources Wales has also provided breeding season reference populations for the BDMPS areas defined in Furness (2015), for use in EIA assessments. These are based on the data in the Appendix A to Furness (2015). This guidance was received in March 2024, after the underpinning calculations for the North Falls assessment had been completed, and is not incorporated here. The breeding season BDMPS population estimates for each species scoped into the North Falls assessment are given in the notes to Table 13.10. Notes are included throughout the assessment in relation to any change in a predicted effect that would result from application of the new breeding season reference populations. In all cases predicted increases in baseline mortality from displacement and / or collision would be less or the same as than those given. In no case would the new BDMPS breeding populations make a difference to the outcome of the assessment as presented here.

13.5.3 Abundance of key species within the North Falls array area and relevant buffers

69. Monthly and seasonal population estimates for seabird species within the North Falls array area and relevant buffers (2km; 4km; and 12km for red-throated divers) are given in ES Appendix 13.2 (Document Reference: 3.3.13).

13.5.4 Demographic data for key species

70. Demographic data for species scoped in for assessment for one or more potential effects are provided in Table 13.11. The demographic data are from Horswill and Robinson (2015) for all species except great black-backed gull which is taken from Royal HaskoningDHV (2016).

71. Information on seasonal population age structure is not available from baseline survey data, so an average baseline mortality rate (BMR) for all age classes was calculated for each species screened in for assessment. Average mortality rates are used in the assessment as it is assumed that effects (e.g. collision with WTG blades) act equally on all age classes in a population. The values in Table 13 11 are taken from Royal HaskoningDHV (2023a). These were calculated using empirical information on the survival rates for each age class (as per Table 13.11) and their relative proportions in the population. Each age class survival rate was multiplied by its proportion in the population and the total for all ages summed to give the average survival rate for all ages. Subtracting this value from 1 gives the average mortality rate.

72. Recent interim guidance from Natural England and Natural Resources Wales has provided updated average mortality rates for seabirds for use in EIA assessments. This guidance was received in March 2024, after the underpinning calculations for the North Falls assessment had been completed, and is not incorporated here. The updated average mortality values for each species are given in the notes to Table 13 11 for comparison with the values used in the assessment. These small changes in background mortality would make no difference to the assessment conclusions presented within the ES.

Table 13.10 Biological seasons and non-breeding period BDMPS (Furness 2015) for seabird species at North Falls

Species	Season[1,2]						BDMPS area (non-breeding period[3])
	Breeding[4]	Migration-free breeding	Migration – Autumn[2]	Winter[2]	Migration – Spring[2]	Non-breeding[2]	
Arctic Skua	May-Jul	Jun-Jul	Aug-Oct	-	Apr-May	-	UK North Sea and Channel
Black-headed gull	-	Apr-Jul	-	-	-	Aug-Mar	-
Common gull	-	May-Jul	-	-	-	Aug-Apr	-
Common tern	May-Aug	Jun-mid Jul	Late Jul – early Sept	-	Apr-May	-	UK North Sea and Channel
Cormorant	Apr-Aug	-	-	-	-	Sep-Mar	SW North Sea and Channel
Fulmar	Jan-Aug	Apr-Aug	Sep-Oct	Nov	Dec-Mar	-	UK North Sea
Gannet	Mar-Sep	Apr-Aug	Sep-Nov (456,298)	-	Dec-Mar (248,385)	-	UK North Sea and Channel
Great black-backed gull	Late Mar-Aug	May-Jul	-	-	-	Sep-Mar (91,399)	UK North Seas
Great skua	May-Aug	May-Jul	Aug-Oct	Nov-Feb	Mar-Apr	-	North Sea and Channel
Guillemot	Mar-Jul	Mar-Jun	-	-	-	Aug-Feb (1,617,306)	North Sea and Channel
Herring gull	Mar-Aug	May-Jul	-	-	-	Sep-Feb (466,511)	North Sea and Channel
Kittiwake	Mar-Aug	May-Jul	Aug-Dec (829,937)	-	Jan-Apr (627,816)	-	UK North Sea
Lesser black-backed gull	Apr-Aug	May-Jul	Aug-Oct (209,007)	Nov-Feb (39,314)	Mar-Apr (197,483)	-	North Sea and Channel

Species	Season[1,2]						BDMPS area (non-breeding period[3])
	Breeding[4]	Migration-free breeding	Migration – Autumn[2]	Winter[2]	Migration – Spring[2]	Non-breeding[2]	
Little gull	Apr-Jul	May-Jul	-	-	-	Aug-Apr	-
Puffin	Apr-early Aug	May-Jun	-	-	-	Mid Aug-Mar (231,957)	North Sea and Channel
Razorbill	Apr-Jul	Apr-Jun	Aug-Oct (591,874)	Nov-Dec (218,622)	Jan-Mar (591,874)	-	North Sea and Channel
Red-throated diver	Mar-Aug	May-Aug	Sep-Nov (13,277)	Dec-Jan (10,177)	Feb-Apr (13,277)	-	UK North Sea (migration), SW North Sea (winter)
Sandwich tern	Apr-Aug	Jun	Jul-Sept	-	Mar-May	-	North Sea and Channel

1. Seasons are as defined by Furness (2015) except for black-headed gull, common gull and little gull which are based on Cramp and Simmons (1983).
2. Seasonal BDMPS population sizes from Furness (2015) are included for species scoped into the assessment for collision risk and / or displacement
3. The relevant BDMPS region(s) for the non-breeding season(s), Furness (2015).
4. Breeding season BDMPS population sizes from Natural England and Natural Resources Wales interim guidance provided to North Falls in March 2024 are as follows, for the species scoped into the assessment: gannet (UK North Sea and Channel) 400,326, great black-backed gull (UK North Sea) 28,119, guillemot (UK North Sea and Channel) 2,045,078, kittiwake (UK North Sea) 839,456, lesser black-backed gull (UK North Sea and Channel) 51,233, and razorbill (158,031) (all estimates of the numbers of individual birds of all age classes). These BDMPS populations have not been referred to in this chapter as the underpinning calculations for the assessment had been completed prior to receiving them

Table 13.11 Age-specific demographic rates and average mortality (all age classes) for key seabird species at North Falls (screened in for assessment for one or more effects)

Species	Parameter*	Age class						Productivity	Average mortality
		0-1	1-2	2-3	3-4	4-5	Adult		
Gannet	Survival	0.424	0.829	0.891	0.895	0.895	0.919	0.700	0.187
	Proportion	0.191	0.081	0.067	0.060	0.054	0.547		
Great black-backed gull	Survival	0.798	0.93	0.93	0.93	0.93	0.815	1.139	0.093
	Proportion	0.178	0.142	0.132	0.123	0.114	0.312		
Guillemot	Survival	0.560	0.792	0.917	0.939	0.939	0.939	0.672	0.143
	Proportion	0.173	0.097	0.077	0.071	0.066	0.516		
Kittiwake	Survival	0.790	0.854	0.854	0.854	-	0.854	0.690	0.157
	Proportion	0.168	0.133	0.114	0.097	-	0.488		
Lesser black-backed gull	Survival	0.820	0.885	0.885	0.885	0.885	0.885	0.530	0.125
	Proportion	0.133	0.109	0.096	0.085	0.075	0.501		
Razorbill	Survival	0.630	0.630	0.895	0.895	-	0.895	0.570	0.178
	Proportion	0.17	0.107	0.067	0.06	-	0.596		
Red-throated diver	Survival	0.600	0.620	-	-	-	0.840	0.571	0.233
	Proportion	0.196	0.118	-	-	-	0.686		

* Demographic data from Horswill and Robinson (2015) except for great black-backed gull, from Royal HaskoningDHV (2016); age-class proportions and average mortality calculated as per para 71.

Note that average mortality values have recently been updated by Natural England and Natural Resources Wales in interim guidance provided to North Falls in March 2024. These have not been incorporated here as the underpinning calculations for the assessment had been completed prior to receiving them. The updated values are as follows: gannet 0.1866, great black-backed gull 0.0969, guillemot 0.1405, kittiwake 0.1577, lesser black-backed gull 0.1237, razorbill 0.1302, and red throated diver, 0.2277).

73. Demographic data are used as a reference where a quantitative assessment has been carried out (for collision risk and displacement), to estimate the potential increase in the mortality of ornithological receptors due to the effect. The predicted mortality from an effect can be expressed as a change in the population mortality rate. This has been calculated as a percentage change as follows: (Effect Mortality Rate (EMR) – BMR) / BMR x 100, where EMR is the population mortality rate including additional mortality from a given effect, and BMR is the baseline (average) mortality rate. EMR is calculated as ((BMR x P) + EM) ÷ P, where EM is the estimated mortality from the effect and P is the reference population size. The percentage change in population mortality from an effect resolves to (EM / baseline annual mortality (BM)) x 100, where BM is the baseline (average) annual mortality of the reference population (BMR x P). It is recognised that the use of average baseline mortalities assumes that the age structure of the population that is subject to an impact (e.g. birds in flight and at collision risk within the turbine array of an OWF) is the same as the wider population of the same species.

13.5.5 Future trends in baseline conditions

74. In the event that North Falls is not developed, an assessment of the future conditions for offshore ornithology has been carried out and is described within this section.

75. There are a number of pressures acting on offshore ornithology receptors in the North Sea and beyond which are considered here as context to the assessment. These include changes in prey availability, climate change, bycatch in fisheries, invasive alien species, and pollution, as well as cumulative disturbance, displacement, and collision risk from OWFs (Dias *et al.*, 2019; Mitchell *et al.*, 2020; Royal HaskoningDHV, 2019a) and the recent outbreak of Highly Pathogenic Avian Influenza (HPAI) which began in the summer of 2021. It will take some time for the full effects of the disease on longer-term seabird population trends to become evident. Monitoring activities at some seabird colonies were suspended during the 2022 breeding season to reduce risks of spreading HPAI. Seabird surveys were carried out at selected colonies in 2023 for priority species of conservation concern to provide data for comparison with pre-HPAI counts (Tremlett *et al.,* 2024). It is possible some of the longer-term trends described below may be subject to change.

76. Trends in seabird numbers at breeding colonies in the UK are better known, and better understood, than trends in numbers within particular areas of offshore waters (for which monitoring is logistically challenging and which are likely to be subject to greater temporal variation). Breeding numbers are regularly monitored at many UK colonies (JNCC, 2021), and in Britain and Ireland there have been four comprehensive censuses of breeding seabirds in 1969 – 70, 1985 – 88, 1998 – 2002 (Mitchell *et al.*, 2004) and 2015 – 2021 (Burnell *et al.*, 2023), as well as single-species surveys (such as the decadal counts of breeding gannet numbers, Murray *et al.*, 2015). In contrast, while surveys of seabirds (and cetaceans) at sea have been ongoing since 1979 (JNCC 2020), data have not been systematically collected, albeit that the extent of survey data for UK offshore waters has increased substantially in recent years as a consequence of OWF developments. The European Seabirds at Sea (ESAS)

database[3] is incomplete, and few data have been added to this database since 2000, so that current trends in numbers at sea in areas of the North Sea are not so easy to assess. Available data have been used to assess the potential impacts of oil and gas developments offshore (Tasker *et al*., 1984), in the designation of offshore SPA for birds (JNCC, 2020) and to predict densities across large spatial scales (e.g. Waggitt *et al*., 2019) but such predictions tend to be at very broad-scale (almost by definition), may be subject to various limitations and do not provide information on long-term trends.

77. Breeding numbers of many seabird species in Britain and Ireland are declining, especially in the northern North Sea (Foster and Marrs, 2012; Macdonald *et al.*, 2015; JNCC, 2021). The most recent census (Burnell *et al.*, 2023) has shown that eleven species, including kittiwake and great black-backed gull, have declined by over 10% since the previous seabird census of these countries (1998 – 2002) (Mitchell *et al.*, 2004). On the other hand, populations of five species, including gannet and razorbill, increased by over 10%. For other species, including lesser black-backed gull and herring gull, trends are unclear due to changes in methodology or improved survey coverage so that the latest Seabirds Counts population estimates cannot be compared with previous estimates (Burnell *et al.,* 2023).

78. Key drivers of seabird population size in western Europe are climate change (Daunt *et al.*, 2017; Daunt and Mitchell, 2013; Mitchell *et al.*, 2020; Sandvik *et al.*, 2012; Frederiksen *et al.*, 2004, 2013; Burthe *et al.*, 2014; Macdonald *et al.*, 2015; Furness, 2016; JNCC, 2021, Burnell *et al.*, 2023, Searle *et al.,* 2022), 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), predation by native and invasive predators (Burnell *et al.*, 2023), disease, and loss of nesting habitat also impact on seabird populations but are generally much less important and often act at more local scales (Ratcliffe, 2004; Votier *et al.*, 2005, 2008; JNCC, 2021).

79. 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). Climate change has the potential to impact seabird populations in two main ways; indirectly through changes in prey abundance / availability, and directly through impacts such as mortality or reduced breeding success due to extreme weather events. Whilst effects may not extend to all areas (e.g. some areas where prey recruitment may be less affected (ClimeFish, 2019; Frederiksen *et al.*, 2005), climate models generally predict increased incidences of warming and extreme weather in the future (Palmer *et al.*, 2018). Indeed, such patterns are already occurring (Intergovernmental Panel on Climate Change (IPCC), 2021). Ocean conditions are projected to continue diverging from a pre-industrial state, increasing risk of regional extirpations and global extinctions of marine species (IPCC, 2022). It is therefore highly likely that breeding numbers of most of our seabird species will continue to decline

[3] https://www.ices.dk/data/data-portals/Pages/European-Seabirds-at-sea.aspx

under a scenario with continuing climate change due to increasing levels of greenhouse gases.

80. Fisheries management is also likely to influence future numbers in seabird populations. The Common Fisheries Policy (CFP) Landings Obligation ('discard ban'), progressively introduced from 2015 to 2019 and remaining in place following Brexit, will continue to further reduce 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.

81. 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 'forage-fish' such as sandeels (Frederiksen *et al.*, 2007; Macdonald *et al.*, 2015). Lindegren *et al.* (2018) concluded that sandeel stocks in the North Sea, the most important prey fish stock for North Sea seabirds during the breeding season (Furness and Tasker, 2000), have been depleted by high levels of fishing effort and also affected by climate change.

82. Clear links between kittiwake breeding success and reduced sandeel availability due to fishing activities have been demonstrated (Carroll *et al.*, 2017; Daunt *et al.*, 2008; Frederiksen *et al.*, 2004; Furness and Tasker, 2000; Greenstreet *et al.*, 2010; Hayhow *et al.*, 2017; Lindegren *et al.*, 2018; Wright *et al.*, 2018, Searle *et al.*, 2023). It has been identified that three traits that make kittiwake particularly sensitive to sandeel depletion by fisheries activity are the species' limited ability to dive so that it relies on prey availability at the sea surface, lack of spare time in its daily budget, and its restricted ability to switch diet (Furness and Tasker, 2000).

83. A closure of sandeel fisheries in English and Scottish North Sea waters came into force in March 2024 (Scottish Government 2024, Defra 2024a). These closures recognise the importance of sandeels in the wider marine ecosystem and benefits to the resilience of other marine life including seabirds. At the time of writing, a challenge from the EU, in relation to the EU-UK Trade and Cooperation Agreement is underway.

84. Gannet numbers may continue to increase for some years, but evidence suggests that this increase is already slowing (Murray *et al.*, 2015), and numbers may peak not too far into the future. While the Landings Obligation reduces 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). The flexible behaviour and diet of gannets probably reduces their vulnerability to changes in fishery practices or to climate change impacts on fish communities (Garthe *et al.*, 2012).

85. 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).

86. 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 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.

87. 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 (Mercker *et al.,* 2021), and the Landings Obligation is anticipated to 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.

88. 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.

89. For offshore ornithology, the ecological impact assessment is therefore carried out in a context of declining baseline populations of a number of receptor species. In this context, the emergence of HPAI in UK breeding seabird populations in 2021 / 2022 is a key concern, particularly with outbreaks affecting two species for which the UK hosts more than 50% of the global breeding populations: gannet and great skua. While arrangements are being put in place by the UK Government, SNCBs and non-governmental organisations (NGOs) to monitor the situation and evaluate the consequences, this is expected to take several years and in the interim it will be necessary to work with imperfect knowledge on the long-term effects of HPAI (Natural England, 2022d).

90. However, there are indications that some species have suffered very high levels of adult mortality as well as declines in breeding productivity (Royal HaskoningDHV, 2023; Tremlett *et al*., 2024; BTO, 2024; NatureScot, 2023). As most seabird species are long-lived with low annual reproductive rates, with

immatures taking several years to reach breeding age, recovery of populations that have experienced significant mortality may take some time (NatureScot 2023). At the time of writing, the latest update from Defra (2024b) is that the risk of HPAI in wild birds in Great Britain is assessed as low (i.e. the event is rare but does occur).

91. Baseline surveys for North Falls were completed before the HPAI outbreak, and most reference populations used for the offshore ornithology EIA (e.g. BDMPS populations, Furness 2015) are also based on pre-HPAI data. Therefore, the HPAI outbreak has no implications in terms of the validity of relating baseline survey data to reference populations as undertaken for this assessment.

92. Where a receptor species is declining, the assessment takes into account whether a given impact is likely to exacerbate a decline in the relevant reference population, and prevent a receptor species from recovery should environmental conditions become more favourable. Climate change has been identified as the strongest long-term influence on future seabird population trends. In this context it is noted that a key component of global strategies to reduce climate change is the development of low-carbon renewable energy developments such as OWFs.

13.6 Assessment of significance

93. This section includes the Project-alone assessment of likely significant effects of North Falls on offshore ornithology receptors during the construction, maintenance, operation and decommissioning phases. The worst-case scenarios listed in Table 13.1 for each impact are assessed.

13.6.1 Likely significant effects during construction

13.6.1.1 Effect 1: Direct disturbance and displacement

94. The construction of North Falls has the potential to affect offshore bird populations through disturbance due to 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.

95. Offshore construction of North Falls is expected to take place over approximately two years (Table 13.1).

96. The construction phase would require the mobilisation of vessels, helicopters and equipment and the installation of foundations, turbines, cables, platforms and associated infrastructure. These activities have the potential to disturb and displace birds from within and around the offshore project area. Causes of potential disturbance would comprise the presence of construction vessels and associated human activity, noise and vibration from construction activities and lighting associated with construction sites. 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 construction period.

97. As the focus of construction activity will change throughout the construction period, any impacts resulting from disturbance and displacement from construction activities in a given location would 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.

98. Bird species differ in their susceptibility to anthropogenic disturbance and in their responses to noise and visual disturbance stimuli. The principal source of noise during construction in the offshore project area would be subsea noise from piling works for the installation of foundations. While assessed for marine mammals and fish, subsea noise is not considered a risk factor for diving birds.

99. Seabirds and other diving bird species will spend most of their time above or on the water surface, where hearing will detect sound propagated through the air. Measurements of the underwater hearing capabilities of seabirds are limited and contain quite large sources of error (Johansen *et al.*, 2016). Some diving birds possess specialised anatomical traits that may be associated with improved underwater hearing (Crowell *et al.*, 2015; Johansen *et al.*, 2016), which may render them more sensitive to potential effects resulting from underwater noise. That said, such anatomical adaptations have been shown to include protection against the large pressure changes that may occur while diving, which may actually protect the ear from damage during acoustic overexposure (Dooling and Therrien, 2012).

100. Above water noise disturbance from construction activities is not considered in isolation as a risk factor for birds; but rather, combined with the presence of vessels, man-made structures, and human activity, part of the overall disturbance stimulus that causes birds to avoid boats and other structures – as discussed below.

101. Lighting of construction sites, vessels and other structures at night may potentially be a source of attraction (phototaxis), as opposed to displacement, for birds; however, the areas affected would be very small, and restricted to offshore construction areas which are active at a given time. Phototaxis can be a serious hazard for fledglings of some seabird species (documented mainly for petrels (Procellariiformes) Rodriguez *et al.*, 2017) but tends to occur over short distances (hundreds of metres) in response to bright white light close to breeding colonies. It is not seen over large distances or in older (adult and immature) seabirds (Furness, 2018), or documented for the species scoped in for assessment at North Falls. Construction sites associated with the offshore project area would be far enough removed from any seabird breeding colonies as to render this risk negligible. Phototaxis of nocturnal migrating birds can be a problem, especially in autumn during conditions of poor visibility, but is generally seen where birds are exposed to intense white lighting such as from lighthouses; light from construction sites is likely to be one or two orders of magnitude less powerful than that from lighthouses (Furness, 2018).

102. Considering variation between species in response to anthropogenic disturbance, 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 OWF (GGOW, 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 observed to avoid shipping by

several kilometres (Mitschke *et al.*, 2001 from Exo *et al.*, 2003; Garthe and Hüppop, 2004; Bellebaum *et al.*, 2006; Schwemmer *et al.*, 2011; Mendel *et al.*, 2019).

103. There are a number of reviews relevant to bird disturbance and displacement from areas of sea in response to construction activities associated with an OWF. Garthe and Hüppop (2004) developed a scoring system for such disturbance factors which they applied to seabird species in German sectors of the North Sea. This was refined by Furness and Wade (2012) and Furness *et al.* (2013) with a focus on seabirds using Scottish offshore waters. The approach uses information in the scientific and 'grey' literature, as well as expert opinion to identify disturbance ratings for individual 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. In this context it is noted that the minimum safe altitude for helicopters operating offshore is 1,000ft above the highest known obstacle within 5nm. It is considered that at these altitudes that any disturbance caused by the visual presence or noise of helicopters will be minimal and will not result in significant disturbance of birds in the offshore environment. Helicopters servicing the North Falls project will be taking off and landing on a helipad on the OSPs / OCP within the array area, but this will be a relatively infrequent occurrence (maximum of 100 round trips per annum, Table 13.1).

104. The reviews referred to above and other relevant literature were used to inform a screening exercise to identify those species most likely to be at risk of construction disturbance, to focus the assessment of disturbance and displacement (Table 13.12). Any species recorded with a low sensitivity to displacement or recorded only in very small numbers within the study area (see ES Appendix 13.2 (Document Reference: 3.3.13)) were screened out of further assessment.

Table 13.12 Screening for sensitivity to construction disturbance and displacement.

Species	Sensitivity to Disturbance and Displacement[1]	Screening Result (IN or OUT)	Rationale
Arctic skua	Low	Out	Low susceptibility to disturbance)
Black-headed gull	Low	Out	Low susceptibility to disturbance
Common gull	Low	Out	Low susceptibility to disturbance
Common tern	Low	Out	Low susceptibility to disturbance and recorded in low numbers
Cormorant	Low	Out	Low susceptibility to disturbance and recorded in low numbers
Fulmar	Low	Out	Low susceptibility to disturbance
Gannet	Medium	In	Low susceptibility to disturbance from vessel traffic, but shows high rate of macroavoidance of constructed OWFs, so displacement may occur once WTGs begin to be installed on foundations.

Species	Sensitivity to Disturbance and Displacement[1]	Screening Result (IN or OUT)	Rationale
Great black-backed gull	Low	Out	Low susceptibility to disturbance
Great skua	Low	Out	Low susceptibility to disturbance
Guillemot	Medium	In	Potentially susceptible to disturbance and present in the North Falls array area
Herring gull	Low	Out	Low susceptibility to disturbance
Kittiwake	Low	Out	Low susceptibility to disturbance
Lesser black-backed gull	Low	Out	Low susceptibility to disturbance
Little gull	Low	Out	Low susceptibility to disturbance
Puffin	Medium	Out	Potentially susceptible to disturbance but recorded infrequently and in low numbers at the North Falls array area
Razorbill	Medium	In	Potentially susceptible to disturbance and present in the North Falls array area
Red-throated diver	High	In	High susceptibility to disturbance and displacement and present in the North Falls array area and offshore cable corridor (the latter overlapping with the Outer Thames Estuary SPA for which red-throated diver is a qualifying species)
Sandwich tern	Low	Out	Low susceptibility to disturbance and recorded in low numbers
1. With reference to Garthe and Hüppop, 2004; Furness and Wade, 2012; Furness *et al.*, 2013; Wade *et al.*, 2016, Dierschke *et al.,* 2016.			

105. The species screened in for assessment were gannet, guillemot, razorbill and red-throated diver. For red-throated diver the assessment considered likely significant effects within the array area and offshore cable corridor, as the latter area passes close to and (for 19km of its total length of 57km) through the Outer Thames Estuary SPA which is designated for non-breeding red-throated diver, and red-throated divers show strong avoidance reactions to vessels (more detail on this below).

106. Because of lower sensitivity to shipping activity, the short duration (six months) and small spatial extent (maximum of two cable-laying vessels at any one time) of construction activities, gannet, guillemot and razorbill were not screened in for assessment in relation to the offshore cable corridor. The assessment for gannet, guillemot and razorbill focuses on the array area, where construction activity will be more intense and take place over a longer period of time (two years) (see Table 13.1 for details of construction activities).

107. In their response to the outline method statement for the North Falls EIA (see ES Appendix 13.1 (Document Reference: 3.3.12)), Natural England commented '*The construction phase presents a range of potential drivers that may cause displacement of seabirds. This includes vessel movement and construction activities (which may be both spatially and temporally limited), however the physical presence of the constructed turbines is also likely to cause a*

displacement response. As the construction phase progresses, more turbines are built and the spatial scale increases, until a point when the entire array is constructed, yet not operational, and may present the same displacement stimulus as an operational farm. Therefore, it should not be asserted that displacement will only occur where vessels and construction activities are present; instead we consider that displacement is likely to occur within and around the constructed array areas (due to the presence of turbines) and where construction activities are ongoing. This will represent an increasing spatial impact as construction progresses. For assessment of construction phase displacement, we advise North Falls consider the pragmatic method NE advised for PEIR at Hornsea 4 of calculating operational displacement per species and reducing by 50% during the construction period (to broadly reflect reduced spatial and temporal scale) across the range of displacement mortality advised by Natural England for a particular species. We recommend this approach is taken for construction displacement assessments for red-throated diver, gannet, and auks'.

108. Thus, the assessment of construction disturbance and displacement from the array area assumes that the displacement effects in any one year will be 50% of those predicted during a single year of the operational period (Section 13.6.2.1 below). It is considered by the Applicant, however that this is likely to over-estimate the magnitude of construction disturbance and displacement as, until WTGs are installed on to foundations in the latter part of the construction period, there will be few tall structures above the sea surface (OSPs and OCP, maximum height 116m above MHWS with cranes, are installed early on) from which birds might be displaced. Before the installation of WTGs begins, it is the case that construction disturbance and displacement are likely to be confined to a limited number of areas of activity within the array area at any given time, for example in areas where piling is ongoing and WTG foundations are being installed, and array cabling is being laid.

13.6.1.1.1 Gannet

109. The assessment of operational disturbance and displacement of gannet from North Falls is included in Section 13.6.2.1.1 below. Displacement mortality is predicted by season and year-round, and expressed as a percentage increase in the mortality rate of the appropriate seasonal reference population (BDMPS) and, for the full annual period, in relation to both the largest seasonal BDMPS and the biogeographic population with connectivity to UK waters (Furness, 2015).

110. It is assumed that a maximum of 1% of gannets subject to displacement suffer mortality as a result. Gannet has high habitat flexibility (Furness and Wade, 2013) and an extensive foraging range (Woodward *et al.*, 2019), suggesting that displaced birds will readily find alternative foraging areas. Thus, in reality, there may be no mortality costs for gannets associated with displacement from OWFs during construction or operation.

111. In each season (autumn migration, breeding and spring migration) and year-round the predicted mortality of gannets due to displacement from the OWF during operation is equivalent to a 0.01% or less increase in the mortality rate of the relevant reference population (Section 13.6.2.1.1). Such predicted

magnitudes of increase in mortality would not materially alter the background mortality of any seasonal BDMPS or the biogeographic population of gannet with connectivity to UK waters and would be undetectable in the context of natural variation.

112. The same conclusion would apply if construction disturbance and displacement is estimated as 50% of the magnitude of operational displacement (with the maximum increase in baseline mortality rate of the relevant BDMPS and biogeographic populations estimated as 0.005%). The impact magnitude is therefore defined as negligible. As gannet is of medium sensitivity to displacement and disturbance, the effect significance is minor adverse, which is not significant in EIA terms.

13.6.1.1.2 Guillemot

113. The assessment of operational disturbance and displacement of guillemot from North Falls is included in Section 13.6.2.1.1 below. Displacement mortality is predicted by season (breeding and non-breeding) and year-round and expressed as a percentage increase in the mortality rate of the appropriate seasonal reference population (BDMPS) and, for year-round, both the largest seasonal BDMPS and biogeographic population with connectivity to UK waters (Furness 2015).

114. For guillemot, Natural England has advised that a range of displacement rates of 30 – 70%, and mortality rates of 1-10% for displaced birds, should be considered, and that they agree the mortality is likely to be at the low end of the range. Recent reviews of evidence for guillemot displacement have recommended that 50% displacement and 1% mortality of displaced birds is an appropriate precautionary assumption (MacArthur Green, 2019b; APEM, 2022b; see Section 13.6.2.1.1). For the Hornsea Project Four (HP4) HRA (DESNZ, 2023c), the SoS is understood to have based the consent decision on displacement and mortality rates of 70% and 2% for guillemot. As discussed in Section 13.6.2.1.1 below, 10% mortality of displaced birds is considered to be highly unlikely.

115. In all seasons, and annually, at 50% displacement and 1% mortality, the mean predicted mortality of guillemots due to displacement from the OWF during operation is equivalent to a 0.01% (95% Confidence Limits (CLs) 0 – 0.04%) or less increase in the mortality rate of the relevant reference population (Table 13.13, Section 13.6.2.1.1). At 70% displacement and 2% mortality, displacement mortality would represent a 0.04% (95% CLs 0.01 – 0.10%) or less increase in the baseline mortality rate of the relevant reference population. In fact, all predictions for annual operational displacement mortality represent increases of less than 1% in the baseline mortality rate of the UK North Sea and Channel BDMPS and biogeographic populations (Table 13.13, Section 13.6.2.1.1).

Table 13.13 Seasonal and year-round predicted operational displacement mortality for guillemot (summary from Section 13.6.2.1.1 below)

Season and Scenario[1]	No. of predicted mortalities from operational displacement[2]			% Increase in baseline mortality (with reference to 'average' baseline rate of 0.143)			
	Mean	Lower Confidence Limit (LCL)	Upper Confidence Limit (UCL)	Reference population	Mean	LCL	UCL
Non-breeding							
30% / 1%	16	3	44	1,617,306 individuals, UK North Sea and Channel (Furness 2015)	0.01%	0.00%	0.02%
70% / 10%	376	61	1,027		0.16%	0.03%	0.44%
50% / 1%	27	4	73		0.01%	0.00%	0.03%
70% / 2%	75	12	205		0.03%	0.01%	0.09%
Breeding							
30% / 1%	3	1	7	695,442 individuals (43% of the non-breeding BDMPS, see Section 13.6.2.1.1)[3]	0.00%	0.00%	0.01%
70% / 10%	61	17	164		0.06%	0.02%	0.17%
50% / 1%	4	1	12		0.00%	0.00%	0.01%
70% / 2%	12	3	33		0.01%	0.00%	0.03%
Year-round							
30% / 1%	19	3	51	1,617,306 individuals, largest seasonal BDMPS (as above)[3]	0.01%	0.00%	0.02%
70% / 10%	436	78	1,191		0.19%	0.03%	0.52%
50% / 1%	31	6	85		0.01%	0.00%	0.04%
70% / 2%	87	16	238		0.04%	0.01%	0.10%
30% / 1%	19	3	51	4,125,000 individuals, Biogeographic population with connectivity to UK (Furness 2015)	0.00%	0.00%	0.01%
70% / 10%	436	78	1,191		0.07%	0.01%	0.20%
50% / 1%	31	6	85		0.01%	0.00%	0.01%
70% / 2%	87	16	238		0.01%	0.00%	0.04%

1. The scenarios are the % of birds displaced and the % of displaced birds assumed to suffer mortality, e.g. 30% / 1% is 30% of birds displaced and 1% of displaced birds suffering mortality. 2. Seasonal and year-round totals are rounded to the nearest integer, so the year-round totals may not exactly match the sum of seasonal values. 3. Under the latest guidance from Natural England and NRW on EIA reference populations (see para 68), the breeding season BDMPS is 2,045,078 individuals for the UK North Sea and Channel, and this is also the largest seasonal BDMPS; applying these, the percentage increases in baseline mortality during the breeding season, and year round, would be even smaller than those given in the table.

116. Such predicted magnitudes of increase in mortality would not materially alter the background mortality of any seasonal BDMPS or the biogeographic population of guillemot with connectivity to UK waters, and would be undetectable in the context of natural variation.

117. The same conclusion would apply if construction disturbance and displacement is estimated as 50% of the magnitude of operational displacement (for which the predicted increases in baseline mortality rate would be half the value given for each scenario in Table 13.13).

118. The impact magnitude of disturbance and displacement during construction is therefore defined as negligible. As guillemot is of medium sensitivity to

displacement and disturbance, the effect significance is minor adverse, which is not significant in EIA terms.

13.6.1.1.3 Razorbill

119. The assessment of operational disturbance and displacement of razorbill from North Falls is included in Section 13.6.2.1.1 below. Displacement mortality is predicted by season and year-round and expressed as a percentage increase in the mortality rate of the appropriate seasonal reference population (BDMPS) and, for year-round, both the largest seasonal BDMPS and the biogeographic population with connectivity to UK waters (Furness, 2015).

120. For razorbill, Natural England has advised that a range of displacement rates of 30-70% mortality rates of 1-10% should be considered for displaced birds, and also that they agree that the mortality is likely to be at the low end of the range. Recent reviews of evidence for razorbill displacement have recommended that 50% displacement and 1% mortality of displaced birds is an appropriate precautionary assumption (MacArthur Green, 2019b; APEM, 2022b; see Section 13.6.2.1.1). For the Hornsea Project Four (HP4) HRA (DESNZ, 2023c), the SoS is understood to have based the consent decision on displacement and mortality rates of 70% and 2% for razorbill. As discussed in Section 13.6.2.1.1, 10% mortality of displaced birds is considered to be highly unlikely.

121. At 50% displacement and 1% mortality of displaced birds, the predicted mortality of razorbills due to displacement from the OWF during operation is equivalent to a mean of 0.02% (95% CLs 0 – 0.04%) or less increase in the mortality rate of the relevant reference population (Table 13.14, Section 13.6.2.1.1). At 70% displacement and 2% mortality displacement mortality would represent a 0.06% (95% CLs 0 – 0.11%) or less increase in the mortality rate of the relevant reference population. In fact, all predictions for annual operational displacement mortality represent increases of less than 1% in the baseline mortality rate of the UK North Sea and Channel BDMPS and biogeographic populations (Table 13.14, Section 13.6.2.1.1).

Table 13.14 Seasonal and year-round predicted operational displacement mortality for razorbill (summary from Section 13.6.2.1.1 below)

Season and Scenario[1]	No. of predicted mortalities from operational displacement[2]			% Increase in baseline mortality (with reference to 'average' baseline rate of 0.178)			
	Mean	LCL	UCL	Reference population	Mean	LCL	UCL
Autumn Migration							
30% / 1%	1	0	2	591,874 individuals, UK North Sea and Channel (Furness 2015)	0.00%	0.00%	0.00%
70% / 10%	17	1	42		0.02%	0.00%	0.04%
50% / 1%	1	0	3		0.00%	0.00%	0.00%
70% / 2%	3	0	8		0.00%	0.00%	0.01%
Winter							
30% / 1%	5	4	8	218,622 individuals, UK North Sea and	0.01%	0.01%	0.02%
70% / 10%	125	87	178		0.32%	0.22%	0.46%

Season and Scenario[1]	No. of predicted mortalities from operational displacement[2]			% Increase in baseline mortality (with reference to 'average' baseline rate of 0.178)			
	Mean	LCL	UCL	Reference population	Mean	LCL	UCL
50% / 1%	9	6	13	Channel (Furness, 2015)	0.02%	0.02%	0.03%
70% / 2%	25	17	36		0.06%	0.04%	0.09%
Spring Migration							
30% / 1%	5	1	15	591,874 individuals, UK North Sea and Channel (Furness 2015)	0.00%	0.00%	0.01%
70% / 10%	122	29	343		0.12%	0.03%	0.33%
50% / 1%	9	2	25		0.01%	0.00%	0.02%
70% / 2%	24	6	69		0.02%	0.01%	0.07%
Breeding							
30% / 1%	0	0	1	94,007 individuals, UK North Sea and Channel (Furness 2015)[3]	0.00%	0.00%	0.01%
70% / 10%	7	0	23		0.04%	0.00%	0.14%
50% / 1%	1	0	2		0.00%	0.00%	0.01%
70% / 2%	1	0	5		0.01%	0.00%	0.03%
Year-round							
30% / 1%	12	5	25	591,874 individuals, largest seasonal BDMPS (as above)	0.01%	0.00%	0.02%
70% / 10%	271	116	587		0.03%	0.01%	0.06%
50% / 1%	19	8	42		0.02%	0.01%	0.04%
70% / 2%	54	23	117		0.05%	0.02%	0.11%
30% / 1%	12	5	25	1,707,000 individuals, Biogeographic population with connectivity to UK (Furness 2015)	0.00%	0.00%	0.01%
70% / 10%	271	116	587		0.01%	0.00%	0.02%
50% / 1%	19	8	42		0.01%	0.00%	0.01%
70% / 2%	54	23	117		0.02%	0.01%	0.04%

1. The scenarios are the % of birds displaced and the % of displaced birds assumed to suffer mortality, e.g. 30% / 1% is 30% of birds displaced and 1% of displaced birds suffering mortality. 2. Seasonal and year-round totals are rounded to the nearest integer, so the year-round totals may not exactly match the sum of seasonal values. 3. Under the latest guidance from Natural England and NRW on EIA reference populations (see para 68), the breeding season BDMPS is 158,031 individuals for the UK North Sea and Channel; applying this, the percentage increases in baseline mortality during the breeding season would be even smaller than those given in the table.

122. Such predicted magnitudes of increase in mortality would not materially alter the background mortality of any seasonal BDMPS or the biogeographic population of razorbill with connectivity to UK waters, and would be undetectable in the context of natural variation.

123. The same conclusion would apply if construction disturbance and displacement is estimated as 50% of the magnitude of operational displacement (for which the predicted increases in baseline mortality rate would be half the value given for each scenario in Table 13.14).

124. The impact magnitude is defined as negligible. As razorbill is of medium sensitivity to displacement and disturbance, the effect significance is minor adverse which is not significant in EIA terms.

