# Morecambe Offshore Windfarm: Generation Assets

## Development Consent Order Documents

## Volume 4

## Report to Inform Appropriate Assessment

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## Glossary of Acronyms

| | |
|---|---|
| AC | Alternating Current |
| ADD | Acoustic Deterrent Device |
| AEoI | Adverse Effect on Integrity |
| AfL | Agreement for Lease |
| AIS | Automatic Identification System |
| ALs | Action Levels |
| AON | Apparently Occupied Nests |
| AOS | Apparently Occupied Sites |
| AOT | Apparently Occupied Territories |
| ASCOBANS | Agreement on the Conservation of Small Cetaceans of the Baltic, North East Atlantic, Irish and North Seas |
| AyM | Awel y Môr |
| BDMPS | Biologically Defined Minimum Population Scales |
| BEIS | Department for Business, Energy and Industrial Strategy[1] |
| BTO | British Trust for Ornithology |
| BWM Convention | The International Convention for the Control and Management of Ships' Ballast Water and Sediments |
| CBRA | Cable Burial Risk Assessment |
| CCW | Countryside Council for Wales |
| CD | Chart Datum |
| CEA | Cumulative Effects Assessment |
| Cefas | Centre for Environment, Fisheries and Aquaculture Science |
| CI | Confidence Interval |
| CIS | Celtic and Irish Sea |
| CL | Confidence Limit |
| CRM | Collision Risk Model |
| cSAC | Candidate Special Area of Conservation |
| CSIP | Cetacean Strandings Investigation Programme |
| CV | Coefficient of Variation |
| DAERA | Department of Agriculture, Environment and Rural Affairs |
| DCO | Development Consent Order |
| Defra | Department for Environment, Food & Rural Affairs |

[1] As of February 2023, BEIS is known as the Department for Energy Security and Net Zero (DESNZ)

| | |
|---|---|
| DESNZ | Department for Energy Security and Net Zero |
| DML | Deemed Marine Licence |
| DoE | Department of the Environment |
| DP | Dynamic Positioning |
| EC | European Council |
| ECC | European Economic Commission |
| EDR | Effective Deterrence Radius |
| EEA | European Economic Area |
| EEC | European Economic Community |
| EIA | Environmental Impact Assessment |
| EMF | Electromagnetic Field |
| EMP | Environmental Management Plan |
| EPP | Evidence Plan Process |
| EPS | European Protected Species |
| ERL | Effects Range – Low |
| ES | Environmental Statement |
| ETG | Expert Topic Group |
| EU | European Union |
| EUNIS | European Nature Information System |
| FCS | Favourable Conservation Status |
| GBS | Gravity Based Structure |
| HAT | Highest Astronomical Tide |
| HF | High Frequency |
| HNDR | Holistic Network Design Review |
| HPAI | Highly Pathogenic Avian Influenza |
| HRA | Habitats Regulations Assessment |
| HRGN | Habitats Regulations Guidance Note |
| IAMMWG | Inter-Agency Marine Mammal Working Group |
| IEC | International Electrotechnical Commission |
| IFCA | Inshore Fisheries and Conservation Authority |
| INNS | Invasive Non-Native Species |
| iPCoD | interim Population Consequences of Disturbance |
| IROPI | Imperative Reasons of Overriding Public Interest |
| IS | Irish Sea |
| ISAA | Information to Support an Appropriate Assessment |
| JNCC | Joint Nature Conservation Committee |

| | |
|---|---|
| LAT | Lowest Astronomical Tide |
| LCL | Lower Confidence Limit |
| LSE | Likely Significant Effects |
| MARPOL | International Convention for the Prevention of Pollution from Ships |
| MCA | Maritime and Coastguard Agency |
| MCZA | Marine Conservation Zone Assessment |
| MGN | Marine Guidance Note |
| MHWS | Mean High Water Springs |
| MMMP | Marine Mammal Mitigation Protocol |
| MMO | Marine Management Organisation |
| MPCP | Marine Pollution Contingency Plan |
| MW | Megawatts |
| NGET | National Grid energy transmission |
| NIEA | Northern Ireland Environment Agency |
| NPS | National Policy Statement |
| NPWS | National Parks and Wildlife Service |
| NRW | Natural Resources Wales |
| NSER | No Significant Effects Report |
| NSIP | Nationally Significant Infrastructure Project |
| OSP | Offshore substation platform |
| OSPAR | Oslo-Paris Convention (Convention for the Protection of the Marine Environment of the North-East Atlantic) |
| OTNR | Offshore Transmission Network Review |
| OWF | Offshore Windfarm |
| PBDE | Polybrominated Diphenyl Ether |
| PDE | Project Design Envelope |
| PEIR | Preliminary Environmental Information Report |
| PEMP | Project Environmental Management Plan |
| PINS | Planning Inspectorate |
| pSAC | Potential Special Area of Conservation |
| pSPA | Potential Special Protection Areas |
| PTS | Permanent Threshold Shift |
| RHR | Rotor Height Range |
| RIAA | Report to Inform Appropriate Assessment |
| RPM | Rotations Per Minute |
| RSPB | Royal Society for the Protection of Birds |

| | |
|---|---|
| SAC | Special Area(s) of Conservation |
| SACO | Supplementary Advice on Conservation Objectives |
| SCANS | Small Cetaceans in the European Atlantic and North Sea |
| SCI | Site of Community Importance |
| SCOS | Special Committee on Seals |
| sCRM | Stochastic Collision Risk Model |
| SELcum | Sound Exposure Level Cumulative Exposure |
| SMP | Seabird Monitoring Programme |
| SNCBs | Statutory Nature Conservation Bodies |
| SoCG | Statement of Common Ground |
| SoS | Secretary of State |
| SPA | Special Protection Areas |
| SPA | Special Protection Area(s) |
| SPL | Sound Pressure Level |
| SSC | Suspended sediment concentration |
| SSSI | Site of Special Scientific Interest |
| TH | Trinity House |
| TSHD | Trailing suction hopper dredger |
| TTS | Temporary Threshold Shift |
| UCL | Upper Confidence Limit |
| UK | United Kingdom |
| UXO | Unexploded Ordnance |
| VHF | Very-High Frequency |
| WFD | Water Framework Directive |
| WTG | Wind turbine generator |
| ZoI | Zone of Influence |

## Glossary of Unit Terms

| | |
|---|---|
| km | kilometre |
| kJ | kilojoule |
| kV | kilovolt |
| m | metre |
| MW | Megawatt |

# Glossary of Terminology

| | |
|---|---|
| Agreement for Lease (AfL) | Agreements under which seabed rights are awarded following the completion of The Crown Estate tender process. |
| Applicant | Morecambe Offshore Windfarm Ltd |
| Application | This refers to the Applicant's application for a Development Consent Order (DCO). An application consists of a series of documents and plans which are published on the Planning Inspectorate's (PINS) website. |
| Biologically defined minimum population scale (BDMPS) | The estimated population size of a species within a defined biogeographic area during a biologically relevant season, as defined by Furness (2015). For many seabird species present in UK waters there are two defined biogeographic areas; UK Western waters and UK North Sea and Channel. However, some species have different defined BDMPS areas, dependent on the distribution and movements of the species population through the year. Furness (2015) defines the BDMPS for non-breeding seasons; the breeding season BDMPS is defined as the breeding population within foraging range from the project, plus non-breeders and immatures. |
| Biologically relevant seasons | Defined time periods during the year where a species population will predominantly be present in a certain biogeographic area and/or exhibit particular behaviours in relation to the species' life-cycle. Biologically relevant seasons, as defined by Furness (2015), include breeding, non-breeding, spring migration, autumn migration and winter. In many cases seasons will overlap, and not all seasons are relevant to all species. |
| European sites | Designated nature conservation sites which include the National Site Network (NSN) (designated within the UK) and Natura 2000 sites (designated in any European Union (EU) country). This includes candidate Special Areas of Conservation (cSAC), Sites of Community Importance (SCI), Special Areas of Conservation (SAC) and Special Protection Areas (SPAs). |
| Evidence Plan Process (EPP) | A voluntary consultation process with specialist stakeholders to agree the approach, and information to support, the Environmental Impact Assessment (EIA) and Habitats Regulations Assessment (HRA) for certain topics. The EPP provides a mechanism to agree the information required to be submitted to the Planning Inspectorate as part of the Development Consent Order application. This function of the EPP helps Applicants to provide sufficient information in their application, so that the Examining Authority can recommend to the Secretary of State (SoS) whether or not to accept the application for examination and whether an appropriate assessment is required. |
| Expert Topic Group (ETG) | A forum for targeted engagement with regulators and interested stakeholders through the EPP. |
| Generation Assets (the Project) | Generation assets associated with the Morecambe Offshore Windfarm. This is infrastructure in connection with electricity production, namely the fixed foundation wind turbine generators (WTGs), inter-array cables, offshore substation platform(s) (OSP(s)) and possible platform link cables to connect OSP(s). |
| In-row | The distance separating WTGs in the main rows. |

| | |
|---|---|
| Inter-array cables | Cables which link the WTGs to each other and the OSP(s). |
| Inter-row | The distance between the main rows. |
| Landfall | Where the offshore export cables would come ashore. |
| 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. |
| Morgan and Morecambe Offshore Wind Farms: Transmission Assets | The transmission assets for the Morgan Offshore Wind Project and the Morecambe Offshore Windfarm. This includes the OSP(s)[2], interconnector cables, Morgan offshore booster station, offshore export cables, landfall site, onshore export cables, onshore substations, 400kV cables and associated grid connection infrastructure such as circuit breaker infrastructure.<br>Also referred to in this document as the Transmission Assets, for ease of reading. |
| National Site Network | The national site network encompasses existing SACs and SPAs and new SACs and SPAs designated under the EIA Regulations. |
| Offshore export cables | The cables which would bring electricity from the OSP(s) to the landfall. |
| Offshore substation platform(s) | A fixed structure located within the windfarm site, containing electrical equipment to aggregate the power from the WTGs and convert it into a more suitable form for export to shore. |
| Onshore export cables | The cables which would bring electricity from landfall to the onshore project substation and from the onshore project substation to a National Grid substation. |
| Onshore project substation | Part of an electrical transmission and distribution system. Substations transform voltage from high to low, or the reverse by means of electrical transformers. |
| Permanent threshold shift | Physical or permanent auditory injury causing a permanent shift in the auditory threshold. |
| Platform link cable | An electrical cable which links one or more OSP(s). |

[2] At the time of writing the Environmental Statement (ES), a decision had been taken that the offshore substation platforms (OSP(s)) would remain solely within the Generation Assets application and would not be included within the DCO Application for the Transmission Assets. This decision post-dated the Preliminary Environmental Information Report (PEIR) that was prepared for the Transmission Assets. The OSP(s) are still included in the description of the Transmission Assets for the purposes of this document as the in-combination effects assessment carried out in respect of the Generation/Transmission Assets is based on the information available from the Transmission Assets PEIR and associated Habitat Regulations documentation.