13.6.1.1.4 Red-throated diver

125. Red-throated diver has been identified as being particularly sensitive to human activities in marine areas, including in relation to the disturbance effects of ship and helicopter traffic (Garthe and Hüppop, 2004; Bellebaum *et al.*, 2006; Schwemmer *et al.*, 2011; Furness and Wade, 2012; Furness *et al.*, 2013; Bradbury *et al.*, 2014; Fliessbach *et al.*, 2019; Mendel *et al.*, 2019). A selectivity index derived from aerial surveys in the German North Sea indicated that the numbers of divers (red- and black-throated divers could not be reliably distinguished during the surveys) were significantly lower in shipping lanes than in other areas, although there were insufficient data to estimate flushing distances of divers from approaching vessels (Schwemmer *et al.*, 2011); in this study it was assumed that the responses of red and black-throated divers to disturbance were similar. Fliessbach *et al.* (2019) investigated escape distances of seabirds from vessels in the German and Baltic Seas. They reported distances of 1,374 ± (SD) 416m for individual divers not identified to species and 1,281 ± 424m for flocks of divers not identified to species; 750 ± 437m and 702 ± 348m, respectively, for individuals and flocks of red-throated divers; and 721 ± 616 and 562 ± 450m, respectively, for individuals and flocks of black-throated divers.

126. Observational studies of the responses of marine birds to disturbance in Orkney inshore waters (Jarrrett *et al*. 2018, 2021) found that red-throated and black-throated divers showed similar flushing behaviour from ferries (with respectively 75% (n=88) and 62% (n=21) of birds showing an evasive response within 300m of a passing ferry); for red-throated divers, response rates were 100% within 50m of a ferry, 87% between 50-100m, 60% between 100-200m and 54% within 200-300m.

127. Both Fliessbach *et al.* (2019) and Jarrett *et al.* (2018, 2021) observed that red-throated divers were highly likely to fly in response to vessels whereas black-throated divers were more likely to dive or swim away (in the Orkney study it was suggested these differences may be related to differences in the timing of moult in the two species, which affects flight ability; although also that red-throated divers have a lower wing loading and lower energetic costs of take-off than black-throated divers; Jarret *et al*., 2018, 2021). The Orkney study seems to indicate lesser displacement distances from vessels than those in the German North Sea, although displacement effects may increase with the size and / or speed of vessels.

13.6.1.1.4.1 Array area

128. As for gannet, guillemot and razorbill, the assessment for red-throated diver assumes that displacement and disturbance during construction will be 50% of that predicted during operation, based on advice from Natural England (noting that this is considered to be overly precautionary, as outlined in Section 13.6.1.1 above).

129. The operational assessment of displacement and disturbance for red-throated divers (Section 13.6.2.1.2) considers a mortality rate of 1 – 10% for displaced birds, based on advice from Natural England.

130. There is no direct empirical evidence relating to the effects of displacement on red-throated diver mortality rates. A detailed review of the likely effects of displacement of red-throated divers on mortality during the non-breeding season is included in MacArthur Green (2019c). The annual mortality rate of red-throated divers under baseline conditions is 16% per annum for adults (three years and older), 40% for juveniles (0-1 year) and 38% for immatures (1-2 years) (Horswill and Robinson, 2015; with these rates based on population studies in Sweden and Alaska, published respectively in 2002 and 2014). These rates will include mortality in the breeding and non-breeding seasons due to 'natural' factors such as weather or predation, as well as mortality (if any) from anthropogenic impacts such as disturbance and displacement by vessels. As vessels are mobile and red-throated divers will often fly away from approaching vessels (e.g. Schwemmer *et al.*, 2011, Jarrett *et al.*, 2018) the energy costs of displacement from moving vessels may already be incorporated in the existing estimates of survival. It is considered that the mortality rate of displaced birds is likely to be 1% at most (see Section 13.6.2.1.2 below).

131. In all seasons, and annually, at 100% displacement and 1% mortality, the mean predicted mortality of red-throated divers due to displacement from the OWF during operation is equivalent to a 0.03% (95% CLs 0 – 0.06%) or less increase in the baseline mortality rate of the relevant reference population (Table 13.15, Section 13.6.2.1.2). In fact, all predictions for annual operational displacement mortality represent increases of less than 1% in baseline mortality rate of the relevant BDMPS and the biogeographic population with connectivity to UK waters, even when the mortality amongst displaced birds is assumed to be as high as 10% as opposed to 1%.

132. Such predicted magnitudes of increase in mortality would not materially alter the background mortality rate of any seasonal BDMPS or the biogeographic population of red-throated diver with connectivity to UK waters and would be undetectable in the context of natural variation.

Table 13.15 Seasonal and year-round predicted operational displacement mortality for red-throated diver (summary from Section 13.6.2.1.2 below)

Season and Scenario[1]	No. of predicted mortalities from operational displacement[2]			% Increase in baseline mortality (with reference to 'average' baseline rate of 0.143)			
	Mean	LCL	UCL	Reference population	Mean	LCL	UCL
Autumn migration							
100% / 1%	0	0	0	13,277 individuals, UK North Sea (Furness 2015)	0%	0%	0%
100% / 10%	0	0	0		0%	0%	0%
Winter							
100% / 1%	0	0	0	10,177 individuals, UK south-west	0%	0%	0%
100% / 10%	2	0	4		0.08%	0%	0.19%

Season and Scenario[1]	No. of predicted mortalities from operational displacement[2]			% Increase in baseline mortality (with reference to 'average' baseline rate of 0.143)			
	Mean	LCL	UCL	Reference population	Mean	LCL	UCL
				North Sea (Furness 2015)			
Spring migration							
100% / 1%	1	0	1	13,277 individuals, UK North Sea (Furness 2015)	0.02%	0%	0.05%
100% / 10%	7	1	15		0.21%	0.04%	0.48%
Year round							
100% / 1%	1	0	2	13,277 individuals, UK North Sea (Furness 2015)	0.03%	0%	0.06%
100% / 10%	9	1	19		0.28%	0.04%	0.62%
100% / 1%	1	0	2	27,000 individuals, UK bio-geographic (Furness 2015)	0.01%	0%	0.03%
100% / 10%	9	1	19		0.14%	0.02%	0.31%
1. The scenarios are the % of birds displaced and the % of displaced birds assumed to suffer mortality, e.g. 100% / 1% is 100% of birds displaced and 1% of displaced birds suffering mortality. 2. Seasonal and year-round totals are rounded to the nearest integer, so the year-round totals may not exactly match the sum of seasonal values							

133. This conclusion would hold if construction disturbance and displacement is estimated as 50% of that during operation (for which the predicted increases in baseline mortality rate would be half the value given for each scenario in Table 13.15). The impact magnitude is defined as negligible.

134. In all seasons and year-round, the impact magnitude of construction disturbance on red-throated divers is assessed as negligible. As the species is of high sensitivity to disturbance, the effect significance is minor adverse, which is not significant in EIA terms.

13.6.1.1.4.2 Offshore cable corridor

135. There is potential disturbance and displacement of non-breeding red-throated divers resulting from the vessels installing the offshore export cables, particularly where they pass through the Outer Thames Estuary SPA (Figure 13.1 (Document Reference: 3.2.9)). The offshore cable corridor is 57km long, and 19km of the cable (33.3% of the overall length), passes through the spa. Where it overlaps with the SPA, the offshore cable corridor width is 1km wide, giving a total potential overlap between the offshore cable corridor and the SPA of approximately 19km^2, or 0.48% of the SPA area (although this represents the area of search and the actual cable route itself will be much smaller; see ES Chapter 5 Project Description (Document Reference: 3.1.7)).

136. Cable-laying operations, utilising up to two cable-laying vessels working simultaneously (see Table 13.1), have the potential to displace red-throated divers from an area around each vessel.

137. On a precautionary basis it is assumed that there would be 100% displacement of all birds from a 2km buffer of the focus of cable laying activities, in this case a maximum of two cable laying vessels and associated vessels. This displacement distance is based on the largest displacement distances reported from empirical studies (see Section 13.6.1.1.4 above). However, given that studies from Orkney (Jarett *et al.*, 2018, 2021), found that not all birds were flushed within 300m of vessels suggests that the assumption (as applied here) of total displacement out to 2km is highly precautionary.

138. The most recently available data for the offshore cable corridor where it overlaps with the SPA derives from two aerial surveys undertaken in February 2018 (Irwin *et al.*, 2019). These surveys found respective densities of 3.64 and 7.10 red-throated divers per km^2 in the southern part of the SPA which overlaps with the offshore cable corridor for North Falls. The two surveys, less than two weeks apart, took place within the spring migration period when densities of red-throated diver are expected to be highest within this SPA (Webb *et al*., 2009) and mean densities over the entire non-breeding period (September to April, Table 13.10) would be lower. In addition, most of the offshore cable corridor is outside the SPA (38km of 57km, 66.6% by length) where red-throated diver densities will be lower than within the SPA boundary (O'Brien *et al*., 2012). Thus, using densities during the spring migration period to estimate the number of birds displaced over the course of a non-breeding season (including autumn migration and winter as well as spring migration) is highly precautionary, especially considering that most of the offshore cable corridor is outside the SPA. Thus, the lower density estimate for the southern area of the SPA from the 2018 surveys is selected as a precautionary mean density for the cable corridor.

139. The worst case for the total area from which birds could be displaced was defined as a circle with a 2km radius around each cable laying vessel, equating to $25.2km^2$ (i.e. 2 x $12.6km^2$). If 100% displacement is assumed to occur within this area, then based on a density of 3.64 birds per km^2, 92 divers would be displaced at any given time.

140. It is considered reasonable to assume that birds will reoccupy areas following the passage of the cable laying vessel. The indicative rate of cable installation is 150-400m per hour (ES Chapter 5 Project Description (Document Reference 3.1.7)), which is assumed to be the average speed of the cable laying vessels during this activity. This represents a maximum speed of 6.7m per minute. In context, a modest tidal flow rate for the Outer Thames area is about 30m per minute (derived from Department of Energy and Climate Change (DECC), 2009). The tide would therefore be flowing at least four times faster than the cable laying vessel. Birds on the water surface are likely to be drifting with the tide and moving at the same speed as the tidal flow. Thus, even though they would be moving, during cable-laying the vessels would be effectively stationary as far as the birds are concerned, so the zone of impact around the vessel would be more or less fixed. Consequently, for the purposes of this assessment it can be assumed that the estimated number of red-throated divers displaced at any one time from cable-laying vessels is equivalent to the total number of birds that can be regarded as being displaced over the full duration of a single non-breeding season.

141. Assuming a maximum mortality of 1% of displaced birds (MacArthur Green, 2019c, see para 280 above), a maximum of one red-throated diver would be predicted to suffer mortality over the course of a non-breeding season due to displacement from construction activities in the offshore cable corridor.

142. The relevant reference populations for red-throated divers present at North Falls during the non-breeding season are the UK North Sea BDMPS during Autumn and Spring migration, and the south-west North Sea during winter, respectively estimated as 13,277 and 10,177 individuals (Furness, 2015). Based on the average annual mortality rate across age classes of 0.233 (Table 13.11), respectively 3,094 and 2,371 would be expected to suffer mortality each year. The addition of one individual would represent an increase in mortality rate of <0.1% of either the migration period or winter BDMPS (e.g. for the spring migration BDMPS the maximum increase in mortality rate would be 1 ÷ 3094 x 100 = 0.03%).

143. At 10% mortality of displaced birds, which is considered unrealistically high (para 280 above), 9.2 red-throated divers would suffer mortality as a result of displacement from the offshore cable corridor, equivalent to an increase of 0.3% in the baseline mortality rate of the migration period BDMPS and 0.4% for the winter BDMPS.

144. The predicted magnitudes of increase in mortality at 1- 10% mortality of displaced birds would not materially alter the background mortality of any seasonal BDMPS and would be undetectable in the context of natural variation.

145. Construction disturbance and displacement within the North Falls offshore cable corridor would be a temporary effect, due to take place over approximately six months (Table 13.1). The predicted magnitude of increase in red-throated diver mortality would not materially alter the background mortality of the population and would be undetectable. Thus, this precautionary assessment generates an impact of negligible magnitude. As the species is of high sensitivity to disturbance, the effect significance is minor adverse, which is not significant in EIA terms.

13.6.1.2 Effect 2: Indirect effects through effects on habitats and prey species

146. Indirect disturbance and displacement of birds may occur during the construction phase if there are effects 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 seabird prey species. 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 may result in less prey being available within the construction area to foraging seabirds. Such effects on benthic invertebrates and fish have been assessed in ES Chapter 10 Benthic and Intertidal Ecology (Document Reference: 3.1.12) and ES Chapter 11 Fish and Shellfish Ecology (Document Reference: 3.1.13) and the conclusions of those assessments inform this assessment of indirect effects on ornithology receptors.

147. ES Chapter 11 Fish and Shellfish Ecology (Document Reference: 3.1.13) discusses the effects upon fish relevant to offshore ornithology as prey species of seabirds at North Falls during construction. For species such as herring, sprat and sandeel, which are amongst the main prey items of seabirds such as gannet, kittiwake, auks, and red-throated diver, underwater noise impacts (physical injury or behavioural changes) during construction are considered to be minor or negligible. With regard to changes to the seabed and to suspended sediment levels, ES Chapter 10 Benthic and Intertidal Ecology (Document Reference: 3.1.12) also considers the effects on prey species arising from any change and impacts on the seabed and benthic habitats during construction. Such changes are considered to be temporary, small scale and highly localised with negligible to minor impacts on benthic habitats and species. The consequent effect on fish through physical disturbance and temporary habitat loss is therefore also considered to be minor or negligible for species such as herring, sprat and sandeel.

148. All offshore ornithology receptors are considered to have a medium sensitivity to effects on prey species. This is because, while they depend on the availability of prey, under most environmental conditions, they have the capability to exploit alternative foraging areas if prey is depleted or unavailable in a given foraging area. As above, any such effects are expected to be localised and temporary. As affected seabird species will forage over a wide area (relative the potential impacts on prey) and will necessarily exhibit some flexibility in the areas within which they forage, it is considered very unlikely that these localised, temporary impacts on prey would result in significant effects on seabirds' ability to forage. With a minor or negligible impact on fish that are important bird prey species, it is concluded that the indirect effect significance on seabirds occurring in or around North Falls during the construction phase is similarly a minor or negligible adverse effect significance, which is not significant in EIA terms.

13.6.2 Likely significant effects during operation

13.6.2.1 Effect 1: Direct disturbance and displacement

149. The presence of WTGs and associated infrastructure and operational activities have the potential to directly disturb and displace birds from within and around the array area. This has the potential to reduce the area available to birds for feeding, loafing and moulting, and may result in reduction in survival rates of displaced birds. WTGs, associated ancillary structures, vessel activity and factors such as the lighting of WTGs could also attract certain species of birds.

150. Following installation of the offshore export cables, maintenance activities (in relation to the cable) may have short-term and localised disturbance and displacement impacts on birds using the offshore project area. However, disturbance from operational activities would be temporary and localised, and is unlikely to result in detectable effects at either the local or regional population level. Therefore, no displacement within the offshore cable corridor due to cable operation and maintenance is predicted.

151. During operation, the WTGs and OSPs / OCP will have lights for air safety and navigational safety. There would be other lighting for personnel working at night, however these would not be as bright as air and navigational safety lighting. Air safety lights will be placed high on the WTG structures, and as a minimum on

WTGs at the periphery of the wind farm. Navigational lights for shipping will be placed lower on WTG structures and other offshore structures. A review of the potential effects of operational lighting on birds considered eight categories: disruption of photoperiod physiology; extension of daytime activity; phototaxis of seabirds; phototaxis of nocturnal migrant birds; ability of birds to use artificial light to feed at night or to feed on prey aggregating under artificial lights; increased predation risk for nocturnal migrant birds; birds better able to avoid collision when structures are illuminated; displacement of birds due to avoidance of artificial lights (Furness, 2018). The available evidence suggests that lights on offshore WTGs in European shelf seas are extremely unlikely to have any detectable effect on birds as a consequence of any of the processes listed above (see also para 101 above). The effects of operational lighting are therefore not assessed separately.

152. The assessment below is based on a guidance note on displacement from the UK SNCBs (SNCB, 2017).

153. Displacement is defined as ‘a reduced number of birds occurring within or immediately adjacent to an offshore wind farm’ (Furness *et al.*, 2013) and involves birds present in the air and on the water (SNCB, 2017). Birds that do not intend to utilise a wind farm area but would have previously flown through the area on the way to a feeding, resting or nesting area, and which either stop short or detour around a development, are subject to barrier effects (SNCB, 2017).

154. Birds are considered to be most at risk from operational disturbance and displacement effects when they are resident in an area, for example during the breeding season or wintering season, as opposed to passage or migratory seasons. Birds that are resident in an area may regularly encounter and be displaced by an OWF for example during daily commuting trips to foraging areas from nest sites, whereas birds on passage may encounter (and potentially be displaced from) a particular OWF only once during a given migration journey.

155. For the purposes of assessment of displacement for resident birds, it is usually not possible to distinguish between displacement and barrier effects – for example to define where individual birds may have intended to travel to, or beyond an OWF, even when tracking data are available. Therefore, in this assessment the effects of displacement and barrier effects on the key resident species are considered together.

156. The small risk of impact to migrating birds resulting from flying around rather than through, the WTG array of an OWF is considered a potential barrier effect. 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 through the site (including seabirds, waders and waterbirds on passage) are considered negligible and these have been scoped out of detailed assessment.

157. The focus of this section is therefore on the disturbance and displacement of seabirds due to the presence and operation of WTGs, other offshore infrastructure and any maintenance operations associated with them. The methodology presented in the SNCB Advice Note (SNCB, 2017) recommends a matrix is presented for each key species showing the predicted bird mortality

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 mortalities as a consequence of the proposed project. These potential mortalities are then placed in the context of the relevant population (usually the BDMPS) to determine the impact magnitude.

158. As OWFs are relatively new features in the marine environment, there is limited robust empirical evidence about the disturbance and displacement effects of the operational infrastructure in the long term, although the number of available post-construction monitoring studies is increasing (e.g. Busch *et al.*, 2015; Dierschke *et al.*, 2016; Vallejo *et al.*, 2017; Marine Management Organisation (MMO), 2018). Dierschke *et al.* (2016) reviewed evidence from 20 operational OWFs in European waters. They found strong avoidance by divers, gannet, great crested grebe, and fulmar; less consistent displacement of razorbill, guillemot, little gull and sandwich tern; no evidence of any consistent response by kittiwake, common tern and Arctic tern, evidence of weak attraction to operating OWFs 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. Thaxter *et al.* (2018) also found no evidence of macro-avoidance of OWFs by lesser black-backed gulls. Where species were displaced, Dierschke *et al.* (2016) considered effects were mainly due to the presence of OWF structures and were apparently stronger when WTGs were rotating. For cormorants and shags the presence of structures for roosting and drying plumage is a factor in attraction, while other species appear to benefit from increases in food abundance within operational OWFs (Dierschke *et al.*, 2016).

159. There is no empirical evidence that birds displaced from wind farms, or exposed to barrier effects, suffer increased mortality. Unlike birds which collide with turbines, displaced birds are unlikely to suffer immediate mortality, but individual displacement events could accrue energetic costs or result in reduced foraging efficiency which adversely affect body condition and may increase the likelihood of mortality. Thus, any mortality due to displacement could result from increased energetic costs from avoiding an OWF, and / or (more likely) increased densities of foraging birds in locations outside the affected area, resulting in increased competition for food. The latter would be unlikely for those seabird species that have large areas of alternative foraging habitat available, but would be more likely to affect species with highly specialised habitat requirements that are limited in availability (Furness and Wade, 2012; Bradbury *et al.*, 2014). Impacts of displacement are also likely to be dependent on other environmental factors such as food supply, and are expected to be greater in years of low prey availability (e.g. as could result from unsustainably high fisheries pressures or effects of climatic changes on fish populations).

160. Modelling of the consequences of displacement on the fitness of displaced birds suggests that even in the case of breeding seabirds that are displaced on a daily (or more frequent) basis (due to the frequency with which they have to commute between the colony and the foraging areas for purposes of provisioning chicks), there is likely to be little or no impact on survival unless the OWF is close to the breeding colony (Searle *et al.,* 2014, 2017). For example, modelling of the effects of displacement from OWFs in the outer Forth and Tay Area was carried

out for guillemot, razorbill, puffin and kittiwake, by Searle *et al*. (2014) in relation to nearby SPAs. The OWFs were Seagreen Alpha and Bravo, Inch Cape and Neart na Gaoithe, and the SPAs Buchan Ness to Collieston Coast (71.68km from the nearest OWF), Fowsheugh (27.58km from the nearest OWF), Forth Islands (12.43km from the nearest OWF) and St Abb's Head to Fastcastle (30.01km from the nearest OWF). Three species and SPA combinations were identified for which declines in adult survival of more than 0.5% were predicted under certain scenarios – kittiwakes at Forth Islands and Fowlsheugh and puffins at Forth Islands. There was no suggestion of declines in adult survival of more than 0.5% for razorbills or guillemots, or for kittiwakes at St. Abbs or Buchan Ness. It is noted that this modelling included kittiwake, a species which Natural England, SNCBs (2017) and Dierschke *et al*., 2016, do not advise is susceptible to displacement as based upon the available evidence.

161. In many seabird species, most mortality occurs during the non-breeding rather than the breeding season, with the direct cause often adverse environmental conditions such as winter storms; species such as auks which lose the ability to fly during the moult period may be particularly at risk at this time (MacArthur Green, 2019b). Thus, displacement at any time of year could potentially contribute to the risk that an individual would suffer mortality during critical periods of the non-breeding season, if displacement acts to reduce body condition.

162. During the breeding season, displacement from an OWF might also affect the body condition, and hence survival, of chicks (which depend on parent birds to deliver food until they leave the nest). In the absence of empirical evidence of this effect, and guidance on its incorporation in displacement assessments, the assessment presented here focuses on effects on the survival of adult and subadult birds (as the basis of the approach of the SNCB (2017) guidance).

163. 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 13.16), focusing on the main species described in the Offshore Ornithology Technical Report (ES Appendix 13.2 (Document Reference: 3.3.13)). The species identified as at risk were then assessed within the biological seasons within which effects were likely to occur. The general sensitivity to disturbance and displacement for each species is presented in Table 13.16. Any species with a low sensitivity to displacement, and / or recorded only in very small numbers within the North Falls survey area during the breeding and non-breeding seasons (with reference to ES Appendix 13.2 (Document Reference: 3.3.13), were screened out of further assessment. For species screened in, further context on displacement and mortality rates used in the assessment is provided below.

Table 13.16 Screening for operational disturbance and displacement

Species	Sensitivity to Disturbance and Displacement[1]	Screened IN or OUT	Rationale
Arctic skua	Low	Out	Low susceptibility to displacement from WTGs
Black-headed gull	Low	Out	No evidence of displacement from WTGs

Species	Sensitivity to Disturbance and Displacement[1]	Screened IN or OUT	Rationale
Common gull	Low	Out	No evidence of displacement from WTGs
Common tern	Low	Out	Recorded in very low numbers during baseline surveys and not very susceptible to displacement
Cormorant	Low	Out	Recorded in very low numbers during baseline surveys and not very susceptible to displacement
Fulmar	Low	Out	Considered low in some studies, but possibly some degree of avoidance according to Dierschke *et al.* (2016). The species has a maximum habitat flexibility score of 1 in Furness and Wade (2012), suggesting it utilises a wide range of habitats, and it also ranges over a extensive areas.
Gannet	Medium	In	Considered low in some studies, but Dierschke *et al.* (2016) review suggests strong avoidance, and has a high macro-avoidance rate for wind farms (Cook *et al.,* 2018, Pavat et al. 2023)
Great black-backed gull	Low	Out	No evidence of displacement from WTGs
Great skua	Low	Out	Low susceptibility to displacement from WTGs
Guillemot	Medium	In	Potentially susceptible to displacement from WTGs and abundant during baseline surveys
Herring gull	Low	Out	No evidence of displacement from WTGs
Kittiwake	Low	Out	No evidence of displacement from WTGs
Lesser black-backed gull	Low	Out	No evidence of displacement from WTGs
Little gull	Low	Out	No evidence of displacement from WTGs
Puffin	Medium	Out	Potentially susceptible to displacement from WTGs but recorded in very low numbers during baseline surveys
Razorbill	Medium	In	Potentially susceptible to displacement from WTGs and abundant during baseline surveys
Red-throated diver	High	In	Recorded regularly during baseline surveys outside the breeding season and sensitive to disturbance and displacement
Sandwich tern	Medium	Out	Potentially susceptible to displacement from WTGs but recorded in very low numbers during baseline surveys
1. With reference to Garthe and Hüppop, 2004; Furness and Wade, 2012; Furness *et al.*, 2013; Wade *et al.*, 2016, Dierschke *et al.,* 2016.			

164. The site population estimate used for each species to assess the displacement effects was the relevant seasonal mean peak (i.e. the highest mean abundance value (all birds, flying and sitting) for the months within each season over the two-year survey period). As per SNCBs (2017), for all species except divers, these population estimates were derived for the array area and 2km buffer. For red-throated diver, a number of studies have shown displacement effects up to 10km or more from wind farm areas, with the proportion of birds displaced reducing with distance from the turbine array (SNCBs, 2022). Natural England

advised that for the purposes of the North Falls EIA, displacement of red-throated divers should be assessed for the array area and a 4km buffer (consistent with SNCBs 2017), assuming 100% displacement of birds within this area.

165. For the RIAA Part 4 (Document Reference: 7.1.4), displacement effects for this species in relation to the Outer Thames Estuary SPA are considered in 1km buffers out to 12km from the North Falls array area, taking account of reductions in the proportion of birds displaced with distance from the array area.

166. Seasonal site population estimates for species included in the displacement assessment are included in Table 13.17. Use of mean peaks (the mean of the highest monthly count in a given season over each of the two years of the baseline surveys) to estimate seasonal populations is likely to overestimate the number of birds typically occurring within the array area and buffer during a given season, particularly for species like guillemot and razorbill which have may distinct concentrations pre-breeding and for post-breeding dispersal. Thus the use of mean peak counts for these species at least, builds in precaution to an assessment.

167. For each species and seasonal period assessed, the predicted mortality due to displacement was determined and the effect of this assessed in terms of the change in the baseline mortality rate of the relevant population. It has been assumed that all age classes are equally at risk of displacement in proportion to their presence in the population, and potential changes in mortality rate from displacement have been compared to the average baseline mortality of the species concerned (Table 13.11).

Table 13.17 Seasonal mean peak populations for species assessed for displacement

Species	Area	Mean peak population (95% confidence interval)				
		Breeding	Migration-free breeding	Migration-Autumn	Winter / non-breeding	Migration-Spring
Gannet	Array area +2km buffer	69 (6 – 173)	–	287 (105 – 575)	–	290 (19 – 658)
Guillemot	Array area +2km buffer	–	866 (242 – 2,346)	–	5,365 (868– 14,674)	–
Razorbill	Array area +2km buffer	–	104 (0 – 328)	248 (8 – 607)	1,781 (1,239 – 2,548)	1,741 (413 – 4,907)
Red-throated diver	Array area +4km buffer	–	0	0	20 (0 – 44)	66 (12 – 149)

168. SNCBs (2017) advice 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. SNCBs (2017) acknowledge that summing impacts in this manner almost certainly over-estimates the number of individuals at risk through double counting (e.g. some individuals may potentially be present in more than one season). In addition, 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.

169. As context to the assessment, it is noted that North Falls is located close to several operational OWFs (Figure 13.2 (Document Reference: 3.2.9)). It is adjacent to the GGOW, which in turn is adjacent to the Galloper Wind Farm (GWF). Thus, in terms of displacement effects, North Falls cannot strictly be considered in isolation from adjacent OWFs, and the densities and abundances of seabirds recorded in the baseline surveys will be inclusive of any displacement effects from these existing OWFs.

Gannet

Sensitivity of receptor

170. Gannets show a low level of sensitivity to ship and helicopter traffic (Garthe and Hüppop, 2004; Furness and Wade, 2012; Furness *et al.*, 2013), but strong avoidance and displacement from offshore WTGs (Dierschke *et al.,* 2016, Wade *et al.*, 2016) and on this basis SNCB (2017) indicates that a detailed assessment of potential displacement effects is required for OWF assessments. Gannet is therefore considered to have a medium sensitivity to disturbance and displacement.

Magnitude of impact

171. In a review of a number of studies of displacement of gannets from OWFs, Pavat *et al.* (2023) calculated a mean macro-avoidance rate (i.e. the percentage of birds taking action to avoid entering array areas) of 85.64%. Cook *et al.* (2018) reported (where quantified), macro-avoidance rates of 64 – 100% from various studies. Some studies however reported no displacement response of gannets, possibly in areas where low densities of birds were present. Cook *et al.* (2018) recommended that the lowest of the quantified macro-avoidance rates, 64% for OWEZ (Krijgsveld *et al.*, 2011) was appropriate for this species.

172. A study of seabird flight behaviour at Thanet OWF, not included in the above review, found a macro-avoidance rate of 79.7% for gannets approaching within 3km of the wind farm (Skov *et al.*, 2018).

173. Digital aerial surveys at Beatrice OWF between May and July 2019, the first year of post-construction monitoring, found no overall significant change in gannet abundance compared with pre-construction surveys over the same area in 2015, but a significant decrease in the abundance of gannets within the OWF; only two individuals were recorded in the array area across six surveys in 2019 (MacArthur Green, 2021). No estimate of the displacement rate was provided for Beatrice from the first year of post-construction monitoring. The second year of post-construction monitoring at Beatrice OWF also showed a consistent and clear pattern by gannet in response to the wind farm. Gannet abundance decreased in the wind farm area (only 12 individuals were recorded over six surveys) and increased outside of the wind farm area, showing consistency with the findings of other studies (MacArthur Green, 2023).

174. A global positioning system (GPS) tagging study of 28 gannets breeding at Helgoland in the southern North Sea found that during the breeding season (birds were tracked during the incubation and chick-rearing periods) most (n=25, 89%) avoided entering OWFs within their foraging range over the two years of the study, although a few (n=3, 11%) frequently foraged within or commuted through OWFs (Peschko *et al.*, 2021).

175. A review and meta-analysis of post-construction monitoring studies for gannet, comprising studies from 19 study areas and 25 OWFs in the UK and western Europe, was carried out to support the DCO examination for Hornsea Four OWF (APEM, 2022a). Of the OWFs considered, 26% of OWFs fell into the range of 60 – 80% displacement, 32% greater than 80%, and 42% reported or inferred rates of <60%. The methodologies, data quality and duration of each OWF study were reviewed and each was graded low, moderate or good in terms of the confidence in the reported displacement effect. It was noted that unless observed declines in gannet numbers are large, or survey effort is intense, the likelihood of being able to detect declines and / or displacement of less than about 50% is low, and all studies reporting no significant evidence for displacement fell into the low confidence category. Thus, weak or even moderate levels of displacement may have gone undetected in studies that reported no significant displacement or macro-avoidance. It was suggested that reported displacement rates of >80% should be considered with caution as there may have been factors compounding the displacement effect (either design or location based, such as complex layouts or ongoing construction activities in proximity to the study area, or to do with the data collection or analysis method), and that such high rates may only be applicable to OWFs under certain scenarios.

176. APEM (2022a) also considered variables which may influence gannet displacement rates from OWFs. A number of statistically significant differences were found. Displacement rates were reported as significantly lower during the breeding season (40 – 60%) than the non-breeding (migratory) season (60 – 80%).

177. Considering the non-breeding season only (there were insufficient numbers of OWFs with displacement rates for the breeding season for analysis), maximum distance between turbines and OWF area were negatively correlated with displacement rate, whereas displacement rate increased with WTG density (WTG's per km^2) and distance from shore. When maximum distances between turbines (usually between row distances) were less than 900m an OWF was found to be more likely to have a displacement rate exceeding 75% during the non-breeding season. There was an indication of a threshold size for OWFs below which migrating gannets are more likely to detour around them. Thus, OWFs less than $25km^2$ were more likely to be associated with a displacement rate exceeding 75%. OWFs with turbine density greater than 2.7 per km^2 were more likely to be associated with a displacement rate greater than 75%, as were OWFs situated more than 19km from shore. It was considered that these associations between OWF characteristics and gannet displacement rate could explain the fact that all non-UK OWFs in the study fell into the group with higher displacement rates, whereas UK OWFs were in the lower displacement rate group (with UK OWFs tending to be larger and further from the coast, with lower WTG densities) (APEM, 2022a).

178. The assessment for gannet assumes that 60-80% of birds are displaced from operational OWFs in line with the advice from Natural England (comments on the outline method statement for North Falls, see ES Appendix 13.1 (Document Reference: 3.3.12), Section 1.1.2) as well as the empirical evidence reviewed above. A maximum 1% mortality of displaced birds is assumed, because gannet has high habitat flexibility (Furness and Wade, 2012) and an extensive foraging range in the breeding season (Woodward *et al.*, 2019). This suggests that displaced birds will readily find alternative foraging areas. This is backed-up by a review of the evidence for mortality rates of displaced gannets (APEM, 2022a) which considers studies using simulation models of displacement to predict changes in mortality rates and inferred evidence from increasing numbers of gannets breeding at Heligoland in the German North Sea, where OWFs have been in operation since 2014. The review suggests that mortality rates for displaced gannets are likely to be negligible or less than 1% during the breeding and non-breeding season.

Autumn migration

179. Within the range of 60 – 80% displacement and 1% mortality, the number of gannets which could potentially suffer mortality as a consequence of displacement from the North Falls array area and 2km buffer during the autumn migration period has been estimated as a maximum of two individuals (95% CLs 1 – 5) (Table 13.18).

180. The BDMPS is 456,298 non-breeding individuals (UK North Sea and Channel, Furness, 2015). At the average baseline mortality rate for gannet of 0.187 (Table 13.11) the number of expected individual mortalities annually from the BDMPS population is 85,328. The addition of two birds (95% CLs 1-5) is equivalent to a 0% (95% CLs 0 – 0.01%).

181. 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 impact magnitude is assessed as negligible.

Breeding season

182. The closest gannet breeding colony to North Falls is Bempton Cliffs within the Flamborough and Filey Coast SPA, 266.3km at the nearest point. This is based on a straight-line distance partly across land, so the at-sea distance would be greater, however it is considered that North Falls is within the breeding season foraging range of gannet (Mean Maximum Foraging Range (MMFR) 315.2km ± 194.2km SD, Woodward *et al.,* 2019), and it is possible that breeding adult gannets from Bempton Cliffs might occur at North Falls. Breeding adult gannets from Bempton Cliffs fitted with satellite tracking devices did not however travel as far south as North Falls; 42 birds were tracked during the chick-rearing phase, most foraging trips were within 150km of the colony tracked birds tending to head out into the North Sea to the north-east, east and south-east but none venturing further south than offshore areas off the North Norfolk Coast (Langston *et al.*, 2013).

183. Low numbers of gannets were recorded at North Falls during the breeding season compared with the Spring and Autumn migration periods (Table 13.17). It is most likely that gannets present at North Falls during the breeding season

are sub-adults or non-breeding adults, and any displacement of such birds would not affect the Bempton Cliffs breeding population. On a precautionary basis, however, predicted displacement mortality of gannet during the breeding season has been compared to the SPA reference population. The SPA population at designation was 11,061 pairs, increasing to 13,392 pairs by 2017 (Aitken *et al.*, 2017), 13,125 pairs in 2022 (Clarkson *et al.,* 2022) and 15,233 pairs in 2023 (Butcher *et al.,* 2023). These equate to total population sizes of approximately 40,222, 48,698, 47,727 and 55,393 (designated 2017, 2022 and 2023 count respectively; calculated as individuals and multiplied up to include subadult birds, based on the adult proportion of 0.55 from Furness, 2015). Clarkson *et al.* (2022) suggested that the numbers of breeding gannets at Bempton Cliffs may be stabilising around 13,000 pairs, based on reducing annual growth rates since 1987; however, the 2023 count recorded an increase in breeding numbers. This increase was despite records of HPAI infection in some seabirds at Flamborough and Filey in 2022, which was most evident in gannets, for which declines in breeding numbers were noted in high density areas of the northern colony between successive counts (Clarkson *et al.*, 2022). A mean of the 2022 and 2023 counts: 51,560 individuals, breeding and non-breeding / sub-adult birds (assuming that 55% of the population is comprised of breeding adults, Furness 2015) has been used as a reference population.

184. Within the range of 60 – 80% displacement and 1% mortality, the maximum number of gannets which could potentially suffer mortality as a consequence of displacement from the North Falls array area during the breeding season has been estimated as one individual (95% CLs 0 – 1) (Table 13.19Table 13.19).

185. At the average baseline mortality rate of 0.187 the number of individuals expected to suffer mortality annually from the Bempton Cliffs (Flamborough and Filey Coast SPA) population is 9,642. The addition of one bird increases the mortality rate by 0.01%.

186. 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 impact magnitude is assessed as negligible.

187. Under the latest guidance from Natural England and NRW on EIA reference populations (see para 68), the breeding season BDMPS is 400,326 individuals for the UK North Sea and Channel; applying this, the percentage increases in baseline mortality during the breeding season would be even smaller than that given in the table.

Spring migration

188. Within the range of 60 – 80% displacement and 1% mortality, the maximum number of gannets that could potentially suffer mortality as a consequence of displacement from the North Falls array area during the spring migration period has been estimated as two individuals (95% CLs 0 – 5) (Table 13.20).

189. The BDMPS is 248,385 non-breeding individuals (UK North Sea and Channel, Furness 2015). At the average baseline mortality rate for gannet of 0.187 (Table 13.11) the number of individuals expected to suffer mortality annually from the BDMPS population is 46,448. The addition of two birds (95% CLs 0 – 5) increases the mortality rate by 0% (95% CLs 0 – 0.01%).

190. 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 impact magnitude is assessed as negligible.

Year round

191. Considering the year-round effects, the number of gannets expected to suffer mortality as a result of displacement from the North Falls array area, at a displacement rate of 60 – 80% and maximum mortality of 1%, would be 4 – 5 birds (95% CLs 1 – 11) (Table 13.21).

192. These predictions are assessed against the largest BDMPS, 456,298 (UK North Sea and Channel) during the non-breeding season, and the biogeographic gannet population with connectivity to UK waters, 1,180,000 (Furness 2015). The percentage increase in baseline mortality rates of these populations for 60 – 80% displacement and 1% mortality of displaced birds, at the average annual mortality of 0.187, are shown in Table 13.21.

193. All predictions for annual displacement mortality of gannet represent increases of 0.01% or less in baseline mortality of the North Sea and Channel BDMPS and biogeographic populations. The predicted magnitudes of increase in mortality would not materially alter the background mortality of the BDMPS and biogeographic populations and would be undetectable.

194. The magnitude of predicted year-round displacement mortality to gannets is assessed as negligible.

Table 13.18 Displacement matrix for gannet during the autumn migration period. The cells show the number of predicted bird mortalities (to the nearest integer) at a given rate of displacement and mortality (LCL and UCL = upper and lower 95% CLs). Grey cells identify the range of displacement and mortality rates considered in the assessment

Mean		Mortality										
		1%	2%	3%	4%	5%	10%	20%	30%	50%	80%	100%
Displacement	10%	0	1	1	1	1	3	6	9	14	23	29
	20%	1	1	2	2	3	6	11	17	29	46	57
	30%	1	2	3	3	4	9	17	26	43	69	86
	40%	1	2	3	5	6	11	23	34	57	92	115
	50%	1	3	4	6	7	14	29	43	72	115	143
	60%	2	3	5	7	9	17	34	52	86	138	172
	70%	2	4	6	8	10	20	40	60	100	160	201
	80%	2	5	7	9	11	23	46	69	115	183	229
	90%	3	5	8	10	13	26	52	77	129	206	258
	100%	3	6	9	11	14	29	57	86	143	229	287
LCL		Mortality										
		1%	2%	3%	4%	5%	10%	20%	30%	50%	80%	100%
Displacement	10%	0	0	0	0	1	1	2	3	5	8	10
	20%	0	0	1	1	1	2	4	6	10	17	21
	30%	0	1	1	1	2	3	6	9	16	25	31
	40%	0	1	1	2	2	4	8	13	21	33	42
	50%	1	1	2	2	3	5	10	16	26	42	52
	60%	1	1	2	3	3	6	13	19	31	50	63
	70%	1	1	2	3	4	7	15	22	37	59	73
	80%	1	2	3	3	4	8	17	25	42	67	84
	90%	1	2	3	4	5	9	19	28	47	75	94
	100%	1	2	3	4	5	10	21	31	52	84	105
UCL		Mortality										
		1%	2%	3%	4%	5%	10%	20%	30%	50%	80%	100%
Displacement	10%	1	1	2	2	3	6	11	17	29	46	57
	20%	1	2	3	5	6	11	23	34	57	92	115
	30%	2	3	5	7	9	17	34	52	86	138	172
	40%	2	5	7	9	11	23	46	69	115	184	230
	50%	3	6	9	11	14	29	57	86	144	230	287
	60%	3	7	10	14	17	34	69	103	172	276	345
	70%	4	8	12	16	20	40	80	121	201	322	402
	80%	5	9	14	18	23	46	92	138	230	368	460
	90%	5	10	16	21	26	52	103	155	259	414	517
	100%	6	11	17	23	29	57	115	172	287	460	575

Table 13.19 Displacement matrix for gannet during the breeding season. The cells show the number of predicted bird mortalities (to the nearest integer) at a given rate of displacement and mortality (LCL and UCL = upper and lower 95% CLs). Grey cells identify the range of displacement and mortality rates considered in the assessment. Values in red identify scenarios where the baseline mortality rate would increase by >1%

Mean	Mortality										
Displacement	1%	2%	3%	4%	5%	10%	20%	30%	50%	80%	100%
10%	0	0	0	0	0	1	1	2	3	5	7
20%	0	0	0	1	1	1	3	4	7	11	14
30%	0	0	1	1	1	2	4	6	10	16	21
40%	0	1	1	1	1	3	5	8	14	22	27
50%	0	1	1	1	2	3	7	10	17	27	34
60%	0	1	1	2	2	4	8	12	21	33	41
70%	0	1	1	2	2	5	10	14	24	38	48
80%	1	1	2	2	3	5	11	16	27	44	55
90%	1	1	2	2	3	6	12	19	31	49	62
100%	1	1	2	3	3	7	14	21	34	55	69

LCL	Mortality										
Displacement	1%	2%	3%	4%	5%	10%	20%	30%	50%	80%	100%
10%	0	0	0	0	0	0	0	0	0	0	1
20%	0	0	0	0	0	0	0	0	1	1	1
30%	0	0	0	0	0	0	0	1	1	1	2
40%	0	0	0	0	0	0	0	1	1	2	2
50%	0	0	0	0	0	0	1	1	2	2	3
60%	0	0	0	0	0	0	1	1	2	3	4
70%	0	0	0	0	0	0	1	1	2	3	4
80%	0	0	0	0	0	0	1	1	2	4	5
90%	0	0	0	0	0	1	1	2	3	4	6
100%	0	0	0	0	0	1	1	2	3	5	6

UCL	Mortality										
Displacement	1%	2%	3%	4%	5%	10%	20%	30%	50%	80%	100%
10%	0	0	1	1	1	2	3	5	9	14	17
20%	0	1	1	1	2	3	7	10	17	28	35
30%	1	1	2	2	3	5	10	16	26	42	52
40%	1	1	2	3	3	7	14	21	35	55	69
50%	1	2	3	3	4	9	17	26	43	69	87
60%	1	2	3	4	5	10	21	31	52	83	104
70%	1	2	4	5	6	12	24	36	61	97	121
80%	1	3	4	6	7	14	28	42	69	111	139
90%	2	3	5	6	8	16	31	47	78	125	156
100%	2	3	5	7	9	17	35	52	87	139	173

Table 13.20 Displacement matrix for gannet during the spring migration period. The cells show the number of predicted bird mortalities (to the nearest integer) at a given rate of displacement and mortality (LCL and UCL = upper and lower 95% CLs). Grey cells identify the range of displacement and mortality rates considered in the assessment. Values in red identify scenarios where the baseline mortality rate would increase by >1%.

Mean		Mortality										
		1%	2%	3%	4%	5%	10%	20%	30%	50%	80%	100%
Displacement	10%	0	1	1	1	1	3	6	9	14	23	29
	20%	1	1	2	2	3	6	12	17	29	46	58
	30%	1	2	3	3	4	9	17	26	43	69	87
	40%	1	2	3	5	6	12	23	35	58	93	116
	50%	1	3	4	6	7	14	29	43	72	116	145
	60%	2	3	5	7	9	17	35	52	87	139	174
	70%	2	4	6	8	10	20	41	61	101	162	203
	80%	2	5	7	9	12	23	46	69	116	185	232
	90%	3	5	8	10	13	26	52	78	130	208	261
	100%	3	6	9	12	14	29	58	87	145	232	290
LCL		**Mortality**										
		1%	2%	3%	4%	5%	10%	20%	30%	50%	80%	100%
Displacement	10%	0	0	0	0	0	0	0	1	1	1	2
	20%	0	0	0	0	0	0	1	1	2	3	4
	30%	0	0	0	0	0	1	1	2	3	4	6
	40%	0	0	0	0	0	1	1	2	4	6	7
	50%	0	0	0	0	0	1	2	3	5	7	9
	60%	0	0	0	0	1	1	2	3	6	9	11
	70%	0	0	0	1	1	1	3	4	6	10	13
	80%	0	0	0	1	1	1	3	4	7	12	15
	90%	0	0	0	1	1	2	3	5	8	13	17
	100%	0	0	1	1	1	2	4	6	9	15	19
UCL		**Mortality**										
		1%	2%	3%	4%	5%	10%	20%	30%	50%	80%	100%
Displacement	10%	1	1	2	3	3	7	13	20	33	53	66
	20%	1	3	4	5	7	13	26	39	66	105	132
	30%	2	4	6	8	10	20	39	59	99	158	197
	40%	3	5	8	11	13	26	53	79	132	210	263
	50%	3	7	10	13	16	33	66	99	164	263	329
	60%	4	8	12	16	20	39	79	118	197	316	395
	70%	5	9	14	18	23	46	92	138	230	368	460
	80%	5	11	16	21	26	53	105	158	263	421	526
	90%	6	12	18	24	30	59	118	178	296	473	592
	100%	7	13	20	26	33	66	132	197	329	526	658

Table 13.21 Year-round predicted displacement mortality for gannet (summed seasonal totals from Table 13.18 through Table 13.20)

Statistic	No. of predicted bird mortalities as a result of displacement				% Increase in baseline mortality	
	Autumn migration	Breeding	Spring migration	Total*	UK North Sea and Channel BDMPS, Autumn	Biogeographic
60% displacement, 1% mortality						
Mean	2	0	2	4	0	0
LCL	1	0	0	1	0	0
UCL	3	1	4	8	0.01	0
80% displacement, 1% mortality						
Mean	2	1	2	5	0.01	0
LCL	1	0	0	1	0	0
UCL	5	1	5	11	0.01	0.01
*Seasonal numbers are rounded to the nearest integer, so the totals may not exactly match the sum of seasonal values						

Significance of effect

Autumn migration

195. Due to the negligible impact magnitude and medium sensitivity of gannet to disturbance and displacement, the effect significance during autumn migration is minor adverse, which is not significant in EIA terms.

Breeding season

196. Due to the negligible impact magnitude and medium sensitivity, the effect significance during the breeding season is minor adverse, which is not significant in EIA terms.

Spring migration

197. Due to the negligible impact magnitude and medium sensitivity, the effect significance during spring migration is minor adverse, which is not significant in EIA terms.

Year Round

198. The magnitude of predicted year-round displacement mortality to gannet is assessed as negligible. As the species is of medium sensitivity to displacement, the effect significance is minor adverse, which is not significant in EIA terms.

13.6.2.1.1 Auks: Guillemot and Razorbill

Sensitivity of receptor

199. Auks are considered to have medium sensitivities to disturbance and displacement from operational OWFs based on available monitoring data and information on their responses to man-made disturbance, for example for ship

and helicopter traffic (Garthe and Hüppop, 2004; Schwemmer *et al.*, 2011; Furness and Wade, 2012; Furness *et al.*, 2013; Bradbury *et al.*, 2014, MMO, 2018).

Magnitude of impact

200. Available pre- and post-construction data for OWFs have yielded variable results; they indicate that auks may be displaced to some extent by some wind farms, but displacement is partial and apparently negligible at others (Dierschke *et al.*, 2016).

201. Monitoring at GGOW, adjacent (east) to the North Falls array area, found reduced densities of guillemot and razorbill post-construction compared with pre-construction, indicating displacement from the GGOW array areas, with the effects strongest for guillemot (Grant and Clements, 2015; Elston *et al.*, 2016). No estimates of displacement rates were presented.

202. At the London Array, displacement analyses were carried out based on non-breeding season surveys for auks collectively (guillemot and razorbill could not be reliably distinguished in winter plumage from some of the digital aerial surveys) over a study area including the array, areas adjacent to the array and a separate reference area (APEM, 2021; 2022b). An estimated 68% of auks were displaced from the array area post-construction. However, during the monitoring period there was also a shift in auk distribution within the array area and the reference area suggesting other factors besides the presence of the OWF were affecting auk distribution in the study area.

203. Inconclusive displacement effects were detected for auk species collectively at the Lynn and Inner Dowsing Offshore Wind Farms (LID) / Lincs OWFs (HiDef, 2017). No significant displacement was found for guillemots at Robin Rigg OWF in the Solway Firth (Vallejo *et al.*, 2017).

204. At Beatrice OWF in the Moray Firth post-construction monitoring found no evidence of displacement for guillemot or razorbill from breeding season surveys (MacArthur Green, 2021, 2023). It was concluded that overall, there was very little indication of either positive or negative response to the wind farm by guillemot or razorbill. An analysis of the distribution of auks (and some other seabird species) in 100m buffers within 400m of turbines was carried out, which found no evidence for avoidance of individual turbines (at least for birds which had entered the array area), and no consistent effect of turbine rotation speed on seabird distribution within the OWF.

205. Research involving birds tracked using GPS tags suggested that displacement rates within OWFs north of Helgoland (German North Sea) were 63%, increasing to 75% when turbine blades were turning (Peschko *et al.*, 2020a). Effects outside OWFs were not quantified, though it was noted that some individuals used habitats adjacent to OWFs despite the presence of operational turbines.

206. A study in the same region using a long term dataset of boat-based and visual aerial survey data, found a similar effect level during spring. Displacement from the OWF and 3km buffer was calculated to be 63% relative to the surrounding area (Peschko *et al.*, 2020a), with effects detectable up to 9km from OWF boundaries. During the breeding season, the effect was weaker, at 44%

displacement. There were no statistically significant differences between the effects observed in the two seasons. It was hypothesised that guillemots may possess greater flexibility during the spring, allowing them to avoid the OWF and surrounding area. During the breeding season however, birds are more closely associated with their breeding colonies (in this case located 23km from the nearest OWF) and foraging range is constrained to a greater degree by the requirement to return to the colony to attend nests and provision chicks. This might reduce their flexibility with respect to habitat preferences, including OWF avoidance.

207. Following SNCB (2017) guidance, for each of the two auk species the mean peak abundance estimates within the array area and a 2km buffer for the relevant biological periods (see Table 13.10) are used to derive the potential displacement effects based upon applying a range of potential displacement and mortality rates.

208. For auks, Natural England has advised (comments on the outline method statement for North Falls, see ES Appendix 13.1 (Document Reference: 3.3.12), Section 1.1.2) that a range of mortality rates of 1 – 10% and displacement rates of 30 – 70%, should be considered, with 70% displacement and 10% mortality representing the worst case.

209. This advice is considered in the context of three recent reviews of empirical studies of auk / guillemot displacement from OWFs. MacArthur Green (2019b) concluded that displacement of guillemots and razorbills by OWFs is highly variable among sites, may potentially reduce with habituation (although supporting evidence is very limited), and that OWFs may in the long-term increase food availability to guillemots and razorbills through providing enhanced habitat for fish populations. The variable displacement response may be linked to ecological conditions – as per the example above for Helgoland, birds may be displaced less by (more willing to enter) OWFs at times of year when they are more constrained in their foraging ranges, i.e. during the breeding season when they need to return to nests frequently to feed chicks; when they are less constrained, during the non-breeding season, they may show stronger avoidance of OWFs, as they have access to extensive offshore areas outside OWFs. Variability in response might also be a response to the configuration of an OWF, such as the spacing of turbines. Mortality due to displacement might arise if displacement increased competition for resources in areas of auk foraging habitat outside the wind farm. However, increases in density outside the wind farm area may not occur due to the large extent of available habitat outside OWFs. Thus, there may be no increase in mortality rate due to displacement. The worst-case scenario of 10% mortality amongst displaced birds would equate to a doubling of the annual mortality for adult razorbill (10.5%) and more than a doubling of that for adult guillemot (6%). These mortality rates (from Horswill and Robinson, 2015), will include mortality from 'natural' and 'anthropomorphic' factors, such as adverse environmental conditions and fisheries bycatch. MacArthur Green (2019b) suggested that appropriately precautionary rates of displacement and mortality from operational OWFs would be 50% and 1%, respectively.

210. APEM (2022b) presents a review and meta-analysis of post-construction monitoring studies for guillemot and razorbill, carried out to support the DCO

examination for Hornsea Four OWF. The review considered studies from 21 OWFs in the UK and western Europe, and reported: one OWF with attraction (i.e. greater numbers present post-construction), eight OWFs with no significant effects or weak displacement effects, four with inferred displacement but not statistically tested and eight with clear (statistically significant) displacement effects. Based on the findings of the studies, the range of displacement rates was 25 – 75%. Closer examination of the analysis methods and data sets indicated that not all predicted displacement effects were equally reliable. The power to detect change from monitoring may require more than the three-years of post-construction data typically deployed in monitoring programmes, unless displacement is substantial (e.g. rates of 50% or more), or survey effort is intensive.

211. Variables which may influence auk displacement rates from OWFs were considered in the analyses undertaken by APEM (2022b). Higher displacement rates were reported for OWFs with lower auk abundances during pre-construction, which might reflect either lower competition between birds and a greater potential for individuals to move to alternate areas outside an OWF or, alternatively, issues with the reliability of the statistical analyses undertaken on data sets based on low counts. Comparison of array area and WTG density between OWFs with or without reported displacement effects showed no statistically significant difference, but a significant difference was found when turbine density was represented as total windswept area (understood to be the total area occupied by the rotating blades of turbines) as a percentage of the array area footprint). In the latter case, reported displacement effects were associated with higher WTG density, which may conceivably correlate with the extent of shadow flicker over the array area. Greater distance from shore was also significantly associated with displacement effects at OWFs, which may reflect greater flexibility in habitat choices further from the coast. Significant regional differences were also found; OWFs in Belgian, Dutch, and German waters tended to have reported displacement effects whereas those in the Irish Sea and UK North Sea tended to show no displacement effects. APEM (2022b) recommended that until further monitoring data are available, considering up to 50% displacement for auks would be an appropriate precautionary approach (which is consistent with the conclusions of MacArthur Green 2019b).

212. APEM (2022b) also reviewed evidence for the mortality rates of displaced auks. Two modelling studies, which simulate the responses of auks (as well as other seabird species) to displacement from OWFs (Searle *et al.*, 2014; Van Kooten *et al.*, 2019), indicate that mortality rates for displaced auks would be considerably less than 10%. This is supported by inferred evidence from increasing numbers of guillemots breeding at Heligoland in the German North Sea, where auk displacement rates of 44 – 63% have been reported (Peschko *et al.*, 2020b) and OWFs have been in operation since 2014. It was recommended that considering additional mortality rates from displacement of up to 1% would be appropriately precautionary, in light of the evidence base (and is again consistent with the conclusions reached in the review by MacArthur Green 2019b).

213. Leopold and Verdaat (2018) present a review of guillemot displacement from OWFs in UK and other European waters, noting that this species is an ideal subject for comparative study as it is abundant throughout this area and the

available evidence suggests there is much variation in its response to OWFs. The review is supported by revaluation of available monitoring data for guillemots and re-analysis using new statistical methods, the Integrated Nested Laplace Approximations (INLA) method (Zuur, 2018). As a result of this process, data from some OWFs was not considered suitable for more advanced analysis because of large numbers of zero counts (surveys where few or no guillemots were present).

214. Data for three OWFs, OWEZ, Prinses Amalia Windpark (PAWP) and Robin Rigg (the first two off the Netherlands and the latter in the Solway Firth) were re-analysed. This indicated that guillemot distribution was highly variable between individual surveys, and that '*waves of contracting and expanding concentrations of guillemots pulsed through the wind farm areas, dwarfing any displacement effects the wind farms might have'* (Leopold, 2018). Displacement effects were evident in some surveys but were inconsistent for OWEZ and PAWP (in contrast to original analyses by Leopold *et al.,* (2013) and Zuur *et al.,* (2014) which concluded that displacement was occurring), while very weak displacement was found at Robin Rigg (in contrast to Vallejo *et al.*, 2017 who reported no displacement). It was noted that analysis of combined surveys for the same OWF may be misleading. Guillemots are highly mobile and range over large offshore areas in response to many environmental stimuli (of which prey distribution is likely to be a key factor) as well as the presence of OWFs.

215. In the Project alone assessment of displacement for guillemot and razorbill, a range of 30 – 70% displacement and 1 – 10% mortality is presented, as requested by Natural England. The scenario of 50% displacement and 1% mortality is given, as this is considered as an appropriate precautionary scenario. In addition, for the Hornsea Project Four (HP4) HRA (DESNZ, 2023c), the SoS Is understood to have based the consent decision on displacement and mortality rates of 70% and 2% for guillemot and razorbill, so this scenario is presented also.

216. There are no breeding colonies for guillemot or razorbill within foraging range of the North Falls array area (the closest breeding colony for both species is the Flamborough and Filey Coast SPA, 266.3km at the nearest point, compared with respective foraging ranges (mean maximum plus one SD, Woodward *et al.*, 2019) of 153.7km and 164.6km for guillemot and razorbill). Therefore, it is reasonable to assume that individuals seen during the breeding season are non-breeding birds; comprising largely immature birds not yet of breeding age and also a small proportion of sabbatical adults of breeding age, skipping breeding in a given year. Assuming the vast majority of non-breeding birds are immature, and these immature birds remain within the BDMPS area, 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 (based on modelled age structures for these species populations in Furness, 2015). This gives breeding season populations of non-breeding individuals of 695,442 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, 218,622 x 43%). (See notes below referring to updated guidance from Natural England and Natural Resources Wales regarding breeding season BDMPS values).

217. For guillemot, non-breeding season is defined as August – February; for razorbill the non-breeding season is subdivided into autumn migration (August – October), winter (November – December) and spring migration (January – March); Table 13.10. The number of birds which could potentially be displaced has been estimated for each species-specific relevant season.

Guillemot

Non-breeding season

218. Within the range of 30 – 70% displacement and 1 – 10% mortality, the number of guillemots which could potentially suffer mortality as a consequence of displacement from the North Falls array area and 2km buffer during the non-breeding period has been estimated as 16 – 376 individuals (95% CLs 3 – 1,027) (Table 13.22).

219. At 50% displacement and 1% mortality of displaced birds, the predicted number of birds suffering mortality is 27 (95% CLs, 4 – 73), and at 70% displacement and 2% mortality, 75 (95% CLs, 12-205).

220. The BDMPS is 1,617,306 individuals of all age classes (North Sea and Channel, Furness, 2015). At the average baseline mortality rate for guillemot of 0.143 (Table 13.11) the number of individuals expected to suffer mortality annually from the BDMPS population is 231,275.

221. The predicted level of additional mortality for the range of 30-70% displacement and 1-10% mortality would represent a 0.01% (95% CLs 0 – 0.02%) to 0.16% (95% CLs 0.03 – 0.44%) increase in annual mortality within the BDMPS population.

222. These maximum increases in mortality rate assume up to 10% mortality of displaced guillemots, which, as discussed above, is considered highly unlikely. At 1% mortality of displaced birds, and a maximum displacement rate of 50%, the increase in population mortality rate would be 0.01% (95% CLs 0 – 0.03%), and at 70% displacement and 2% mortality of displaced birds, the corresponding increases in mortality rate would be 0.03% (95% CLs 0.01 – 0.09%).

223. Under all scenarios of displacement and mortality, the predicted increase in mortality rate would not materially alter the background mortality of the population and would be undetectable. Therefore, during the non-breeding season, the impact magnitude is assessed as negligible.

Breeding

224. Within the range of 30 – 70% displacement and 1 – 10% mortality, the number of guillemots which could potentially suffer mortality as a consequence of displacement from the North Falls array area during the breeding period has been estimated as 3 – 61 individuals (95% CLs 1 – 164) (Table 13.23).

225. At 50% displacement and 1% mortality of displaced birds, the predicted number of birds suffering mortality is four (95% CLs, 1 – 12), and at 70% displacement and 2% mortality, 12 (95% CLs, 3–- 33).

226. The BDMPS is 695,442 non-breeding individuals (see paragraph 216 above). At the average baseline mortality rate for guillemot of 0.143 (Table 13.11) the number of individuals expected to suffer mortality annually from the BDMPS population is 99,448.

227. The predicted level of additional mortality for the range of 30-70% displacement and 1-10% mortality would represent a 0% (95% CLs 0 – 0.01%) to 0.06% (95% CLs 0.02 – 0.17%) increase in annual mortality within the BDMPS population.

228. These increases in mortality rate assume up to 10% mortality of displaced guillemots, which, as discussed above, is considered highly unlikely. At 1% mortality of displaced birds, and a maximum displacement rate of 50%, the increase in population mortality rate would be 0% (95% CLs 0 – 0.01%), and at 70% displacement and 2% mortality of displaced birds, the corresponding increases in mortality rate would be 0.01% (95% CLs 0 – 0.03%).

229. Under all scenarios of displacement and mortality, the predicted increase in mortality rate would not materially alter the background mortality of the population and would be undetectable. Therefore, during the breeding season, the impact magnitude is assessed as negligible.

230. Under the latest guidance from Natural England and Natural Resources Wales on EIA reference populations (see para 68), the breeding season BDMPS is 2,045,078 individuals for the UK North Sea and Channel; if this were applied, the percentage increases in baseline mortality during the breeding season, would be even smaller than those given above.

Table 13.22 Displacement matrix for guillemot during the non-breeding period. The cells show the number of predicted bird mortalities (to the nearest integer) at a given rate of displacement and mortality (LCL and UCL = upper and lower 95% CLs). Grey cells identify the range of displacement and mortality rates considered in the assessment. Values in red identify scenarios where the baseline mortality rate would increase by >1%

Mean		Mortality										
		1%	2%	3%	4%	5%	10%	20%	30%	50%	80%	100%
Displacement	10%	5	11	16	21	27	54	107	161	268	429	537
	20%	11	21	32	43	54	107	215	322	537	858	1,073
	30%	16	32	48	64	80	161	322	483	805	1,288	1,610
	40%	21	43	64	86	107	215	429	644	1,073	1,717	2,146
	50%	27	54	80	107	134	268	537	805	1,341	2,146	2,683
	60%	32	64	97	129	161	322	644	966	1,610	2,575	3,219
	70%	38	75	113	150	188	376	751	1,127	1,878	3,004	3,756
	80%	43	86	129	172	215	429	858	1,288	2,146	3,434	4,292
	90%	48	97	145	193	241	483	966	1,449	2,414	3,863	4,829
	100%	54	107	161	215	268	537	1,073	1,610	2,683	4,292	5,365
LCL		**Mortality**										
		1%	2%	3%	4%	5%	10%	20%	30%	50%	80%	100%
Displacement	10%	1	2	3	3	4	9	17	26	43	69	87
	20%	2	3	5	7	9	17	35	52	87	139	174
	30%	3	5	8	10	13	26	52	78	130	208	260
	40%	3	7	10	14	17	35	69	104	174	278	347
	50%	4	9	13	17	22	43	87	130	217	347	434
	60%	5	10	16	21	26	52	104	156	260	417	521
	70%	6	12	18	24	30	61	122	182	304	486	608
	80%	7	14	21	28	35	69	139	208	347	556	694
	90%	8	16	23	31	39	78	156	234	391	625	781
	100%	9	17	26	35	43	87	174	260	434	694	868
UCL		**Mortality**										
		1%	2%	3%	4%	5%	10%	20%	30%	50%	80%	100%
Displacement	10%	15	29	44	59	73	147	293	440	734	1,174	1,467
	20%	29	59	88	117	147	293	587	880	1,467	2,348	2,935
	30%	44	88	132	176	220	440	880	1,321	2,201	3,522	4,402
	40%	59	117	176	235	293	587	1,174	1,761	2,935	4,696	5,870
	50%	73	147	220	293	367	734	1,467	2,201	3,669	5,870	7,337
	60%	88	176	264	352	440	880	1,761	2,641	4,402	7,044	8,804
	70%	103	205	308	411	514	1,027	2,054	3,082	5,136	8,217	10,272
	80%	117	235	352	470	587	1,174	2,348	3,522	5,870	9,391	11,739
	90%	132	264	396	528	660	1,321	2,641	3,962	6,603	10,565	13,207
	100%	147	293	440	587	734	1,467	2,935	4,402	7,337	11,739	14,674

Table 13.23 Displacement matrix for guillemot during the breeding period. The cells show the number of predicted bird mortalities (to the nearest integer) at a given rate of displacement and mortality (LCL and UCL = upper and lower 95% CLs). Grey cells identify the range of displacement and mortality rates considered in the assessment. Values in red identify scenarios where the baseline mortality rate would increase by >1%

Mean	Mortality										
Displacement	1%	2%	3%	4%	5%	10%	20%	30%	50%	80%	100%
10%	1	2	3	3	4	9	17	26	43	69	87
20%	2	3	5	7	9	17	35	52	87	139	173
30%	3	5	8	10	13	26	52	78	130	208	260
40%	3	7	10	14	17	35	69	104	173	277	346
50%	4	9	13	17	22	43	87	130	217	346	433
60%	5	10	16	21	26	52	104	156	260	416	520
70%	6	12	18	24	30	61	121	182	303	485	606
80%	7	14	21	28	35	69	139	208	346	554	693
90%	8	16	23	31	39	78	156	234	390	624	779
100%	9	17	26	35	43	87	173	260	433	693	866

LCL	Mortality										
Displacement	1%	2%	3%	4%	5%	10%	20%	30%	50%	80%	100%
10%	0	0	1	1	1	2	5	7	12	19	24
20%	0	1	1	2	2	5	10	15	24	39	48
30%	1	1	2	3	4	7	15	22	36	58	73
40%	1	2	3	4	5	10	19	29	48	77	97
50%	1	2	4	5	6	12	24	36	61	97	121
60%	1	3	4	6	7	15	29	44	73	116	145
70%	2	3	5	7	8	17	34	51	85	136	169
80%	2	4	6	8	10	19	39	58	97	155	194
90%	2	4	7	9	11	22	44	65	109	174	218
100%	2	5	7	10	12	24	48	73	121	194	242

UCL	Mortality										
Displacement	1%	2%	3%	4%	5%	10%	20%	30%	50%	80%	100%
10%	2	5	7	9	12	23	47	70	117	188	235
20%	5	9	14	19	23	47	94	141	235	375	469
30%	7	14	21	28	35	70	141	211	352	563	704
40%	9	19	28	38	47	94	188	282	469	751	938
50%	12	23	35	47	59	117	235	352	587	938	1,173
60%	14	28	42	56	70	141	282	422	704	1,126	1,408
70%	16	33	49	66	82	164	328	493	821	1,314	1,642
80%	19	38	56	75	94	188	375	563	938	1,501	1,877
90%	21	42	63	84	106	211	422	633	1,056	1,689	2,111
100%	23	47	70	94	117	235	469	704	1,173	1,877	2,346

Year round

231. Considering the year-round effects (summed seasonal totals from the tables above), the numbers of guillemots expected to suffer mortality as a result of displacement from the North Falls array area and 2km buffer, at a range of displacement and mortality rates are shown in Table 13.24.

232. These predictions are assessed against the largest (North Sea and Channel) BDMPS, 1,617,306 during the non-breeding season, and the biogeographic guillemot population with connectivity to UK waters, 4,125,000 (Furness, 2015). The percentage increase in baseline mortality rates of these populations are also shown in Table 13.24.

Table 13.24 Year-round predicted displacement mortality for guillemot

Statistic	No. of predicted bird mortalities as a result of displacement			% Increase in baseline mortality	
	Non-breeding	Breeding	Total*	North Sea and Channel BDMPS, nonbreeding season	Biogeographic
30% displacement, 1% mortality					
Mean	16	3	19	0.01	0.00
LCL	3	1	3	0.00	0.00
UCL	44	7	51	0.02	0.01
30% displacement, 10% mortality					
Mean	161	26	187	0.08	0.03
LCL	26	7	33	0.01	0.01
UCL	440	70	511	0.22	0.09
50% displacement, 1% mortality					
Mean	27	4	31	0.01	0.01
LCL	4	1	6	0.00	0.00
UCL	73	12	85	0.04	0.01
70% displacement, 1% mortality					
Mean	38	6	44	0.02	0.01
LCL	6	2	8	0.00	0.00
UCL	103	16	119	0.05	0.02
70% displacement, 2% mortality					
Mean	75	12	87	0.04	0.01
LCL	12	3	16	0.01	0.00
UCL	205	33	238	0.10	0.04
70% displacement, 10% mortality					
Mean	376	61	436	0.19	0.07
LCL	61	17	78	0.03	0.01
UCL	1,027	164	1,191	0.52	0.20
*Seasonal numbers are rounded to the nearest integer, so the totals may not exactly match the sum of seasonal values					

233. All predictions for annual displacement mortality of guillemot represent increases of less than 1% increase in baseline mortality of the UK North Sea and Channel BDMPS and biogeographic populations. As discussed above, a 10% mortality rate of displaced guillemots is considered highly unlikely.

234. Under the latest guidance from Natural England and Natural Resources Wales on EIA reference populations (see para 68), the largest annual BDMPS would be for the breeding season, 2,045,078 individuals for the UK North Sea and Channel; applying this, the percentage increases in baseline mortality during the breeding season, would be even smaller than those given above in Table 13.24.

235. Under all scenarios, the predicted increases in mortality would not materially alter the background mortality of the BDMPS and biogeographic populations and would be undetectable.

236. The magnitude of predicted year-round displacement mortality to guillemots is assessed as negligible.

Razorbill

Autumn migration

237. Within the range of 30 – 70% displacement and 1 – 10% mortality, the number of razorbills which could potentially suffer mortality as a consequence of displacement from the North Falls array area and 2km buffer during the non-breeding period has been estimated as 1 – 17 individuals (95% CLs 0 – 42) (Table 13.25).

238. At 50% displacement and 1% mortality of displaced birds, the predicted number of birds suffering mortality is one (95% CLs 0–- 3), and at 70% displacement and 2% mortality, three (95% CLs 0–- 8).

239. The BDMPS is 591,874 individuals (North Sea and Channel, Furness 2015). At the average baseline mortality rate of 0.178 (Table 13.11) the number of razorbills expected to suffer mortality annually from the BDMPS population is 105,354.

240. The predicted level of additional mortality for the range of 30-70% displacement and 1-10% mortality would represent a 0% (95% CLs 0 – 0%) to 0.02% (95% CLs 0 – 0.04%) increase in annual mortality within the BDMPS population

241. These maximum increases in mortality rate assume 10% mortality of displaced razorbills, which, as discussed above, is considered highly unlikely. At 50% displacement and 1% mortality of displaced birds, the increase in mortality rate would be 0% (95% CLs 0-0%). At 70% displacement and 2% mortality of displaced birds, corresponding increases in mortality rate would be 0% (95% CLs 0.00 – 0.01%).

242. The magnitude of increase in mortality would not materially alter the background mortality of the population and would be undetectable. Therefore, during the autumn migration season, the impact magnitude is assessed as negligible.

Winter

243. Within the range of 30 – 70% displacement and 1 – 10% mortality, the number of razorbills which could potentially suffer mortality as a consequence of displacement from the North Falls array area during the winter period has been estimated as 5 – 125 individuals (95% CLs 4 – 178) (Table 13.26).

244. At 50% displacement and 1% mortality of displaced birds, the predicted number of birds suffering mortality is 9 (95% CLs 6–- 13), and at 70% displacement and 2% mortality, 25 (95% CLs 17–- 36).

245. The BDMPS is 218,622 non-breeding individuals (UK North Sea and Channel, Furness 2015). At the average baseline mortality rate of 0.178 (Table 13.11) the number of individuals expected to suffer mortality annually from the BDMPS population is 38,915.

246. The predicted level of additional mortality for the range of 30-70% displacement and 1-10% mortality would represent a 0.01% (95% CLs 0.01 – 0.02%) to 0.32% (95% CLs 0.22 – 0.46%) increase in annual mortality within the BDMPS population.

247. These maximum increases in mortality rate assume 10% mortality of displaced razorbills, which, as discussed above, is considered highly unlikely. At 1% mortality of displaced birds, and a maximum displacement rate of 50%, the predicted increase in mortality rate would be 0.02% (95% CLs 0.02–- 0.03%). At 70% displacement and 2% mortality of displaced birds, the corresponding increases in mortality rate would be 0.06% (95% CLs 0.04 – 0.09%).

248. Under all scenarios of displacement and mortality, the predicted increase in mortality rate would not materially alter the background mortality of the population and would be undetectable. Therefore, during the winter period, the impact magnitude is assessed as negligible.

Spring migration

249. Within the range of 30 – 70% displacement and 1 – 10% mortality, the number of razorbills which could potentially suffer mortality as a consequence of displacement from the North Falls array area during the spring migration period has been estimated as 5 – 122 individuals (95% CLs 1 – 343) (Table 13.27).

250. At 50% displacement and 1% mortality of displaced birds, the predicted number of birds suffering mortality is 9 (95% CLs 2–- 25), and at 70% displacement and 2% mortality, 24 (95% CLs 6–- 69).

251. The BDMPS is 591,874 individuals (UK North Sea and Channel, Furness 2015). At the average baseline mortality rate for razorbill of 0.178 (Table 13.11) the number of individuals expected to suffer mortality annually from the BDMPS population is 105,354.

252. The predicted level of additional mortality for the range of 30-70% displacement and 1-10% mortality would represent a 0% (95% CLs 0 – 0.01%) to 0.12% (95% CLs 0.03 – 0.33%) increase in annual mortality within the BDMPS population.

253. These increases in mortality rate assume up to 10% mortality of displaced razorbills, which, as discussed above, is considered highly unlikely. At 1% mortality of displaced birds, and a maximum displacement rate of 50%, the increase in population mortality rate would be 0.01% (95% CLs 0 – 0.02%). At

70% displacement and 2% mortality of displaced birds, the corresponding increases in mortality rate would be 0.02% (95% CLs 0.01 – 0.07%).

254. Under all scenarios of displacement and mortality, the predicted increase in mortality would not materially alter the background mortality of the population and would be undetectable. Therefore, during the spring migration period, the impact magnitude is assessed as negligible.

Breeding

255. Within the range of 30 – 70% displacement and 1 – 10% mortality, the number of razorbills which could potentially suffer mortality as a consequence of displacement from the North Falls array area during the breeding season has been estimated as 0 – 7 individuals (95% CLs 0 – 23) (Table 13.28).

256. At 50% displacement and 1% mortality of displaced birds, the predicted number of birds suffering mortality is one (95% CLs 0–- 2), and at 70% displacement and 2% mortality, one (95% CLs 0–- 5).

257. The BDMPS is 94,007 non-breeding individuals (see paragraph 216 above). At the average baseline mortality rate for razorbill of 0.178 (Table 13.11) the number of individuals expected to suffer mortality annually from the BDMPS population is 16,733.

258. The predicted level of additional mortality for the range of 30-70% displacement and 1-10% mortality would represent a 0% (95% CLs 0 – 0.01%) to 0.04% (95% CLs 0 – 0.14%) increase in annual mortality within the BDMPS population.

259. These increases in mortality rate assume up to 10% mortality of displaced razorbills, which, as discussed above, is considered highly unlikely. At 1% mortality of displaced birds, and a maximum displacement rate of 50%, the increase in mortality rate would be 0% (95% CLs 0–- 0.01%). At 70% displacement and 2% mortality of displaced birds, the corresponding increases in mortality rate would be 0.01% (95% CLs 0 – 0.03%).

260. Under all scenarios of displacement and mortality, the 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 impact magnitude is assessed as negligible.

261. Under the latest guidance from Natural England and Natural Resources Wales on EIA reference populations (see para 68), the breeding season BDMPS is 158,031 individuals for the UK North Sea and Channel; if this were applied, the percentage increases in baseline mortality during the breeding season would be even smaller than that given above.