| | |
|---|---|
| Safety zones | An area around a structure or vessel which should be avoided, as set out in Section 95 of the Energy Act 2004 and the Electricity (Offshore Generating Stations) (Safety Zones) (Application Procedures and Control of Access) Regulations 2007. |
| Scour protection | Protective materials to avoid sediment being eroded away from the base of the foundations due to the flow of water. |
| Steering Group | The Applicant and key stakeholders responsible for overseeing EPP. |
| Study area | This is an area which is defined for each topic which includes the windfarm site as well as potential spatial and temporal considerations of the impacts on relevant receptors. The study area for each topic is intended to cover the area within which an effect can be reasonably expected. |
| Technical stakeholders | Technical consultees are considered to be organisations with detailed knowledge or experience of the area within which the Project is located and/or receptors which are considered in the EIA and HRA. Examples of technical stakeholders include Marine Management Organisation (MMO), local authorities, Natural England and Royal Society for the Protection of Birds (RSPB). |
| Temporary threshold shift | Auditory injury causing a short-term shift in the auditory threshold. |
| Windfarm site | The area within which the WTGs, inter-array cables, OSP(s) and platform link cables will be present |
| Wind turbine generator (WTG) | A fixed structure located within the windfarm site that converts the kinetic energy of wind into electrical energy. |
| Zone of Influence (ZoI) | The maximum anticipated spatial extent of a given potential impact. |

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# 1 Introduction

## 1.1 The Project

1. Morecambe Offshore Windfarm: Generation Assets (hereafter referred to as the “Project”) is a proposed offshore windfarm located in the Eastern Irish Sea, with an expected nominal capacity of 480 megawatts (MW). The Project is located approximately 30km off the Lancashire coast, as illustrated in **Figure 1.1**. It is being developed by Morecambe Offshore Windfarm Ltd (the Applicant).

2. As the Project windfarm is an offshore generating station of over 100MW, it is defined under the Planning Act 2008 as a Nationally Significant Infrastructure Project (NSIP) and as such it requires a Development Consent Order (DCO).

3. A Government-initiated review of offshore windfarm transmission connections has concluded that the Morecambe Offshore Windfarm would share a grid connection location at Penwortham in Lancashire with the Morgan Offshore Wind Project, also located in the Eastern Irish Sea, as shown in **Figure 1.2.** Given this, the Applicant intends to deliver a coordinated grid connection with the Morgan Offshore Wind Project and submit a separate DCO application for the Morgan and Morecambe Offshore Wind Farms: Transmission Assets (referred to as the “Transmission Assets”). For the purposes of this document the “Project” refers only to the generation assets of the Morecambe Offshore Windfarm.

4. As illustrated in **Plate 1.1**, the Project includes the generation assets to be located within the windfarm site (wind turbine generators (WTGs), inter-array cables, offshore substation platform(s) (OSP(s)) and possible platform link cables to connect OSP(s)). The Environmental Impact Assessment (EIA) of the Transmission Assets, including offshore export cables to landfall and onshore infrastructure, is part of a separate DCO application, as outlined in **Chapter 1 Introduction** of the Environmental Statement (ES) (Document Reference 5.1.1).

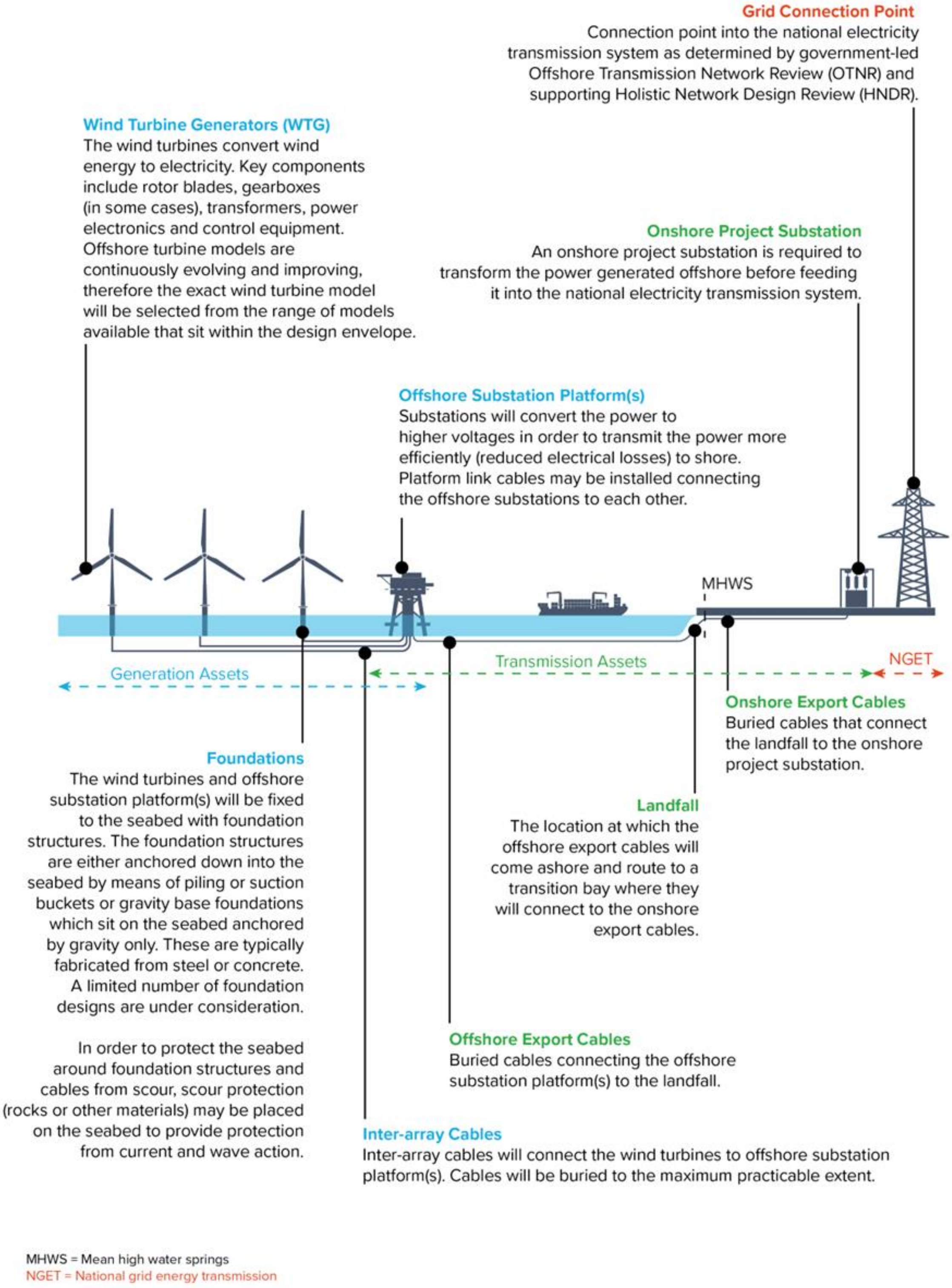
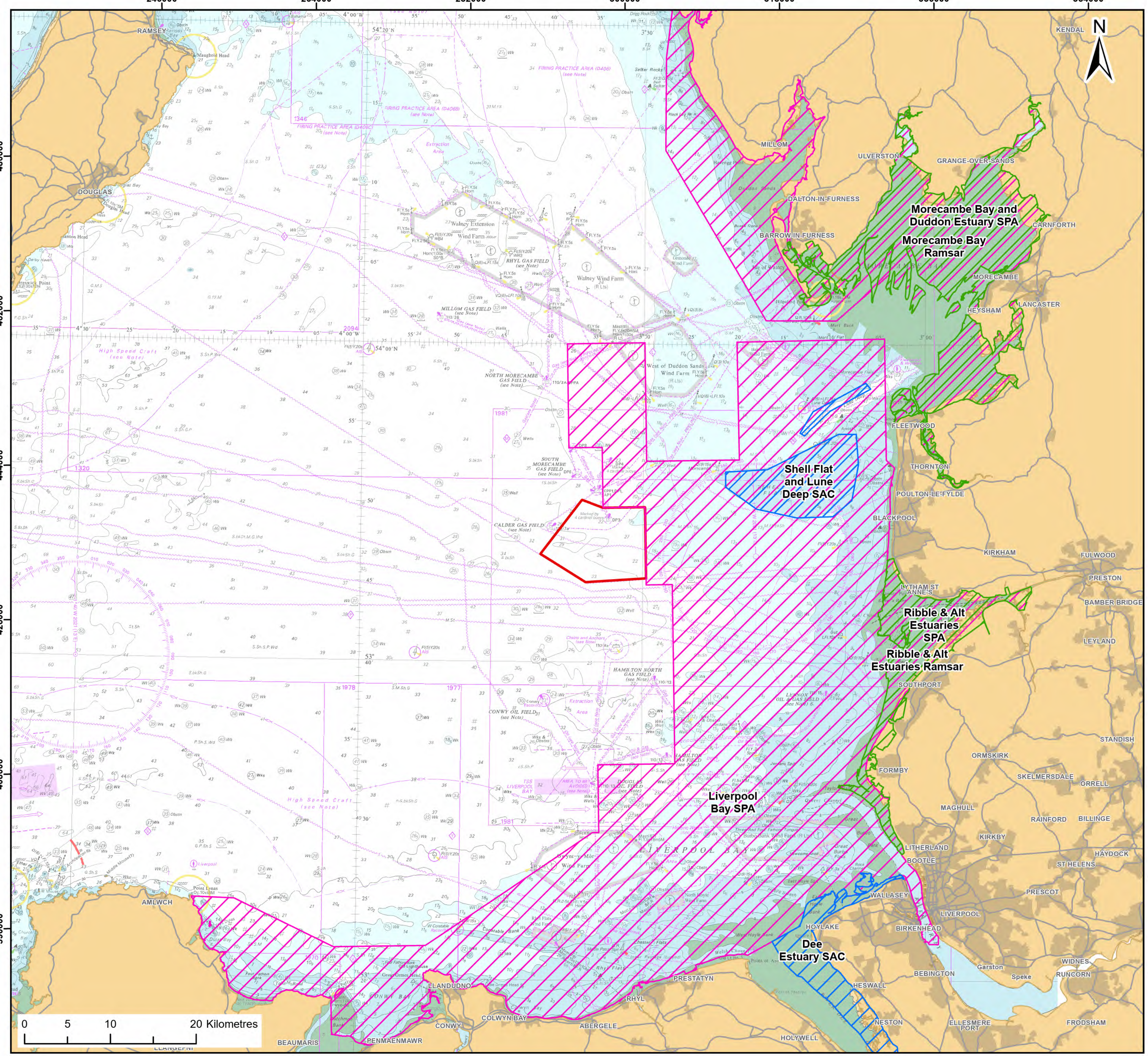