Table 13.25 Displacement matrix for razorbill during the autumn migration period. The cells show the number of predicted bird mortalities (to the nearest integer) at a given rate of displacement and mortality (LCL and UCL = upper and lower 95% CLs). Grey cells identify the range of displacement and mortality rates considered in the assessment. Increase in baseline mortality rate is <1% for all scenarios. The equivalent increase in baseline mortality rate is <1% for all scenarios

Mean	Mortality										
Displacement	1%	2%	3%	4%	5%	10%	20%	30%	50%	80%	100%
10%	0	0	1	1	1	2	5	7	12	20	25
20%	0	1	1	2	2	5	10	15	25	40	50
30%	1	1	2	3	4	7	15	22	37	60	74
40%	1	2	3	4	5	10	20	30	50	79	99
50%	1	2	4	5	6	12	25	37	62	99	124
60%	1	3	4	6	7	15	30	45	74	119	149
70%	2	3	5	7	9	17	35	52	87	139	174
80%	2	4	6	8	10	20	40	60	99	159	198
90%	2	4	7	9	11	22	45	67	112	179	223
100%	2	5	7	10	12	25	50	74	124	198	248
LCL	**Mortality**										
Displacement	1%	2%	3%	4%	5%	10%	20%	30%	50%	80%	100%
10%	0	0	0	0	0	0	0	0	0	1	1
20%	0	0	0	0	0	0	0	0	1	1	2
30%	0	0	0	0	0	0	0	1	1	2	2
40%	0	0	0	0	0	0	1	1	2	3	3
50%	0	0	0	0	0	0	1	1	2	3	4
60%	0	0	0	0	0	0	1	1	2	4	5
70%	0	0	0	0	0	1	1	2	3	4	6
80%	0	0	0	0	0	1	1	2	3	5	6
90%	0	0	0	0	0	1	1	2	4	6	7
100%	0	0	0	0	0	1	2	2	4	6	8
UCL	**Mortality**										
Displacement	1%	2%	3%	4%	5%	10%	20%	30%	50%	80%	100%
10%	1	1	2	2	3	6	12	18	30	49	61
20%	1	2	4	5	6	12	24	36	61	97	121
30%	2	4	5	7	9	18	36	55	91	146	182
40%	2	5	7	10	12	24	49	73	121	194	243
50%	3	6	9	12	15	30	61	91	152	243	304
60%	4	7	11	15	18	36	73	109	182	291	364
70%	4	8	13	17	21	42	85	127	212	340	425
80%	5	10	15	19	24	49	97	146	243	388	486
90%	5	11	16	22	27	55	109	164	273	437	546
100%	6	12	18	24	30	61	121	182	304	486	607

Table 13.26 Displacement matrix for razorbill during the winter period. The cells show the number of predicted bird mortalities (to the nearest integer) at a given rate of displacement and mortality (LCL and UCL = upper and lower 95% CLs). Grey cells identify the range of displacement and mortality rates considered in the assessment. Values in red identify scenarios where the baseline mortality rate would increase by >1%

Mean	Mortality										
Displacement	1%	2%	3%	4%	5%	10%	20%	30%	50%	80%	100%
10%	2	4	5	7	9	18	36	53	89	142	178
20%	4	7	11	14	18	36	71	107	178	285	356
30%	5	11	16	21	27	53	107	160	267	427	534
40%	7	14	21	28	36	71	142	214	356	570	712
50%	9	18	27	36	45	89	178	267	445	712	891
60%	11	21	32	43	53	107	214	321	534	855	1,069
70%	12	25	37	50	62	125	249	374	623	997	1,247
80%	14	28	43	57	71	142	285	427	712	1,140	1,425
90%	16	32	48	64	80	160	321	481	801	1,282	1,603
100%	18	36	53	71	89	178	356	534	891	1,425	1,781

LCL	Mortality										
Displacement	1%	2%	3%	4%	5%	10%	20%	30%	50%	80%	100%
10%	1	2	4	5	6	12	25	37	62	99	124
20%	2	5	7	10	12	25	50	74	124	198	248
30%	4	7	11	15	19	37	74	112	186	297	372
40%	5	10	15	20	25	50	99	149	248	396	496
50%	6	12	19	25	31	62	124	186	310	496	620
60%	7	15	22	30	37	74	149	223	372	595	743
70%	9	17	26	35	43	87	173	260	434	694	867
80%	10	20	30	40	50	99	198	297	496	793	991
90%	11	22	33	45	56	112	223	335	558	892	1,115
100%	12	25	37	50	62	124	248	372	620	991	1,239

UCL	Mortality										
Displacement	1%	2%	3%	4%	5%	10%	20%	30%	50%	80%	100%
10%	3	5	8	10	13	25	51	76	127	204	255
20%	5	10	15	20	25	51	102	153	255	408	510
30%	8	15	23	31	38	76	153	229	382	612	764
40%	10	20	31	41	51	102	204	306	510	815	1,019
50%	13	25	38	51	64	127	255	382	637	1,019	1,274
60%	15	31	46	61	76	153	306	459	764	1,223	1,529
70%	18	36	54	71	89	178	357	535	892	1,427	1,784
80%	20	41	61	82	102	204	408	612	1,019	1,631	2,038
90%	23	46	69	92	115	229	459	688	1,147	1,835	2,293
100%	25	51	76	102	127	255	510	764	1,274	2,038	2,548

Table 13.27 Displacement matrix for razorbill during the spring migration period. The cells show the number of predicted bird mortalities (to the nearest integer) at a given rate of displacement and mortality (LCL and UCL = upper and lower 95% CLs). Grey cells identify the range of displacement and mortality rates considered in the assessment. Values in red identify scenarios where the baseline mortality rate would increase by >1%

Mean		Mortality										
		1%	2%	3%	4%	5%	10%	20%	30%	50%	80%	100%
Displacement	10%	2	3	5	7	9	17	35	52	87	139	174
	20%	3	7	10	14	17	35	70	104	174	279	348
	30%	5	10	16	21	26	52	104	157	261	418	522
	40%	7	14	21	28	35	70	139	209	348	557	696
	50%	9	17	26	35	44	87	174	261	435	696	871
	60%	10	21	31	42	52	104	209	313	522	836	1,045
	70%	12	24	37	49	61	122	244	366	609	975	1,219
	80%	14	28	42	56	70	139	279	418	696	1,114	1,393
	90%	16	31	47	63	78	157	313	470	783	1,254	1,567
	100%	17	35	52	70	87	174	348	522	871	1,393	1,741

LCL		Mortality										
		1%	2%	3%	4%	5%	10%	20%	30%	50%	80%	100%
Displacement	10%	0	1	1	2	2	4	8	12	21	33	41
	20%	1	2	2	3	4	8	17	25	41	66	83
	30%	1	2	4	5	6	12	25	37	62	99	124
	40%	2	3	5	7	8	17	33	50	83	132	165
	50%	2	4	6	8	10	21	41	62	103	165	207
	60%	2	5	7	10	12	25	50	74	124	198	248
	70%	3	6	9	12	14	29	58	87	145	231	289
	80%	3	7	10	13	17	33	66	99	165	264	330
	90%	4	7	11	15	19	37	74	112	186	297	372
	100%	4	8	12	17	21	41	83	124	207	330	413

UCL		Mortality										
		1%	2%	3%	4%	5%	10%	20%	30%	50%	80%	100%
Displacement	10%	5	10	15	20	25	49	98	147	245	393	491
	20%	10	20	29	39	49	98	196	294	491	785	981
	30%	15	29	44	59	74	147	294	442	736	1,178	1,472
	40%	20	39	59	79	98	196	393	589	981	1,570	1,963
	50%	25	49	74	98	123	245	491	736	1,227	1,963	2,454
	60%	29	59	88	118	147	294	589	883	1,472	2,355	2,944
	70%	34	69	103	137	172	343	687	1,030	1,717	2,748	3,435
	80%	39	79	118	157	196	393	785	1,178	1,963	3,140	3,926
	90%	44	88	132	177	221	442	883	1,325	2,208	3,533	4,416
	100%	49	98	147	196	245	491	981	1,472	2,454	3,926	4,907

Table 13.28 Displacement matrix for razorbill during the breeding period. The cells show the number of predicted bird mortalities (to the nearest integer) at a given rate of displacement and mortality (LCL and UCL = upper and lower 95% CLs). Grey cells identify the range of displacement and mortality rates considered in the assessment. Values in red identify scenarios where the baseline mortality rate would increase by >1%

Mean	Mortality										
Displacement	1%	2%	3%	4%	5%	10%	20%	30%	50%	80%	100%
10%	0	0	0	0	1	1	2	3	5	8	10
20%	0	0	1	1	1	2	4	6	10	17	21
30%	0	1	1	1	2	3	6	9	16	25	31
40%	0	1	1	2	2	4	8	12	21	33	42
50%	1	1	2	2	3	5	10	16	26	42	52
60%	1	1	2	2	3	6	12	19	31	50	62
70%	1	1	2	3	4	7	15	22	36	58	73
80%	1	2	2	3	4	8	17	25	42	67	83
90%	1	2	3	4	5	9	19	28	47	75	94
100%	1	2	3	4	5	10	21	31	52	83	104
LCL	Mortality										
Displacement	1%	2%	3%	4%	5%	10%	20%	30%	50%	80%	100%
10%	0	0	0	0	0	0	0	0	0	0	0
20%	0	0	0	0	0	0	0	0	0	0	0
30%	0	0	0	0	0	0	0	0	0	0	0
40%	0	0	0	0	0	0	0	0	0	0	0
50%	0	0	0	0	0	0	0	0	0	0	0
60%	0	0	0	0	0	0	0	0	0	0	0
70%	0	0	0	0	0	0	0	0	0	0	0
80%	0	0	0	0	0	0	0	0	0	0	0
90%	0	0	0	0	0	0	0	0	0	0	0
100%	0	0	0	0	0	0	0	0	0	0	0
UCL	Mortality										
Displacement	1%	2%	3%	4%	5%	10%	20%	30%	50%	80%	100%
10%	0	1	1	1	2	3	7	10	16	26	33
20%	1	1	2	3	3	7	13	20	33	52	66
30%	1	2	3	4	5	10	20	30	49	79	98
40%	1	3	4	5	7	13	26	39	66	105	131
50%	2	3	5	7	8	16	33	49	82	131	164
60%	2	4	6	8	10	20	39	59	98	157	197
70%	2	5	7	9	11	23	46	69	115	184	230
80%	3	5	8	10	13	26	52	79	131	210	262
90%	3	6	9	12	15	30	59	89	148	236	295
100%	3	7	10	13	16	33	66	98	164	262	328

Table 13.29 Year-round predicted displacement mortality for razorbill (summed seasonal totals from Table 13.25 through Table 13.28)

Statistic	No. of predicted bird mortalities from displacement					% Increase in baseline mortality	
	Autumn migration	Winter	Spring migration	Breeding	Total*	North Sea and Channel BDMPS, Autumn migration	Biogeographic
30% displacement, 1% mortality							
Mean	1	5	5	0	12	0.01	0.00
LCL	0	4	1	0	5	0.00	0.00
UCL	2	8	15	1	25	0.02	0.01
30% displacement, 10% mortality							
Mean	7	53	52	3	116	0.11	0.04
LCL	0	37	12	0	50	0.05	0.02
UCL	18	76	147	10	252	0.24	0.08
50% displacement, 1% mortality							
Mean	1	9	9	1	19	0.02	0.01
LCL	0	6	2	0	8	0.01	0.00
UCL	3	13	25	2	42	0.04	0.01
70% displacement, 1% mortality							
Mean	2	12	12	1	27	0.03	0.01
LCL	0	9	3	0	12	0.01	0.00
UCL	4	18	34	2	59	0.06	0.02
70% displacement, 2% mortality							
Mean	3	25	24	1	54	0.05	0.02
LCL	0	17	6	0	23	0.02	0.01
UCL	8	36	69	5	117	0.11	0.04
70% displacement, 10% mortality							
Mean	17	125	122	7	271	0.26	0.09

Statistic	No. of predicted bird mortalities from displacement					% Increase in baseline mortality	
	Autumn migration	Winter	Spring migration	Breeding	Total*	North Sea and Channel BDMPS, Autumn migration	Biogeographic
LCL	1	87	29	0	116	0.11	0.04
UCL	42	178	343	23	587	0.56	0.19
*Seasonal numbers are rounded to the nearest integer, so the annual totals may not exactly match the sum of seasonal values							

Year Round

262. The number of razorbills expected to suffer mortality year-round as a result of displacement from the North Falls array area, at a range of displacement and mortality rates, are shown in Table 13.29.

263. These predictions are assessed against the largest BDMPS, 591,874 (UK North Sea and Channel) during the non-breeding (autumn migration) season, and the biogeographic population with connectivity to UK waters, 1,707,000 (Furness 2015). The percentage increase in baseline mortality rates of these populations for each scenario of displacement and mortality of displaced birds, at the average annual mortality of 0.178, is shown in Table 13.29.

264. These increases in mortality rate assume up to 10% mortality of displaced razorbills, which, as discussed above, is considered highly unlikely. At 1% mortality of displaced birds, and a maximum displacement rate of 50%, the increase in population mortality rate for the largest BDMPS would be 0.02% (95% CLs 0.01 – 0.04%). At 70% displacement and 2% mortality of displaced birds, the corresponding increase in mortality rate would be 0.05% (95% CLs 0.02 – 0.11%).

265. All predictions for annual displacement mortality of razorbill represent increases of less than 1% increase in baseline mortality of the North Sea and Channel BDMPS and biogeographic populations.

266. The predicted increases in mortality would not materially alter the background mortality of the BDMPS and biogeographic populations and would be undetectable.

267. The magnitude of predicted year-round displacement mortality to razorbills is assessed as negligible.

Significant of effect

Guillemot

Non-breeding season

268. During the non-breeding season, the impact magnitude is assessed as negligible. As the species is of medium sensitivity to disturbance, the effect significance is minor adverse, which is not significant in EIA terms.

Breeding season

269. During the breeding season, the impact magnitude is assessed as negligible. As the species is of medium sensitivity to disturbance, the effect significance is minor adverse, which is not significant in EIA terms.

Year round

270. The magnitude of predicted year-round displacement mortality to guillemots is assessed as negligible. As the species is of medium sensitivity to disturbance, the effect significance is minor adverse, which is not significant in EIA terms.

Razorbill

Autumn migration

271. During the autumn migration season, the impact magnitude is assessed as negligible. As the species is of medium sensitivity to disturbance, the effect significance is minor adverse, which is not significant in EIA terms.

Winter

272. During the winter period, the impact magnitude is assessed as negligible. As the species is of medium sensitivity to disturbance, the effect significance is minor adverse, which is not significant in EIA terms.

Spring migration

273. During the spring migration period, the impact magnitude is assessed as negligible. As the species is of medium sensitivity to disturbance, the effect significance is minor adverse, which is not significant in EIA terms.

Breeding

274. During the breeding season, the impact magnitude is assessed as negligible. As the species is of medium sensitivity to disturbance, the effect significance is minor adverse, which is not significant in EIA terms.

Year Round

275. The magnitude of predicted year-round displacement mortality to razorbills is assessed as negligible. As the species is of medium sensitivity to disturbance, the effect significance is minor adverse, which is not significant in EIA terms.

13.6.2.1.2 Red-throated diver

276. Displacement from OWFs could influence the survival of individual red-throated divers through increased energetic costs and / or decreased energy intake (via reduced foraging efficiency). The former could arise if birds had to fly greater distances to avoid OWFs or to reach more distant foraging areas. The latter could arise if birds were displaced to lower quality habitat where food capture rates were reduced, and / or if displacement resulted in an increase in the density of divers in foraging areas outside the OWF and a consequent increase in intra-specific competition. Alternatively, displacement may have no effect on individuals if birds are displaced into equally suitable habitat so that their energy budget is unaffected, or if birds could buffer any impact on energy budget by adjusting their time budget (for example by spending a higher proportion of the time foraging rather than resting in order to compensate for an increase in energy budget) (MacArthur Green, 2019c).

Sensitivity of receptor

277. Red-throated divers are considered to have a very high general sensitivity to anthropogenic disturbance and displacement. A range of studies and reviews indicate avoidance of OWFs and associated shipping, and shipping lanes (Garthe and Hüppop, 2004; Bellebaum *et al.*, 2006; Petersen *et al.*, 2006; Schwemmer *et al.*, 2011 Furness and Wade, 2012; Furness *et al.*, 2013;

Bradbury *et al.*, 2014; Percival, 2014; Dierschke *et al.*, 2017; Mendel *et al.*, 2019; Irwin *et al.*, 2019; SNCBs, 2022; Vilela *et al.*, 2021, 2022).

278. Post-construction monitoring studies of OWFs have shown that displacement effects on red-throated diver can occur at considerable distances from OWFs. The joint (UK) SNCBs (2022) advice on displacement of red-throated diver includes a summary of studies from OWFs in the UK, Danish and German North Sea, indicating displacement extending from 0 – 2 to 20km from the turbine array of an OWF. These studies report 55 – 100% (mean of 86% based on eight studies) displacement within the array area of an OWF, and provide evidence (particularly from studies which consider greater distances from OWFs) that the proportion of red-throated divers displaced does not remain constant but declines with distance from the OWF. For example, displacement rates reduce to 12.6% at a distance of 11.5km from the London Array (APEM, 2021). Mendel *et al*. (2019) attributed displacement of divers from OWFs to the combined effect of the wind turbines and shipping traffic associated with the turbine array area, and found that these effects could not be separated out in modelling diver distribution. The assessment of operational displacement for North Falls assumes that this is caused by the turbine array and associated shipping traffic.

Magnitude of impact

279. Natural England has advised for red-throated diver that the ES assessment for displacement from North Falls is based on a displacement rate of 100% within the array area and a 4km buffer, and a mortality rate of 1 – 10% for displaced birds.

280. A recent review (MacArthur Green 2019c) considered that displacement could influence the survival of individual red-throated divers through increased energy costs and / or decreased energy intake. The former could arise if birds had to fly / travel further to avoid OWFs or to reach more distant foraging areas. The latter could arise if birds were displaced to lower quality habitat where food capture rates were reduced, and / or if displacement resulted in localised increases in the density of divers and, hence, increased intra-specific competition for food. Alternatively, displacement may have no effect on individuals if birds are displaced into equally good habitat so that their energy budget is unaffected, or if birds could buffer any impact on energy budget by adjusting their time budgets (for example by spending a higher proportion of time foraging rather than resting in order to compensate for an increase in energetic costs or reduced food intake rate).

281. From the range of 1-10% mortality advised by Natural England, it was considered that a 1% mortality rate for displaced birds is an appropriate precautionary estimate. This is for a number of reasons: red-throated divers appear to utilise a range of offshore habitats and prey species and occur at relatively low densities rather than in large aggregations; they are also highly mobile during the non-breeding season. This flexibility in diet and habitat use indicates displacement from OWFs is unlikely to result in inter-specific competition for prey that might deplete prey resources and affect body condition and survival. The adult mortality rate is estimated at 16% per annum, which will include mortality from existing anthropogenic sources of disturbance and displacement such as shipping traffic. As red-throated divers tend to fly away from approaching vessels, it is likely that the energy costs of this behaviour

exceed the costs of avoiding fixed structures such as OWFs. Thus, it seems biologically implausible that OWF displacement would add substantially to the existing mortality rate of this species.

282. This is supported by long-term studies of red-throated and black-throated divers in the German North Sea, where no changes in the population size during spring migration have been found over the period 2001-2021, despite the construction of 20 OWFs in this area (Vilela *et al*., 2021, 2022); although the divers changed their distribution to avoid OWFs, the population size remained stable, suggesting no or minimal consequences of displacement in increasing mortality amongst displaced birds.

283. Natural England has stated they consider there is insufficient evidence to categorically state that there have been no changes in the red-throated diver population size during spring migration in the German North Sea over the stated period, since there have been changes to survey platform, and presumably survey efficiency, during that period. Furthermore, Leemans & Collier (2022) point out that "*the main construction period of offshore wind farms in the German Bight started in 2012 and the most relevant wind farms (closest to the core area of the birds) became operational in 2014 / 2015. Population level effects may thus not yet have been visible*".

284. Vilela *et al*. (2022), report fluctuations but no trend in RTD population size in spring between 2001-2021, which includes a seven-year period since OWFs became operational in 2014 / 15. If the observed displacement from OWFs in this area were to affect the survival of adult birds using this area during the non-breeding season it might be expected that population level effects would have manifested in this seven-year period. Vilela *et al*. (2022) suggest that in this area, the carrying capacity of the available habitat has not been reached. The effects of displacement on RTDs, if any, may be via body condition and perhaps breeding success. This and earlier studies in the same area (Vilela *et al*., 2021, 2020), use data from visual aerial and digital aerial surveys. It is reported that it was possible to incorporate differences in detection rate between techniques in the statistical analysis. Ship survey data were not included In the analysis as density estimates were considered to have large uncertainties and they were not considered comparable with aerial survey data.

285. Similarly for the Outer Thames Estuary, there is no evidence of population decline since the SPA was classified in 2010; the population estimate has increased by 180% during the period in which five OWFs (including extensions) have been constructed and become operational within 12km of the SPA. Given changes in the survey platforms, from visual aerial to digital aerial (the latter with a higher detection rate for divers), it is not possible to say whether there has been a genuine increase over this period but, nonetheless, it is the case that there is no evidence for a decline in population size.

286. In recognition of the sensitivity of red-throated divers to displacement from OWFs, the time budgets of this species during the non-breeding season have been investigated through fitting time-depth recorder (TDR) and global location sensor (GLS) tags to birds breeding in Finland, Scotland and Iceland (Thompson *et al*., 2023, Duckworth *et al*., 2022, 2020).

287. Birds tagged in Finland migrated through the Baltic Sea during the early part of the non-breeding season, and the southern North Sea, including the Outer Thames Estuary, during the latter part of the non-breeding season. Birds tagged in Iceland remained in Icelandic coastal waters and those from Scotland showing a partial migration, some remaining in Scottish waters and others moving southwest or south-east into coastal waters of north and west Britain and Northern Ireland. Thus, in this study, only birds from Finland were likely to use the Outer Thames Estuary SPA during the non-breeding season and their behaviour is taken to be representative of birds using the SPA. Assuming that other red-throated divers breeding in FennoScandia follow a similar migration pattern to those from Finland (i.e. migrating through the Baltic Sea to the southern North Sea in the latter part of the non-breeding season), this accords with the findings that red-throated diver numbers in the Outer Thames Estuary are highest during the late winter and early spring migration period (January / February, Webb *et al*., 2009).

288. Thompson *et al*. (2023) combined the TDR and GLS data to classify red-throated diver activity into five behaviours: foraging, resting, flight, active on water (e.g. preening) and swimming. During the non-breeding season birds from Finland spent an average of 3.6 (SE (standard error) 0.3) hours foraging per day, varying throughout the season with the shortest foraging time per day in October (when birds were in the Baltic Sea) and the longest time in December and January (when birds were in the southern North Sea); due to limitations of the tags, data was not available for the latter part of the non-breeding season. Foraging occurred almost exclusively during daylight hours. Thompson *et al*. (2023) concluded that temporal and spatial variation in foraging behaviour suggests that during the non-breeding season, red-throated divers may have the capacity to adapt their foraging behaviour to potentially accommodate the energetic costs (if any) of displacement from OWFs, although this is likely to be constrained by factors such as available daylight and food availability. The availability of suitable alternative habitat is important in terms of accommodating the foraging needs of any displaced birds.

289. Natural England (2023b) commented on their review of Thompson *et al*. (2023), that '*data from Finnish tagged birds (that winter in the southern North Sea) shows that from the end of October onwards the percentage of available daylight hours spent foraging increases from 29% in mid-November to 72% in mid-January. This represents an increase from ca 2.5 hours a day spent foraging in November to ca 6 hours a day in January, when there are only 8-8.5 hours of daylight. We also note that tagged birds are breeding adults, i.e., experienced individuals. Juvenile and immature birds may need to devote even more time to foraging if their success rate is lower. Ultimately, the energetic costs of this level of foraging in the depths of winter need to be investigated further, but it appears plausible that in fact red-throated divers are already operating at or close to sustainable limits. Thus, we urge caution in an optimistic reading of the general conclusions made by Thompson et al (2023)*'.

290. Tracking studies of red-throated divers captured in the German North Sea indicate that non-breeding season home ranges are extensive (several thousand square kilometres) such that displacement effects of OWFs will affect only a very small part of individual home ranges (Kleinschmidt *et al*., 2022,

Nehls *et al*., 2018), and divers have access to extensive alternative areas if displaced from part of their home range. Distribution maps indicate that some of the birds captured in the German North Sea subsequently moved to the UK southern North Sea including the Outer Thames area (Kleinschmidt *et al*., 2022, Diverlog 2022 and 2023). Red-throated divers tagged at breeding grounds in Finland also moved extensively during the non-breeding season, through the east and west Baltic Sea to the southern North Sea and the east coast of England (Duckworth *et al*., 2022). Thus there is evidence that red-throated divers using the Outer Thames Estuary during the non-breeding season also have extensive home ranges, such that displacement effects from OWFs would only affect a very small proportion of the area available to these birds. Given these extensive areas used by red-throated divers during the non-breeding season, it seems likely that the effects of displacement, if any, will be minimal and may be via body condition and perhaps subsequent breeding success rather than direct mortality.

291. In the context of possible energetic constraints during the non-breeding season, it is perhaps of note that red-throated divers are rarely reported to suffer mass mortality during seabird ‘wrecks’ (e.g. Clairbaux *et al*., 2021, Camphuysen *et al*., 1999, Harris and Wanless 1996, Underwood and Stowe 1984). Such wrecks are often associated with severe storms which appear to cause starvation due to interfering with the ability to forage and / or affecting the availability of prey to seabirds (Clairbaux *et al*., 2021). A review of the causes of mass mortalities of seabirds reported four wrecks involving red-throated divers in the North Atlantic, compared to 34 for guillemot, 25 for seaduck, 21 for razorbill, and 20 for little auk (all species with a similar ecology to red-throated divers, diving for food from the sea surface); the causes of red-throated diver wrecks were all related to oil contamination, as opposed to food, storms or other causes (Camphuysen *et al*., 1999). This may suggest that red-throated divers are less energetically constrained during the non-breeding season than other seabird species.

292. At North Falls, the largest numbers of red-throated divers were recorded during the late winter and spring migration period (Table 13.17), at which time there is likely to be a turnover of individuals passing through the area. For example, Irwin *et al*. (2019) recorded almost 50% more red-throated divers in a survey on 17th February 2018, compared with 4th February 2018; and APEM (2013) recorded 27% more red-throated divers in surveys on 9th to 12th February 2013 compared with 26th to 27th January 2013). Individuals passing through the area might only be displaced once from on OWF, as opposed to being displaced multiple times if they were resident over a longer period. This might suggest lesser effects of displacement for passage birds, although this would depend on the energy requirements of birds on migration and the available resources for ‘refuelling’ at staging areas. Even if birds spend only a short time in a given area, this could be a critical stopover in terms of their energy needs.

293. The displacement matrices in Table 13.31 and Table 13.31 have been populated with data for red-throated diver during the winter and spring migration periods, within the site and a 4km buffer (mean and 95% CLs), in line with guidance (SNCB, 2017) and advice from Natural England during the EPP.

Autumn migration

294. No red-throated divers were recorded in the North Falls array area and 4km buffer during the autumn migration period (Table 13.17). Thus, there is no predicted additional mortality from displacement, and there would be no increase in the population mortality rate of the relevant BDMPS, the UK North Sea (Furness 2015).

295. Therefore, during the autumn migration period, the impact magnitude is assessed as negligible.

Winter

296. At 100% displacement and 1% mortality, the number of individual red-throated divers which could potentially suffer mortality as a consequence of displacement from the North Falls array area during the winter period has been estimated as zero individuals (95% CLs 0 – 0); at 100% displacement and 10% mortality this is estimated as 2 individuals (95% CLs 0 – 4) (Table 13.30).

297. The relevant BDMPS for red-throated divers is 10,177 (south-west North Sea, Table 13.10; Furness, 2015). At the average baseline mortality rate for red-throated diver of 0.233 (Table 13.11) the number of individuals expected to suffer mortality in the winter BDMPS is 2,371 (10,177 x 0.233). At 1% mortality of displaced birds, there would be no increase in the population mortality rate. At 10% mortality the increase in mortality rate would be 0.08% (95% CLs 0 – 0.19)%. As discussed above, a maximum 1% mortality of displaced birds is considered most likely based on expert opinion.

298. This magnitude of impact would be negligible as the increase in mortality would not materially alter the background mortality of the population and would be undetectable.

Spring migration

299. At 100% displacement and 1% mortality, the number of individual red-throated divers which could potentially suffer mortality as a consequence of displacement from the North Falls array area during the spring migration period has been estimated as 1 individual (95% CLs 0 – 1); at 100% displacement and 10% mortality this is estimated as 7 individuals (95% CLs 1 – 15) (Table 13.31).

300. The relevant BDMPS for red-throated divers is 13,277 (UK North Sea, Table 13.10, Furness, 2015). At the average baseline mortality rate for red-throated diver of 0.233 (Table 13.11) the number of individuals expected to suffer mortality in the spring migration season BDMPS is 3,094 (13,277 x 0.233). At 1% mortality of displaced birds the increase in mortality rate would be 0.02% (95% CLs 0 – 0.05%); at 10% mortality the increase would be 0.21% (95% CLs 0.04– 0.48%). As discussed above, a maximum 1% mortality of displaced birds is considered most likely based on expert opinion.

301. This magnitude of increase in mortality would not materially alter the background mortality of the population and would be undetectable. The impact magnitude is assessed as negligible.

Table 13.30 Displacement matrix for red-throated diver during the winter period. The cells show the number of predicted bird mortalities (to the nearest integer) at a given rate of displacement and mortality (LCL and UCL = upper and lower 95% CLs). Grey cells identify the range of displacement and mortality rates considered in the assessment. Values in red identify scenarios where the baseline mortality rate would increase by >1%

Mean		Mortality										
		1%	2%	3%	4%	5%	10%	20%	30%	50%	80%	100%
Displacement	10%	0	0	0	0	0	0	0	1	1	2	2
	20%	0	0	0	0	0	0	1	1	2	3	4
	30%	0	0	0	0	0	1	1	2	3	5	6
	40%	0	0	0	0	0	1	2	2	4	6	8
	50%	0	0	0	0	1	1	2	3	5	8	10
	60%	0	0	0	0	1	1	2	4	6	10	12
	70%	0	0	0	1	1	1	3	4	7	11	14
	80%	0	0	0	1	1	2	3	5	8	13	16
	90%	0	0	1	1	1	2	4	5	9	14	18
	100%	0	0	1	1	1	2	4	6	10	16	20

LCL		Mortality										
		1%	2%	3%	4%	5%	10%	20%	30%	50%	80%	100%
Displacement	10%	0	0	0	0	0	0	0	0	0	0	0
	20%	0	0	0	0	0	0	0	0	0	0	0
	30%	0	0	0	0	0	0	0	0	0	0	0
	40%	0	0	0	0	0	0	0	0	0	0	0
	50%	0	0	0	0	0	0	0	0	0	0	0
	60%	0	0	0	0	0	0	0	0	0	0	0
	70%	0	0	0	0	0	0	0	0	0	0	0
	80%	0	0	0	0	0	0	0	0	0	0	0
	90%	0	0	0	0	0	0	0	0	0	0	0
	100%	0	0	0	0	0	0	0	0	0	0	0

UCL		Mortality										
		1%	2%	3%	4%	5%	10%	20%	30%	50%	80%	100%
Displacement	10%	0	0	0	0	0	0	1	1	2	4	4
	20%	0	0	0	0	0	1	2	3	4	7	9
	30%	0	0	0	1	1	1	3	4	7	11	13
	40%	0	0	1	1	1	2	4	5	9	14	18
	50%	0	0	1	1	1	2	4	7	11	18	22
	60%	0	1	1	1	1	3	5	8	13	21	27
	70%	0	1	1	1	2	3	6	9	15	25	31
	80%	0	1	1	1	2	4	7	11	18	28	35
	90%	0	1	1	2	2	4	8	12	20	32	40
	100%	0	1	1	2	2	4	9	13	22	35	44

Table 13.31 Displacement matrix for red-throated diver during the spring migration period. The cells show the number of predicted bird mortalities (to the nearest integer) at a given rate of displacement and mortality (LCL and UCL = upper and lower 95% CLs). Grey cells identify the range of displacement and mortality rates considered in the assessment. Values in red identify scenarios where the baseline mortality rate would increase by >1%

Mean		Mortality										
		1%	2%	3%	4%	5%	10%	20%	30%	50%	80%	100%
Displacement	10%	0	0	0	0	0	1	1	2	3	5	7
	20%	0	0	0	1	1	1	3	4	7	11	13
	30%	0	0	1	1	1	2	4	6	10	16	20
	40%	0	1	1	1	1	3	5	8	13	21	26
	50%	0	1	1	1	2	3	7	10	17	26	33
	60%	0	1	1	2	2	4	8	12	20	32	40
	70%	0	1	1	2	2	5	9	14	23	37	46
	80%	1	1	2	2	3	5	11	16	26	42	53
	90%	1	1	2	2	3	6	12	18	30	48	60
	100%	1	1	2	3	3	7	13	20	33	53	66

LCL		Mortality										
		1%	2%	3%	4%	5%	10%	20%	30%	50%	80%	100%
Displacement	10%	0	0	0	0	0	0	0	0	1	1	1
	20%	0	0	0	0	0	0	0	1	1	2	2
	30%	0	0	0	0	0	0	1	1	2	3	4
	40%	0	0	0	0	0	0	1	1	2	4	5
	50%	0	0	0	0	0	1	1	2	3	5	6
	60%	0	0	0	0	0	1	1	2	4	6	7
	70%	0	0	0	0	0	1	2	3	4	7	8
	80%	0	0	0	0	0	1	2	3	5	8	10
	90%	0	0	0	0	1	1	2	3	5	9	11
	100%	0	0	0	0	1	1	2	4	6	10	12

UCL		Mortality										
		1%	2%	3%	4%	5%	10%	20%	30%	50%	80%	100%
Displacement	10%	0	0	0	1	1	1	3	4	7	12	15
	20%	0	1	1	1	1	3	6	9	15	24	30
	30%	0	1	1	2	2	4	9	13	22	36	45
	40%	1	1	2	2	3	6	12	18	30	48	60
	50%	1	1	2	3	4	7	15	22	37	60	75
	60%	1	2	3	4	4	9	18	27	45	72	89
	70%	1	2	3	4	5	10	21	31	52	83	104
	80%	1	2	4	5	6	12	24	36	60	95	119
	90%	1	3	4	5	7	13	27	40	67	107	134
	100%	1	3	4	6	7	15	30	45	75	119	149

Year Round

302. Considering the year-round effects (summed seasonal totals from tables above), which for this species equates to the non-breeding period, the number of red-throated divers expected to suffer mortality as a result of displacement from the North Falls array area and 4km buffer, at a displacement rate of 100% and mortality of 1%, would be one (95% CLs 0 – 2), and at 10% mortality of displaced birds 9 (1 – 19) (Table 13.32). As discussed above, a maximum 1% mortality of displaced birds is considered most likely based on expert opinion.

303. These predictions are assessed against the largest BDMPS, 13,277 during spring and autumn migration, and the biogeographic red-throated diver population with connectivity to UK waters, 27,000 (Furness, 2015).

Table 13.32 Year-round predicted displacement mortality for red-throated diver

Statistic	No. of predicted bird mortalities as a result of displacement				
	Breeding	Autumn migration	Winter	Spring Migration	Total
100% displacement, 1% mortality					
Mean	n/a	0	0	1	1
LCL	n/a	0	0	0	0
UCL	n/a	0	0	1	2
100% displacement, 10% mortality					
Mean	n/a	0	2	7	9
LCL	n/a	0	0	1	1
UCL	n/a	0	4	15	19

304. At the average baseline mortality rate for red-throated diver of 0.233, the number of individuals expected to suffer mortality from the BDMPS over one year is 3,094. At 1% mortality of displaced birds the increase in mortality rate would be 0.03 (95% CLs 0 – 0.06)%; at 10% mortality the increase would be 0.28 (95% CLs 0.04 – 0.62)%.

305. In relation to the biogeographic population, the number of individuals expected to suffer mortality over one year is 6,291. At 1% mortality of displaced birds the increase in mortality rate would be 0.01 (95% CLs 0 – 0.14)%; at 10% mortality the increase would be 0.14 (95% CLs 0.02 – 0.31)%.

306. This magnitude of increase in mortality would not materially alter the background mortality of the population and would be undetectable. The impact magnitude of predicted year-round displacement mortality to red-throated divers is assessed as negligible.

Significance of effect

Autumn migration

307. Due to the negligible impact magnitude and high sensitivity of red-throated diver to disturbance, the effect significance during the autumn migration is minor adverse, which is not significant in EIA terms.

Winter

308. Due to the negligible impact magnitude and high sensitivity of red-throated diver to disturbance, the effect significance during the winter migration is minor adverse, which is not significant in EIA terms.

Spring migration

309. Due to the negligible impact magnitude and high sensitivity of red-throated diver to disturbance, the effect significance during the spring migration is minor adverse, which is not significant in EIA terms.

310. This conclusion is further supported by the likelihood that the spring migration BDMPS population for red-throated divers is an underestimate and therefore the assessment over-estimates the effects on population mortality rate. The UK North Sea Migration BDMPS of 13,277 is less than the current population estimate for the Outer Thames Estuary SPA of 18,079 individuals, the latter based on digital aerial surveys of the SPA in January and February during 2013 and February 2018 (APEM, 2013; Irwin *et al.*, 2019). The UK North Sea BDMPS for red-throated divers includes three additional SPAs for this species during the non-breeding season, at the Greater wash (SPA population estimate 1,407 individuals, the Outer Firth of Forth and St Andrews Bay complex (SPA population estimate 851 individuals) and the Moray Firth (SPA population estimate 324 individuals) (RIAA Section 4.4.1.4.1).

Year Round

311. The impact magnitude of predicted year-round displacement mortality to red-throated divers is assessed as negligible. As discussed above, red-throated diver has high sensitivity to disturbance and displacement, and therefore the effect significance is minor adverse, which is not significant in EIA terms.

312. As noted above, this conclusion is further supported by the likelihood that the spring migration BDMPS population for red-throated divers is an underestimate and therefore the assessment over-estimates the effects on population mortality rate.

13.6.2.2 Effect 2: Collision Risk

313. Birds flying through the WTG arrays of OWFs, whilst foraging for food, commuting between breeding or roosting sites and foraging areas, or during migration, may collide with rotor blades. Collisions are assumed always to result in fatality.

13.6.2.2.1 Sensitivity of receptors

314. The sensitivity to collision risk of seabirds recorded at North Falls is summarised in Table 13.33. The table indicates which species have been screened in and out for assessment and the reasons why.

Table 13.33 Screening for collision risk

Species	Sensitivity to collision risk	Screened in or out	Rationale[1]
Black-headed gull	Medium	In	Species does not appear to be displaced by operational OWFs (Dierschke *et al.*, 2016), and occurs regularly at flight heights where collisions are possible (Johnston *et al.*, 2014a & b).

Species	Sensitivity to collision risk	Screened in or out	Rationale[1]
Common gull	Medium	In	Species does not appear to be displaced by operational OWFs (Dierschke *et al.*, 2016), and occurs regularly at flight heights where collisions are possible (Johnston *et al.*, 2014a & b). Recorded flying in the array area in February and April only during baseline surveys.
Common tern	Medium	In	Species does not appear to be displaced by operational OWFs (Dierschke *et al.*, 2016), and occurs regularly at flight heights where collisions are possible (Johnston *et al.*, 2014a & b). Recorded flying in the array area in baseline surveys in August only.
Cormorant	Low	Out	Species attracted to OWFs (Dierschke *et al.*, 2016) but infrequently flies at heights where collision with rotor blades is possible (Johnston *et al.*, 2014a & b).
Fulmar	Low	Out	Species shows weak avoidance of OWFs (Dierschke *et al.,* 2016) and infrequently flies at heights where collision with rotor blades is possible (Johnston *et al.*, 2014a & b).
Gannet	Medium-low	In	Species quite regularly flies at heights where collision with rotor blades is possible (Johnston *et al.*, 2014a & b), but shows high macro-avoidance of OWFs (Dierschke *et al.,* 2016, Cook *et al.*, 2018, 2021).
Great black-backed gull	Medium	In	Species does not appear to be displaced by operational OWFs (Dierschke *et al.*, 2016), and occurs regularly at flight heights where collisions are possible (Johnston *et al.*, 2014a&b).
Great skua	Medium-low	Out	Unclear whether species shows avoidance of OWFs; sometimes flies at heights where collisions are possible. Not recorded flying in the array area during baseline surveys. Migratory collision risk model run for RIAA, predicts collision mortality per annum of <1 individual (ES Appendix 13.2, Table 3.41 (Document Reference: 3.3.13)).
Guillemot	Low	Out	Species shows some avoidance of OWFs (Dierschke *et al.*, 2016) and infrequently flies at heights where collision with rotor blades is possible (Johnston *et al.*, 2014a & b).
Herring gull	Medium	In	Species does not appear to be displaced by operational OWFs (Dierschke *et al.*, 2016), and occurs regularly at flight heights where collisions are possible (Johnston *et al.*, 2014a & b). Recorded flying in the array area in baseline surveys in April only.
Kittiwake	Medium	In	Species does not appear to be displaced by operational OWFs (Dierschke *et al.*, 2016), and occurs quite regularly at flight heights where collisions are possible (Johnston *et al.*, 2014a & b).
Lesser black-backed gull	Medium	In	Species does not appear to be displaced by operational OWFs (Dierschke *et al.*, 2016), and occurs regularly at flight heights where collisions are possible (Johnston *et al.*, 2014a & b).
Little gull	Medium	In	Species shows weak avoidance of OWFs (Dierschke *et al.*, 2016) but occurs regularly at heights where collision with rotor blades is possible (Johnston *et al.*, 2014a & b). Recorded flying in the array area in baseline surveys in February and November only.

Species	Sensitivity to collision risk	Screened in or out	Rationale[1]
Puffin	Low	Out	Species shows some avoidance of OWFs (Dierschke *et al.*, 2016) and does not fly regularly at heights where collision with rotor blades is possible (Johnston *et al.*, 2014a & b). Not recorded flying in the array area during baseline surveys.
Razorbill	Low	Out	Species shows some avoidance of OWFs (Dierschke *et al.*, 2016) and infrequently flies at heights where collision with rotor blades is possible (Johnston *et al.*, 2014a & b).
Red-throated diver	Low	Out	Species shows strong avoidance of OWFs (Dierschke *et al.*, 2016) and infrequently flies at heights where collision with rotor blades is possible (Johnston *et al.*, 2014a & b).
Sandwich tern	Medium	In	Species shows weak avoidance of OWFs (Dierschke *et al.*, 2016) but flies regularly at heights where collision with rotor blades is possible (Johnston *et al.*, 2014a & b). Recorded flying in the array area in baseline surveys in September only.
1. Species recorded infrequently in the North Falls Array area (present only in 2 or less months) are identified – see ES Appendix 13.2 (Document Reference: 3.3.13) for further information.			

13.6.2.2.2 Magnitude of impact

Collision Risk Modelling

315. CRM has been used in this assessment to predict the risk to seabird populations associated with collisions at the North Falls array area. Based on advice from Natural England, the stochastic CRM (sCRM) (McGregor *et al.*, 2018) has been used to generate predictions of collision mortality for seabird species scoped in for this effect at North Falls. These assessments are made across the relevant biological seasons (see Table 13.10) and annually. The approach to CRM is summarised here and further details are provided in ES Appendix 13.2 (Document Reference: 3.3.13).

316. Collision risk models generate estimates of the number of birds likely to collide with an OWF based on a range of inputs that encompass turbine parameters; turbine numbers and wind farm location; flight density of birds within the OWF array area; and flight height and species-specific characteristics (e.g, body size and flight speed). Full details of the input parameters are provided in ES Appendix 13.2 (Document Reference: 3.3.13). The models generate estimates assuming birds take no action to avoid turbines. A species-specific avoidance rate is then applied to the number of predicted collisions, reflecting empirical evidence that birds change their behaviour to avoid collision (Skov *et al.*, 2018). The avoidance rate is the best available estimate of the percentage of birds of a given species that take action to avoid collision with turbines (and also incorporates model error in the number of birds estimated to pass through the rotor swept area of an OWF over a given time period (flux), and the probability of a birds passing through the rotor swept area and colliding with a blade; Cook 2021). Avoidance rates applied to the North Falls assessment are taken from

the interim guidance update email on CRM parameters that was issued by Natural England (2023).

Stochastic CRM

317. The collision risk assessment for the seabird species screened in for this effect (Table 13.33) for North Falls, is based on Option 2 of the sCRM (in accordance with Natural England (2022b) advice). The sCRM[4] has been run separately for both turbine scenarios (MiRD and MaRD, Table 13.1). Option 2 of the sCRM uses generic estimates of flight height for each species to calculate the percentage of birds flying at Potential Collision Height (PCH). The flight height data are derived from surveys of 32 potential OWF development sites (i.e. pre-construction of turbines) located around the UK and elsewhere in northwestern Europe, as presented in Johnston *et al.* (2014a).

318. The sCRM was run for all species recorded in the North Falls array area that were identified as being of medium and medium-low sensitivity to collision risk on the basis of the criteria set out in Table 13.33..

319. The sCRM allows for the incorporation of statistical variation in the input parameters. Within the sCRMs undertaken for this assessment, measures of variability were incorporated for the following parameters: flight densities (see below); species biometrics, flight speed, avoidance rates and Nocturnal activity factor (NAF) (all based on Natural England (2023) interim guidance update email), as well as flight height data (using the option to select data from Johnston *et al.* (2014a, 2014b) in the sCRM model).

320. Mean monthly densities for birds in flight within the North Falls array area, and the associated SDs, were generated from the baseline aerial survey data as described in ES Appendix 13.2 (Document Reference: 3.3.13). The sCRM was initially run using the ‘truncated normal’ distribution option, inputting mean and SDs of flight densities for each month over the two years of surveys. After the sCRM had been run for all species, updated advice was received from Natural England (2023b), on methodology, advising that monthly flight densities and the associated statistical variability should be estimated directly from the bootstrap samples (with data from surveys in the same calendar months pooled by year) using the ‘distribution samples’ option within the sCRM (rather than the truncated normal distribution). sCRM was re-run for the key species scoped into the assessment (gannet, great black-backed gull, kittiwake and lesser black-backed gull) using the distribution samples method (ES Appendix 13.2 (Document Reference: 3.3.13)).

321. Further background on incorporating statistical variation other input parameters for sCRM is described in ES Appendix 13.2 (Document Reference: 3.3.13); commentary is included below on two key parameters which have been subject to considerable debate and are important in terms of influencing the magnitude of CRM predictions: avoidance rate and nocturnal activity.

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[4] Online Shiny App https://dmpstats.shinyapps.io/avian_stochcrm/

Avoidance rate

322. The avoidance rates used for species for which sCRM was run are set out in Table 13.34. These are based on Natural England (2022c) interim guidance and updates from the in-preparation SNCB CRM guidance note provided by email on 13 September 2023 (Natural England, 2023a); with the only difference between these being that the latter expressed the avoidance rates to four decimal places rather than three.

Table 13.34 Avoidance rates used in sCRM

Species	Avoidance rate (SD) (Natural England 2022b; Natural England 2023a)[1]
Black-headed gull Common gull	0.9949 (±0.0002)
Gannet	0.9928 (±0.0003)
Herring gull Lesser black-backed gull Great black-backed gull	0.9939 (±0.0004)
Kittiwake	0.9928 (±0.0003)
Sandwich tern	0.9907 (±0.0004)
All other species	0.9907 (±0.0004)
1. Avoidance rates from both sources are the same, but given to three decimal places in Natural England (2022b), and four decimal places in Natural England (2023a), with the latter used in the sCRM for North Falls.	

323. As stated above, further review of bird species avoidance rates for use in CRM for OWFs is ongoing and interim guidance has been issued followed by an email with further updates (Natural England, 2022c, Natural England, 2023a). For gannet, a number of studies of behaviour in relation to constructed OWFs suggest this species consistently shows high macro-avoidance (i.e., individuals tend to avoid entering turbine arrays) (see Section 13.6.2.1.1). Therefore, Natural England (2022c) recommends that a correction factor is applied to the density of gannets in flight within the array area, as estimated from the baseline survey data, to account for the effects of macro-avoidance. Pavat *et al*. (2023) determined that although the rate could be applied at various stages of the CRM process, it was most appropriate to apply the rate to the input densities, before the CRM is run. Thus, for the purposes of estimating collision mortality in this assessment, the densities gannets in flight were reduced by 70% prior to undertaking the sCRM.

324. A two-year study at Aberdeen Offshore Wind Farm (AOWFL, 2023), using both radar and video analysis to investigate avoidance behaviour within the wind farm, found that in over 10,000 bird videos, no collisions occurred. This suggests that avoidance rates could be even higher than the recommended guidance.

325. An earlier bird collision avoidance study was conducted at Thanet OWF between 2014 and 2016 (Skov *et al.,* 2018). A detection system including daylight and thermal imaging cameras recorded six collisions of birds with turbine rotor blades during the course of the study. These were all gulls including one kittiwake and one great black-backed gull (the others were not identified to species). Empirical avoidance rates were estimated for five seabird species as

follows: 99.9% for gannet and herring gull, 99.8% for kittiwake and lesser black-backed gull, and 99.6% for great black-backed gull. Bowgen and Cook (2019) reviewed the findings of the above study, and recommended avoidance rates for use in the sCRM of 99.7% (95% CLs 99.2-99.9%) for gannet and large gulls and 99.4% (95% CLs 97.6– 99.8%) for kittiwake, higher than the latest Natural England advice .

326. The latest recommendations from Natural England (2023, 2022b) are based on a review and subsequent analysis of data to calculate avoidance rates for seabird species for use with CRM outputs (Ozsanlav-Harris *et al*., 2023). This includes the Thanet study and a number of earlier studies at coastal OWFs in England and northern Europe where data on flight activity was collected alongside carcass searches to assess collision rates. However, the estimates on which the latest advised avoidance rates are based exclude data from the avoidance behaviour study at the Aberdeen OWF, which along with the data from the bird collision avoidance study at the Thanet OWF, arguably represent some of the most relevant data that are currently available for estimating avoidance rates at OWFs.

Nocturnal Activity Factor

327. The nocturnal activity parameter defines the level of nocturnal flight activity of each seabird species, expressed in relation to daytime flight activity levels. For example, a value of 50% for the NAF is appropriate for a species which is half as active at night as during the day. 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 and night.

328. The starting point for identifying nocturnal activity rates was a review of seabird activity reported in Garthe and Hüppop (2004), which ranked species from 1 to 5 (1 low, 5 high) for relative nocturnal activity based on limited existing evidence, and consultation with a panel of experts. These scores were subsequently modified for the purposes of CRM into 1 = 0% to 5 = 100%, which assumes a linear ranking scale (Band 2012). 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 quantitative 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'.

329. A review of evidence for nocturnal activity of gannet from tracking studies (Furness *et al.*, 2018) found average rates of 7.1% for the breeding and 2.3% for the non-breeding season. The review recommended precautionary nocturnal flight activity rates for gannet in the breeding and non-breeding seasons of 8% and 4% respectively. This compares with a NAF of two assigned to the species by Garthe and Hüppop (2004), which (as it has been used in the CRM) translates into 25% nocturnal activity. For kittiwake, review and analysis of activity data from tracking studies (Furness, 2019) identified a nocturnal activity rate for the non-breeding seasons of 17%. This compares with a NAF of three assigned by Garthe and Hüppop (2004), translating into 50% nocturnal activity (in terms of how it has been applied in the CRM).

330. Thus comparing the nocturnal activity scores of two species with empirical evidence, has indicated that the scores derived from the Garthe and Hüppop (2004) ranking are precautionary, in terms of estimating nocturnal activity relative to daytime. 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 one (e.g. from 3 to 2 for kittiwake) has been investigated previously (APEM, 2015). This predicted reductions in annual mortality estimates of between 14.5% (lesser black-backed gull) and 28.6% (gannet). This suggests NAFs based on arbitrary conversions of the Garthe and Hüppop (2004) scores into percentages are over-estimated, and consequently CRM outputs based on these factors are highly precautionary.

331. As the relative proportion of daytime to night-time varies considerably during the year at the UK's latitude, it is also the case that the effect of changes in the NAF for CRM outputs depends on the relative abundance of birds throughout the year.

332. Nocturnal activity rates used for North Falls sCRM are as recommended in Natural England (2023a) interim guidance update email where available for a given species, or otherwise based on Garthe and Hüppop (2004) (Table 13.35). The value of 0.375 ± 0.0637) for gulls captures NAFs of 0.25 and 0.5 in the 95% CLs.

Table 13.35 NAFs used in sCRM.

Species	NAF (±SD) for sCRM (from Natural England 2022b)	Nocturnal Activity Score Garthe & Hüppop (2004)
Black-headed gull	0.5*	2
Common gull	0.5*	3
Common tern	0*	1
Little gull	0.375 (±0.0637)	2
Gannet	0.08 (±0.1)	2
Great black-backed gull	0.375 (±0.0637)	3
Herring gull	0.375 (±0.0637)	3
Kittiwake	0.375 (±0.0637)	3
Lesser black-backed gull	0.375 (±0.0637)	3
Sandwich tern	0*	1
*Species with no specific recommendation in Natural England Interim guidance (2023), based on Garthe and Hüppop (2004) (except common gull where 0.5 has been used rather than 0.75)		

sCRM predictions

333. Monthly collision estimates for North Falls with associated statistics, for all turbine design scenarios and each species for which sCRM was run, are included in ES Appendix 13.2 (Document Reference: 3.3.13).

334. Annual collision risk estimates for species scoped in are included in Table 13.36 below along with 95% CLs. In each case the predicted annual collision risk is very similar for the two turbine scenarios, with the maximum (turbine size) design being higher for half of the species' (34 turbines with rotor diameter of

337m), and the minimum scenario (57 turbines with a rotor diameter of 236m) (see Table 13.1) for the other half. The highest predicted annual collision risk for each species is denoted by grey shading in the tables below.

335. Collision risk assessments are presented by species below, based on predicted collision mortality and the percentage increase in mortality rate for the relevant reference population (increases in mortality rates calculated as described previously (Section 13.5.4). Although initially scoped in for collision risk assessment (Table 13.33), species where the mean annual collision risk at North Falls was subsequently estimated to be very low (<2 birds per annum) were scoped out of further assessment. This applies to black-headed gull, common gull, common tern, herring gull, little gull and Sandwich tern (Table 13.36). This approach was also adopted for offshore ornithology at PEIR.

336. The 95% CLs associated with annual collision risk estimates in Table 13.36 are derived from the sCRM outputs and are understood to be based on bootstrapped estimates for annual collision risk generated by the sCRM. The sCRM does not however generate bootstrapped results for seasonal collision risk, so the seasonal 95% CLs in the species tables below (Table 13.37 to Table 13.40) are derived from summing the 95% CLs from monthly estimates (see tables in ES Appendix 13.2, Section 3.1.2 (Document Reference: 3.3.13)), thus the sum of seasonal values for LCL and UCL does not match the annual LCL and UCL values.

Table 13.36 Annual collision risk for seabirds at North Falls for all turbine layouts (grey shading indicates design with highest estimate)

Species	Annual Collision Risk, Mean (95% CLs)	
	MiRD Scenario	MaRD Scenario
Black-headed gull	0.659 (0.145 – 1.891)	0.725 (0.145 – 2.002)
Common gull	1.689 (0.376 – 3.896)	1.675 (0.377 – 3.935)
Common tern	0.186 (0.01 – 0.477)	0.214 (0.013 – 0.595)
Gannet	2.097 (0.585 – 4.786)	2.015 (0.587 – 4.289)
Great black-backed gull	3.04 (0 – 7.989)	3.02 (0 – 7.664)
Herring gull	0.688 (0.031 – 1.947)	0.703 (0.042 – 1.862)
Kittiwake	19.163 (7.387 – 39.341)	20.235 (7.368 – 43.593)
Lesser black-backed gull	8.531 (1.63 – 21.32)	8.549 (1.669 – 20.969)
Little gull	1.541 (0 – 20.48)	1.013 (0 – 15.543)
Sandwich tern	0.201 (0.016 – 0.707)	0.13 (0.012 – 0.44)

Gannet

337. Predicted seasonal and annual mortality of gannets at North Falls for both turbine scenarios are shown in Table 13.37, along with corresponding percentage increases in mortality rates of reference populations. For the breeding season, as for operational displacement, the predicted mortality has been assigned to the Flamborough and Filey Coast SPA population (51,560 breeding adults and associated sub-adult birds see Section 13.6.2.1.1). For all

other seasons the reference population is the BDMPS as defined by Furness (2015) (see Table 13.10). The annual collision risk is presented as a percentage increase in the largest seasonal BDMPS (autumn migration) and the biogeographic population with connectivity with connectivity to UK waters (Furness, 2015).

338. For both turbine scenarios, the predicted seasonal and annual mortality from collisions at North Falls represents less than a 1% increase in the predicted mortality rate of the corresponding reference population. The highest predicted collision risk is for the MiRD (57 turbines with rotor diameter of 236m) (shown in grey in Table 13.37).

339. All magnitudes of increase in mortality would not materially alter the background mortality of the population and would be likely to be undetectable against natural variation. Therefore, for all seasons and annually, the impact magnitude is assessed as negligible.

340. Under the latest guidance from Natural England and NRW on EIA reference populations (see para 68), the breeding season BDMPS is 400,326 individuals for the UK North Sea and Channel, applying this, the percentage increases in baseline mortality during the breeding season, would be even smaller than those given in the table.

Table 13.37 Seasonal and annual collisions for gannet at North Falls and increase in population mortality rates

WTG scenario	Statistic	Predicted collisions (sCRM)				% increase in population mortality rate				
		Breed-full	Aut-mig	Spr-mig	Annual	Breed-full	Aut-mig	Spr-mig	Annual Max BDMPS	Annual Biogeographic
MiRD	Mean	0.57	0.89	0.64	2.10	0.01	0.00	0.00	0.00	0.00
	LCL	0.03	0.13	0.03	0.59	0.00	0.00	0.00	0.00	0.00
	UCL	1.78	2.35	2.11	4.79	0.02	0.00	0.00	0.00	0.00
MaRD	Mean	0.55	0.85	0.61	2.02	0.01	0.00	0.00	0.00	0.00
	LCL	0.03	0.15	0.03	0.59	0.00	0.00	0.00	0.00	0.00
	UCL	1.74	2.19	1.97	4.29	0.02	0.00	0.00	0.00	0.00
Reference populations (all from Furness (2015) except breeding season, see para 337.						51,560	456,298	248,385	456,298	1,180,000
Annual mortality at 'average' rate of 0.187 (Table 13.11)						9,600	85,328	46,448	85,328	220,660

Great black-backed gull

341. Predicted seasonal and annual mortality of greater black-backed gulls at North Falls for both turbine scenarios is shown in Table 13.38, along with corresponding percentage increases in mortality rates of reference populations. There are no large breeding colonies of great black backed gull within foraging range of North Falls (Maximum Foraging Range (no SD) 74km) (Woodward *et al.*, 2019; only one study available so no value for MMFR). According to the SMP[2] database, 0 – 4 pairs have been recorded annually at the Alde-Ore Estuary SPA from 2000 – 2018 (the species is not an SPA qualifying feature), and seven pairs at Felixstowe Docks in 2013. For the breeding season (migration free), the reference population has been estimated as 52,829 individuals (57.8% of the non-breeding BDMPS, based on the proportions of adults (44%), juveniles (25%) and older immatures (31%) from the population model in Furness, 2015). For the non-breeding season, the reference population is the UK North Sea BDMPS as defined by Furness (2015) (91,399 individuals, Table 13.10). The annual collision risk is presented as a percentage increase in the largest seasonal BDMPS (non-breeding) and the biogeographic population with connectivity with connectivity to UK waters (Furness, 2015).

342. For all scenarios, the predicted seasonal and annual mortality from collisions at North Falls represents less than a 1% increase in the predicted mortality rate of the corresponding reference population. The highest annual predicted collision risk, 3.04 (95% CI 0 – 7.99) is for the MiRD (57 turbines with rotor diameter of 236m) (shown in grey in Table 13.38) (although there is very little difference between the two turbine scenarios). This represents an increase of 0.04% (0 – 0.09%) in the mortality rate of the largest (non-breeding season) BDMPS and 0.01% (0 – 0.04%) for the biogeographical population with connectivity to UK waters.

343. All magnitudes of increase in mortality would not materially alter the background mortality of the population and would be undetectable. Therefore, for all seasons and annually, the impact magnitude is assessed as negligible.

344. Under the latest guidance from Natural England and Natural Resources Wales on EIA reference populations (see para 68), the breeding season BDMPS is 25,917 individuals for the UK North Sea; if this were to be applied, there would however be no change to percentage increases in baseline mortality during the breeding season, as no collisions were predicted.

Table 13.38 Seasonal and annual collisions for great black-backed gull at North Falls and increase in population mortality rates

WTG scenario	Statistic	Predicted collisions (sCRM)			% increase in population mortality rate			
		Non-Breeding	Breed – Migration free	Annual	Non-Breeding	Breed-Migration free	Annual Max BDMPS	Annual Biogeo
MiRD	Mean	3.04	0.00	3.04	0.04	0.00	0.04	0.01
	LCL	0.00	0.00	0.00	0.00	0.00	0.00	0.00
	UCL	13.53	0.00	7.99	0.16	0.00	0.09	0.04
MaRD	Mean	3.02	0.00	3.02	0.04	0.00	0.04	0.01
	LCL	0.00	0.00	0.00	0.00	0.00	0.00	0.00
	UCL	13.38	0.00	7.66	0.16	0.00	0.09	0.04
Reference populations (see para 341)					91,399	52,829	91,399	235,000
Annual mortality at 'average' rate of 0.093 (Table 13.11)					8,500	4,913	8,500	21,855

Kittiwake

345. Predicted seasonal and annual mortality of kittiwakes at North Falls for both turbine scenarios is shown in Table 13.39, along with corresponding percentage increases in mortality rates of reference populations.

346. Consideration was given to potential connectivity with the Flamborough and Filey Coast SPA during the breeding season. Assuming that 53% of the population comprises adults (Furness 2015), the total SPA population is estimated as 168,204 adult and associated subadult individuals.

347. The SPA is the nearest large breeding colony of kittiwakes to North Falls, a by-sea distance of 279km north-west of the array area at the nearest point, As the SPA boundary includes a 2km marine extension, North Falls is approximately 299km from the nearest coastal area within the SPA where kittiwakes might nest. The MMFR plus 1SD for kittiwake is 300.6km (Woodward *et al.*, 2019). While strictly applying this distance would mean the North Falls array is just within foraging range of kittiwakes breeding at Flamborough and Filey Coast SPA, the evidence from tracking studies of kittiwake at the SPA indicates that kittiwakes from SPA colonies do not travel as far south as North Falls. For example, a tracking study of kittiwakes breeding at Flamborough and Filey Coast SPA in 2017 found an average foraging range of 88.65km (range 3.2 – 324km), with birds travelling into the North Sea, north-east and south-east of the breeding colony (Wischnewski *et al.*, 2017) although none as far south as the North Falls array area. (See RIAA Part 4 Section 4.4.4.5.2 (Document Reference: 7.1.4) for further detail on kittiwake tracking studies from Flamborough and Filey Coast).

348. While RSPB's Future of the Atlantic Marine Environments (FAME) studies have shown some extremely long foraging trips for this species (as reported in various publications such as Fair Isle Bird Observatory annual reports) those extreme values tend to occur at colonies where food supply is extremely poor and breeding success is low (for example in 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 zero chicks per nest, but less than 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.

349. It is concluded that there is no breeding season connectivity with the Flamborough and Filey Coast SPA. It is therefore considered likely that kittiwakes present at the North Falls array area during the breeding season comprise a combination of non-breeding adult birds (Horswill and Robinson (2015) estimate that 18.0 – 20.8% of adult kittiwakes opt out of breeding in a

given year), some breeding birds from smaller colonies on the Suffolk coast closer to North Falls (Sizewell Rigs and Lowestoft Harbour), and subadult birds (noting that kittiwakes adopt adult plumage by their third year but (on average) do not start to breed until four years old (Coulson 2011), and so a proportion of birds recorded in adult plumage during offshore surveys will be immatures). As for guillemot and razorbill, the breeding season BDMPS is estimated as the proportion of the non-breeding BDMPS comprised of subadult birds (47% based on the modelled age structure for this species, Furness, 2015), noting that this will probably underestimate the size of the reference population because it includes only one of the above three categories of birds that are likely to use the North Falls array area. For kittiwake this is taken as 390,070 individuals, 47% of the largest non-breeding BDMPS for the UK North Sea (Table 13.10).

350. Outside the breeding season, the reference populations in Table 13.39 are the seasonal BDMPS's as defined by Furness (2015) (see Table 13.10). The annual collision risk is presented as a percentage increase in the largest seasonal BDMPS (autumn migration) and the biogeographic population with connectivity with connectivity to UK waters (Furness, 2015).

351. For all scenarios, the predicted seasonal and annual mortality from collisions at North Falls represents a less than 1% increase in the predicted mortality rate of the corresponding reference population (Table 13.39). The highest predicted collision risk is for the MaRD (34 turbines with rotor diameter of 337m) (shown in grey in Table 13.39, with estimated increases in seasonal mortality rates of 0 – 0.03% (95% CI 0 – 0.09%), and annual mortality rates by 0.02% (95%CI 0 – 0.03%) for the largest seasonal BDMPS and 0% for the biogeographic population.

352. All predicted magnitudes of increase in mortality would not materially alter the background mortality of the population and would be undetectable. Therefore, for all seasons and annually, the impact magnitude is assessed as negligible.

353. Under the latest guidance from Natural England and Natural Resources Wales on EIA reference populations (see para 68), the breeding season BDMPS is 839,456 individuals for the UK North Sea, and this is also the largest seasonal BDMPS; if this were applied, the percentage increases in baseline mortality during the breeding season, and year round, would be even smaller than those given in the table.

Table 13.39 Seasonal and annual collisions for kittiwake at North Falls and increase in population mortality rates

WTG scenario	Statistic	Predicted collisions (sCRM)				% increase in population mortality rate				
		Aut-mig	Spr-mig	Breed-full	Annual	Aut-mig	Spr-mig	Breed-full	Annual Max BDMPS	Annual Biogeographic
MiRD	Mean	3.42	7.62	8.13	19.16	0.00	0.01	0.01	0.01	0.00
	LCL	0.39	0.85	0.46	7.39	0.00	0.00	0.00	0.01	0.00
	UCL	9.30	29.93	23.36	39.34	0.01	0.03	0.04	0.03	0.00
MaRD	Mean	3.64	7.83	8.76	20.24	0.00	0.01	0.01	0.02	0.00
	LCL	0.48	0.94	0.45	7.37	0.00	0.00	0.00	0.01	0.00
	UCL	9.86	30.43	24.56	43.59	0.01	0.03	0.04	0.03	0.00
Reference populations, Furness (2015) except breeding season see para 349)						829,937	627,816	360,070	829,937	5,100,000
Annual mortality at 'average' rate of 0.157 (Table 13.11)						130,300	98,567	61,241	130,300	800,700

Lesser black-backed gull

354. Predicted seasonal and annual mortality of lesser black-backed gulls at North Falls for both turbine scenarios and lower and upper nocturnal activity rates is shown in Table 13.40, along with corresponding percentage increases in mortality rates of reference populations. The estimates of collision risk for both turbine scenarios are very similar, being only slightly larger for the maximum turbine scenario (shown in grey in the table).

355. The MMFR plus 1SD of lesser black-backed gull is 236km (Woodward *et al.*, 2019). Thus, birds present during the breeding season could potentially include breeding adults from nesting colonies within this distance from North Falls. This includes breeding birds from the Alde-Ore Estuary SPA, 39.1km from North Falls at the nearest point (a tracking study of lesser black-backed gulls breeding at Orfordness within the SPA indicated that the foraging ranges overlapped with the North Falls array area in one out of three breeding seasons; Thaxter *et al.,* 2015). There are also a number of urban nesting colonies of lesser black-backed gulls within potential foraging range of North Falls. The breeding season reference population has been estimated based on monitoring data of breeding birds within the SPA from the SMP and available data on urban-nesting gulls (principally a survey of lesser black-backed gulls in Suffolk and south Norfolk in 2012; Piotrowski, 2013). The reference population is estimated as 15,873 birds of all age classes, comprising the estimated breeding population from the Alde-Ore Estuary SPA (1,880 pairs, 2014 – 2023) plus the estimated breeding population from urban sites in Suffolk and south Norfolk in 2012 (2,882 pairs) (See RIAA, Section 4.4.2.5.1 for background) multiplied up to include associated sub-adult birds based on stable population proportions of 60% adults, 15% juveniles and 25% immatures (Furness, 2015).

356. While the counts of urban gulls are from 12 years ago, more recent count data for such sites are not consistently available in the SMP database[2] (as of September 2022). Given the continuing increase in occupation of urban habitats by nesting lesser black-backed gulls (Burnell 2021), it is assumed that the 2012 data represent at least a minimum estimate of the current urban nesting population in the surveyed area. Gulls nesting in urban environments (often on elevated surfaces such as flat roofs) are more difficult to count accurately than in natural sites (Burnell 2021, Ross *et al.,* 2016), and abnormally wet and cold weather in April and May 2012 was likely to have caused premature failure of some nests, so the estimate from Piotrowski (2013) is likely to be conservative.

357. Outside the breeding season, the reference populations in Table 13.40 are the seasonal BDMPS' as defined by Furness (2015) (see Table 13.10). The annual collision risk is presented as a percentage increase in the mortality rate of the largest seasonal BDMPS (autumn migration) and the biogeographic population with connectivity to UK waters (Furness 2015).

Table 13.40 Seasonal and annual collisions for lesser black-backed gull at North Falls and increase in population mortality rates

WTG scenario	Statistic	Predicted collisions (sCRM)					% increase in population mortality rate					
		Breed-full	Aut-mig	Winter	Spr-mig	Annual	Breed-full	Aut-mig	Winter	Spr-mig	Annual Max BDMPS	Annual Biogeographic
MiRD	Mean	6.41	0.80	1.32	0	8.53	0.32	0.00	0.03	0	0.03	0.01
	LCL	0.00	0.00	0.00	0	0.00	0.00	0.00	0.00	0	0.01	0.00
	UCL	22.01	4.73	5.84	0	32.58	1.11	0.02	0.12	0	0.08	0.02
MaRD	Mean	6.52	0.80	1.23	0	8.55	0.33	0.00	0.02	0	0.03	0.01
	LCL	0.00	0.00	0.00	0	0.00	0.00	0.00	0.00	0	0.01	0.00
	UCL	22.22	5.28	5.75	0	33.25	1.12	0.02	0.12	0	0.08	0.02
Reference populations, Furness (2015) except breeding season see para 355.							15,873	209,007	39,314	197,483	209,007	864,000
Annual mortality at 'average' rate of 0.125 (Table 13.11)							1,984	26,126	4,914	24,685	26,126	108,000

358. For all scenarios except the upper 95% CLs associated with both scenarios during the breeding season, the predicted seasonal and annual mortality from collisions at North Falls represents less than a 1% increase in the predicted mortality rate of the corresponding reference population (Table 13.40). It is considered highly unlikely that collisions would reach the upper 95% CL rate in multiple years, or indeed at all. All other predicted magnitudes of increase in mortality would not materially alter the background mortality of the population and would be undetectable. For all seasons and annually, the impact magnitude is assessed as low.

359. Under the latest guidance from Natural England and Natural Resources Wales on EIA reference populations (see para 68), the breeding season BDMPS is 51,233 individuals for the UK North Sea and Channel; if this were applied, the percentage increases in baseline mortality during the breeding season, would be smaller than those given in the table, and predictions for the 95% UCL would represent a <1% increase in baseline mortality.

13.6.2.2.3 Significance of effect

Gannet

360. For all seasons and annually, the impact magnitude is assessed as negligible. As gannet is of medium-low sensitivity to collision risk, the effect significance is minor adverse, which is not significant in EIA terms.

Great black-backed gull

361. For all seasons and annually, the impact magnitude is assessed as negligible. As great black-backed gull is of medium sensitivity to collision risk, the effect significance is minor adverse, which is not significant in EIA terms.

Kittiwake

362. For all seasons and annually, the impact magnitude is assessed as negligible. As kittiwake is of medium sensitivity to collision risk, the effect significance is minor adverse, which is not significant in EIA terms.

Lesser black-backed gull

363. For all seasons and annually, the impact magnitude is assessed as low. As lesser black-backed gull is of medium sensitivity to collision risk, the effect outcome is minor adverse, which is not significant in EIA terms.

13.6.2.3 Combined operational collision risk and displacement (Effects 1 and 2)

364. For species subject to collision risk and displacement from OWFs, these effects could combine to adversely affect populations. Obviously, collision and displacement would not act on the same individuals, as birds which do not enter a wind farm cannot be subject to mortality from collision, and vice versa. Thus, birds which exhibit macro-avoidance (do not enter an OWF turbine array) could be subject to mortality from displacement. Only one species, gannet, has been scoped in for effects of both collision and displacement. This section considers the potential combined effects.

13.6.2.3.1 Gannet

365. The combined seasonal and annual collision and displacement risks for gannets at North Falls are shown in Table 13.41. A range for the combined effects is given, based on the worst case collision mortality (MiRD scenario, 57 turbines, 236m rotor diameter) and the two alternative displacement scenarios of 60% displacement and 1% mortality of displaced birds, and 80% displacement and 1% mortality.

366. Under all scenarios the seasonal and annual predicted mortality for collision and displacement represents less than a 1% increase in the predicted annual mortality of the corresponding reference population (Table 13.41) (reference populations are as previously defined, e.g., see para 337). All magnitudes of increase in mortality would not materially alter the background mortality of the population and would be undetectable. Therefore, for all seasons and annually, the impact magnitude is assessed as negligible. As gannet is of medium-low sensitivity to collision risk, and medium sensitivity to displacement, the effect significance is minor adverse, which is not significant in EIA terms.

367. Under the latest guidance from Natural England and Natural Resources Wales on EIA reference populations (see para 68), the breeding season BDMPS is 400,326 individuals for the UK North Sea and Channel; applying this, the percentage increases in baseline mortality during the breeding season, would be even smaller than those given in the table.

Table 13.41 Year-round predicted displacement and collision mortality for gannet.

Statistic	No. of predicted bird mortalities				% Increase in baseline mortality of reference population				
	Autumn Migration	Spring Migration	Breeding	Annual	Autumn Migration	Spring Migration	Breeding	Annual BDMPS	Biogeo-graphic
(i) 60% Displacement, 1% mortality									
Mean	2	2	0	4	0.00	0.00	0.00	0.00	0.00
LCL	1	0	0	1	0.00	0.00	0.00	0.00	0.00
UCL	3	4	1	8	0.00	0.01	0.01	0.01	0.00
(ii) 80% Displacement, 1% mortality									
Mean	2	2	1	5	0.00	0.00	0.01	0.01	0.00
LCL	1	0	0	1	0.00	0.00	0.00	0.00	0.00
UCL	5	5	1	11	0.01	0.01	0.01	0.01	0.01
(iii) Collision: MiRD scenario									
Mean	1	1	1	2	0.00	0.00	0.01	0.00	0.00
LCL	0	0	0	1	0.00	0.00	0.00	0.00	0.00
UCL	2	2	2	5	0.00	0.00	0.02	0.00	0.00
Collision MiRD + displacement 60%/1% (i+iii)									
Mean	3	2	1	6	0.00	0.01	0.01	0.01	0.00
LCL	1	0	0	1	0.00	0.00	0.00	0.00	0.00
UCL	6	6	3	13	0.01	0.01	0.03	0.02	0.01
Collision MiRD + displacement 80%/1% (ii+iii)									
Mean	3	3	1	7	0.00	0.01	0.01	0.01	0.00
LCL	1	0	0	2	0.00	0.00	0.00	0.00	0.00
UCL	7	7	3	16	0.01	0.02	0.03	0.02	0.01
*Seasonal numbers are rounded to the nearest integer, so the annual totals may not exactly match the sum of seasonal values, and similarly the sums of collision and displacement for a given season, or annually may not exactly match the component values.									

13.6.2.4 Effect 3: Indirect effects via effects on habitats and prey species

368. Indirect disturbance and displacement of birds may occur during the operational phase of North Falls 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 WTGs), electromagnetic fields (EMF) and the generation of suspended sediments (e.g. due to scour and 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 including invasive non-native species) may also occur, and changes in fishing activity could influence the communities present.

369. ES Chapter 11 Fish and Shellfish Ecology (Document Reference: 3.1.13) discusses the likely significant effects upon fish relevant to ornithology as prey species. The sensitivity of fish and shellfish species to operational noise is considered to be low and the impact magnitude negligible. A negligible adverse effect on fish is concluded. With regard to changes to the seabed and to suspended sediment levels, ES Chapter 10 Benthic and Intertidal Ecology (Document Reference: 3.1.12) discusses the nature of any change and impact. It identifies that the small quantities of sediment released due to maintenance activities and scour processes would rapidly settle within a few hundred metres of each WTG or cable protection structure. The magnitude of the impact is negligible and the effect significance minor adverse. With regard to EMF effects, the magnitude of impact is considered negligible on benthic invertebrates (ES Chapter 10 Benthic and Intertidal Ecology, (Document Reference: 3.1.12)), negligible on fish and low on lobster, crab and elasmobranchs (ES Chapter 11 Fish and Shellfish Ecology (Document Reference: 3.1.13)).

370. All offshore ornithology receptors are considered to have a medium sensitivity to effects on prey species. This is because, while they depend on the availability of prey, under most environmental conditions, they have the capability to exploit alternative foraging areas if prey is depleted or unavailable in a given foraging are. A As above, any such effects are expected to be negligible and localised. As affected seabird species will forage over a wide area (relative the potential impacts on prey) and will necessarily exhibit some flexibility in the areas within which they forage, it is considered very unlikely that these small, localised, impacts would result in significant effects on seabirds’ ability to forage. Based on this, it is concluded that the indirect impact on seabirds occurring in or around the North Falls array area during the operational phase is similarly a negligible adverse effect, which is not significant in EIA terms.