*Plate 1.1 Components of Morecambe Offshore Windfarm (note the components in blue are Generation assets and those in green are anticipated Transmission assets)*





Legend:

- Morecambe Offshore Windfarm Site
- Special Protection Areas (SPA)
- Special Area of Conservation (SAC)
- Ramsar



Report:
Morecambe Offshore Windfarm: Generation Assets
Habitat Regulations Report to Inform Appropriate Assessment

Title:
Morecambe Offshore Windfarm Site location and designated sites

| Figure: 1.1 | | Drawing No: PC1165-RHD-ES-OF-DR-Z-0062 | | | |
|---|---|---|---|---|---|
| Revision: | Date: | Drawn: | Checked: | Size: | Scale: |
| P01 | 19/04/2024 | JH | SB | A3 | 1:450,000 |

Co-ordinate system: WGS 1984 UTM Zone 30N

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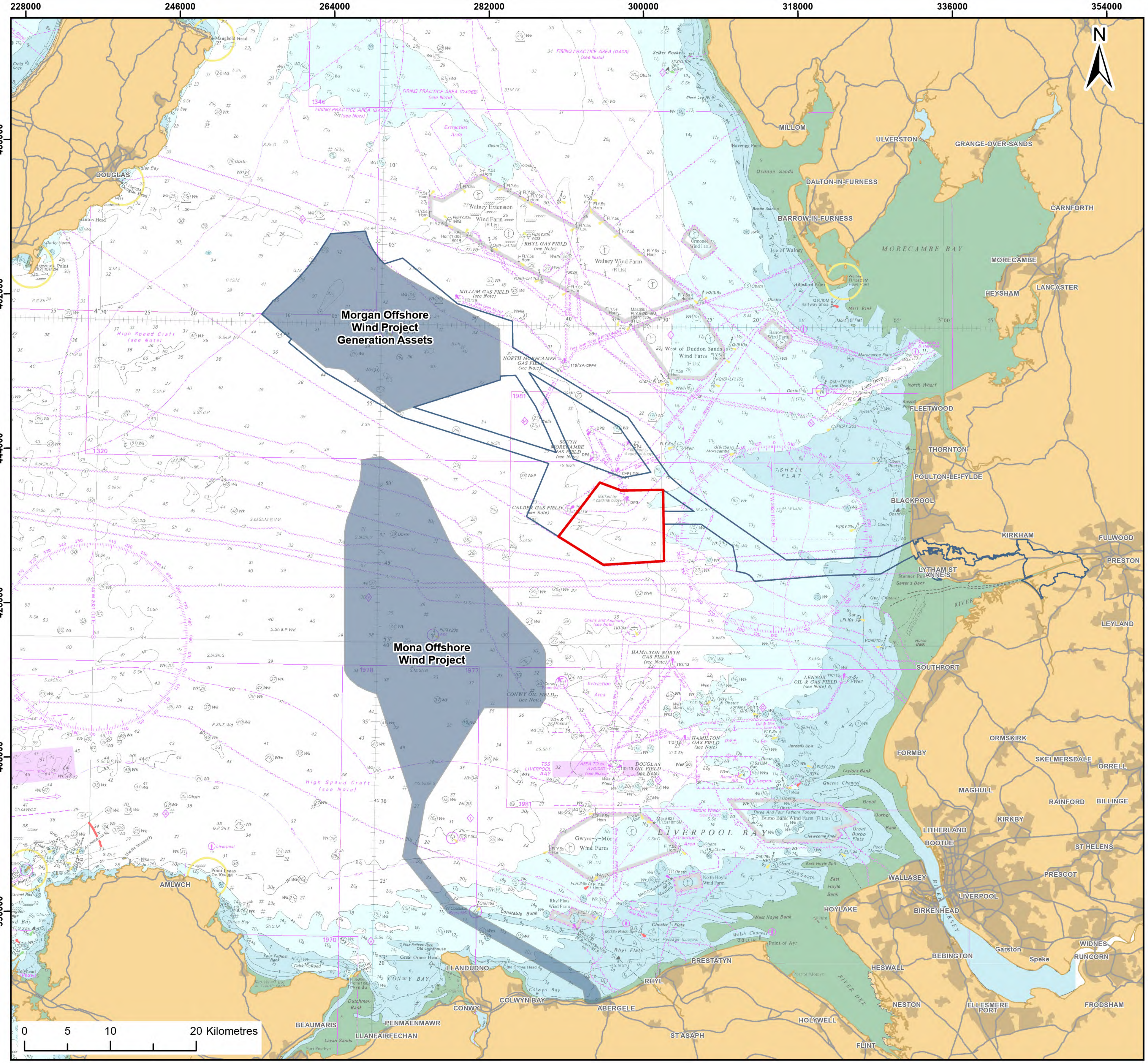


Legend:

Morecambe Offshore Windfarm Site

Morgan and Morecambe Offshore Wind Farms: Transmission Assets (In Planning)

Windfarm status

In Planning



Report: Morecambe Offshore Windfarm: Generation Assets Habitat Regulations Report to Inform Appropriate Assessment

Title: Morecambe Offshore Windfarm location with other Round 4 Projects

Figure: 1.2 | Drawing No: PC1165-RHD-ES-OF-DR-Z-0063

| Revision: | Date: | Drawn: | Checked: | Size: | Scale: |
|---|---|---|---|---|---|
| P02 | 15/01/2024 | JH | AS | A3 | 1:450,000 |
| P03 | 09/04/2024 | JH | SB | A3 | 1:450,000 |

Co-ordinate system: WGS 1984 UTM Zone 30N

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## 1.2 Purpose of this document

5. This document has been produced to provide the competent authority with information on the potential for Adverse Effect on the Integrity (AEoI) of European designated sites as a result of the Project. The HRA process derives from the requirements of specific European Directives and the United Kingdom (UK) Regulations that implement their requirements in national law, as outlined in **Section 2** of this report.

6. This report is intended to inform the process of undertaking an Appropriate Assessment and is submitted alongside the ES as part of the DCO Application, having been updated to reflect comments received from consultation on the draft RIAA. This report has also been updated with further survey data (year two of site-specific aerial surveys), updated underwater noise modelling and associated updated assessments and relevant changes to the Project Design Envelope (PDE).

7. The HRA process has to be applied as a matter of law or policy to the following ‘European sites’ (referred to as ‘Natura 2000’ sites in the European Union (EU) or ‘National Site Network’ sites in the UK):

- Special Areas of Conservation (SACs)
- Special Protection Areas (SPAs)
- Sites of Community Importance (SCI)
- Potential SPAs (pSPAs)
- Possible SACs (pSACs)
- Candidate SACs (cSACs)
- Listed and proposed Ramsar sites (internationally important wetlands designated under the Ramsar Convention 1971)

8. This RIAA therefore covers potential effects upon the following receptors:

- Benthic ecology – Habitats Directive Annex I Habitats (SACs, cSACs and SCIs, as appropriate)
- Fish ecology – Habitats Directive Annex II Species (SACs, cSACs and SCIs, as appropriate)
- Offshore ornithology – features of National Site Network sites (SPAs, pSPAs and Ramsar sites), including rare and vulnerable birds (as listed on Annex I of the Birds Directive) and regularly occurring migratory species.
- Marine mammals – Habitats Directive Annex II Species (SACs, cSACs and SCIs, as appropriate)

## 1.3 Structure of this document

9. The structure of this report is as follows:

- Introduction: provides an introduction to the report and the structure of the assessment (**Section 1**)
- Relevant legislation, policy and guidance: (**Section 2**)
- Description of the Project (**Section 3**)
- Overview of the HRA process: provides an overview of the HRA process and the approach taken (**Section 4**)
- Screening conclusions (**Section 5**): summary of the conclusions reached in the HRA Screening Report (Morecambe Offshore Windfarm Ltd, 2023a). The Screening Report is provided as part of the DCO Application (Document Reference 4.10).
- Assessment of each relevant receptor (**Sections 6 – 9**)
- Summary of the RIAA (**Section 10**)
- References (**Section 11**)

# 2 Legislation, policy and guidance

10. The Conservation of Habitats and Species Regulations 2017 (2017 No. 1012) (as amended) and The Conservation of Offshore Marine Habitats and Species Regulations 2017 (2017 No. 1013) (as amended) are the principal pieces of secondary legislation which, prior to the UK's departure from the EU, transposed the terrestrial and offshore marine aspects of the EU Habitats Directive (Council Directive 92/43/European Economic Community (EEC)) and certain elements of the EU Wild Birds Directive (Directive 2009/147/ European Commission (EC)) into the domestic law. Together, these regulations are collectively known as the "Habitats Regulations".

11. The Conservation of Habitats and Species (Amendment) (EU Exit) Regulations 2019 (2019 No. 579) set out the changes that apply since the UK left the European Union. These confirmed that:

- All protected sites and species retain the same level of protection.
- Among other things, the requirement for HRA to be undertaken continues to apply. Unless the UK government implements further legislative changes, the obligations, process and terminology of the Habitats Regulations will, for the purposes of this document, remain as set out in existing legislation and regulations. The role of the European Commission is now exercised by UK Ministers.

## 2.1 European sites (post-EU exit)

12. The Europe-wide network of nature conservation areas that are the subject of the HRA process was established under the Habitats Directive. The Habitats Directive established a network of internationally important sites, designated for their ecological status. For EU member states (and formerly for the UK), SACs are designated under the Habitats Directive and promote the protection of flora, fauna and habitats. SPAs are designated under the Birds Directive to protect rare, vulnerable and migratory birds. European sites located within an EU Member State combine to create a Europe-wide network of designated sites (the Natura 2000 network) and may be referred to as Natura 2000 Sites.

13. Following the UK's exit from the EU, European sites located within the UK are no longer part of the Natura 2000 network (nor Natura Sites) but instead combine to form the UK's "National Site Network". The National Site Network comprises European sites in the UK that already existed (i.e., were established under the Nature Directives) on 31st December 2020 (or proposed to the EC before that date) and any new sites designated under the Habitats Regulations under an amended designation process. Hereafter, sites within the UK and the EU have been both referred to as 'European sites'.

14. Ramsar sites designated under the Convention on Wetlands of International Importance especially as Waterfowl Habitat, as amended in 1982 and 1987 (the 'Ramsar Convention') were not included within the National Site Network but have still been included within the HRA as they remain protected in the same way as SACs and SPAs.

## 2.2 Guidance

15. A description of the guidance documents relevant to the HRA has been provided in the HRA Screening Report (Morecambe Offshore Windfarm Ltd, 2023a; Document Reference 4.10).