371. The impact of the colonisation of introduced hard substrate is assessed as a minor adverse effect in terms of benthic ecology (ES Chapter 10 Benthic and Intertidal Ecology (Document Reference: 3.1.12)), and of negligible significance for fish and shellfish (ES Chapter 11 Fish and Shellfish Ecology (Document Reference: 3.1.13)). The consequences for seabirds may be positive or negative locally (for example, new substrates could provide new or additional

habitat for prey species, or conversely result in localised loss of suitable habitat) but are not predicted to be significant (either beneficially or adversely) in EIA terms, at a wider scale.

Likely significant effects during decommissioning

372. Two effects on bird populations during the decommissioning phase of North Falls have been screened in. These are:

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

373. As a worst-case, any effects generated during the decommissioning phase are expected to be similar to those generated during the construction phase. This is because decommissioning would generally involve a reverse of the construction phase through the removal of some structures and materials installed.

374. It is anticipated that decommissioning activities would be programmed in close consultation with the relevant statutory marine and nature conservation bodies, to allow any future guidance and industry good practice to be incorporated to minimise likely significant effects.

13.6.2.5 Effect 1: Direct disturbance and displacement

375. Disturbance and displacement is likely to occur due to the presence of working vessels and crews and the movement, noise and light associated with these. Such activities have already been assessed for relevant bird species in the construction section above and have been assessed as negligible to minor adverse.

376. As discussed above, impacts during the decommissioning phase of North Falls are expected, as a worst-case, to be of similar magnitude compared to construction and therefore, the magnitude of impact is predicted to be negligible. This magnitude of impact, applied to the range of species scoped in for the construction-phase assessment (Section 13.6.1.1), of medium to high sensitivity to disturbance, gives an assessment outcome of negligible to minor adverse, which is not significant in EIA terms.

13.6.2.6 Effect 2: Indirect effects through effects on habitats and prey species

377. Indirect effects such as displacement of seabird prey species are likely to occur as structures are removed. As for the construction-phase assessment, these indirect effects include those resulting from the production of underwater noise and the generation of suspended sediments, that may alter the behaviour or availability of seabird prey species. The effects on benthic habitats and prey species (i.e. fish species such as herring, sprat and sandeel) have been considered in ES Chapter 10 Benthic and Intertidal Ecology (Document Reference: 3.1.12) and ES Chapter 11 Fish and Shellfish Ecology (Document Reference: 3.1.13).

378. Any impacts generated during the decommissioning phase of the proposed project are, as worst-case, expected to be similar to those during the construction phase, as described in Section 13.6.1.2, above. All offshore ornithology receptors are considered to have a medium sensitivity to effects on prey species (see para 370 above). Any such effects are expected to be localised and temporary, acting over very small areas compared with the

foraging ranges of seabird species such that is highly unlikely that these impacts on prey would result in significant effects on seabirds' ability to forage; therefore, the magnitude of effect is predicted to be negligible. This magnitude of impact on seabird species of medium sensitivity to effects on prey species, gives an assessment outcome of minor adverse, which is not significant in EIA terms.

13.7 Monitoring

379. Monitoring requirements for North Falls are outlined in the In-Principle Monitoring Plan (IPMP) (document reference 7.10).

380. Monitoring is proposed to:

- Determine whether there is a change in abundance and distribution of red-throated diver within the array area and appropriate buffer zones following construction of the wind farm; and
- Record potential collisions with wind turbine blades.

13.8 Cumulative effects

13.8.1 Identification of potential cumulative effects

381. The first step in the CEA process is the identification of which residual effects assessed for North Falls on their own have the potential for a cumulative effect with other plans, projects and activities. This information is set out in Table 13.42.

Table 13.42 Potential cumulative effects

Impact	Potential for cumulative effect	Rationale
Construction		
Direct disturbance and displacement:	Yes (for offshore cable corridor)	On the advice of Natural England, the disturbance and displacement effects of the North Falls Array area during construction have been assessed as 50% of the predictions for operational displacement. Cumulative operational displacement effects for North Falls and other OWFs in the UK North Sea are assessed in Section 13.8.3.1 below for the offshore ornithology receptors screened in for displacement (gannet, guillemot, razorbill and red-throated diver). In each case the assessment has concluded no significant ecological effect of cumulative operational displacement. In the context of the conclusions of the operational displacement assessment, the same conclusion would apply if the construction effects of North Falls are estimated as 50% of operational displacement. Thus cumulative effects of disturbance and displacement in relation to the North Falls array area, and other OWFs, are screened out. However, as there is overlap between the offshore cable corridors for North Falls and Five Estuaries there is potential for cumulative disturbance to occur in this area during the construction phase, and this impact has been screened in.
Indirect impacts through effects on habitats and prey species	No	The likelihood that there would be a cumulative impact is low because the contribution from the proposed project is very small-scale and temporary, and would make no material contribution to any cumulative effect.

Impact	Potential for cumulative effect	Rationale
Operation		
Displacement / barrier effect	Yes	There is a sufficient likelihood of a cumulative impact to justify a detailed, quantitative cumulative impact assessment. The cumulative assessment considers cumulative displacement / barrier effects on the seabird species screened in for the Project alone assessment (gannet, guillemot, and red-throated diver). Cumulative operational barrier effect for non-seabird migratory species is also screened in (although barrier effect for non-migratory seabirds was not screened in for the Project Alone assessment), This is based on Natural England's comments on North Falls Scoping Report. Natural England agreed that '*species would be likely to encounter the turbine array only once during a given migration journey if North Falls is situated within their flight corridor, meaning they could potentially encounter the site and hence any barrier effect up to twice per year*' and that '*the energetic costs of such one-off avoidance events can be considered to be negligible for the North Falls project alone. However, we recommend that the impact of cumulative barrier effects [of OWFs] on migratory species is not scoped out of the assessment at this stage*'.
Indirect impacts through effects on habitats and prey species	No	The likelihood that there would be a cumulative impact is low because the contribution from the proposed project is very small-scale, and would make no material contribution to any cumulative effect.
Collision risk	Yes	There is a sufficient likelihood of a cumulative impact to justify a detailed, quantitative cumulative impact assessment.
Decommissioning		
Direct disturbance and displacement:	No	As a worst-case, any effects generated during the decommissioning phase are expected to be similar to those generated during the construction phase. This is because decommissioning would generally involve a reverse of the construction phase through the removal of some structures and materials installed. As for construction disturbance and displacement above, cumulative effects of disturbance and displacement in relation to the North Falls array area, and other OWFs, are screened out. However, as there is overlap between the offshore cable corridors for North Falls and Five Estuaries there is potential for cumulative disturbance to occur in this area during the decommissioning phase, if the two projects are decommissioned at the same time, and this impact has been screened in.
Indirect impacts through effects on habitats and prey species	No	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 effects from other plans or projects.

13.8.2 Other plans, projects and activities

382. The second step in the cumulative assessment

383. is the identification of the other plans, projects and activities that may result in cumulative effects for inclusion in the CEA (described as 'project screening').

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

- OWFs;
- Marine aggregate extraction;
- Oil and gas exploration and extraction;
- Subsea cables and pipelines; and
- Commercial shipping.

385. Of these, only OWFs are considered to have potential to contribute to cumulative operational displacement and collision risk, the key effects screened in for cumulative assessment. This is because only operational OWFs have large-scale permanent infrastructure above the sea surface (i.e. turbine arrays), which birds may collide with or be displaced by. Thus, the cumulative assessment is mainly focused on OWFs.

386. It is recognised that Natural England disagrees with this view, commenting on the North Falls PEIR that '*the exclusion of displacement causing activities from the CEA on the grounds that they do not have large scale permanent infrastructure does not consider the fact that aggregate extraction and busy commercial shipping lanes can lead to long-term displacement of birds*', and '*the submitted ES should consider other displacement-generating projects (including relevant aggregate extraction) projects in the CEA*'. However, the view of North Falls is that both shipping and aggregate extraction are activities which have been ongoing within the Outer Thames Estuary area in the long term, and were therefore part of the baseline conditions when the Digital Aerial Surveys for the Project were undertaken. Including these activities in the CEA would be to effectively double count their impacts.

387. It is acknowledged that a new aggregate production area (aggregate production area 524, Chapter 18, Figure 18.2 (Document Reference: 3.2.14) adjacent to the southern boundary of the North Falls array area became operational in April 2023, and thus aggregate extraction activities in this area will not be reflected in the baseline surveys. The species most-likely to be affected by aggregate extraction is red-throated diver, given the evidence of displacement from areas of vessel activity such as shipping lanes (Section 13.6.1.1.4 above). However, the new aggregate extraction area is almost entirely (97% of total area) within the 4km buffer of North Falls, from which it is assumed in the project alone and cumulative assessments that 100% of red-throated divers are displaced. Thus, this new aggregate production area would not add to the predicted cumulative numbers of red-throated divers displaced from North Falls and other OWFs.

388. In relation to red-throated diver, the cumulative assessment of disturbance during construction of the offshore cable corridor also considers the Sea Link cable (Section 13.8.3.1 below).

389. It is considered valid to argue that there is a distinction between permanent infrastructure above the sea surface, within OWF turbine arrays, where displacement would be ongoing as long as the WTGs are in place, and aggregate extraction areas where disturbance would take place only when extraction is ongoing and would be spatially limited to areas in the vicinity of

extraction vessels. Previous cumulative assessments of seabird displacement from OWFs have considered only other OWFs (for example Sheringham Shoal and Dudgeon Extension Projects, East Anglia ONE North and TWO) and not other activities.

390. A list of OWFs considered for inclusion in the CEA for North Falls is set out in Table 13.43 below, together with relevant details, including current status (e.g. under construction), closest distance to North Falls, status of available data and rationale for including or excluding from the assessment. The overall area of search for these OWFs is based on the largest seasonal BDMPS – the UK North Sea and Channel (as defined by Furness 2015) – for the offshore ornithological receptors screened in for assessment for one or more effects at North Falls. Wind farms are assigned to Tiers following the approach proposed by Natural England (2022c) as follows:

1. Built and operational projects;
2. Projects under construction;
3. Projects that have been consented (but construction has not yet commenced);
4. Projects that have an application submitted to the appropriate regulatory body that have not yet been determined;
5. Projects that have produced a PEIR and have characterisation data within the public domain;
6. Projects that the regulatory body are expecting to be submitted for determination (e.g. projects listed under the Planning Inspectorate programme of projects); and
7. Projects that have been identified in relevant strategic plans or programmes.

391. The project screening has been informed by the development of a CEA project list which forms an exhaustive list of plans, projects and activities within the CEA area of search (Section 13.7) relevant to North Falls. The list has been appraised, based on the confidence in being able to undertake an assessment from the information and data available.

392. As the relevant areas for CEA vary between bird species, not all OWFs in Table 13.43 are included in the CEA for every species scoped in for assessment below.

393. For each assessment it is only possible to include OWFs where quantitative information on effects is available in the public domain at the time or writing. Thus, projects in Tier 5 are included but no projects in Tiers 6 and 7.

13.8.3 Assessment of cumulative effects

394. Cumulative effects of displacement and collision risk are assessed quantitatively based on the sum of estimates of mean effect (mean predicted levels of mortality) from all OWF projects for which data are available within the appropriate zone of influence for a given species. While in the Project alone assessment for North Falls, means and 95% CLs have been presented for assessment, 95% CLs are not included in the cumulative assessment which is based on mean values for all OWF projects, including North Falls. This is

because CLs are not available for all OWFs included in the CEA, and it is also considered that summing of 95% CLs would produce substantially over-estimated (in the case of upper 95% CLs) or underestimated (in the case of lower 95% CLs) predicted impacts.

395. The cut off for inclusion of other OWFs into the CEA was the end of March 2024. This means that updates are not included for OWFs for which PEIRs became available or the ES was submitted beyond this date.

396. It is noted that since this cut-off, Green Volt and Sheringham Shoal and Dudgeon Extension Projects have been consented; and the Ess for three OWFs, Dogger Bank South, Five Estuaries and Outer Dowsing, have been submitted. It is understood that no changes to the predicted displacement and collision mortalities for the two consented sites have been made after March 2024. However, for Five Estuaries, Dogger Bank South and Outer Dowsing, the cumulative assessment here is based on predicted displacement and collision mortalities from the PEIR, and has not been updated to reflect any changes in the Ess that accompanied the DCO submission.

Table 13.43 Projects included in the CEA for offshore ornithology

Project	Tier	Status	Closest distance (km) from: Array area (km)	Closest distance (km) from: Offshore cable corridor	Confidence in Data	Included in the CEA (Y/N)	Rationale
Beatrice (demonstrator)	1	Built and operational, fully commissioned July 2007	774.46	753.88	Complete but limited quantitative species assessment	Yes	Included as an operational project. Due to be decommissioned between 2024 and 2027
Beatrice	1	Built and operational, fully commissioned May 2019	774.46	753.88	Complete for the ornithology receptors being assessed	Yes	Included as an operational project.
Blyth Demonstration	1	Built and operational, fully commissioned Oct 2017	429.39	405.74	Complete but limited quantitative species assessment	Yes	Included as a operational project.
Dudgeon	1	Built and operational, fully commissioned November 2017	160.92	147.07	Complete but limited quantitative species assessment	Yes	Included as an operational project.
East Anglia ONE	1	Built and operational, fully commissioned July 2020	53.08	59.24	Complete for the ornithology receptors being assessed	Yes	Included as an operational project.
EOWDC (Aberdeen OWF)	1	Built and operational, fully commissioned Sep 2018	653.11	632.86	Complete for the ornithology receptors being assessed	Yes	Included as an operational project.
GGOW	1	Built and operational, fully commissioned August 2013	0.00	3.91	Complete for the ornithology receptors being assessed	Yes	Included as an operational project.
Gunfleet Sands	1	Built and operational, fully commissioned Jun 2010	39.00	6.00	Complete for the ornithology receptors being assessed	Yes	Included as an operational project.
GWF	1	Built and operational, fully commissioned March 2018	0.00	6.35	Complete for the ornithology receptors being assessed	Yes	Included as an operational project.

Project	Tier	Status	Closest distance (km) from:		Confidence in Data	Included in the CEA (Y/N)	Rationale
			Array area (km)	Offshore cable corridor			
Hornsea Project One	1	Built and operational, fully commissioned 2020	225.84	216.80	Complete for the ornithology receptors being assessed	Yes	Included as an operational project.
Hornsea Project Two	1	Built and operational, fully commissioned August 2022	227.55	216.59	Complete for the ornithology receptors being assessed	Yes	Included as an operational project.
Humber Gateway	1	Built and operational, fully commissioned May 2015	229.65	207.31	Complete but limited quantitative species assessment	Yes	Included as an operational project.
Hywind	1	Built and operational, fully commissioned Oct 2017	665.57	647.09	Complete but limited quantitative species assessment	Yes	Included as an operational project.
Kentish Flats	1	Built and operational, fully commissioned Dec 2005	54.59	38.08	Complete but limited quantitative species assessment	Yes	Included as an operational project.
Kentish Flats Extension	1	Built and operational, fully commissioned Oct 2015	54.59	39.70	Complete but limited quantitative species assessment	Yes	Included as an operational project.
Kincardine	1	Built and operational, fully commissioned Oct 2021	626.20	605.83	Complete but limited quantitative species assessment	Yes	Included as an operational project.
Lincs	1	Built and operational, fully commissioned Sep 2013	176.80	153.31	Complete but limited quantitative species assessment	Yes	Included as an operational project.
London Array	1	Built and operational, fully commissioned Apr 2013	20.59	15.52	Complete but limited quantitative species assessment	Yes	Included as an operational project.

Project	Tier	Status	Closest distance (km) from:		Confidence in Data	Included in the CEA (Y/N)	Rationale
			Array area (km)	Offshore cable corridor			
Lynn and Inner Dowsing (LID)	1	Built and operational, fully commissioned Mar 2009	177.34	153.75	Complete but limited quantitative species assessment	Yes	Included as an operational project.
Moray East	1	Built and operational, fully commissioned April 2022	761.98	742.04	Complete for the ornithology receptors being assessed	Yes	Included as an operational project.
Race Bank	1	Built and operational, fully commissioned February 2018	173.38	153.52	Complete but limited quantitative species assessment	Yes	Included as an operational project.
Rampion	1	Built and operational, fully commissioned November 2018	177.82	158.76	Complete for the ornithology receptors being assessed	Yes	Included as an operational project.
Scroby Sands	1	Built and operational, fully commissioned Dec 2004	92.78	84.41	Complete but limited quantitative species assessment	Yes	Included as an operational project.
Sheringham Shoal	1	Built and operational, fully commissioned Sep 2012	152.60	135.65	Complete but limited quantitative species assessment	Yes	Included as an operational project.
Teesside	1	Built and operational, fully commissioned Aug 2013	373.71	349.33	Complete but limited quantitative species assessment	Yes	Included as an operational project.
Thanet	1	Built and operational, fully commissioned Sep 2010	24.92	36.16	Complete for the ornithology receptors being assessed	Yes	Included as an operational project.
Triton Knoll	1	Built and operational, fully commissioned Oct 2021	190.27	172.38	Complete for the ornithology receptors being assessed	Yes	Included as an operational project.

Project	Tier	Status	Closest distance (km) from: Array area (km)	Offshore cable corridor	Confidence in Data	Included in the CEA (Y/N)	Rationale
Westermost Rough	1	Built and operational, fully commissioned May 2015	250.38	228.07	Complete for the ornithology receptors being assessed	Yes	Included as an operational project.
Dogger Bank A and B (formerly Creyke Beck A and B)	2	Offshore construction began April 2022, Doggerbank A partially generation from October 2023	318.50	309.98	Complete for the ornithology receptors being assessed	Yes	Included as a consented project.
Dogger bank C and Sofia (formerly Dogger bank Teeside A and B)	2	Sofia onshore works began mid-2021, converter station and export cable route 2022, offshore works due 2023, completion due 2026.	339.14	332.23	Complete for the ornithology receptors being assessed	Yes	Included as a consented project.
Moray West	2	Offshore construction began Feb 2023. Due to be fully operational by 2025	763.65	742.98	Complete for the ornithology receptors being assessed	Yes	Included as a consented project.
Neart na Gaoithe	2	Offshore construction began 2020, completion due 2024	559.53	536.96	Complete for the ornithology receptors being assessed	Yes	Included as a consented project.
Seagreen (Alpha and Bravo)	2	114 turbines fully operational October 2023. S36 consent variation for 36 additional turbines	572.35	552.01	Complete for the ornithology receptors being assessed	Yes	Included as a consented project.
East Anglia ONE North	3	Consented March 2022. Onshore construction due to start 2025, offshore construction due to start 2027	63.07	67.43	Complete for the ornithology receptors being assessed	Yes	Included as a consented project.
East Anglia THREE	3	Consented August 2017. No construction start date	98.77	104.22	Complete for the ornithology receptors being assessed	Yes	Included as a consented project.
East Anglia TWO	3	Consented March 2022. No construction start date	31.49	36.67	Complete for the ornithology receptors being assessed	Yes	Included as a consented project.

Project	Tier	Status	Closest distance (km) from:		Confidence in Data	Included in the CEA (Y/N)	Rationale
			Array area (km)	Offshore cable corridor			
Green Volt*	3	Consented 19 April 2024	691.50	675.24	Complete for the ornithology receptors being assessed	Yes	Outputs from the ES have been included
Hornsea Project Three	3	Consented Dec 2020. Construction due to start 2024	218.40	217.21	Complete for the ornithology receptors being assessed	Yes	Included as a consented project.
Hornsea Project Four	3	Consented July 2023. No construction start date	229.15	216.62	Complete for the ornithology receptors being assessed	Yes	Included as a consented project.
Inch Cape	3	Consented Sep 2014, revised June 2019. No construction start date.	579.15	557.03	Complete for the ornithology receptors being assessed	Yes	Included as a consented project.
Methil (Forthwind Demonstration)	3	Consented December 2016. New consent authorised March 2023.	584.74	559.18	Complete but limited quantitative species assessment	Yes	Included as a consented project.
Norfolk Boreas	3	Consented Dec 2021. Work paused until further notice	132.04	134.56	Complete for the ornithology receptors being assessed	Yes	Included as a consented project.
Norfolk Vanguard	3	Consented Feb 2022. Construction was due to begin Sep 2023	95.76	117.16	Complete for the ornithology receptors being assessed	Yes	Included as a consented project.
Sheringham and Dudgeon Extension Projects*	3	Consented 18 April 2024	150.51	134.07	Complete for the ornithology receptors being assessed	Yes	Outputs from the ES / DCO examination have been included.
Berwick Bank	4	Application submitted August 2023	522.45	501.36	Complete for the ornithology receptors being assessed	Yes	Outputs from the ES have been included.

Project	Tier	Status	Closest distance (km) from:		Confidence in Data	Included in the CEA (Y/N)	Rationale
			Array area (km)	Offshore cable corridor			
Dogger Bank South*	4	DCO Application submitted 12 June 2024, accepted 10 July	285.19	276.33	Complete for the ornithology receptors being assessed	Yes	Outputs from the PEIR have been included.
Five Estuaries*	4	DCO application accepted 30 April 2024, at pre-examination stage	0.00	12.93	Complete for the ornithology receptors being assessed	Yes	Outputs from the PEIR have been included.
Outer Dowsing*	4	DCO application accepted 16 April 2024, at pre-examination stage	193.46	177.64	Complete for the ornithology receptors being assessed	Yes	Outputs from the PEIR have been included.
Rampion 2	4	Application submitted. Project in examination.	171.11	155.25	Complete for the ornithology receptors being assessed	Yes	Outputs from the ES have been included.
West of Orkney	4	Application submitted Sept 2023	869.58	847.96	Complete for the ornithology receptors being assessed	Yes	Outputs from the ES have been included.
South & East Anglia (SEA) Link	5	Pre-application	5.4	0	Medium	Yes (Red-throated diver only)	Outputs from the PEIR have been included.
* Status changed after the cut-off date of the end of March 2024 for updates to the cumulative assessment. Green Volt and Sheringham Shoal and Dudgeon Extension Projects have been consented (changing from Tier 4 to 3); and the Ess for two OWFs, Five Estuaries and Outer Dowsing, have been submitted (moving from Tier 5 to 4). It is understood that no changes to the predicted displacement and collision mortalities for the two consented sites have been made after March 2024. However, for Five Estuaries and Outer Dowsing, the cumulative assessment here is based on predicted displacement and collision mortalities from the PEIR, and has not been updated to reflect any changes in the Ess that accompanied the DCO submission							

13.8.3.1 Cumulative effect 1: Construction Disturbance / Displacement, Offshore Cable Corridor

397. As there is overlap between the offshore cable corridors of North Falls and Five Estuaries OWFs (Figure 13.2 (Document Reference: 3.2.9)), cumulative impacts of construction disturbance and displacement may occur in this area if the construction phase of North Falls overlaps with Five Estuaries (Table 13.42). This cumulative effect has been screened in for red-throated diver only, as the offshore cable corridors pass close to and through the Outer Thames Estuary SPA and thus areas of high densities of this species. As for the project alone assessment, because of lower sensitivity to shipping activity, and the short duration and small spatial extent (at any one time) of construction activities, gannet, guillemot and razorbill were not screened in for assessment in relation to the offshore cable corridor (Section 13.6.1.1).

398. North Falls has applied to the Offshore Coordination Support Scheme (OCSS) for an offshore connection to Sea Link, a marine cable between Suffolk and Kent proposed by NGET as part of their Great Grid Upgrade. Construction of the offshore cable component of this scheme may be ongoing at the same time as the offshore cable corridor for North Falls. However, to avoid cumulative effects with other projects, the PEIR for Sea Link states that except at the landfall areas, all other construction works will be timed outside the months of January – March to avoid the core overwintering period of red-throated diver (AECOM 2023). These months coincide with the peak numbers of red-throated divers at North Falls (ES Appendix 13.2 (Document Reference: 3.3.13)). Thus, this project is not considered in the cumulative assessment here.

13.8.3.1.1 Red-throated diver

399. The worst-case scenario would be five cable laying vessels operating at one time: North Falls (two vessels) and Five Estuaries (three vessels). As cable laying vessels are static for long periods of time and move slowly over short distances as cable installation takes place, and assuming that red-throated divers reoccupy areas of suitable habitat once the vessels (source of disturbance) have moved on the zone of impact around each vessel would be relatively fixed as far as the birds are concerned (see para 140 above). Thus, the worst-case area of a 2km radius around each vessel, would equate to a total of 63km^2 (5 x 12.6km^2) from which birds could be displaced.

400. Assuming 100% displacement and a density of 3.64 birds per km^2 (para 138 above), a cumulative total of 229 red-throated divers would be displaced at any one time.

401. Assuming a maximum mortality of 1% of displaced birds (MacArthur Green, 2019c, para 280), a maximum of 2.3 red-throated divers would be predicted to suffer mortality over the course of a non-breeding season due to displacement from cumulative construction activities in the offshore cable corridor.

402. The relevant reference populations for red-throated divers present at North Falls during the non-breeding season are the UK North Sea BDMPS during Autumn and Spring migration, and the south-west North Sea during winter, respectively estimated as 13,277 and 10,177 individuals (Furness, 2015). Based on the average annual mortality rate across age classes of 0.233 (Table 13.11), respectively 3,094 and 2,371 birds would be expected to suffer mortality each year. The addition of 2.3 individuals would represent an increase in mortality

rate of <0.1% of either the migration period or winter BDMPS (e.g. for the spring migration BDMPS the maximum increase in mortality rate would be 2 ÷ 3094 x 100 = 0.06%).

403. At 10% mortality of displaced birds, which is considered unrealistically high (para 281), 22.9 red-throated divers would suffer mortality as a result of displacement from the offshore cable corridor, equivalent to an increase of 0.74% in the baseline mortality rate of the migration period BDMPS and 0.97% for the winter BDMPS. Even at this unrealistic mortality rate, therefore, the increase would be below the 1% threshold likely to be detectable against background mortality.

404. Cumulative construction disturbance and displacement within the North Falls offshore cable corridor would be a temporary effect, expected to take place over approximately six months (Table 13.1). The predicted magnitude of increase in red-throated diver mortality would not materially alter the background mortality of the population and would be undetectable. Cumulative construction disturbance during the construction of the offshore cable corridors for North Falls and Five Estuaries is assessed as an impact of negligible magnitude. As the species is of high sensitivity to disturbance, the effect significance is minor adverse, and not significant in EIA terms.

13.8.3.2 Cumulative effect 2: Operational Displacement

405. The species assessed for project alone operational displacement effects (and the relevant seasons) were gannet (autumn, spring), guillemot (breeding, non-breeding), razorbill (breeding, autumn, winter, spring) and red-throated diver (autumn, winter, spring). These species have also been scoped in for cumulative effect assessment.

13.8.3.2.1 Gannet

406. The number of birds at risk of displacement from all OWFs in the UK North Sea and Channel BDMPS is provided for all relevant projects in ES Appendix 13.3 (Document Reference: 3.3.14). The seasonal totals for all OWFs in Tiers 1 to 3, along with the contribution made by all known relevant OWFs in Tier 4 and above for which data were available, is presented in Table 13.44. Whilst 2km was the preferred buffer where it was available (as recommended for gannet by SNCBs, 2017), the buffer zones for which data were presented for OWFs included in the assessment varied between 0 – 4km. The total number of birds at risk of displacement annually is 63,304, towards which North Falls contributes 646 birds, or 1% of the cumulative total.

Table 13.44 Cumulative numbers of gannet potentially at risk of displacement for all OWFs included in CEA

Tiers / development	Breeding	Autumn migration	Spring migration	Annual
1 to 3[1]	21,343	21,334	5,412	48,089
Green Volt (Tier 3)	120	16	49	185
Sheringham and Dudgeon Extension Projects (Tier 3)	440	638	58	1,136
Berwick Bank (Tier 4)	4,735	1,500	269	6,504
Dogger Bank South (Tier 4)	1038	1020	17	2075
Five Estuaries (Tier 4)[2]	233	640	67	940

Tiers / development	Breeding	Autumn migration	Spring migration	Annual
Outer Dowsing (Tier 4)[2]	847	169	172	1,187
Rampion 2 (Tier 4)	111	102	123	336
West of Orkney (Tier 4)	958	1,171	77	2,206
North Falls	69	287	290	646
Totals	**29,894**	**26,877**	**6,535**	**63,304**
1. Data for individual OWFs in Tiers 1-3 included in ES Appendix 13.3 (Document Reference: 3.3.14). Total does not include Green Volt and Sheringham and Dudgeon Extension Projects, consented after the cut-off date of end March 2024 for updates to the North Falls assessment. These OWFs are therefore listed separately. 2. ES submitted (and publicly available) after the cut off date for inclusion in this assessment, end March 2024. The values in the table are taken from the PEIR.				

407. The displacement matrix for annual cumulative gannet mortality is presented in Table 13.45.

408. At displacement rates of 60 – 80% and a 1% mortality rate of displaced birds (Section 13.6.2.1.1), 380 – 507 gannets are predicted to suffer mortality annually from cumulative displacement from operational OWFs.

409. To assess the magnitude of the year-round impact of cumulative operational OWF displacement on gannet, two background populations are considered. Firstly, the largest relevant BDMPS population autumn migration UK North Sea and Channel BDMPS, consisting of 456,298 individuals (Furness, 2015)). Based on the published baseline mortality of 18.7%- across all age classes (Table 13.11), 85,328 individual gannets from this population would be expected to suffer mortality annually. Based on the published baseline mortality of 18.7% across all age classes ((Table 13.11), 85,328 individual gannets from this population would be expected to suffer mortality annually. Secondly, the biogeographic population with connectivity to UK waters of 1,180,000 (Furness, 2015), from which 220,660 individuals would be expected to suffer mortality annually using the same all age class mortality rate.

410. The predicted level of additional mortality would represent a 0.44% to 0.59% increase in annual mortality within the largest BDMPS population, or a 0.17% to 0.23% increase in annual mortality within the annual biogeographic population with connectivity to UK waters. These mortality increases would not be detectable at the population level within the context of natural variation. In addition, as discussed in Section 13.6.2.1.1, the assumption of 1% mortality of displaced gannets is considered a precautionary prediction.

Table 13.45 Cumulative operational OWF displacement matrix for year round impacts on gannet. The cells show the number of predicted bird mortalities (to the nearest integer) at a given rate of displacement and mortality. Grey cells identify the range of displacement and mortality rates considered in the assessment. Values in red identify scenarios where the baseline mortality rate would increase by >1%

Mean		Mortality										
		1%	2%	3%	4%	5%	10%	20%	30%	50%	80%	100%
Displacement	10%	63	127	190	254	317	634	1,268	1,901	3,169	5,070	6,338
	20%	127	254	380	507	634	1,268	2,535	3,803	6,338	10,140	12,675
	30%	190	380	570	761	951	1,901	3,803	5,704	9,506	15,210	19,013
	40%	254	507	761	1,014	1,268	2,535	5,070	7,605	12,675	20,280	25,350
	50%	317	634	951	1,268	1,584	3,169	6,338	9,506	15,844	25,350	31,688
	60%	380	761	1,141	1,521	1,901	3,803	7,605	11,408	19,013	30,420	38,025
	70%	444	887	1,331	1,775	2,218	4,436	8,873	13,309	22,181	35,490	44,363
	80%	507	1,014	1,521	2,028	2,535	5,070	10,140	15,210	25,350	40,560	50,700
	90%	570	1,141	1,711	2,282	2,852	5,704	11,408	17,111	28,519	45,630	57,038
	100%	634	1,268	1,901	2,535	3,169	6,338	12,675	19,013	31,688	50,700	63,375

411. The cumulative annual total of gannets predicted to be at risk of year-round displacement from OWFs in UK North Sea and Channel BDMPS (63,375) is 14% of the largest seasonal BDMPS population. As noted previously, it is acknowledged by SNCBs (2017) that summing seasonal peak means for individual OWFs almost-certainly over-estimates the number of birds of a given species at risk of displacement annually, as some birds may be present in more than one season. Further, summing totals from all OWFs within the BDMPS will build in further over-estimation as many individuals would be expected to encounter more than one OWF. In addition, assessing against the BDMPS almost certainly under-estimates the population from which they are drawn (as the BDMPS population must be at least this size and is likely to be considerably larger as a consequence of turnover of individuals). Thus, the cumulative assessment of gannet displacement is considered highly pre-cautionary. The overall proportion of the largest seasonal BDMPS predicted to be present in OWFs and 2km buffers is rather lower for gannet than for guillemot and razorbill, below.

412. The year-round magnitude of cumulative operational displacement on gannet is assessed as negligible. As gannet is of medium sensitivity to displacement, the effect significance is minor adverse.

13.8.3.2.2 Guillemot

413. The number of birds at risk of displacement from OWFs in the UK North Sea and Channel BDMPS is included in ES Appendix 13.3 (Document Reference: 3.3.14). The seasonal totals for all OWFs in Tiers 1 to 3, along with the contribution made by all known relevant OWFs in Tier 4 and above for which data were available, is presented in Table 13.46. The total number of guillemots predicted to be at risk of displacement annually is 658,312, towards which North Falls contributes 6,231 birds, or 0.9% of the cumulative total.

414. The displacement matrix for annual cumulative guillemot mortality is presented in Table 13.47. As for the Project Alone assessment, displacement is considered within the range of 30-70% displacement and 1-10% mortality of displaced birds, 50% displacement and 1% mortality, considered a realistic precautionary scenario, and 70% displacement and 2% mortality, the rates on which the consent decision for HP4 is understood to have been made (see para 215 above).

415. At displacement rates of 30 – 70% and mortality rates of 1 – 10% of displaced birds (Section 13.6.2.1.1), between 1,975 and 46,082 guillemots are predicted to suffer mortality from cumulative operational OWF displacement annually. At 50% displacement and 1% mortality, 3,292 individuals would suffer mortality, and at 70% displacement and 2% mortality, 9216 individuals.

416. To assess the magnitude of the year-round impact of operational OWF displacement on guillemot, two background populations are considered. Firstly, the largest relevant BDMPS population (UK North Sea and Channel BDMPS, consisting of 1,617,306 birds (Furness, 2015)). Assuming a published all age class baseline mortality rate of 14.3% (Table 13.11), 231,275 guillemots from this population would be expected to suffer mortality annually. Secondly, the biogeographic population with connectivity to UK waters of 4,125,000 (Furness, 2015). A total of 589,875 individuals would be expected to suffer mortality annually from this population, using the same all age class mortality rate.

Table 13.46 Cumulative numbers of guillemot potentially at risk of displacement for all OWFs Included in CEA

Tiers / development	Breeding	Non-breeding	Annual
1 to 3[1]	170,621	170,874	387,842
Hornsea Project Four (Tier 3)[2]	9,382	36,965	46,347
Green Volt (Tier 3)	4,429	16,105	20,534
Sheringham and Dudgeon Extension Projects (Tier 3)	4,934	15,972	20,906
Berwick Bank (Tier 4)	44,171	74,154	118,325
Dogger Bank South (Tier 4)	31,587	25,342	56,929
Five Estuaries (Tier 4)[3]	1,201	3,698	4,899
Outer Dowsing (Tier 4)[3]	23,173	22,248	45,421
Rampion 2 (Tier 4)	134	5,723	5,857
West of Orkney (Tier 4)	4,861	4,275	9,136
North Falls	866	5,365	6,231
Totals	**288,631**	**369,681**	**658,312**

Notes: 1. Data for individuals OWFs in Tiers 1-3 included in ES Appendix 13.3 (Document Reference: 3.3.14). Excludes Hornsea Project Four and recently consented sites Green Volt and Sheringham and Dudgeon Extension Projects which are listed separately. 2. Natural England Approach (APEM and GoBe, 2022). 3. ES submitted (and publicly available) after the cut off date for inclusion in this assessment, end March 2024. The values in the table are taken from the PEIR.

Table 13.47 Cumulative operational OWF displacement matrix for year round impacts on guillemot. The cells show the number of predicted bird mortalities (to the nearest integer) at a given rate of displacement and mortality. Grey cells identify the range of displacement and mortality rates considered in the assessment. Values in red identify scenarios where the baseline mortality rate would increase by >1%

Mean		Mortality										
		1%	2%	3%	4%	5%	10%	20%	30%	50%	80%	100%
Displacement	10%	658	1317	1975	2633	3292	6583	13166	19749	32916	52665	65831
	20%	1317	2633	3950	5266	6583	13166	26332	39499	65831	105330	131662
	30%	1975	3950	5925	7900	9875	19749	39499	59248	98747	157995	197493
	40%	2633	5266	7900	10533	13166	26332	52665	78997	131662	210660	263325
	50%	3292	6583	9875	13166	16458	32916	65831	98747	164578	263325	329156
	60%	3950	7900	11850	15799	19749	39499	78997	118496	197493	315990	394987
	70%	4608	9216	13825	18433	23041	46082	92164	138245	230409	368654	460818
	80%	5266	10533	15799	21066	26332	52665	105330	157995	263325	421319	526649
	90%	5925	11850	17774	23699	29624	59248	118496	177744	296240	473984	592480
	100%	6583	13166	19749	26332	32916	65831	131662	197493	329156	526649	658312

417. The predicted level of additional mortality for the range of 30-70% displacement and 1-10% mortality would represent a 0.9% to 19.9% increase in annual mortality within the largest BDMPS population. Annual mortality rate increases in the biogeographic population with connectivity to UK waters would be 0.3% to 7.8%.

418. At 50% displacement and 1% mortality, the increase in the mortality rate of the largest BDMPS population would be 1.4%, and 0.6% for the biogeographic population with connectivity to UK waters. At 70% displacement and 2% mortality, the increase in the mortality rate of the largest BDMPS population would be 4.0%, and 1.6% for the biogeographic population with connectivity to UK waters.

419. The cumulative annual total of guillemots predicted to be at risk of displacement from OWFs in the UK North Sea and Channel BDMPS (658,312 individuals) is 41% of the largest seasonal population estimate for this area, indicating that a large proportion of the BDMPS population will utilise areas within an OWF array, and / or within 2km of an OWF array, each year. As noted previously, it is acknowledged by SNCBs (2017) that summing seasonal peak means for an individual OWF to produce an annual total of birds predicted to be at risk of displacement, almost-certainly over-estimates the number of birds of a given species at risk of displacement annually, for example, as some birds may be present in more than one season. Further, summing annual totals from all OWFs within the BDMPS will build in further over-estimation as many individuals would be expected to encounter more than one OWF. In addition, 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). Thus, the cumulative assessment of guillemot displacement is considered overly pre-cautionary.

420. The maximum increases in mortality rate assume 10% mortality of displaced guillemots, which, as discussed above (Section 13.6.2.1.1), is considered highly unlikely and 1% mortality is considered an appropriate precautionary rate.

421. Hornsea Project Four has been consented subject to derogation and compensation for predicted mortality for guillemot displacement, Hornsea Project Four (DESNZ, 2023c) (and also Sheringham and Dudgeon Extension Projects, consented after the cut-off date of end March 2024 for inclusion in this assessment). The aim of the compensation is to reduce the net effect of an OWF on displacement mortality of guillemot to zero. Assuming that compensation would effectively remove the predicted displacement mortality from Hornsea Project Four, this would reduce the total annual number of guillemots predicted to be at risk of displacement mortality to 611,965. On a precautionary basis this reduction has not been applied to the cumulative assessment here.

422. The UK population of guillemot declined by 11% between the Seabird 2000 and the Seabirds Count censuses, with the overall trend driven by a decrease in the Scottish population, while populations in England, Wales and Northern Ireland increased (Burnell *et al*., 2023). As for other seabird species, the impact of HPAI on guillemots in the UK is currently uncertain. HPAI mortality of 3,775 individuals was recorded in England in 2022, representing about 0.7% of the England breeding population (Royal HaskoningDHV, 2023). A 'minimum loss' of 1,908 individuals to HPAI is reported for Scotland in 2022 (NatureScot 2023). The source of this number is not clearly stated, but it seems to be based on numbers of dead guillemots reported to NatureScot and considered a minimum estimate as not all dead birds will have been recorded or reported (and it seems likely that only a small proportion of actual deaths from HPAI would be encountered by people). A total of 24 breeding sites or groups of sites throughout the UK were included in seabird colony counts carried out in 2023 to assess changes following the 2021-22 HPIA outbreak (Tremlett *et al.,* 2024). The total numbers of individual guillemots recorded across all sites decreased by 7% compared with pre-HPAI baseline counts, although the decline was not consistent across all sites surveyed, and some sites increased. Tremlett *et al.,* 2024 do not present any evidence or comment in relation to the role of HPAI in these declines.

423. The year-round magnitude of cumulative operational displacement on guillemot is assessed as low. This takes into account the sources of precaution in the cumulative totals of birds at risk of assessment, the fact that compensation for guillemot displacement mortality is required for Hornsea Project Four (which is designed to reduce the net effect of this project to zero) and assumes a maximum 1% increase in the mortality rate of displaced birds. As guillemot is of medium sensitivity to displacement, the effect significance is minor adverse, which is not significant in EIA terms.

424. Under the latest guidance from Natural England and Natural Resources Wales on EIA reference populations (see para 68), the largest seasonal BDMPs would increase to 2,045,078 individuals for the breeding season (UK North Sea and Channel); applying this, the percentage increases in baseline mortality of the BDMPS would be smaller than those stated above (equivalent to 0.7% at 30% displacement and 1% mortality, 1.1% at 50% displacement and 1% mortality,

3.2% at 70% displacement and 2% mortality, and 15.8% at 70% displacement and 10% mortality).

13.8.3.2.3 Razorbill

425. The number of birds at risk of displacement from all OWFs in the UK North Sea and Channel BDMPS is included by development in ES Appendix 13.3 (Document Reference: 3.3.14). The seasonal totals for all OWFs in Tiers 1 to 3, along with the contribution made by all known relevant OWFs in Tier 4 and above for which data were available, is presented in Table 13.48. The total number of razorbills predicted to be at risk of displacement annually is 208,169, towards which North Falls contributes 3,874 birds, or 1.9% of the cumulative total.

426. The displacement matrix for annual cumulative razorbill mortality is presented in Table 13.48.

Table 13.48 Cumulative numbers of razorbill potentially at risk of displacement for all OWFs Included in CEA

Tiers / development	Breeding	Autumn migration	Winter	Spring migration	Annual
1 to 3[1]	32,510	39,411	25,549	32,979	129,449
Green Volt (Tier 3)	457	56	15	28	556
Sheringham and Dudgeon Extension Projects (Tier 3)	1,239	4,500	1,531	464	7,734
Berwick Bank (Tier 4)	4,040	8,849	1,399	7,480	21,768
Dogger Bank South (Tier 6)	5,313	1,238	4,117	8,628	19,296
Five Estuaries (Tier 4)[2]	90	284	1,046	756	2,177
Outer Dowsing (Tier 4)[2]	5,163	2,339	2,570	5,229	15,301
Rampion 2 (Tier 4)	32	26	1,193	6,303	7,554
West of Orkney (Tier 4)	141	167	19	132	459
North Falls	104	248	1,781	1,741	3,874
Totals	**49,090**	**57,118**	**38,221**	**63,741**	**208,169**
Notes: 1. Data for individual OWFs in Tiers 1-3 included in ES Appendix 13.3 (Document Reference: 3.3.14). Total does not include Green Volt and Sheringham and Dudgeon Extension Projects, consented after the cut-off date of end March 2024 for updates to the North Falls assessment. These OWFs are therefore listed separately. 2. ES submitted (and publicly available) after the cut off date for inclusion in this assessment, end March 2024. The values in the table are taken from the PEIR.					

427. At displacement rates of 30 – 70% and mortality rates of 1 – 10% of displaced birds (Section 13.6.2.1.1), between 625 and 14,572 razorbills are predicted to suffer mortality annually from cumulative displacement from operational OWFs. At 50% displacement and 1% mortality, 1,041 individuals would suffer mortality, and at 70% displacement and 2% mortality, 2,914 individuals.

428. To assess the magnitude of the year-round impact of operational OWF displacement on razorbill, two background populations are considered. Firstly, the largest relevant BDMPS population (UK North Sea and Channel BDMPS during autumn and spring migration seasons, consisting of 591,874 birds (Furness, 2015). Assuming an all age class baseline mortality rate of 17.8% (Table 13.11), 105,354 razorbills from this population would be expected to

suffer mortality annually. Secondly, the biogeographic population with connectivity to UK waters of 1,707,000 (Furness, 2015). 303,846 individuals would be expected to suffer mortality annually from this population, using the same all age class mortality rate.

Table 13.49 Cumulative operational OWF displacement matrix for year round impacts on razorbill. The cells show the number of predicted bird mortalities (to the nearest integer) at a given rate of displacement and mortality. Grey cells identify the range of displacement and mortality rates considered in the assessment. Values in red identify scenarios where the BDMPS baseline mortality rate would increase by >1%

Mean		Mortality										
		1%	2%	3%	4%	5%	10%	20%	30%	50%	80%	100%
Displacement	10%	208	416	625	833	1,041	2,082	4,163	6,245	10,408	16,653	20,817
	20%	416	833	1,249	1,665	2,082	4,163	8,327	12,490	20,817	33,307	41,634
	30%	625	1,249	1,874	2,498	3,123	6,245	12,490	18,735	31,225	49,960	62,451
	40%	833	1,665	2,498	3,331	4,163	8,327	16,653	24,980	41,634	66,614	83,267
	50%	1,041	2,082	3,123	4,163	5,204	10,408	20,817	31,225	52,042	83,267	104,084
	60%	1,249	2,498	3,747	4,996	6,245	12,490	24,980	37,470	62,451	99,921	124,901
	70%	1,457	2,914	4,372	5,829	7,286	14,572	29,144	43,715	72,859	116,574	145,718
	80%	1,665	3,331	4,996	6,661	8,327	16,653	33,307	49,960	83,267	133,228	166,535
	90%	1,874	3,747	5,621	7,494	9,368	18,735	37,470	56,206	93,676	149,881	187,352
	100%	2,082	4,163	6,245	8,327	10,408	20,817	41,634	62,451	104,084	166,535	208,169

429. The predicted level of additional mortality for the range of 30-70% displacement and 1-10% mortality would represent a 0.6% to 13.8% increase in annual mortality within the largest BDMPS population. Annual mortality rate increases in the biogeographic population with connectivity to UK waters would be 0.2% to 4.8%.

430. At 50% displacement and 1% mortality, the increase in the mortality rate of the largest BDMPS population would be 1.0%, and 0.3% for the biogeographic population with connectivity to UK waters. At 70% displacement and 2% mortality, the increase in the mortality rate of the largest BDMPS population would be 2.8%, and 1.0% for the biogeographic population with connectivity to UK waters.

431. The cumulative annual total of razorbills at risk of displacement from OWFs in the UK North Sea and Channel BDMPS (208,169 individuals) is 35% of the largest seasonal population estimate for this area, indicating that more than one third of the BDMPS population will utilise areas within an OWF array, and / or within 2km of an OWF array, each year. As noted previously, it is acknowledged by SNCBs (2017) that summing seasonal peak means for individual OWFs almost certainly over-estimates the number of birds of a given species at risk of displacement annually, for example, as some birds may be present in more than one season. Further, summing totals from all OWFs within the BDMPS will build in further over-estimation as many individuals would be expected to encounter more than one OWF. In addition, 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). Thus, the cumulative assessment of razorbill displacement is considered highly pre-cautionary.

432. The UK population of razorbill increased by 18% between the Seabird 2000 and the Seabirds Count censuses, with the overall trend driven by a decrease in the Scottish and Isle of Man populations, while populations in England, Wales and Northern Ireland increased (Burnell *et al*., 2023). Razorbill has not been identified as a species at high risk of HPAI and was not identified for surveillance in seabird colony counts carried out in 2023 to assess changes following the 2021-22 HPAI outbreak (Tremlett *et al*., 2024). A total HPAI mortality of 43 individuals was recorded in England in 2022 representing about 0.39% of the England breeding population (Royal HaskoningDHV, 2023). NatureScot (2023) reports a 'minimum loss' of 38 individuals in 2022. The source of this number is not clearly stated, but it seems to be based on numbers of dead razorbills reported to NatureScot and considered a minimum estimate as not all dead birds will have been recorded or reported (and it seems likely that only a small proportion of actual deaths from HPAI would be encountered by people).

433. The maximum increases in mortality rate assume 10% mortality of displaced razorbills, which, as discussed above (Section 13.6.2.1.1), is considered highly unlikely and 1% mortality is considered an appropriate precautionary rate.

434. The year-round magnitude of cumulative operational displacement on razorbill is assessed as low taking into account the sources of precaution in the cumulative totals of birds at risk of displacement, and a likely maximum mortality rate of 1% of displaced birds. As razorbill is of medium sensitivity to disturbance, the effect significance is minor adverse, which is not significant in EIA terms.

13.8.3.2.4 Red-throated diver

435. Red-throated divers were present at North Falls (OWF plus 4km buffer) during the winter and spring migration periods only, so the Project will contribute to a cumulative displacement effect during this time. During the spring migration period the relevant BDMPS is the UK North Sea, and during the winter period the south-west North Sea (Furness, 2015). The assessment considers the largest non-breeding BDMPS (UK North Sea, 13,277 individuals) and the biogeographic population with connectivity to UK waters (27,000 individuals), as reference populations.

436. All OWF projects within the UK North Sea have been considered for inclusion in the cumulative assessment. However, none of the operational and consented OWFs in Scottish waters are in areas of importance for red-throated diver, and as a consequence no projects have undertaken assessments of potential displacement for this species. The cumulative assessment for North Falls therefore considers OWFs in the English North Sea only.

437. Assessments for many existing OWFs in this area have not considered red-throated diver displacement effects at all, or have considered them in a qualitative manner. Thus, seasonal and annual estimates of the number of individuals at risk of displacement, and predictions of mortality from displacement, are not available for all OWFs within the cumulative assessment area. This is partly because red-throated divers overwintering in the UK generally occur in nearshore waters, so few or no individuals may have been recorded in baseline surveys of OWFs distant from the coast, and the species

has not, therefore, been scoped in for displacement assessment (e.g. the Hornsea and Dogger Bank projects). Also, since the first UK OWFs were consented in the early 2000s, baseline survey and assessment methodologies have evolved to reflect increasing understanding of the responses of red-throated diver to OWFs from empirical studies, and assessments have become more extensive and complex. Thus, limited information on red-throated divers is available for many older projects, including those situated in offshore areas of importance for this species. Finally, as baseline surveys for OWFs have been carried out over different time periods, it is possible that at some sites the numbers and distribution recorded during baseline surveys have been affected by displacement from nearby OWFs which were operational at the time the surveys were undertaken. A summary of available project-specific information on red-throated diver displacement and estimates of seasonal and annual mortality due to displacement for OWFs where this is available, is included in ES Appendix 13.3 (Document Reference: 3.3.14).

438. The variation in available information on red-throated diver displacement for existing OWFs means that the ‘standard’ approach of extracting quantitatively expressed predicted effects from the assessments of OWFs within the area of search (Section 13.8.3) may underestimate cumulative displacement effects for red-throated diver. For this reason, in addition to the standard approach, the relative contribution of OWFs within the UK North Sea to cumulative displacement of this species has been considered based on modelled at-sea density estimates from the Seabird Mapping and Sensitivity Tool (SeaMAST) (Bradbury *et al.*, 2014).

439. The assessment has been conducted using the precautionary rates of displacement and mortality recommended by the SNCBs (100% displacement and 1 – 10% mortality within the 4km buffer). However, for some OWFs the available displacement predictions are provided for a range of 90 – 100% displacement (rather than just 100%), as well as 1 – 10% mortality.

Standard assessment based on OWF baseline data

440. The number of red-throated divers predicted to suffer mortality due to operational phase displacement from all OWFs in the UK North Sea BDMPS is included by development (where data are available) in ES Appendix 13.3 (Document Reference: 3.3.14). In each case the data presented are a range based on 90 – 100% birds displaced within the OWF and a buffer, and 1 – 10% mortality of displaced birds. Whilst 4km was the preferred buffer where it was available, the buffer zones used for OWFs included in this assessment varied between 0km and 4km depending on the data available. Seasonal totals for all OWFs where data on displacement of red-throated divers are available are included in Table 13.50.

441. The cumulative mortality predictions are assessed against the largest BDMPS, 13,277 during spring and autumn migration, and the biogeographic red-throated diver population with connectivity to UK waters, 27,000 (Furness 2015).

442. At the average baseline mortality rate for red-throated diver of 0.233, the number of individuals expected to suffer mortality from the BDMPS over one year is 3,094. The addition of 35 – 381 individuals increases the mortality rate by 1.1 – 12.3%.

443. In relation to the biogeographic population, the number of individuals expected to suffer mortality over one year is 6,291. The addition of 35 – 381 birds) increases the mortality rate by 0.6 – 6.1%

Table 13.50 Cumulative predicted displacement mortality for red-throated diver from OWFs

Site (tier)	No. of predicted bird mortalities as a result of displacement (90 – 100% displacement within OWF and 4km buffer, 1 – 10% mortality of displaced birds)			
	Autumn migration	Winter	Spring Migration	Annual
Galloper (1)	-	-	-	1 – 14
Greater Gabbard (1)	-	-	-	4 – 40
Thanet (1)	-	-	-	1 – 2
East Anglia ONE (1)	0.4 – 5	1 – 10	1.4 – 15	2.8 – 30
East Anglia ONE North (3)[1]	-	-	-	0.1 – 1
East Anglia THREE (3)	0.4 – 5	0.2 – 2	2 – 20	2.6 – 27
East Anglia TWO (3)	0	0 – 2	2 – 25	3 – 28
Norfolk Boreas (3)	0 – 1	1 – 15	5 – 62	6 – 78
Norfolk Vanguard (3)	0.4 – 8	3.2 – 39	3 – 32	6.6 – 79
Sheringham Shoal & Dudgeon Extension Projects (3)	2 – 14	0 – 2	3 – 23	4 – 39
Five Estuaries (4)[2]	0	0 – 2	0 – 3	0 – 5
Outer Dowsing (4)[2]	0.3 – 2.5	0.2 – 2.4	2.2 – 21.7	2.8 – 28.2
North Falls (mean values)	0	0 – 2	1 – 7	1 – 9
			Total	35 – 381

Notes: 1. For East Anglia ONE North, the boundary was amended at consent, with its western extent moved from 2km away from the Outer Thames Estuary SPA at the nearest point, to 8km from the SPA boundary (Department for Business, Energy and Industrial Strategy (BEIS), 2022). Thus, the number of red-throated divers predicted to die from displacement will have been reduced compared with estimates presented in the ES accompanying the DCO Examination submission. Revised seasonal or annual abundance estimates of red-throated divers for the consented boundary of East Anglia ONE North appear not to have been published, so the seasonal and annual mortality predictions cannot be updated. Revision five of offshore ornithology without prejudice compensation measures for East Anglia ONE North (MacArthur Green and Royal HaskoningDHV, 2022 (Table 10.4)) provides estimates of the number of individuals displaced for the consented boundary of between 0 – 10.3, based respectively on a model of red-throated diver displacement developed by the applicant, and a straight-line approach recommended by Natural England, assuming a linear gradient in red-throated diver displacement from 100% at the OWF to 0% at 10km. It is assumed this range estimates the number of individuals to be displaced annually within the Outer Thames Estuary SPA, so that at 10% mortality of displaced individuals 0 – 1 red-throated divers would be predicted to die, and at 1% mortality 0 – 0.1 individual. While the cumulative assessment should consider birds displaced within a 4km buffer of East Anglia ONE North, these have been used as the best-available data. 2. ES submitted (and publicly available) after the cut off date for inclusion in this assessment, end March 2024. The values in the table are taken from the PEIR.

Assessment based on SeaMAST data

444. SeaMAST (Bradbury *et al.*, 2014) provides a common dataset covering most English offshore waters, describing modelled seabird densities in 3 x 3km squares based on both boat-based and visual aerial surveys. As both survey

methods may under-estimate the number of red-throated divers present compared with Digital Aerial Surveys, this dataset was used to assess the potential relative contribution of UK OWFs in the southern North Sea to displacement of red-throated divers during the non-breeding season, rather than provide robust estimates of the numbers of birds present in individual OWFs and 4km buffers. The methodology for producing estimates from SeaMAST data is described in ES Appendix 13.3 (Document Reference: 3.3.14). Whilst more recent evidence indicates that displacement effects of operational OWFs on red-throated divers frequently exceed 4km (SNCBs, 2022), a larger buffer was not used, primarily because Natural England advice for the North Falls EIA was to assess displacement for the OWF and a 4km buffer. In addition, to incorporate larger buffers with the SeaMAST data would cause complications due to extensive overlap of buffers at one OWF with buffers (and red-line boundaries) of other OWFs. Further, if displacement were assessed to a 10km buffer then it would be appropriate to apply a gradient to account for decreasing effect from the array area, while for the 4km buffer, 100% displacement is assumed.

445. The predicted number of red-throated divers within OWFs and 4km buffers, and therefore at risk of displacement, within the English North Sea is shown in Table 13.51. These are OWFs and buffers which overlap (at least partly), with the available SeaMAST data (further details in ES Appendix 13.3 (Document Reference: 3.3.14)). The number of birds predicted to be displaced is expressed as a percentage of the total reference population of non-breeding red-throated divers in the English North Sea, as estimated from SeaMAST data (19,978 individuals).

446. Overall, assuming 100% of displacement of red-throated divers from OWFs and 4km buffers, it is predicted from SeaMAST data that 15.3% of the total population of red-throated divers in the English North Sea would be potentially displaced by OWFs. For North Falls alone the predicted displacement from the OWF and 4km buffer is equivalent to 0.3% of the reference population, thus making a very small relative contribution to the cumulative total.

447. The SeaMAST predictions of the percentage of the reference population likely to be displaced can be used to estimate the increase in mortality rate of the reference population under scenarios of 1 – 10% mortality of displaced birds. If R represents the size of the reference population (to avoid making an assumption about the absolute number), and 1% of displaced birds are predicted to suffer mortality as a result of displacement, then the proportion of the population predicted to suffer mortality is $(1 \div 100) \times (15.3 \div 100) \times R$, which resolves to 0.00153R. Based on an 'average' annual mortality of 0.233 (Table 13.11) the increase in mortality rate of the reference population is $(0.00153 \times R) \div (0.233 \times R) \times 100$, which equals 0.7%. Performing a similar calculation for 10% mortality of displaced birds, the increase in mortality rate would be 7%.

Assessment conclusion

448. Predictions of the extent of cumulative displacement of red-throated divers in the North Sea and the estimated increase in mortality rate for red-throated divers have been made using two methods, the 'standard' method of collating available data on the number of red-throated divers predicted to suffer mortality as a result of displacement from OWF assessments within the area of search,

and using modelled at-sea density estimates from SeaMAST to estimate the relative contribution of OWFs to potential displacement of red-throated divers within a similar area.

449. The standard methodology may underestimate the effects of cumulative displacement as predictions of displacement effects are not available for all OWFs within the area of search. From this methodology, assuming 90 – 100% displacement of red-throated divers from OWFs, and 1 – 10% mortality of displaced birds, the estimated increase in population mortality from displacement is 1.2 – 13.3% of the UK North Sea BDMPS and 0.6 – 6.6% of the biogeographic population with connectivity to UK waters.

450. As noted in the Project alone assessment (para 310), the largest BDMPS population for red-throated divers (spring and autumn migration) is an underestimate and therefore the assessment over-estimates the effects on population mortality rate. The UK North Sea Migration BDMPS of 13,277 is less than the current population estimate for the Outer Thames Estuary SPA of 18,079 individual alone, the latter based on digital aerial surveys of the SPA during 2013 and 2018 (APEM, 2013; Irwin *et al.*, 2019). Thus, the BDMPS must be at least the same as the population of the OTE SPA, and may be larger. Adding together the latest population estimate for the Outer Thames Estuary SPA, with the cited population estimates of other SPAs within the UK North Sea BDMPS (Greater Wash, Outer Firth of Forth and St Andrews Bay complex and Moray Firth) gives a total of 20,661 birds (see RIAA Part 4 Section 4.4.1.4.1, Table 4.6). This could be a minimum estimate as it doesn't include birds using areas outside SPA, or may include some double-counting if birds move around within the BDMPS area and use more than one SPA and areas outside SPAs in a given migratory season). If, on a conservative basis, the OTE SPA population estimate was to be used in lieu of the BDMPS population, the estimated increase in population mortality from displacement would be 0.2 – 2.3%.

451. Based on SeaMAST predictions of the numbers of red-throated divers at risk of displacement from OWFs in the area of search, 100% displacement of birds from OWFS and 4km buffers, and 1 – 10% mortality of displaced birds, the predicted increase in mortality rate of the reference population is 0.7 – 7%.

452. Assuming that the mortality rate of displaced birds is a maximum of 1%, based on a review of evidence and scientific judgement (MacArthur Green, 2019c, see para 130), then for both the standard approach (using the population of the Outer Thames Estuary SPA as a proxy for the BDMPS) and relative approaches, the predicted increase in the mortality rate of the reference population is less than 1%. This magnitude of increase in mortality would not materially alter the background mortality of the population and would be undetectable.

453. Trends in the GB and North Sea BDMPS wintering populations of red-throated divers are unknown. The most recent GB population estimate of 17,000 individuals is based on largely on visual aerial surveys between 2001-2006; previous estimates ranged from 1,000 to 15,000 – 20,000 individuals (O'Brien *et al*., 2008). The population of the Outer Thames Estuary SPA originally cited as 6,466 individuals, based on visual aerial surveys between 1989 and 2007 (Natural England and JNCC 2010, 2015), has recently been revised to 18,079

individuals based on digital aerial surveys in 2013 and 2018. Natural England (2023a) states that '*these increases are thought to reflect improved survey methods and techniques, namely the use of digital aerial surveys, which has provided more accurate counts and suggests that previous counts [from visual aerial surveys] have been significant underestimates*'.

454. Red-throated diver does not appear to be a species at high risk for HPAI. Few reports of red-throated diver mortality from HPAI have been found; Defra (2023) reports one individual testing positive, and NatureScot (2023) reports 11 dead birds recorded between 17 October 2023 and 8 January 2023, although not all of these birds may have been tested for HPAI.

455. The impact magnitude of cumulative year-round displacement mortality from OWFs to red-throated divers is assessed as negligible. As the species is of high sensitivity to disturbance, the effect outcome is minor adverse, which is not significant in EIA terms.

Table 13.51 Predicted abundance of red-throated diver within OWFs in the North Sea from SeaMAST data and percentage of reference population

Project (Tier)	OWF	% reference population	4km buffer	% reference population	OWF + 4km buffer	% reference population
Blyth Demonstration (1)	0	0.00	0.5	0.00	0.6	0.00
East Anglia ONE (1)	5.8	0.03	16.1	0.08	21.9	0.11
Greater Gabbard and Galloper (1)	35.4	0.18	77.9	0.39	113.3	0.57
Gunfleet Sands (1)	54	0.27	487.2	2.44	541.2	2.71
Humber Gateway (1)	0.1	0.00	0.7	0.00	0.8	0.00
Kentish Flats (1)	48.6	0.24	343.7	1.72	392.3	1.96
Lincs, Lynn and Inner Dowsing (1)	3.1	0.02	18.4	0.09	21.5	0.11
London Array (1)	337.4	1.69	1165.1	5.83	1502.6	7.52
Race Bank (1)*	0.7	0.00	2.7	0.01	3.4	0.02
Scroby Sands (1)	9.7	0.05	80	0.40	89.6	0.45
Sheringham Shoal (1)*	0.1	0.00	0.6	0.00	0.7	0.00
Teesside (1)	0	0.00	0.8	0.00	0.9	0.00
Thanet (1)	5.7	0.03	34.8	0.17	40.5	0.20
Westermost Rough (1)*	0.1	0.00	0.8	0.00	0.9	0.00
East Anglia ONE North (consented boundary) (3)	31.7	0.2	89.1	0.4	120.8	0.6
East Anglia THREE (3)	5.9	0.03	13.2	0.07	19.1	0.10
East Anglia TWO (3)	19	0.10	71.4	0.36	90.4	0.45
Norfolk Boreas (3)*	2.9	0.01	3.5	0.02	4.6	0.02
Norfolk Vanguard (3)*	9.4	0.05	13.5	0.07	24.6	0.12
Sheringham Shoal Extension Project (3)	0.0	0.0	0.6	0.0	0.6	0.0
Five Estuaries (4)**	1.9	0.0	3.1	0.0	5.0	0.0
North Falls	5.5	0.03	45.8	0.23	51.3	0.26

Project (Tier)	OWF	% reference population	4km buffer	% reference population	OWF + 4km buffer	% reference population
Totals	577	2.9	2469.5	12.4	3046.6	15.3
Reference population as estimated from SeaMAST data = 19,978 individuals. * OWF and / or 4km buffer partly outside the coverage of SeaMAST data, see ES Appendix 13.3 (Document Reference: 3.3.14); ** ES submitted (and publicly available) but after the cut off date for inclusion in this assessment, end March 2024. The values in the table are taken from the PEIR.						

13.8.3.3 *Cumulative effect 3: Operational collision risk*

13.8.3.3.1 Gannet

456. The number of birds predicted to suffer mortality due to collision at all OWFs in the UK North Sea and Channel BDMPS is provided for all relevant projects in ES Appendix 13.3 (Document Reference: 3.3.14). The seasonal totals for all OWFs in tiers 1 to 3, along with the contribution made by all known relevant OWFs in tier 4 and above for which data were available, is presented in Table 13.52. For North Falls the means for the worst case are included (MiRD scenario, Table 13.37). The total annual mortality for all projects is 494 birds, towards which North Falls contributes only two birds, or 0.4% of the total predicted annual collisions.

457. As noted above (Section 13.6.2.2.3), Natural England's (2022c) interim advice and the subsequent update email (Natural England, 2023) on CRM parameters recommends for gannet sCRM: that densities from baseline surveys within OWF array areas should be reduced by 70% to account for high macro-avoidance; that the avoidance rate is increased from 0.989 to 0.9928 (±0.0003) for the stochastic (MacGregor *et al*., 2018) model, and 0.9924 (±0.0001) for the deterministic Band (2012) model; and that the NAF is reduced from 0.1-0.2 to 0.08. These changes will all reduce the predicted collision risk for a given OWF. Where appropriate, the seasonal and annual predicted collisions for other OWFs included in the cumulative assessment, have been adjusted to reflect the updated macro-avoidance and avoidance rates (see ES Appendix 13.3 (Document Reference: 3.3.14) for full details of the calculations). The reduction in the NAF ((the proportion of birds estimated to be active at night compared with daytime) would also result in a reduction of collision risk, but has not been accounted for as this would require collision risk models to be re-run.

Table 13.52 Cumulative Collision Mortality Predictions for Gannet for all OWFs in CEA, incorporating 70% macroavoidance and latest revised avoidance rate

Tiers / development	Breeding	Autumn migration	Spring migration	Annual
1 to 3[1]	212	151	52	414
Green Volt (Tier 3)	5	0	1	5
Sheringham and Dudgeon Extension Projects (Tier 3)	0	1	0	1
Berwick Bank (Tier 4)	29	3	0	32
Dogger Bank South (Tier 4)	8	3	0	10
Five Estuaries (Tier 4)[1]	1	1	0	1
Outer Dowsing (Tier 4)[1]	3	0	0	4
Rampion 2 (Tier 4)	3	1	1	5
West of Orkney (Tier 4)	12	2	0	15
North Falls	1	1	1	2
Total	**274**	**165**	**55**	**494**
Note that the seasonal and annual totals presented are rounded to the nearest integer, but sums are based on the actual values, so that annual totals may not always match the seasonal totals. 1. Values for individual OWFs in ES Appendix 13.3 (Document Reference: 3.3.14). Excludes two recently consented sites listed below. 2. ES submitted (and publicly available) after the cut off date for inclusion in this assessment, end March 2024. The values in the table are taken from the PEIR.				

458. To assess the magnitude of the year-round impact of cumulative OWF collision on gannet, two background populations are considered. Firstly, the largest relevant BDMPS population of 456,298 individuals (UK North Sea and Channel BDMPS, autumn migration; Furness, 2015). Based on the published baseline mortality of 18.7% across all age classes (Table 13.11), 85,328 individual gannets from this population would be expected to suffer mortality annually. Secondly, the biogeographic population with connectivity to UK waters of 1,180,000 (Furness, 2015), from which 220,660 individuals would be expected to suffer mortality annually from this population using the same all age class mortality rate.

459. The predicted level of additional mortality, assuming 70% macro-avoidance, would represent a 0.6% ((494 ÷ 85328) x 100) increase in annual mortality within the largest BDMPS population, or a 0.2% ((494 ÷ 220660) x 100) increase in annual mortality within the annual biogeographic population with connectivity to UK waters. These mortality increases would not be detectable at the population level within the context of natural variation.

460. It is also worth noting that there are levels of precaution built into these collision mortality predictions. A number of OWFs in English waters have been built out, or will be built out (subject to non-material change), to designs with a lower collision risk than the worst-case consented design envelope. However, on the advice of Natural England, the worst-case estimate of collision risk is used in the cumulative assessment as it is considered to be legally secured. The use of consented rather than as-built OWF parameters may lead to the overestimation of cumulative collision predictions for UK OWFs by up to 14% for this species (MacArthur Green, 2017). For Scottish OWFs the values for the as-built designs, if different from consented designs (and if available), are used, as these are accepted by Marine Scotland and NatureScot. Also, a reduced nocturnal activity rate (of 0.08, compared with previous values of 0.1-0.2, and 0-0.25, at which CRM has been run for other OWFs in the cumulative assessment) would result in lower estimates of collision risk, although it is not possible to estimate the extent of any reduction without running comparative models for a sample of OWFs.

461. The UK population of gannet has undergone a long-term increase (JNCC, 2021). Recently, however, some gannet breeding colonies have been severely affected by HPAI (Lane *et al*., 2024, Tremlett *et al*., 2024). The effect of HPAI on future population trends and conservation status of breeding gannet populations at UK breeding colonies is therefore uncertain. HPAI mortality of 802 gannets was recorded in England during 2022, with adult mortality representing approximately 3% of the England breeding population (Royal HaskoningDHV, 2023). In Scotland, a 'minimum loss' of 11,175 birds is reported for 2022 – by far the largest number of individuals reported for any seabird species (NatureScot, 2023). The source of this number is not clearly stated, but it seems to be based on numbers of dead gannets reported to NatureScot and considered a minimum estimate as not all dead birds will have been recorded or reported (and it seems likely that only a small proportion of actual deaths from HPAI would be encountered by people). In Wales there were an estimated 5,000 mortalities at Grassholm (Tremlett *et al*., 2024). A total of 13 gannet breeding sites throughout the UK were included in seabird colony counts carried out in

2023 to assess changes following the 2021-22 HPIA outbreak (Tremlett *et al*., 2024). The total numbers of Apparently Occupied Sites / Nests (AOS / AONs) decreased by 25% across all sites surveyed compared with pre-HPAI baseline counts, although the decrease was not consistent across sites, with some sites increasing. Tremlett *et al*. (2024) do not present any evidence or comment in relation to the role of HPAI in these changes. Serological investigation of a sample of 17 gannets at the Bass Rock found apparently healthy birds with antibodies for HPAI, indicating that some birds had recovered from infection (most of these recovered birds had black or mottled irises, rather than the normal pale blue colour, which appears to be an indicator of previous HPAI infection; Lane *et al*., 2024).

462. Taking into account the small predicted increase in baseline mortality rate for gannet from cumulative collision risk, and the potential for over-estimation of collisions at some English OWFs, the year-round effect of collisions is considered to be of low magnitude. Gannets are considered to be of medium sensitivity to collision mortality and the effect significance is therefore minor adverse, which is not significant in EIA terms.

13.8.3.3.2 Great black-backed gull

463. The number of birds predicted to suffer mortality due to collision at all OWFs in the UK North Sea and Channel BDMPS is provided for all relevant projects in ES Appendix 13.3 (Document Reference: 3.3.14). The seasonal totals for all OWFs in tiers 1 to 3, along with the contribution made by all known relevant OWFs in tier 4 and above for which data were available, is presented in Table 13.53. For North Falls the mean for the worst case is included (MiRD scenario, Table 13.38). The total annual mortality for all projects is 1,501 birds, towards which North Falls would contribute three birds, representing only 0.2% of the predicted collisions.

Table 13.53 Cumulative collision mortality predictions for great black-backed gull for all OWFs in CEA, incorporating latest revised avoidance rate

Tiers / development	Breeding	Non-breeding	Annual
1 to 3[1]	269	1,164	1,434
Green Volt (Tier 3)	0	7	7
Sheringham and Dudgeon Extension Projects (Tier 3)	1	9	10
Berwick Bank (Tier 4)	0	0	0
Dogger Bank South (Tier 4)	1	5	6
Rampion 2 (Tier 4)	6	14	20
West of Orkney (Tier 4)	2	12	13
Five Estuaries (Tier 4)[1]	1	2	4
Outer Dowsing (Tier 4)[1]	3	1	4
North Falls	0	3	3
Total	**284**	**1,217**	**1,501**
Note that the seasonal and annual totals presented are rounded to the nearest integer, but sums are based on the actual values, so that annual totals may not always match the seasonal totals. 1. Values for individual OWFs in ES Appendix 13.3 (Document Reference: 3.3.14). Excludes two recently consented sites listed			

Tiers / development	Breeding	Non-breeding	Annual
below. 2. ES submitted (and publicly available) after the cut off date for inclusion in this assessment, end March 2024. The values in the table are taken from the PEIR.			

464. As noted above (Section 13.6.2.2.3), Natural England's (2022c) interim advice and the subsequent update (Natural England, 2023) on CRM parameters recommends for that avoidance rate for great black-backed gull is reduced from 0.995 to 0.9939 (±0.0004) for the stochastic (MacGregor *et al.*, 2018) model, and to 0.9936 (±0.0001) for the deterministic Band (2012) model. The sCRM for the North Falls ES reflects this newer guidance. Where appropriate, the seasonal and annual predicted collisions for other OWFs included in the cumulative assessment, have been adjusted to reflect the revised avoidance rates (see ES Appendix 13.3 (Document Reference: 3.3.14) for full details of the calculations).

465. Not all projects included in the CEA provided a seasonal breakdown of collision impacts for this species. Natural England has previously advised that an 80:20 split between the non-breeding and breeding seasons is appropriate for lesser black-backed gull in terms of apportioning collision estimates to biologically relevant seasons where this is not split by the original assessment. This is also considered to be appropriate for great black-backed gull, and has been applied for OWFs where only an annual collision mortality estimate is available.

466. To assess the magnitude of the year round impact of cumulative OWF collision on great black-backed gull, two background populations are considered. Firstly, the largest relevant BDMPS population of 91,399 individuals (non-breeding season UK North Sea BDMPS; Furness, 2015)). Based on the published baseline mortality of 9.3% across all age classes (Table 13.11), 8,500 individuals from this population would be expected to suffer mortality annually. Secondly, the biogeographic population with connectivity to UK waters of 235,000 individuals (Furness, 2015), from which 21,855 birds would be expected to suffer mortality annually using the same all age class mortality rate.

467. The predicted level of additional mortality would represent a 17.7% (1501 ÷ 8500 x 100) increase in annual mortality within the largest BDMPS population, or a 6.9% (1501 ÷ 21855 x 100) increase in annual mortality within the annual biogeographic population with connectivity to UK waters. These predicted mortality increases could be detectable at the population level within the context of natural variation in both background populations.

468. There are substantial levels of precaution built into these mortality predictions, notably in two areas. Firstly, consented parameters have been used for English OWFS. The use of consented, rather than as-built, parameters may lead to the overestimation of cumulative collision predictions for UK OWFs by up to 30% for this species (MacArthur Green, 2017). Secondly, the recommended nocturnal activity rate of 37.5% (Natural England, 2023) may be an overestimate. Whilst no species-specific information for great black-backed gull is available, available information for lesser black-backed gull suggests that nocturnal activity values of 25% or less may be more realistic (see para 489).

469. A density dependent population model for great black-backed gull, at the scale of the UK North Sea BDMPS (Furness, 2015), was developed during the East Anglia THREE assessment (Royal HaskoningDHV, 2016). An additional annual mortality of 900 birds resulted in impacted populations after 25 years which were 6.1% to 7.7% smaller than predicted populations in the absence of OWF collision risk impacts. Great black-backed gull has been subject to relatively little research and estimates of demographic rates have been categorised as low quality (Horswill and Robinson, 2015).

470. The breeding population of great black-backed gull in the UK declined 52% between the Seabird 2000 and the Seabirds Count censuses (Burnell *et al*., 2023). Most of the decline took place in Scotland, with a small decrease in England, and increases in the Wales and Northern Ireland populations.

471. As for other seabird species, the impact of HPAI on great black-backed gulls in the UK is currently uncertain. HPAI mortality of 26 individuals was recorded in this species in England in 2022, representing about 0.7% of the England breeding population (Royal HaskoningDHV, 2023). A ‘minimum loss’ of 507 large gulls (herring, lesser black-backed and great black-backed gull) to HPAI is reported for Scotland in 2022 (NatureScot 2023). The source of this number is not clearly stated, but it seems to be based on numbers of dead birds reported to NatureScot and considered a minimum estimate as not all dead birds will have been recorded or reported (and it seems likely that only a small proportion of actual deaths from HPAI would be encountered by people and reported). A total of 27 breeding sites or groups of sites throughout the UK were included in seabird colony counts carried out in 2023 to assess changes following the 2021-22 HPIA outbreak (Tremlett *et al*., 2024). The total numbers of AONs decreased by 20% across all sites surveyed compared with pre-HPAI baseline counts, although the decline was not consistent across all sites surveyed, and some sites increased. Tremlett *et al*., (2024) do not present any evidence or comment in relation to the role of HPAI in these declines.