# 3 Description of the Project

16. This section provides an overview of the main components of the Project, which, for the purposes of this RIAA, covers the Generation Assets (WTGs, inter-array cables, OSP(s) and possible platform link cables to connect OSP(s)). It also summarises the main activities that would occur during construction, operation and maintenance, and decommissioning.

17. A separate HRA assessment is being undertaken for the Transmission Assets associated with the Project (which is subject to a separate consent application process along with the Transmission Assets for the Morgan Offshore Wind Project (Morgan Offshore Wind Limited, 2023). As such, only a summary of this associated infrastructure has been described in this section.

## 3.1 Design envelope approach

18. The PDE has been developed in parallel with the EIA with the Project design outlined in **Chapter 5 Project Description** (Document Reference 5.1.5) of the ES.

19. The PDE provides maximum and minimum parameters, where appropriate, to ensure the worst-case scenario can be quantified and assessed, whilst maintaining design flexibility. Therefore, the description of the Project provided here is indicative at this stage and intended to provide context for the wider document and the basis of the assessment.

20. The PDE reported in this RIAA is based on a design envelope approach in accordance with the National Policy Statement (NPS) for Renewable Energy Infrastructure (NPS EN-3 (Department for Energy Security and Net Zero (DESNZ), 2023a)); paragraph 2.8.74 which recognises that: "*Owing to the complex nature of offshore wind farm development, many of the details of a proposed scheme may be unknown to the applicant at the time of the application to the Secretary of State. Such aspects may include:*

- *the precise location and configuration of turbines and associated development;*
- *the foundation type and size;*
- *the installation technique or hammer energy;*
- *the exact turbine blade tip height and rotor swept area;*
- *the cable type and precise cable or offshore transmission route;*
- *the exact locations of offshore and/or onshore substations*".

21. NPS EN-3 (paragraph 2.6.1 – 2.6.2) recognises: "*Where details are still to be finalised, applicants should explain in the application which elements of the proposal have yet to be finalised, and the reason why this is the case. Where*

*flexibility is sought in the consent as a result, applicants should, to the best of their knowledge, assess the likely worstcase environmental, social and economic effects of the proposed development to ensure that the impacts of the project as it may be constructed have been properly assessed.*[3]

22. This approach has been widely successful in the consenting of offshore wind farms and is consistent with the Planning Inspectorate (PINS) Advice Note Nine: Rochdale Envelope (PINS, 2018) which states that: "*The Rochdale Envelope assessment approach is an acknowledged way of assessing a Proposed Development comprising Environmental Impact Assessment (EIA) development where uncertainty exists and necessary flexibility is sought*."

23. The PDE therefore provides maximum and minimum parameters, where appropriate, to ensure that the worst-case scenario can be quantified and assessed in the EIA and HRA while maintaining design flexibility.

24. The parameters described in this section represent the PDE for the Project and have been derived from the range of designs, technologies and methodologies under consideration. The assessments set out in **Sections 6 - 9** were based on the realistic worst-case scenario for receptors (as set out in these sections), noting that the worst-case scenario will vary depending on the receptor and impact being considered.

## 3.2 Project infrastructure overview

### 3.2.1 Windfarm site

25. The windfarm site would contain all generation infrastructure. The key characteristics of the windfarm site are summarised in **Table 3.1**.

*Table 3.1 Morecambe Offshore Windfarm site overview*

| Area | Parameters | Values |
|---|---|---|
| Windfarm site | Area | $87km^2$ |
| | Closest distance to shore | 30km (approximate) |
| | Water depth | 18 - 40m |

[3] Case law, beginning with R v Rochdale MBC Ex p. Tew [2000] Env.L.R.1 establishes that while it is not necessary or possible in every case to specify the precise details of development, the information contained in the ES should be sufficient to fully assess the project's impact on the environment and establish clearly defined worst-case parameters for the assessment. This is sometimes known as 'the Rochdale Envelope'. See https://infrastructure.planninginspectorate.gov.uk/legislation-andadvice/advice-notes/advice-note-nine-rochdale-envelope/

26. The Agreement for Lease (AfL) area awarded by The Crown Estate spans 125km$^2$. Following consultation on the PEIR, the proposed windfarm site was reduced to approximately 87km$^2$, as further described in **Chapter 4 Site Selection and Assessment of Alternatives** (Document Reference 5.1.4).

### 3.2.2 Wind turbine generators

27. The WTG PDE is outlined in **Table 3.2,** illustrated in **Plate 3.1** and subsequently described, noting this considers both up to 30 'larger' WTG and up to 35 'smaller' WTGs.

28. The information presented in **Table 3.2** includes a range of WTGs with varying parameters and capacity, to accommodate the ongoing rapid development in WTG technology. Accounting for this range, there could be up to 30 'larger' or 35 'smaller' WTGs installed within the windfarm site to generate the nominal export capacity of 480MW.

*Table 3.2 WTG design envelope*

| Parameter | Smaller WTGs | Larger WTGs |
|---|---|---|
| Maximum number of WTGs | 35 | 30 |
| Maximum rotor diameter (m) | 260 | 280 |
| Blade tip height (m) above highest astronomical tide (HAT) | 290 | 310 |
| Maximum hub height (m above HAT) | 160 | 170 |
| Minimum rotor clearance above sea level (m above HAT) | 25[4] | |
| Indicative rotor speed range (rotations per minute (RPM)) | 8.42 | 7.09 |
| Maximum rotor swept area for total windfarm site (km$^2$) | 1.858 | |
| Minimum separation between WTGs (m) in-row | 1,060 | 1,260 |
| Minimum separation between WTGs (m) inter-row | 1,410 | 1,680 |

[4] Equivalent to 34.56m above LAT; 26.07m above MHWS; 29.82m above mean sea level (MSL)

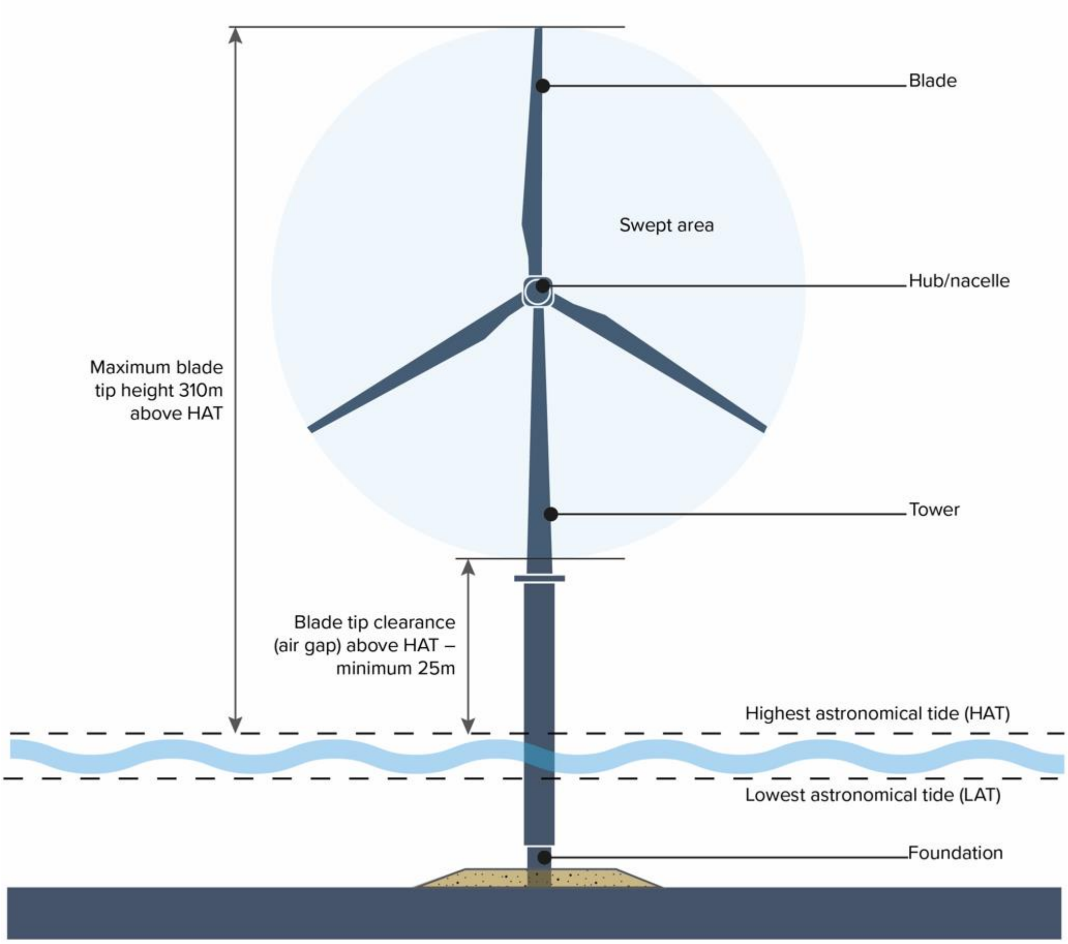



*Plate 3.1 Schematic of a WTG*

29. The layout of WTGs would be finalised post-consent in consideration of design rules (as detailed in Marine Guidance Note (MGN) 654) and in consultation with relevant authorities e.g., Marine Management Organisation (MMO), Maritime and Coastguard Agency (MCA) and Trinity House (TH). The required lighting and navigational markings would also be agreed post-consent.

### 3.2.3 Offshore substation platform(s)

30. The Project would require up to a maximum of two OSPs, depending on the electrical system voltage and final layout. The OSP(s) provide a centralised connection point for the inter-array cable circuits and contain primary electrical equipment and ancillary components that are required to transform the voltage of the electricity generated at the WTGs to a higher voltage suitable for transporting power to the onshore electrical transmission network.

31. The OSP(s) would be situated within the windfarm site and would comprise the following components:

- Transformers
- Batteries

- Generators
- Switchgear
- Fire systems
- Modular facilities for operational and maintenance activities

32. The design of the OSP(s) would include a platform ‘topside’, supported above sea level on a foundation structure.

33. The typical deck plan of the OSP(s) would be a maximum of 50m by 50m, with the topsides comprising several layers/decks stacked on top of another, as required. **Plate 3.2** shows a schematic of a typical OSP.

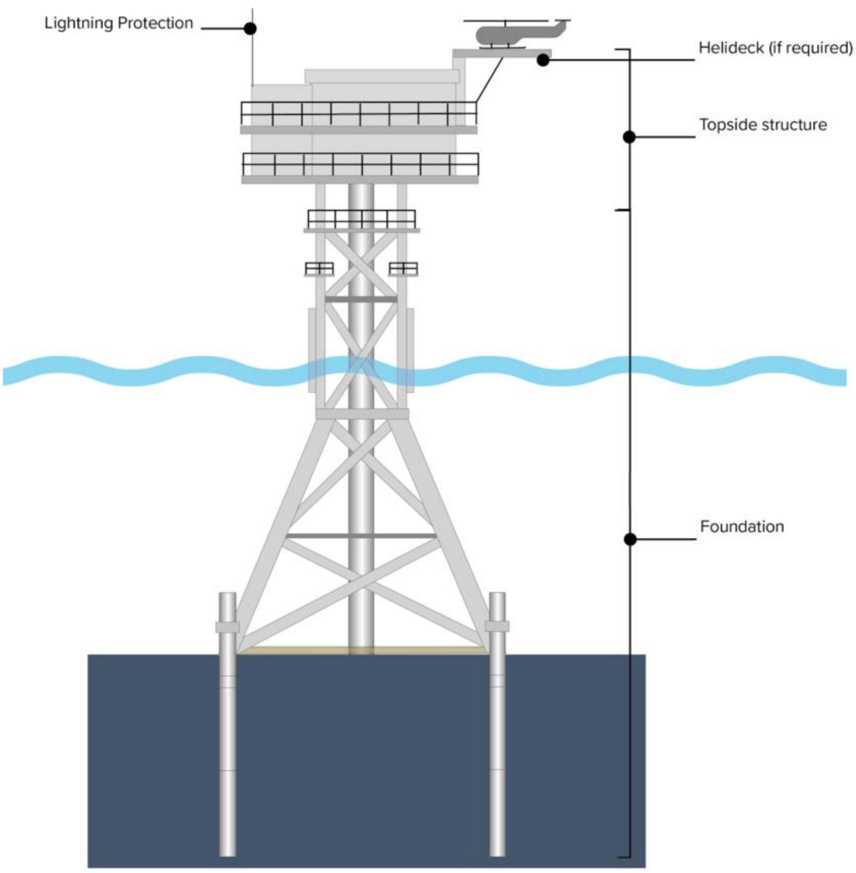



*Plate 3.2 Schematic of an OSP. Note: The schematic shows a 'jacket on pin piles' foundation, however, the actual foundation type may differ e.g. monopile.*

34. The topside design envelope for the OSP(s) is given in **Table 3.3.**

*Table 3.3 OSP(s) topside design envelope*

| Parameter | Value |
|---|---|
| Maximum number of OSP(s) | 2 |
| Maximum topside width (m) | 50 |

| Parameter | Value |
|---|---|
| Maximum topside length (m) | 50 |
| Highest point of topside above HAT (m) (excluding helideck and lightning protection) | 50 |
| Highest point of topside above HAT (m) (including helideck and lightning protection) | 70 |

### 3.2.4 Foundations

35. This section provides an overview of the foundations and substructures that have been considered and assessed for the Project WTGs and OSP(s). The decision on the types of foundation and substructure to support the WTGs and OSP(s) would be made post-consent.

36. The WTG/OSP(s) foundation types and parameters are listed in **Table 3.4** and illustrated in **Plate 3.3**. Options have been described in detail in **Chapter 5 Project Description** of the ES, and briefly described below:

- Gravity based structures (GBS). GBS usually comprise a base supporting a conical section, which tapers to an upper cylindrical section (shaft)
- Multi-legged pin-piled jacket (three-legged or four-legged jackets). A steel lattice construction (tubular steel and welded joints) secured to the seabed by hollow steel pin piles
- Monopile foundations are welded hollow tubular steel structures
- Multi-legged suction bucket jacket (three-legged jackets). A jacket that would be installed on three suction bucket ‘legs’

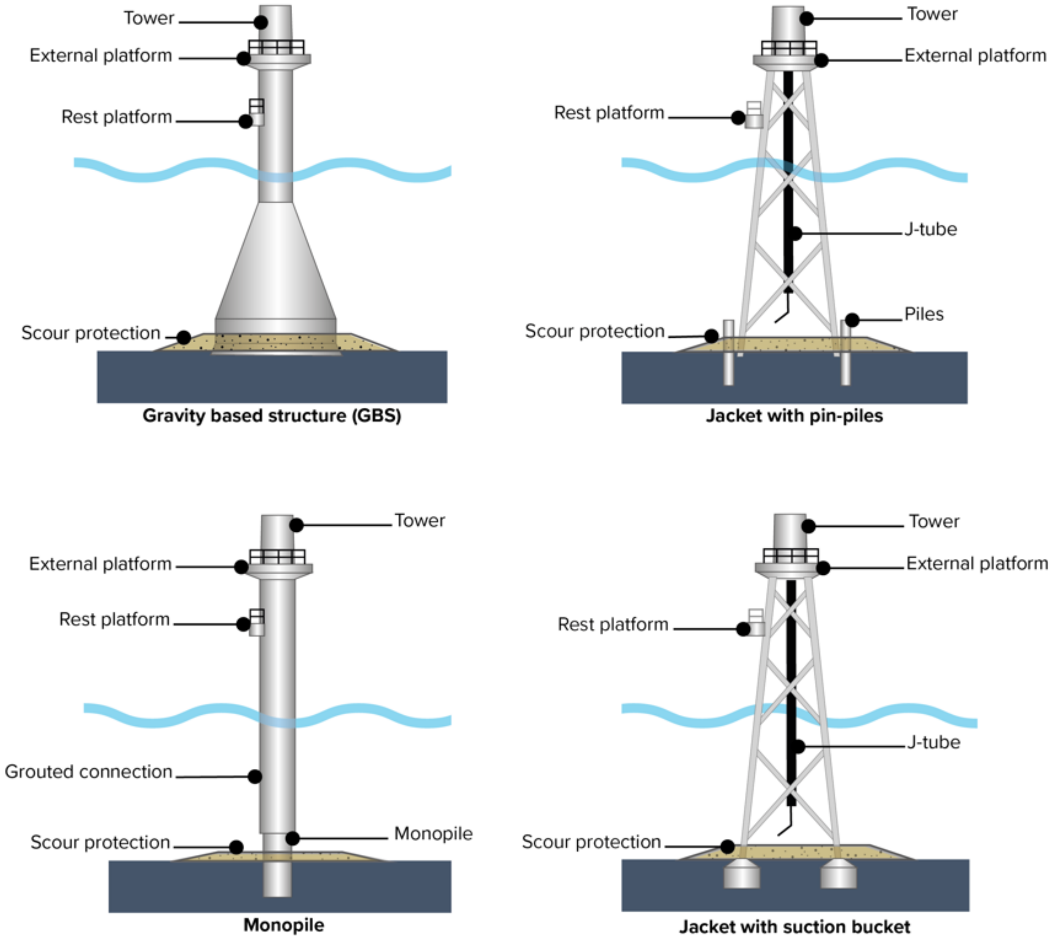



*Plate 3.3 WTG/OSP foundation options*

*Table 3.4 WTG/OSP foundation design envelope*

| Foundation types | Parameter | Maximum values |
|---|---|---|
| Monopile | Maximum pile diameter (m) | 12 |
| | Maximum footprint on the seabed per WTG/OSP ($m^2$) | 114 |
| | Maximum footprint on the seabed for WTGs/OSP(s) ($m^2$) | 3,648 (3,420$m^2$ for 30 x WTGs and 228$m^2$ for 2 x OSPs) |
| | Maximum pile penetration depth (m) | 56 |
| Multi-legged pin-piled jacket (four-legged jacket) | Maximum legs per jacket foundation | 4 |
| | Maximum pile diameter (m) | 3 |
| | Maximum leg spacing at seabed (m) | 35 |
| | Maximum footprint on the seabed, pile-edge to pile-edge, per WTG/OSP ($m^2$) | 28.5 |
| | Maximum footprint on the seabed for WTGs/OSPs ($m^2$) | 1,055 (998$m^2$ for 35 x WTGs and 57$m^2$ for 2 x OSPs) |
| Multi-legged suction bucket jacket (three-legged jackets) | Maximum legs per suction bucket (jacket) foundation | 3 |
| | Maximum bucket diameter (m) | 20 |
| | Maximum leg spacing at seabed (m) | 35 |
| | Maximum footprint on the seabed per WTG/OSP ($m^2$) | 945 |
| | Maximum footprint on the seabed for WTGs/OSPs ($m^2$) | 34,965 (33,075$m^2$ for 35 x WTGs and 1,890$m^2$ for 2 x OSPs) |
| GBS | Maximum base slab diameter (m) | 65 |
| | Maximum cone bottom diameter (m) | 55 |
| | Maximum cone top/shaft diameter (m) | 15 |
| | Maximum cone height (m) | 40 |
| | Maximum footprint on the seabed per WTG/OSP[5] ($m^2$) | 3,318 |
| | Maximum footprint on the seabed for WTGs/OSPs ($m^2$) | 122,766 (116,130$m^2$ for 35 WTGs[6] and 6,636$m^2$ for 2 x OSPs) |

[5] A circular base is assumed as a worst-case.

[6] Noting that both smaller and larger WTGs have the same GBS foundation footprint.

37. Foundation types would be selected following detailed design, based on suitability of the ground conditions, water depths and WTG/OSP models or design. There may be only one type used, or a combination of foundation types may be used across the windfarm site.

### 3.2.5 Inter-array cables

38. Subsea inter-array cables would be installed to connect the individual WTGs and also connect the WTGs to the OSP(s).

39. Where possible, inter-array cables would be buried, with a target burial depth of 1.5m where ground conditions allow and a burial range expected to be between 0.5m and 3m. Where cable burial is not possible, alternative cable protection measures could be used. This may include rock placement, grout/sandbags, concrete mattresses, and polyethylene ducting. The appropriate level of protection would be determined based on an assessment of the risks posed to the Project, in specific areas.

40. It is assumed that 10% of the inter-array cable length would require additional cable protection due to ground conditions. Protection would also be required at the entry points of each WTG and OSP(s) foundation, and at cable crossings. These are outlined in more detail in **Chapter 5 Project Description** (Document Reference 5.1.5) of the ES.

41. The inter-array cables are expected to operate at 66kV or 132kV alternating current (AC). It is expected that 132kV AC cables may not be sufficiently ready or available, on an industry-wide level, for installation, but this higher voltage has been retained, pending further electrical studies.

42. The diameter of the inter-array cables may be up to 220mm. The design envelope for inter-array cables, crossings and entry to WTGs/OSP(s) is given in **Table 3.5**.