472. Accounting for the precaution included in the assessments, the year-round magnitude of cumulative operational collision on great black-backed gull is assessed as medium. This conclusion is considered appropriate because potential mortality increases of more than 1% are predicted due to cumulative OWF collision mortality, because of the sources of precaution that have been identified in the collision predictions, and because of the potentially low reliability of demographic parameters for this species in relation to PVAs. Great black-backed gull is considered to be of medium sensitivity to collision mortality. The effect significance is moderate adverse. This predicted impact is potentially significant in EIA terms.

473. It is noted that the Project has provided mitigation that has reduced collision risk to this species (i.e. through increasing the air gap from 22m to 26.6m above HAT, Table 13.2), and also the very small contribution of the Project (less than 1% of total predicted mortality) to the cumulative effect. It is considered that the North Falls does not make any material contribution to the cumulative total.

13.8.3.3.3 Kittiwake

474. The number of birds predicted to suffer mortality due to collision at all OWFs in the UK North Sea and Channel BDMPS is provided for all relevant projects in

ES Appendix 13.3 (Document Reference: 3.3.14). The seasonal totals for all OWFs in tiers 1 to 3, along with the contribution made by all relevant OWFs in tier 4 and above for which data were available, is presented in Table 13.54. For North Falls the means for the worst case are included (MaRD scenario, Table 13.39). The total annual mortality for all projects is 3,344 birds, towards which North Falls would contribute 20 birds, representing only 0.6% of the predicted collisions.

475. .As noted above (Section 13.6.2.2.3), Natural England's (2022c) interim advice and the subsequent update email (Natural England, 2023) on CRM parameters recommends for kittiwake that the avoidance rate is increased from 0.989 to 0.9928 (±0.0003) for the stochastic (MacGregor *et al*., 2018) model, and 0.9924 (±0.0001) for the deterministic Band (2012). Where appropriate, the seasonal and annual predicted collisions for other OWFs included in the cumulative assessment, have been adjusted to reflect the updated avoidance rates (see ES Appendix 13.3 (Document Reference: 3.3.14) for full details of the calculations).

476. To assess the magnitude of the year-round impact of cumulative OWF collision on kittiwake, two background populations are considered. Firstly, the largest relevant BDMPS population of 829,937 individuals (autumn migration season UK North Sea BDMPS; Furness, 2015). Based on the published baseline mortality of 15.7% across all age classes (Table 13.11), 130,300 individuals from this population would be expected to suffer mortality annually. Secondly, the biogeographic population with connectivity to UK waters of 5,100,000 (Furness, 2015), of which 800,700 individuals would be expected to suffer mortality annually using the same all age class mortality rate.

Table 13.54 Cumulative collision predictions for kittiwake for all OWFs included in CEA, incorporating the latest revised avoidance rate

Tiers / development	Breeding	Autumn migration	Spring migration	Annual
1 to 3[1]	948	796	639	2,369
Green Volt (Tier 3)	7	6	1	14
Sheringham and Dudgeon Extension Projects (Tier 3)	7	4	1	12
Berwick Bank (Tier 4)	294	107	72	473
Dogger Bank South (Tier 4)	164	48	30	242
Five Estuaries (Tier 4)[2]	16	11	8	35
Outer Dowsing (Tier 4)[2]	29	19	52	99
Rampion (Tier 4)	1	10	18	29
West of Orkney (Tier 4)	34	16	4	55
North Falls	9	4	8	20
Totals	**1,510**	**1,020**	**833**	**3,348**
Note that the seasonal and annual totals presented are rounded to the nearest integer, but sums are based on the unrounded values, so that annual totals may not always match the seasonal totals. 1. Values for individual OWFs in ES Appendix 13.3 (Document Reference: 3.3.14). Excludes two recently consented sites listed below. 2. ES submitted (and publicly available) after the cut off date for inclusion in this assessment, end March 2024. The values in the table are taken from the PEIR.				

477. The cumulative annual collisions would represent a 2.6% (3,344 ÷ 130,300 x 100) increase in the annual mortality of the largest BDMPS population, and a 0.4% (3,344 ÷ 800,700 x 100) increase in the annual mortality of the annual biogeographic population with connectivity to UK waters. The predicted mortality increase within the largest BDMPS population could be detectable at the population level within the context of natural variation, though this is not the case for the annual biogeographic population with connectivity to UK waters.

478. There are, however, substantial levels of precaution built into these mortality predictions, notably in two areas. Firstly, consented parameters have been used for English OWFS. The use of consented rather than as-built OWF parameters may lead to the overestimation of cumulative collision predictions for UK OWFS by up to 17% for this species (MacArthur Green, 2017). Secondly, the assumed maximum nocturnal activity of 37.5% may be an overestimate. A review of nocturnal activity from studies of kittiwakes fitted with geolocator (GLS) tags estimated a 17% nocturnal activity rate for the non-breeding seasons (Royal HaskoningDHV, 2019b, Furness 2019). Royal HaskoningDHV (2019b) refers to a similar analysis for the breeding season which estimated a 20% nocturnal activity rate.

479. Density dependent population models assessing the effects of cumulative OWF collision mortality on the kittiwake BDMPS populations indicate that an annual mortality of 4,000 birds would result in a population 3.6% to 4.4% smaller after 25 years than that predicted in the absence of the additional mortality (MacArthur Green, 2015). To place this predicted magnitude of change in context, over three approximately 15-year periods between censuses, the British kittiwake population changed by +24% (1969 to 1988), -25% (1988 to 2002), -42% (2002-2021) (JNCC, 2021; Burnell *et al*, 2024). When considered within this context, it seems likely that declines of up to 4.4% across a longer (25 year) period against a background of changes an order of magnitude larger will be undetectable. It is possible that the longer-term decline observed in the UK kittiwake population will continue, and that recovery over this period is unlikely on the basis that climate change seems to be a key driver in kittiwake declines (Descamps *et al.*, 2017).

480. In addition, the impact of HPAI on UK kittiwake populations is currently uncertain and may affect the population trends and conservation status of the species. HPAI mortality of 925 kittiwakes was recorded in England during 2022, with adult mortality representing approximately 0.5% of the England breeding population (Royal HaskoningDHV, 2023). In Scotland, a 'minimum loss' of 760 birds is reported for 2022 (NatureScot, 2023). The source of this number is not clearly stated, but it seems likely it is based on numbers of dead seabirds reported to NatureScot and considered a minimum estimate as not all dead birds will have been recorded and reported (and it seems likely that only a small proportion of actual deaths from HPAI would be encountered by people and reported). A total of 29 kittiwake breeding sites or groups of sites throughout the UK were included in seabird colony counts carried out in 2023 to assess changes following the 2021-22 HPAI outbreak (Tremlett *et al*., 2024). The total numbers of AONs increased by 8% across all sites surveyed compared with pre-HPAI baseline counts, although the overall increase was driven by a 21%

increase in numbers at sites in Scotland (in contrast to the long-term decline that has taken place in Scotland, Burnell *et al.,* 2023), with sites in England, Wales and Northern Ireland decreasing by respectively -18%, -17% and -29%. Tremlett *et al.* (2024) do not present any evidence or comment in relation to the role of HPAI in these changes.

481. In relation to the status of the UK North Sea kittiwake population, it is unclear whether estimates of additional mortality due to cumulative OWF collisions could significantly increase the rate of the ongoing long term decline or prevent the population from recovering should the wider-scale environmental conditions become more favourable for the species. Agreed compensation measures for a number of recently consented OWFs in the southern North Sea (i.e. East Anglia ONE North, East Anglia TWO, Hornsea Projects Three and Four, Norfolk Boreas and Norfolk Vanguard, as well as Sheringham Shoal and Dudgeon Extension Projects, consented after the cut-off date of end March 2024 for inclusion in this assessment) include measures to increase the number of breeding kittiwakes on the east coast of England through the creation of new artificial nesting colonies to boost the breeding numbers and productivity of the population. The aim of the compensation is to reduce the net effect of an OWF on collision mortality of kittiwake to zero. Excluding predicted mortality from the OWFs with compensation would reduce the total annual predicted kittiwake collision mortality to 3055 (not including Sheringham and Dudgeon Extension Projects). On a precautionary basis this reduction has not been applied to the cumulative assessment here.

482. The year-round magnitude of cumulative collision on kittiwake is assessed as medium. Kittiwakes are considered to be of medium sensitivity to collision mortality and the effect outcome is therefore moderate adverse. Thus, the year-round cumulative predictions in Table 13.54 could represent a significant impact in EIA terms, although collisions are considered to be overestimated for the reasons described above.

483. Under the latest guidance from Natural England and Natural Resources Wales on EIA reference populations (see para 68), the largest seasonal BDMPS is 839,456 individuals for the UK North Sea; if this were applied, the percentage increases in baseline mortality would be a little smaller than those given above.

13.8.3.3.4 Lesser black-backed gull

484. The number of birds predicted to suffer mortality due to collision at all OWFs in the UK North Sea and Channel BDMPS is provided for all relevant projects in ES Appendix 13.3 (Document Reference: 3.3.14). The seasonal totals for all OWFs in tiers 1 to 3, along with the contribution made by all known relevant OWFs in tier 4 and above for which data were available, is presented in Table 13.55. The total annual mortality for all projects is 751 birds, towards which North Falls would contribute nine birds, representing 1.2% of the total predicted collisions.

485. Not all projects included in the CEA provided a seasonal breakdown of collision impacts for this species. Natural England has previously advised that an 80:20 split between the non-breeding and breeding seasons is appropriate for lesser

black-backed gull in terms of apportioning collision estimates to biologically relevant seasons where this is not split by the original assessment, so this ratio has been applied where relevant.

486. As noted above (Section 13.6.2.2.3), Natural England's (2022c) interim advice and the subsequent update (Natural England, 2023) on CRM parameters recommends that the avoidance rate for lesser black-backed gull is reduced from 0.995 to 0.9939 (±0.0004) for the stochastic (MacGregor *et al*., 2018) model, and to 0.9936 (±0.0001) for the deterministic Band (2012) model. The sCRM for the North Falls ES reflects this newer guidance. Where appropriate, the seasonal and annual predicted collisions for other OWFs included in the cumulative assessment, have been adjusted to reflect the revised avoidance rates (see ES Appendix 13.3 (Document Reference: 3.3.14) for full details of the calculations).

487. To assess the magnitude of the year-round impact of cumulative OWF collision on lesser black-backed gull, two background populations are considered. Firstly, the largest relevant BDMPS population of 209,007 individuals (autumn migration season UK North Sea BDMPS; Furness, 2015). Based on the published baseline mortality of 12.5% across all age classes (Table 13.11), 26,125 individuals from this population would be expected to suffer mortality annually. Secondly, the biogeographic population with connectivity to UK waters of 1,707,000 (Furness, 2015), of which 213,375 individuals would be expected to suffer mortality annually using the same all age class mortality rate.

Table 13.55 Cumulative collision predictions for lesser black-backed gull for all OWFs Included in CEA, incorporating latest revised avoidance rates

Tiers / development	Breeding	Non-breeding	Annual
1 to 3[1]	204	475	680
Green Volt (Tier 3)	0	0	0
Sheringham and Dudgeon Extension Projects (Tier 3)	2	0	2
Berwick Bank (Tier 4)	8	0	8
Dogger Bank South (Tier 4)	1	0	1
Five Estuaries (Tier 4)[2]	38	6	44
Outer Dowsing (Tier 4)[2]	3	1	4
Rampion 2 (Tier 4)	3	1	4
West of Orkney (Tier 4)	0	0	0
North Falls	7	2	9
Totals	**265**	**486**	**751**
Note that the seasonal and annual totals presented are rounded to the nearest integer, but sums are based on the actual values, so that annual totals may not always match the seasonal totals. 1. Values for individual OWFs in ES Appendix 13.3 (Document Reference: 3.3.14). Excludes two recently consented sites listed below. 2. ES submitted (and publicly available) after the cut off date for inclusion in this assessment, end March 2024. The values in the table are taken from the PEIR.			

488. The predicted level of additional mortality would represent a 2.9% (751 ÷ 26125 x100) increase in annual mortality within the largest BDMPS population, and a 0.4% (751 ÷ 213375 x100) increase in annual mortality within the annual

biogeographic population with connectivity to UK waters. The predicted mortality increase within the largest BDMPS population could be detectable at the population level within the context of natural variation, though this is not the case for the annual biogeographic population with connectivity to UK waters.

489. There are substantial levels of precaution built into these mortality predictions, notably in two areas. Firstly, consented parameters have been used for English OWFS. The use of consented rather than as-built OWF parameters may lead to the overestimation of cumulative collision predictions for UK OWFs by up to 40% for this species (MacArthur Green, 2017). Secondly, the NAF recommended by Natural England (2023), (0.375 ± 0.0637) is a central value for use in sCRM which captures a range of 25-50% nocturnal activity, based on the assumption that flight activity is 25-50% of that during the daytime. This may be an over-estimate. A review of seabird nocturnal activity carried out for East Anglia THREE (MacArthur Green 2015a&b) cites a study of migration behaviour (a time where flight activity might be expected to be high) where an average of 48% of daylight and 12% of night was spent in flight (Klaassen *et al*., 2012), equivalent to 25% nocturnal activity. Ross-Smith *et al*. (2016) found that GPS-tracked lesser black-backed gulls breeding at Orford Ness spent relatively little time flying at night (0.3% of their total time), and also that birds flew at lower altitudes at night, especially over the sea. If this is representative of the behaviour of this species during the breeding season it suggests that the risk of collisions with OWFs at night may actually very small and may even be over-estimated by a NAF of 0.25.

490. The status of the UK population of lesser black-backed gull is unclear. Historically coastal breeding populations increased by 29% between seabird censuses in 1969 to 1970 (Operation Seafarer) and 1985 to 1988 (Seabird Colony Register), by 40% between 1985 to 1988 and 1998 to 2002 (Seabird 2000) and decreased by 43% between 1998 to 2002 and 2015 to 2021 (Burnell *et al.*, 2023). Seabird 2000 attempted for the first time to include inland and urban breeding colonies, whereas these were not included, or incompletely covered in previous censuses (JNCC, 2021), the most recent census (Burnell *et al.*, 2023) also included the inland colonies. Since 2002, a number of large coastal colonies of lesser black-backed gulls in the UK have declined, while the numbers nesting in urban locations (coastal and inland) has increased. When separated into coastal and inland colonies the most recent census showed a decline of 54% and an increase of 5% respectively (Burnell *et al.,* 2023).

491. As for other seabird species, the impact of HPAI on lesser black-backed gulls in the UK is currently uncertain. HPAI mortality of 207 individuals was recorded in this species in England in 2022, representing about 0.3% of the England breeding population (Royal HaskoningDHV, 2023). A ‘minimum loss’ of 507 large gulls (herring, lesser black-backed and great black-backed gull) to HPAI is reported for Scotland in 2022 (NatureScot 2023). The source of this number is not clearly stated, but it seems to based on numbers of dead gulls reported to NatureScot and considered a minimum estimate as not all dead birds will have been recorded or reported (and it seems likely that only a small proportion of actual deaths from HPAI would be encountered by people and reported). A total of 27 breeding sites or groups of sites throughout the UK were included in seabird colony counts carried out in 2023 to assess changes following the 2021-

22 HPAI outbreak (Tremlett *et al.,* 2024). The total numbers of AONs decreased by 25% across all sites surveyed compared with pre-HPAI baseline counts, although the decline was not consistent across all sites surveyed, and some sites increased. Tremlett *et al.,* 2024 do not present any evidence or comment in relation to the role of HPAI in these declines, and note that continuing declines in this species reflect the trend reported by the Seabirds Count census (Burnell *et al*., 2023).

492. The BDMPS and biogeographic populations of lesser black-backed gulls which are used as reference populations for the CEA include both birds from UK and overseas populations, so overall trends will be influenced by changes in all component populations.

493. A number of OWFs in the southern North Sea have recently been consented subject to compensation measures to enhance breeding numbers of lesser black-backed gull at the Alde-Ore Estuary SPA. Agreed compensation measures for a number of recently consented OWFs in the southern North Sea (East Anglia ONE North, East Anglia TWO, Norfolk Boreas and Norfolk Vanguard) include measures to increase the number of breeding lesser black-backed gull within the Alde-Ore SPA through provision of a predator proof breeding area and associated management to boost the breeding numbers and productivity of the population. The aim of the compensation is to reduce the net effect of an OWF on collision mortality of lesser black-backed gull to zero. Excluding predicted mortality from the OWFs with compensation would reduce the total annual predicted collision mortality to 710 individuals. On a precautionary basis this reduction has not been applied to the cumulative assessment here.

494. The predicted cumulative impact on lesser black-backed gulls due to OWF collisions, both year-round and within individual seasons (Table 13.55), may be overestimated as no account has been taken of OWFs consented subject to compensation; as-consented rather than lower as-built mortality estimates are included in the cumulative totals for English OWFs; and nocturnal activity may have been over-estimated in running CRM (which will produce higher collision estimates). There is however uncertainty about the trends of the UK and BDMPS populations of this species which seems to be declining. On a precautionary basis, given that cumulative predicted collisions represent more than a 1% increase in the BDMPS population, the impact is predicted to be of medium magnitude. Lesser black-backed gulls are considered to be of medium sensitivity to collision mortality and the effect outcome is therefore moderate adverse and could represent a significant impact in EIA terms.

13.8.3.3.5 Cumulative operational collision risk and displacement

Gannet

495. As discussed in Section 13.6.2.3 above, Gannets are considered at risk of operational displacement and collision risk.

496. At displacement rates of 60 to 80% and 1% mortality of displaced birds, 380 – 507 gannets are predicted to suffer mortality annually from cumulative displacement from operational OWFs within the UK North Sea and Channel (Table 13.45).

497. A total of 494 gannets are estimated to suffer mortality each year from collisions with OWFs in the same area (Table 13.52).

498. Combining the displacement and collision predictions gives an estimate of 870 – 1,001 gannet deaths each year.

499. This increased mortality is compared against the largest relevant BDMPS population of 456,298 individuals (UK North Sea and Channel BDMPS, autumn migration; Furness, 2015) from which 85,328 individual gannets would be expected to suffer mortality annually, and the biogeographic population with connectivity to UK waters of 1,180,000 (Furness, 2015), from which 220,660 individuals would be expected to suffer mortality annually based on an 'average' 18.7% mortality across age classes (Table 13.11).

500. The combined mortality from displacement and collision would represent a 1.0% (870 ÷ 85328 x 100) to 1.2% (1001 ÷ 85328 x 100) increase in the mortality rate of the largest BDMPS; and a 0.4% (870 ÷ 220,660 x 100) to 0.5% (1001÷ 220,660 x 100) increase in the mortality rate of the biogeographic population. Increases of 1% or more in the baseline mortality could be detectable at the population level.

501. As discussed previously, there are however a number of areas of precaution built into these estimates:

- The assumption that 1% of displaced gannets suffer mortality is considered likely to be an over-estimate (Section 13.6.2.1);
- Collision risk for English OWFs is based on consented designs and does not take account of built-out designs with lower collision risk; and
- Collisions for all OWFs in the cumulative assessment have been adjusted to account for the latest advice on macro-avoidance and avoidance rate, which has reduced the collision risk both cumulatively and for individual OWFs, but it has not been possible to account for the reduction in the NAF which will also reduce collision risk predictions (see para 457 above).

502. Also as noted previously, a density independent population model for the British gannet population (WWT Consulting *et al.*, 2012) concluded that population growth, on average, would remain positive until additional mortality exceeded 10,000 individuals per year while the lower 95% CI on population growth remained positive until additional mortality exceeded 3,500 individuals. Both values are substantially greater than the current cumulative collision and displacement total, which as described is considered to be highly precautionary. The risk of a 5% population decline was less than 5% for additional annual mortalities below 5,000, indicating a high probability that currently predicted cumulative collision mortalities, even when high precaution is applied, will not result in population declines.

503. The UK population of gannet has undergone a long-term increase (JNCC, 2021). Some gannet breeding colonies have however been severely affected by HPAI (Lane *et al*., 2024, Tremlett *et al*., 2024). The effect of HPAI on future population trends and conservation status of breeding gannet populations at UK breeding colonies is therefore uncertain. HPAI mortality of 802 gannets was recorded in England during 2022, with adult mortality representing

approximately 3% of the England breeding population (Royal HaskoningDHV, 2023); in Scotland, a 'minimum loss' (the source of this number is not clearly stated, but it seems to be based on numbers of dead gannets reported to NatureScot and considered a minimum estimate as not all dead birds will have been recorded or reported) of 11,175 birds is reported for 2022 – by far the largest number of individuals reported for any seabird species (NatureScot, 2023); in Wales there were an estimated 5,000 mortalities at Grassholm (Tremlett *et al*., 2024). A total of 13 gannet breeding sites throughout the UK were included in seabird colony counts carried out in 2023 to assess changes following the 2021-22 HPIA outbreak (Tremlett *et al*., 2024). At Bempton Cliffs, within the Flamborough and Filey Coast SPA, a whole colony count undertaken in 2022, coinciding with an outbreak of HPAI in gannet and other seabird species, reported 13,125 pairs, a decrease of 2% compared to the previous whole colony count of 13,392 pairs in 2017. A repeat count in 2023 found 15,233 pairs, suggesting recovery from HPAI and a continuation of the long term trend of increase at this colony (Aitken *et al*., 2017, Clarkson *et al*., 2022, Butcher *et al*., 2023). The total numbers of AOS / AONs decreased by 25% across all sites surveyed compared with pre-HPAI baseline counts, although the decrease was not consistent across sites, with some sites increasing. Tremlett *et al*. (2024) do not present any evidence or comment in relation to the role of HPAI in these changes.

504. Predicted levels of collision and displacement mortality are equivalent to a 1 – 1.2% increase in baseline mortality for gannet, albeit that the estimates are precautionary, particularly for displacement where there may in reality be no mortality impacts on gannets due to their high habitat flexibility and the extensive distances over which they forage during the breeding season (para 178 above). Thus the actual increase in population mortality rate may be less than 1%. There is uncertainty associated with impacts of HPAI on the UK breeding population of gannet, although monitoring at Bempton Cliffs within the Flamborough and Filey Coast SPA (the nearest breeding colony to North Falls) suggest the potential of UK gannet colonies to recover from HPAI in the short term and return to the long-term trend of increase.

505. The cumulative impact of collisions and displacement is considered to be of low magnitude. Gannets are identified as of medium sensitivity to collision mortality and displacement, and the effect outcome is therefore minor adverse. This predicted impact is not significant in EIA terms.

13.8.3.4 Cumulative effect 4: operational barrier effect on migratory bird species

506. In the Scoping Opinion (Planning Inspectorate, 2021) Natural England's comments on the likely significant effects scoped in for offshore ornithology receptors set out in the Scoping Report NFOW (2021), state that it is agreed that '*migratory species would be likely to encounter the turbine array only once during a given migration journey if North Falls is situated within their flight corridor, meaning they could potentially encounter the site and hence any barrier effect up to twice per year*' and that '*the energetic costs of such one-off avoidance events can be considered to be negligible for the North Falls project alone. However, we recommend that the impact of cumulative barrier effects* [of OWFs] *on migratory species is not scoped out of the assessment at this stage'*.

507. This section therefore considers the potential for cumulative barrier effects North Falls and other OWFs on migratory bird species other than seabirds. The area of potential cumulative effect is considered to be the UK North Sea and Channel offshore area, where migratory birds breeding, wintering, or stopping over in the UK might make sea crossings to Europe and beyond during passage flights. Potentially all OWFs within this area might contribute to a cumulative barrier effect. OWFs off the west coast of the UK are less likely to contribute to a cumulative barrier effect on migratory species passing through the North Sea and Channel, although some species may cross the North Sea and Great Britain en route to Ireland.

508. Whether or not a given migratory bird species is subject to cumulative barrier effect, and the extent of any effect, would depend on the number of OWFs encountered and avoided during a migratory flight, which in turn would depend on the flight path. A migratory bird approaching an OWF within the height range of turbines might choose to fly through, fly around, or fly over. If the bird was flying above the height of the turbines, it might also potentially change direction to avoid flying over the turbines, or change altitude. A barrier effect would be considered to occur if a bird decided to fly around or over an OWF, whereas any birds choosing to fly through a turbine array would be considered at risk of collision rather than barrier effect.

509. A review of the risk of OWFs to UK migratory birds (Wright *et al.*, 2012) found that, although a large number of species regularly migrate across UK offshore waters, there is limited knowledge of the routes taken and flight heights over the sea. In addition, flight heights are known to vary with weather conditions, and migratory routes might also potentially be affected by weather. Thus, estimating the number of OWFs which might be encountered by a given individual bird (or flock) of a particular species during an offshore passage flight is difficult. Further little empirical data is available which would predict the proportion of individuals (or flocks) of a given species which might choose to avoid rather than fly through an OWF, and thus be subject to barrier effect (as opposed to collision risk)

510. Some tracking studies of migratory birds are available, where individual birds are fitted with tracking devices so that migratory routes can be plotted. These tend to be larger species such as geese and swans (e.g. Griffin *et al.*, 2011; 2016, which consider geese and swan migrations in relation to wind farms) due to current limitations on the size and weight of suitable tracking devices which mean that they can often only be fitted on medium to large birds (although the weight of tags is reducing with advances in technology and a number of smaller species are now subject to migratory tagging studies).

511. A tracking study of Bewick's swans migrating across the North Sea between south-east England and continental Europe (WWT Consulting, 2016) found a mean of 4.4 (±3.8) OWF areas was crossed per migration (n=18 migratory tracks). The OWFs included developments in UK, German, Dutch, Belgian and Swedish waters, and considered both operational sites and footprints of sites which were not yet constructed.

512. Radar has also been used to study the behaviour of migrating birds in relation to individual OWFs. A four-year radar monitoring study was carried out between 2007 – 2010 at the Lynn and Inner Dowsing OWFs, off the south-east coast of

England, (Plonczkier and Simms, 2012), which lie on the migration path of pink-footed geese wintering in Norfolk. The study began during construction and completed during operation. A total of 979 goose flocks were recorded passing through the study area during this time. During construction (pre-installation of turbine blades on foundations) almost half of all migrating flocks passed through the wind farm footprint. During operation, there was a growing tendency of geese to avoid the wind farms, with an estimated 94.5% of flocks exhibiting avoidance in the last two years of the study. Avoidance behaviour included flying around the OWF arrays or increasing height to fly over (vertical and horizontal avoidance).

513. Masden *et al.* (2010, 2012) and Speakman *et al.* (2009) calculated that the costs of one-off avoidances of wind farms by birds during migration were small, accounting for less than 2% of available fat reserves. Extrapolating from the studies cited above, if it is considered that individuals of a given migratory bird species crossing the North Sea twice per year encountered four OWFs on each migratory crossing, and chose to fly around all sites, this would potentially account for up to 8% of available fat reserves per migration on average. These assumptions would not hold for migratory species which do not avoid entering wind farms and may even be attracted to man-made structures such as turbines during certain environmental conditions. However, in these scenarios the species would be at risk of collision rather than barrier effect (e.g. see Welcker and Vilela 2019; Fijn *et al.*, 2015; Brabant *et al.*, 2015; WWT Consulting 2014; Wright *et al.,* 2012).

514. While there is little empirical evidence in relation to cumulative barrier effects of OWFs on migratory birds, species other than seabirds are considered to be of medium to low sensitivity to this effect. It is considered likely that any additional energy costs incurred from avoiding actions are of low to negligible magnitude. The effect significance is assessed as minor adverse to negligible.

13.8.3.5 Cumulative effect 5: Decommissioning Disturbance / Displacement, Offshore Cable Corridor

515. As there is overlap between the offshore cable corridors of North Falls and Five Estuaries OWFs (Figure 13.2 (Document Reference: 3.2.9)), cumulative impacts of decommissioning disturbance and displacement may occur in this area if the construction phase of North Falls overlaps with Five Estuaries (Table 13.42).

516. As for the construction phase, this effect has been screened in only for red-throated diver (Section 13.8.3.1 above).

517. As a worst-case, any effects generated during the decommissioning phase are expected to be similar to those generated during the construction phase. This is because decommissioning would generally involve a reverse of the construction phase through the removal of some structures and materials installed.

518. Cumulative disturbance during the decommissioning of the offshore cable corridors for North Falls and Five Estuaries is assessed as an impact of negligible magnitude. As the species is of high sensitivity to disturbance, the effect significance is minor adverse, and not significant in EIA terms.

13.9 Transboundary effects

519. A transboundary effect could occur when a project (in this case North Falls) within the UK or an European Economic Area (EEA) state could affect the environment within another EEA state. As many seabird species have large foraging and migratory ranges, it is possible that seabird populations outside of the UK and within an EEA state could be impacted by North Falls. This is most likely to occur during the operational phase of the Project as a result of collision with wind turbines or disturbance and displacement / barrier effects. Collisions and displacement of offshore ornithology receptors will also occur at OWFs located outside UK territorial waters. This means that the cumulative effects may be greater than those quantitatively assessed in the CEA presented in Section 13.8.3, when projects outside of the UK are taken into account.

520. It is considered that the spatial scale, and hence seabird reference populations sizes for a transboundary assessment, would be very large; considerably larger than those presented in this assessment. Within UK waters, the reference populations against which the assessment is based are defined by the relevant BDMPS (Furness, 2015) or colony-specific data during the breeding season; however, comparable information on the sizes of these wider populations is not currently available. In addition, the methods used to assess potential OWF impacts varies by country, and typically, the outputs of impact assessments are not directly comparable. This makes quantitative transboundary impact assessment challenging. A limited attempt at quantifying this has recently been made as part of the Strategic Environmental Assessment North Seas Energy (SEANSE) project (DHI, 2020a; 2020b). It provides a useful indicator of the level of potential impacts on offshore ornithology receptors beyond UK waters, and suggests that in the majority of cases, impacts on offshore ornithology receptors are largest in UK waters. However, there are a range of limitations that make the approach unsuitable for quantitative impact assessment purposes in its current form.

521. While there may be theoretical connectivity between non-UK seabird colonies and North Falls, in reality most colonies will be at the outer limits of foraging range during the breeding season, and are accounted for within the BDMPS estimates during the non-breeding season. The probability of significant populations of birds from non-UK breeding colonies occurring at North Falls is low. Because of the increased reference populations that would result from the expansion of the area of search, it is anticipated that the inclusion of non-UK OWFs is highly likely to reduce the cumulative effect assessed for each species presented in Section 13.8.3. Accordingly, no significant transboundary effects as a result of North Falls are predicted.

13.10 Interactions

522. Interactions between effects could occur if the same receptor or receptor group is subject to two or more impacts effects together, which could give rise to synergistic effects.

523. A screening for potential interactions between offshore ornithology effects (for example, the extent to which effects from collision risk could interact with displacement effects) is included in Table 13.56.

524. No potential interactions have been identified between the effects which have been assessed.

13.11 Inter-relationships

525. The construction, operation and decommissioning of North Falls would cause a range of effects on offshore ornithology receptors which may be inter-related with other receptor groups. With respect to the impacts assessed for offshore ornithology receptors at North Falls, this is considered to be the case for indirect impacts through effects on habitats and prey species only.

526. Inter-relationships are summarised in Table 13.57, which indicates where assessments carried out in other ES chapters have been used to inform the offshore ornithology assessment.

Table 13.56 Screening for interaction between impacts

<table>
<tr><th colspan="4">Screening matrix</th></tr>
<tr><td colspan="4">Construction</td></tr>
<tr><td>Impacts</td><td colspan="2">Disturbance and displacement from construction activities</td><td>Indirect effects vis prey species and / or their habitats</td></tr>
<tr><td>Disturbance and displacement from construction activities</td><td colspan="2"></td><td>No. Birds that are subject to displacement effects will not be impacted by prey availability / prey habitat effects, which are highly localized.</td></tr>
<tr><td>Indirect effects vis prey species and / or their habitats</td><td colspan="2">No. Birds that are subject to prey availability effects, which are highly localised, have not been displaced by construction activities.</td><td></td></tr>
<tr><td colspan="4">Operation</td></tr>
<tr><td>Impacts</td><td>Displacement and barrier effects from offshore infrastructure</td><td>Collision risk</td><td>Indirect effects vis prey species and / or their habitats</td></tr>
<tr><td>Displacement and barrier effects from offshore infrastructure</td><td></td><td>No. Birds that are displaced by the operational OWF would not be at risk of collision.</td><td>No. Birds that are displaced by the operational OWF would not be subject to prey availability effects as spatial magnitude of the latter is predicted to be small</td></tr>
<tr><td>Collision risk</td><td>No. Birds involved in collisions would not be susceptible to displacement.</td><td></td><td>No. Birds involved in collisions would not be susceptible to indirect effects.</td></tr>
<tr><td>Indirect effects vis prey species and / or their habitats</td><td>No. Birds that are subject to prey availability effects, which are highly localised, have not been displaced by the operational OWF.</td><td>No. Birds subject to indirect effects have not been involved in collisions.</td><td></td></tr>
<tr><td colspan="4">Decommissioning</td></tr>
<tr><td colspan="4">It is anticipated that the decommissioning impacts will be similar in nature to those of construction.</td></tr>
</table>

Table 13.57 Offshore ornithology inter-relationships

Effect	Related chapter (Volume 3.1)	Where addressed in this chapter	Rationale
Construction			
Indirect effects vis prey species and / or their habitats	ES Chapter 11 Fish and Shellfish Ecology ES Chapter 10 Benthic and Intertidal Ecology	Section 13.6.1.2	Effects on fish, shellfish and benthic ecology during construction could affect prey resource for offshore ornithology receptors
Operation			
Indirect effects vis prey species and / or their habitats	ES Chapter 11 Fish and Shellfish Ecology ES Chapter 10 Benthic and Intertidal Ecology	Section 13.6.2.4	Effects on fish, shellfish and benthic ecology during operation could affect prey resource for offshore ornithology receptors
Decommissioning			
Indirect effects vis prey species and / or their habitats	ES Chapter 11 Fish and Shellfish Ecology ES Chapter 10 Benthic and Intertidal Ecology	Section 13.6.2.6	Effects on fish, shellfish and benthic ecology during decommissioning could affect prey resource for offshore ornithology receptors

13.12 Summary

527. This chapter provides an assessment of the likely significant effects on offshore ornithology receptors that may arise from the construction, operation and maintenance and decommissioning of the offshore components of North Falls.

528. The assessment has been subject to extensive consultation with stakeholders (principally Natural England and RSPB) through the ornithology ETG, as well as the Scoping Opinion and Section 42 consultation. This has included detailed discussions regarding the overall approach to the impact assessment on offshore ornithology receptors, through to highly technical discussions on a range of key aspects of the assessment.

529. The chapter sets out the scope and methodology of the assessment, and the baseline state of the study area.

530. The assessment takes account of embedded mitigation including a protocol for reducing disturbance to red-throated divers (provided in Appendix B of the Outline PEMP (Document Reference: 7.6), and raising of the draught heights of WTGs from the minimum height of 22m so the lower blade tip is 26.6m above HAT, to reduce collision risk.

531. The outcome of the assessment of effects for the Project alone assessment is summarised in Table 13.58, and for the cumulative assessment in Table 13.59.

Table 13.58 Summary of project alone effect assessment for offshore ornithology receptors

Impact	Receptor	Sensitivity	Magnitude of impact	Significance of effect	Additional mitigation measures	Residual effect
Construction						
Direct disturbance and displacement during construction of the export cable	Red-throated diver	High	Negligible	Minor adverse	None	Minor adverse
Direct disturbance and displacement from construction activity on array area	Gannet	Medium	Negligible	Minor adverse	None	Minor adverse
	Guillemot	Medium	Negligible	Minor adverse	None	Minor adverse
	Razorbill	Medium	Negligible	Minor adverse	None	Minor adverse
	Red-throated diver	High	Negligible	Minor adverse	None	Minor adverse
Indirect effects due to effects on prey species and habitats	All species	Low to High	Negligible	Negligible to Minor Adverse	None	Negligible to Minor Adverse
Operation						
Disturbance and displacement	Gannet	Medium	Negligible	Minor adverse	None	Minor-adverse
	Guillemot	Medium	Negligible	Minor adverse	None	Minor-adverse
	Razorbill	Medium	Negligible	Minor adverse	None	Minor-adverse
	Red-throated diver	High	Negligible	Minor adverse	None	Minor-adverse
Collision risk	Gannet	Medium-low	Negligible	Minor adverse	None	Minor-adverse
	Great black-backed gull	Medium	Negligible	Minor adverse	None	Minor-adverse
	Kittiwake	Medium	Negligible	Minor adverse	None	Minor-adverse
	Lesser black-backed gull	Medium	Low	Minor adverse	None	Minor-adverse
Collision risk and displacement	Gannet	Medium	Negligible	Minor adverse	None	Minor-adverse
Indirect effects due to effects on prey species and habitats	All species	Low to High	Negligible	Negligible to Minor Adverse	None	Negligible to Minor Adverse

Impact	Receptor	Sensitivity	Magnitude of impact	Significance of effect	Additional mitigation measures	Residual effect
Decommissioning						
Direct disturbance and displacement from decommissioning activities	All species	Low to High	Negligible	Negligible to Minor Adverse	None	Negligible to Minor Adverse
Indirect effects due to effects on prey species and habitats	All species	Low to High	Negligible	Negligible to Minor Adverse	None	Negligible to Minor Adverse

Table 13.59 Summary of cumulative effect assessment for offshore ornithology receptors

Potential impact	Receptor	Sensitivity	Magnitude of impact	Significance of effect	Mitigation measures proposed	Residual effect
Construction						
Direct disturbance and displacement during construction of the export cable	Red-throated diver	High	Negligible	Minor adverse	None	Minor adverse
Operation						
Displacement	Gannet	Medium	Negligible	Minor adverse	None	Minor adverse
	Guillemot	Medium	Negligible	Minor adverse	None	Minor adverse
	Razorbill	Medium	Negligible	Minor adverse	None	Minor adverse
	Red-throated diver	High	Negligible	Minor adverse	None	Minor adverse
Collision risk	Gannet	Medium	Low	Minor adverse	None	Minor adverse
	Great black-backed gull	Medium	Medium	Moderate adverse	None	Moderate adverse
	Kittiwake	Medium	Medium	Moderate adverse	None	Moderate adverse
	Lesser black-backed gull	Medium	Medium	Moderate adverse	None	Moderate adverse
Collision and displacement	Gannet	Medium	Low	Minor adverse	None	Minor adverse

Potential impact	Receptor	Sensitivity	Magnitude of impact	Significance of effect	Mitigation measures proposed	Residual effect
Barrier effect to migratory bird species	Migratory bird species	Low to Medium	Low to negligible	Minor adverse to negligible	None	Minor adverse to negligible

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