*Table 3.5 Inter-array cable design envelope*

| Parameter | Value |
|---|---|
| Maximum length of inter-array cables (km) | 70 |
| Burial depth range (m) | 0.5 – 3<br>(target burial depth of 1.5) |
| Maximum installation corridor disturbance width (m) | 25 |
| **Unburied cable parameters** | |
| Maximum height protection (m) | 2 |
| Maximum width protection (m) | 13 |

| Parameter | Value |
|---|---|
| Anticipated % cable unburied due to ground conditions[7] | 10 |
| Estimated total length of unburied cable due to ground conditions (km) | 7 |
| **Cable protection at entry of cables to WTG/OSP(s)** | |
| Number of entry points to WTGs and OSP(s) | 70 |
| Maximum length of cable protection required at each entry point (m) | 50 |
| Maximum length of protected cable (m) | 3,500 |
| Maximum width of rock berm protection at the bottom (m) | 13 |
| Maximum width at top of rock berm protection (m) | 1 |

### 3.2.6 Platform link cables

43. Should the Project require two OSPs, then platform link cables would be required to connect each of the OSPs, to enable transfer of generated power from one OSP to the other, and to ensure that electricity transmission can continue in the event of one cable failing. The platform link cables are expected to operate at up to 275kV AC.

44. Cables may require protection where they cannot be buried due to ground conditions. Additionally, cables would require protection at cable crossings and at entry points to OSP(s). The exact requirements would be identified post-consent, prior to the start of construction, based on the final WTG and OSP locations and detailed site surveys.

45. The design envelope for the platform link cables is given in **Table 3.6**.

*Table 3.6 OSP(s) platform link cable and crossings design envelopes*

| Parameter | Value |
|---|---|
| **General parameters** | |
| Maximum number of cables | 2 |
| Maximum length of cable (per cable) (km) | 5 |
| Maximum number of cable trenches | 2 |

[7] The percentage of cable that remains unburied due to ground conditions is dependent on the results of a cable burial survey. As such, 10% has been used a worst-case assumption.

| Parameter | Value |
|---|---|
| Maximum total length of all cable trenches (km) | 10 |
| Burial depth range (m) | 0.5 – 3 (target burial depth of 1.5) |
| Maximum installation corridor disturbance width (m) | 25 |
| **Unburied cable parameters** | |
| Maximum height protection (m) | 2 |
| Maximum width protection (m) | 13 |
| Anticipated % cable unburied due to ground conditions[8] | 10 |
| Estimated total length of unburied cable due to ground conditions (km) | 1 |

### 3.2.7 Cable/pipelines crossings

46. It is anticipated that there could be up to nine cable/pipeline crossings required for inter-array cables, and up to six crossings for platform link cables within the windfarm site. Cable protection would be required at the crossings (as outlined in **Table 3.7**) and is additional to the cable protection requirements set out in **Table 3.6**.

*Table 3.7 Cable/pipeline crossings design envelope*

| Parameter | Value |
|---|---|
| Maximum number of cable/pipeline crossings | 15 (9 for inter-array cables, 6 for platform link cables) |
| Maximum cable/pipeline crossing height per crossing (m) | 2.8 |
| Maximum side slope | 3:1 |
| Maximum cable/pipeline crossing top width (m) | 1 |
| Maximum cable/pipeline crossing bottom width per crossing (m) | 17.8 |

[8] The percentage of cable that remains unburied due to ground conditions is dependent on the results of a cable burial survey. As such, 10% has been used a worst-case assumption.

| Parameter | Value |
|---|---|
| Maximum cable/pipeline crossing length per crossing (m) | 250 |

## 3.3 Construction

47. Construction activities may include seabed preparation, Unexploded Ordnance (UXO) clearance[9], foundation installation (which may include pile driving), assembly of WTGs components (tower and the nacelle, which contains the generator), installation of the OSP(s) (including foundations and topside), cable installation and deployment of cable protection and scour protection. The works would require a range of vessel types, including dynamic positioning (DP) and jack-up barges, which could require anchoring.

48. Construction would typically be performed on a 24-hour basis, depending on suitable construction weather windows. During the construction phase, there would be 500m radius safety zones around installation vessels, foundation structures, WTGs and OSP(s).

49. Offshore construction is anticipated over a 2.5 year construction programme.

## 3.4 Operation and maintenance

50. During the operation and maintenance period, scheduled and unscheduled monitoring and maintenance of infrastructure would be required. During the Project life, it is likely that some refurbishment or replacement of offshore infrastructure would be required. Activities such as cable repair or reburial were also anticipated. All offshore infrastructure, including WTGs, foundations, cables and OSP(s) would be included in monitoring and maintenance programmes (see **Chapter 5 Project Description** of the ES).

51. For this RIAA, it has been assumed the operation and maintenance duration would be 35 years from the date of first commercial export, which would then be followed by decommissioning activities.

## 3.5 Decommissioning

52. At the end of the operational lifetime of the Project, offshore decommissioning would include the removal of all of the WTG and OSP(s) components and cutting of foundations to below seabed level. Cables, cable protection, some parts of the foundations and scour protection may be left *in situ*.

[9] Permissions for UXO removal would be sought in a future Marine Licence application and European Protected Species (EPS) licence post-consent.

53. The detail and scope of the decommissioning works would be determined by the relevant legislation and guidance at the time of decommissioning and agreed with the regulator.

## 3.6 Transmission Assets

54. As described in **Section 1.1**, a separate DCO is being sought for the Transmission Assets for the Morecambe and Morgan projects. The key components of the Transmission Assets (as presented in the Transmission Asset PEIR (Morgan Offshore Wind Limited and Morecambe Offshore Windfarm Ltd, 2023a) include:

- OSP(s) - to transform electricity generated by the Morgan and Morecambe Generation Assets to a higher voltage, allowing the power to be efficiently transmitted to shore from each windfarm site (noting that the OSP(s) are also included in the Application for the Project and the Morgan Generation Assets[10])
- Interconnector cables (also known as platform link cables) - to connect OSP(s) within each windfarm site to each other
- Morgan offshore booster station – a potential mid-point reactive power compensation substation
- Offshore export cables – to link the Generation Assets of each windfarm site to the landfall site
- Landfall – where the offshore export cables are joined to the onshore cables
- Onshore export cables - to link the landfall with the onshore substations
- Onshore substations - substations (containing the components for transforming the power supplied via the onshore export cables) and associated grid connection infrastructure

55. The Transmission Assets PEIR red line boundary (including both the offshore and onshore elements) is approximately 697.8km$^2$ in area. The offshore elements of the Transmission Assets are located in the Eastern Irish Sea. The offshore elements connect the Morgan and Morecambe array areas to the coast, south of Blackpool. The onshore elements of the Transmission Assets are located within the local authority areas of Fylde Council, Blackpool Council, South Ribble Borough Council, Preston City Council (and Lancashire County Council, at the County level).

[10] At the time of writing the ES, a decision had been taken that the OSP(s) would remain solely within the Generation Assets application and would not be included within the DCO application for the Transmission Assets. This decision post-dated the PEIR that was prepared for the Transmission Assets. The OSP(s) are still included in the description of the Transmission Assets for the purposes of this document as the CEA carried out in respect of the Generation/Transmission Assets is based on the information available from the Transmission Assets PEIR.

# 4 Approach to HRA

56. The Habitats Regulations place an obligation on ‘competent authorities’ to carry out an Appropriate Assessment of any proposal likely to affect a European site. The HRA process is informed and assisted by the Applicant. It is the responsibility of the Applicant to include ‘sufficient information’ within the application to inform the HRA. The HRA process consists of four stages, as further described within the Defra (2021) and the PINS advice note 10 guidance. These have been detailed within the HRA Screening Report (Morecambe Offshore Windfarm Ltd, 2023a; Document Reference 4.10), summarised below.

- Stage 1 - For all plans and projects which are not wholly, directly connected with or necessary to the conservation management of a site’s qualifying features (such as the proposed Project), Stage 1 screening is required, as a minimum. In Stage 1, European sites are screened for Likely Significant Effect (LSE) arising from the plan or project (either alone or in-combination with other plans or projects). Stage 1 screening for the Project is provided in within the HRA Screening Report (Morecambe Offshore Windfarm Ltd, 2023a; Document Reference 4.10).
- Stage 2 - For those designated sites where LSE cannot be excluded in Stage 1, further information to inform the assessment is prepared. This RIAA provides an assessment on whether the Project-alone or in-combination could adversely affect the integrity of screened in European sites in view of their conservation objectives. This report has been updated following stakeholder feedback on the draft RIAA (FLO-MOR-REP-0005; Morecambe Offshore Windfarm Ltd, 2023b) and incorporates additional survey and project information. This RIAA is submitted alongside the ES as part of the DCO Application.
- Stages 3 and 4 – Consider alternatives, imperative reasons of overriding public interest and compensatory measures where the Competent Authority concludes in the Appropriate Assessment that an AEoI on a European site cannot be ruled out beyond reasonable scientific doubt.

## 4.1 In-combination assessment

57. The in-combination assessment considers effects that may arise from the Project in-combination with other plans and projects.

58. The separate consenting process for the Project and the Transmission Assets has not impacted the conclusions drawn for the Project in the HRA Screening Report (Morecambe Offshore Windfarm Ltd, 2023; Document Reference 4.10) or this RIAA. Where there was a pathway for in-combination effects between

the Project and the Transmission Assets, a separate 'combined' assessment has been undertaken as an additional step within the in-combination assessment.

59. The Transmission Assets Information to Support an Appropriate Assessment (ISAA) (Morgan Offshore Wind Limited and Morecambe Offshore Windfarm Ltd, 2023b), issued at PEIR stage, informed the in-combination assessment.

## 4.2 Consultation

60. This report has been informed by consultation with Statutory Nature Conservation Bodies (SNCBs) and other stakeholders over a number of stages. The key elements of consultation in relation to the Project have been:

- The Scoping Report (submitted in June 2022) and request for a Scoping Opinion (received in August 2022)
- The Evidence Plan Process (EPP), which was ongoing throughout the pre-application phase, including consultation on the draft HRA Screening Report
- Statutory 42 consultation responses received on the draft RIAA, HRA Screening Report and PEIR which were published for statutory consultation in April 2023

### 4.2.1 EIA scoping and HRA screening

61. Consultation has been undertaken with the appropriate authorities as part of the scoping stage of the EIA process. The Scoping Report was submitted to PINS on 23$^{rd}$ June 2022 and a Scoping Opinion received on 2$^{nd}$ August 2022. Scoping established the potential effects of the Project that have been assessed by the EIA and, where applicable, the HRA.

### 4.2.2 Evidence Plan Process

62. The EPP is a non-statutory, voluntary process that aims to encourage upfront agreement on what information an applicant needs to supply to the PINS as part of a DCO Application. It aims to ensure EIA and HRA requirements are met and to reduce the risk of major infrastructure projects being delayed at (or before) the examination phase.

63. The EPP aims to identify and agree the scope of the assessment, the baseline used, methodologies used to collect and analyse data, the interpretation of information, and the conclusions presented (including any LSE). As part of the Project EPP, Expert Topic Groups (ETGs) have been established where it is relevant for multiple agencies to collectively engage in topic specific technical discussions, including those related to the Project HRA process.

64. The EPP also enables consultation on proposed mitigation and/or compensation measures. Agreements and areas where disputes remain between the Applicant and the relevant SNCB have been documented in agreement logs and used to inform a Statement of Common Ground (SoCG) where possible.

65. A summary of the consultation relevant to the HRA process is provided in **Table 4.1**. Specific comments on the HRA screening report (Morecambe Offshore Windfarm Ltd, 2023a; Document Reference 4.10) are listed for each technical topic in the following tables (**Table 6.1**; **Table 7.1**; **Table 8.2**; and **Table 9.1**).

*Table 4.1 Consultation relevant to HRA*

| Dates | Topic | Organisation consulted |
|---|---|---|
| October 2021 – June 2022 | Introductory meetings | Blackpool Airport, Cumbria Local Enterprise Partnership, Environment Agency, Isle of Man Government, Isle of Man Steam Packet Company, Historic England, Isle of Man Harbours and Coastguard, Lancaster City Council, Lancashire County Council, MMO, MCA, Natural England, Ministry of Defence, The National Federation of Fishermen's Organisations, North West Inshore Fisheries and Conservation Authority (IFCA), North West Wildlife Trusts (Cumbria, Lancashire & Cheshire), Peel Ports, Associated British Ports, Port of Barrow, Royal Society for the Protection of Birds, Royal Yachting Association, Sea Truck Ferries, Stena Line Ferries, TH, PINS, UK Chamber of Shipping, the Welsh Government, Wyre Council, |
| March 2022 | EPP Steering Group Meeting 1 | Natural England, MMO, Environment Agency, Historic England, PINS. |
| May 2022 | Marine Mammal ETG 1 | Natural England, MMO, Cumbria Wildlife Trust, Centre for Environment, Fisheries and Aquaculture Science (Cefas) |
| May 2022 | Offshore Ornithology ETG 1 | Natural England, MMO |
| June 2022 | Marine Ecology ETG 1 | Natural England, MMO, Wildlife Trusts, North West IFCA, Environment Agency, Cefas |
| August/ September 2022 | Marine Mammal ETG 2 | Natural England, MMO, Cumbria Wildlife Trust, Cefas |
| September 2022 | Offshore Ornithology ETG 2 | Natural England, MMO, Royal Society for the Protection of Birds (RSPB) |
| September 2022 | Marine Ecology ETG 2 | Natural England, MMO, NW Wildlife Trust, Environment Agency, Cefas |

| Dates | Topic | Organisation consulted |
|---|---|---|
| September 2022 | EPP Steering Group Meeting 2 | Natural England, MMO, Historic England |
| November 2022 | Marine Mammal ETG 3 | Natural England, Wildlife Trusts, MMO, Isle of Man Government |
| November 2022 | Offshore Ornithology ETG 3 | Natural England, MMO, RSPB, Isle of Man Government |
| November 2022 | Marine Ecology ETG 3 | Natural England, MMO, Wildlife Trusts, NW IFCA, Environment Agency, Isle of Man Government |
| June 2023 | EPP Steering Group Meeting 3 | MMO, Environment Agency, Historic England, PINS. |
| June 2023 | Marine Mammal ETG 4 | MMO, NW Wildlife Trust, Isle of Man Government. |
| June 2023 | Offshore Ornithology ETG 4 | MMO, Natural England, RSPB, Isle of Man Government. |
| June 2023 | Marine Ecology ETG 4 | MMO, Cefas, NW Wildlife Trust, Isle of Man Government and NW IFCA. |
| October 2023 | Offshore Ornithology ETG 5 | MMO, NE, RSPB, Isle of Man Government and Merseyside Environmental Advisory Service (MEAS). |
| October 2023 | Marine Ecology ETG 5 | MMO, Natural England, Cefas Isle of Man Government, NW IFCA and MEAS. |
| October 2023 | Marine Mammal ETG 5 | NE, MMO, Cefas, Isle of Man Government and MEAS. |
| January 2024 | Marine Mammal ETG 6 | Natural England, MMO, Cefas and NW Wildlife Trust |
| January 2024 | Offshore Ornithology ETG 6 | MMO, Natural England, RSPB and Isle of Man Government |
| January 2024 | Marine Ecology ETG 6 | Natural England, MMO, Cefas, NW Wildlife Trust and Isle of Man Government |
| February 2024 | EPP Steering Group Meeting 4 | PINS, Natural England, Historic England and MMO |
| February – March 2024 | E mail correspondence | National Parks and Wildlife Service (NPWS) (Ireland)<br>Department of Agriculture, Environment and Rural Affairs (Northern Ireland)<br>Marine Scotland, Nature Scot (Scotland)<br>National Resources Wales (Wales) |

# 5 Screening conclusion

66. The HRA Screening process for the Project has been undertaken following analysis of the Zone of Influence (ZoI) of impacts and in consultation with relevant stakeholders through the EPP. The screening is provided in the HRA Screening Report (Morecambe Offshore Windfarm Ltd, 2023a; Document Reference 4.10).

67. Since the publication of the draft RIAA further environmental survey and assessment work, changes to European sites, consultee responses, refinements to the Project design and re-assessment of cumulative projects have been taken into consideration and any such changes reflected within this RIAA. The following updates to the RIAA have been made as a result:

- Addition of Ynys Seiriol/Puffin Island SPA for great cormorant
- Removal of shag, herring gull, kittiwake and puffin as screened in features (due to distance) and addition of common guillemot for Canna and Sanday SPA
- Addition of Ballaugh Curragh Ramsar site for hen harrier
- Addition of River Ehen SAC for Annex II fish
- Addition of River Eden SAC for Annex II fish
- Addition of Solway Firth SAC for Annex II fish
- Addition of River Derwent and Bassenthwaite Lake SAC for Annex II fish
- Assessment of Pembrokeshire Marine SAC for grey seal
- Assessment of grey seal for Cardigan Bay SAC

68. The following sub-sections identify the sites and features screened into the Appropriate Assessment in relation to the Projects alone or in-combination (together with other plans, activities and projects), with the features and sites screened in summarised in **Table 5.1** and **Table 5.2**.

*Table 5.1 Summary of European sites and features screened in*

| Site | Features | Rationale |
|---|---|---|
| **SACs** | | |
| Shell Flat and Lune Deep SAC | 1110 Sandbanks which are slightly covered by sea water all the time | Potential for the following indirect effects (overlap with the ZoI):<br>▪ Increased suspended sediment concentrations (SSCs)<br>▪ Smothering due to increased SSCs |

| Site | Features | Rationale |
|---|---|---|
| | | ▪ Re-mobilisation of contaminated sediments and changes to water quality<br>The Lune Deep part of the SAC is 18km from the windfarm site and therefore beyond the ZoI of any effects from SSCs increases and subsequent deposition. |
| Dee Estuary/ Aber Dyfrdwy SAC | 1095 Sea lamprey<br>1099 River lamprey | Species range may overlap with the Project ZoI e.g. noise and suspended sediments |
| River Dee and Bala Lake/ Afon Dyfrdwy a Llyn Tegid SAC | 1106 Atlantic salmon<br>1095 Sea lamprey<br>1099 River lamprey | Species range may overlap with the Project ZoI e.g. noise and suspended sediments |
| Afon Gwyrfai a Llyn Cwellyn SAC | 1106 Atlantic salmon | Species range may overlap with the Project ZoI e.g. noise and suspended sediments |
| Afon Eden - Cors Goch Trawsfynydd SAC | 1106 Atlantic salmon | Species range may overlap with the Project ZoI e.g. noise and suspended sediments ZoI |
| River Ehen SAC | 1106 Atlantic salmon<br>1095 Sea lamprey<br>1099 River lamprey | Species range may overlap with the Project ZoI e.g. noise and suspended sediments |
| River Eden SAC | 1106 Atlantic salmon<br>1095 Sea lamprey<br>1099 River lamprey<br>1096 Brook lamprey | Species range may overlap with the Project ZoI e.g. noise and suspended sediments |
| Solway Firth SAC | 1095 Sea lamprey<br>1099 River lamprey | Species range may overlap with the Project ZoI e.g. noise and suspended sediment |
| River Derwent and Bassenthwaite Lake SAC | 1106 Atlantic salmon<br>1095 Sea lamprey<br>1099 River lamprey<br>1096 Brook lamprey | Species range may overlap with the Project ZoI e.g. noise and suspended sediments |
| North Anglesey Marine SAC | Harbour porpoise | Potential for connectivity. It has been assumed that harbour porpoise could be present in the windfarm site or ZoI |
| North Channel SAC | Harbour porpoise | Potential for connectivity. It has been assumed that harbour porpoise could be present in the windfarm site or ZoI |

| Site | Features | Rationale |
|---|---|---|
| West Wales Marine SAC | Harbour porpoise | Potential for connectivity. It has been assumed that harbour porpoise could be present in the windfarm site or ZoI |
| Rockabill to Dalkey Island SAC | Harbour porpoise | Potential for connectivity. It has been assumed that harbour porpoise could be present in the windfarm site or ZoI |
| Bristol Channel Approaches SAC | Harbour porpoise | Potential for connectivity. It has been assumed that harbour porpoise could be present in the windfarm site or ZoI |
| Pen Llŷn a`r Sarnau SAC | Bottlenose dolphin<br>Grey seal | Potential for connectivity. It has been assumed that both bottlenose dolphin and grey seal could be present in the windfarm site or ZoI |
| Cardigan Bay SAC | Bottlenose dolphin<br>Grey seal | Potential for connectivity. Same population of bottlenose dolphins found at Pen Llŷn a`r Sarnau SAC, have been known to travel to Cardigan Bay. It has been assumed that grey seal could be present in the windfarm site or ZoI |
| Pembrokeshire Marine SAC | Grey seal | Potential for connectivity. It has been assumed that grey seal could be present in the windfarm site or ZoI |
| Strangford Lough SAC | Harbour seal | Potential for connectivity. It has been assumed that harbour seal could be present in the windfarm site or ZoI |
| **SPAs and Ramsar** | | |
| See **Table 5.2** in **Section 5.3.** | | |

## 5.1 Offshore sites designated for Annex I habitats

69. The HRA screening exercise considered sites which met the following criteria:

- A component of the Project directly overlaps a site whose qualifying features include benthic habitats
- The distance between the Project windfarm site and the offshore habitat qualifying feature is within the range for which there could be an interaction (i.e. within a ZoI for a physical process change resulting from the Project)

70. With regard to the latter point, the screening assessment (for offshore Annex I habitat) took into account a conservative 15km ZoI, based on the excursion distance of one spring tidal ellipse (see Document Reference 4.10).

71. The outcome of the screening exercise (and subsequent consultation) concluded that one site was screened in for Appropriate Assessment: Shell

Flat and Lune Deep SAC (noting only the Shell Flat part of the SAC has been screened in).

72. The sites screened out of the need for an Appropriate Assessment due to the conclusion of no LSE are listed in Document Reference 4.10.

## 5.2 Offshore sites designated for Annex II fish species

73. The HRA screening exercise considered sites which met the following criteria:

- The Project windfarm site directly overlapped a site whose qualifying features includes an Annex II migratory fish species
- The distance between the Project windfarm site and a site with a fish qualifying feature is within the range for which there could be an interaction e.g. the distance of the site from the source of suspended sediment is within the range at which sediment deposition could occur
- The distance between the Project windfarm site and resources on which the qualifying feature depends (i.e., an indirect effect acting though prey or access to habitat) is within the range for which there could be an interaction
- The likelihood that a foraging area or a migratory route occurs within the windfarm site

74. As a result, the sites screened in for further assessment were:

- Dee Estuary/Aber Dyfrdwy SAC
- River Dee and Bala Lake/Afon Dyfrdwy a Llyn Tegid SAC
- Afon Gwyrfai a Llyn Cwellyn SAC
- Afon Eden - Cors Goch Trawsfynydd SAC
- Solway Firth SAC
- River Ehen SAC
- River Eden SAC
- River Derwent and Bassenthwaite Lake SAC

75. It is noted that brook lamprey *Lampetra planeri* are a designated feature of some of the above sites, but given they are an entirely freshwater species they would not interact with noisy activities from the Project given that the maximum range of effect was 33km (see **Section 7.4.2.1** for noise modelling results). Brook lampreys are therefore screened out of further assessment.

76. The sites screened out of the need for an Appropriate Assessment due to the conclusion of no LSE are listed in the Screening Report (Document Reference 4.10).

## 5.3 Offshore ornithology (Birds Directive Annex 1 and migratory species)

77. Birds potentially affected by the Project were predominantly seabirds (defined for this report as auks, gulls, terns, gannets, skuas, shearwaters, petrels and divers). These species have the potential to be present during the breeding season, non-breeding season and the spring/autumn migration/passage periods. Other bird species that may be affected by the Project include waterfowl (swans, geese, ducks and waders) and other bird species which may fly through the windfarm site during spring and/or autumn migration/passage periods.

78. For offshore ornithology receptors during the breeding season, the HRA screening focused primarily on the potential for connectivity between seabirds breeding at colonies classified as SPAs, and the Project.

79. Outside the breeding season, seabirds breeding at SPAs located beyond the breeding season foraging range of the Project may disperse from the area around the breeding colony. Therefore, these birds may spend part or all of the non-breeding season in the vicinity of the Project, either wintering or migrating through on spring and/or autumn passage to wintering areas. During this time the number of SPAs with potential connectivity to the Project would increase.

80. The HRA screening exercise considered sites which either overlapped with or were in close proximity to the Project elements, or were within the relevant species' foraging range/Biologically Defined Minimum Population Scales (BDMPS) area during the breeding and non-breeding season (Furness, 2015).

81. **Table 5.2** presents a summary of the sites and associated qualifying bird species that were screened into the Appropriate Assessment.

82. Full details of sites and features considered in the HRA Screening Report (including those screened out; i.e. where no LSE was concluded), including rationale, have been provided in the HRA Screening Report (Morecambe Offshore Windfarm Ltd, 2023a; Document Reference 4.10). This also includes a list of all species considered, together with scientific names.

*Table 5.2 Ornithology screening – summary of European sites screened in*

| European site | Qualifying feature |
|---|---|
| Liverpool Bay/Bae Lerpwl SPA | Red-throated diver |
| | Black (common) scoter |
| | Little gull |
| | Common tern |
| Morecambe Bay and Duddon Estuary SPA and Ramsar sites | Little egret |
| | Whooper swan |
| | Pink-footed goose |
| | Common shelduck |
| | Northern pintail |
| | Eurasian oystercatcher |
| | Ringed plover |
| | European golden plover |
| | Grey plover |
| | Ruff |
| | Red knot |
| | Sanderling |
| | Bar-tailed godwit |
| | Eurasian curlew |
| | Common redshank |
| | Ruddy turnstone |
| | Mediterranean gull |
| | Lesser black-backed gull |
| | Black-tailed godwit |
| | Dunlin |
| | Lesser black-backed gull |
| | Herring gull |
| | Sandwich tern |
| | Common tern |
| | Seabird assemblage |

| European site | Qualifying feature |
|---|---|
| | Waterbird assemblage |
| Ribble and Alt Estuaries SPA and Ramsar | Tundra swan |
| | Whooper swan |
| | Pink-footed goose |
| | Common shelduck |
| | Eurasian wigeon |
| | Eurasian teal |
| | Northern pintail |
| | Eurasian oystercatcher |
| | Ringed plover |
| | European golden plover |
| | Grey plover |
| | Red knot |
| | Sanderling |
| | Bar-tailed godwit |
| | Common redshank |
| | Black-tailed godwit |
| | Dunlin |
| | Ruff |
| | Lesser black-backed gull |
| | Common tern |
| | Seabird assemblage |
| | Waterbird assemblage |
| Mersey Narrows and North Wirral Foreshore SPA and Ramsar | Bar-tailed godwit |
| | Little gull |
| | Common tern |
| | Red knot |
| | Common tern |
| | Waterbird assemblage |

| European site | Qualifying feature |
|---|---|
| Martin Mere SPA and Ramsar | Tundra swan |
| | Whooper swan |
| | Pink-footed goose |
| | Eurasian teal |
| | Northern pintail |
| | Eurasian wigeon |
| | Waterbird assemblage |
| The Dee Estuary SPA and Ramsar | Common shelduck |
| | Eurasian teal |
| | Northern pintail |
| | Eurasian oystercatcher |
| | Grey plover |
| | Red knot |
| | Bar-tailed godwit |
| | Eurasian curlew |
| | Common redshank |
| | Sandwich tern |
| | Black-tailed godwit |
| | Dunlin |
| | Common tern |
| | Waterbird assemblage |
| Anglesey Terns/Morwenoliaid Ynys Môn SPA | Sandwich tern |
| | Common tern |
| | Arctic tern |
| Bowland Fells SPA | Hen harrier |
| | Merlin |
| | Lesser black-backed gull |
| Mersey Estuary SPA and Ramsar | Great crested grebe |
| | Common shelduck |
| | Eurasian wigeon |

| European site | Qualifying feature |
|---|---|
| | Eurasian teal |
| | Northern pintail |
| | Ringed plover |
| | European golden plover |
| | Grey plover |
| | Northern lapwing |
| | Eurasian curlew |
| | Common redshank |
| | Black-tailed godwit |
| | Dunlin |
| | Waterbird assemblage |
| Ynys Seiriol/Puffin Island SPA | Great cormorant |
| Leighton Moss Ramsar | Waterbird assemblage |
| | Wetland bird assemblage |
| Traeth Lafan/Lavan Sands, Conway Bay SPA | Great crested grebe |
| | Red-breasted merganser |
| | Eurasian oystercatcher |
| | Eurasian curlew |
| | Common redshank |
| Solway Firth SPA | Red-throated diver |
| | Great cormorant |
| | Whooper swan |
| | Pink-footed goose |
| | Barnacle goose |
| | Common shelduck |
| | Eurasian teal |
| | Northern pintail |
| | Northern shoveler |
| | Greater scaup |
| | Black (common) scoter |

| European site | Qualifying feature |
|---|---|
| | Common goldeneye |
| | Goosander |
| | Eurasian oystercatcher |
| | Ringed plover |
| | European golden plover |
| | Grey plover |
| | Northern lapwing |
| | Red knot |
| | Sanderling |
| | Bar-tailed godwit |
| | Eurasian curlew |
| | Common redshank |
| | Ruddy turnstone |
| | Black-headed gull |
| | Mew gull |
| | Herring gull |
| | Dunlin |
| Migneint-Arenig-Dduallt SPA | Hen harrier |
| | Merlin |
| | Peregrine falcon |
| Berwyn SPA | Red kite |
| | Hen harrier |
| | Merlin |
| | Peregrine falcon |
| South Pennine Moors Phase 2 SPA | Merlin |
| | European golden plover |
| | Short-eared owl |
| North Pennine Moors SPA | Hen harrier |
| | Merlin |
| | Peregrine falcon |

| European site | Qualifying feature |
|---|---|
| | European golden plover |
| Glannau Aberdaron ac Ynys Enlli/ Aberdaron Coast and Bardsey Island SPA | Manx shearwater |
| Strangford Lough SPA and Ramsar | Sandwich tern |
| | Common tern |
| Copeland Islands SPA | Manx shearwater |
| Larne Lough SPA and Ramsar | Sandwich tern |
| Ailsa Craig SPA | Northern gannet |
| | Lesser black-backed gull |
| | Black-legged kittiwake* |
| | Herring gull* |
| | Common guillemot* |
| | Seabird assemblage |
| Coquet Island SPA | Common tern |
| | Seabird assemblage |
| Flamborough and Filey Coast SPA | Northern gannet |
| | Black-legged kittiwake |
| | Seabird assemblage |
| Rathlin Island SPA | Black-legged kittiwake |
| | Common guillemot |
| | Razorbill |
| | Seabird assemblage |
| Sheep Island SPA | Great cormorant |
| Farne Islands SPA | Seabird assemblage |
| Forth Islands SPA | Northern gannet |
| | Atlantic puffin |
| | Seabird assemblage |
| Skomer, Skokholm and the Seas off Pembrokeshire/Sgomer, Sgogwm a Moroedd Penfro SPA | Manx shearwater |
| | European storm-petrel |
| | Atlantic puffin |

| European site | Qualifying feature |
|---|---|
| | Lesser black-backed gull |
| | Seabird assemblage |
| Grassholm SPA | Northern gannet |
| North Colonsay and Western Cliffs SPA | Black-legged kittiwake |
| | Common guillemot |
| | Seabird assemblage |
| Treshnish Isles SPA | European storm-petrel |
| Fowlsheugh SPA | Northern fulmar |
| | Black-legged kittiwake |
| | Seabird assemblage |
| Rum SPA | Manx shearwater |
| | Seabird assemblage |
| Canna and Sanday SPA | Common guillemot |
| | Seabird assemblage |
| Buchan Ness to Collieston Coast SPA | Northern fulmar |
| | Black-legged kittiwake |
| | Seabird assemblage |
| Mingulay and Berneray SPA | Northern fulmar |
| | Common guillemot |
| | Razorbill |
| | Seabird assemblage |
| Troup, Pennan and Lion's Heads SPA | Northern fulmar |
| | Black-legged kittiwake |
| | Seabird assemblage |
| Isles of Scilly SPA | European shag |
| | Lesser black-backed gull |
| | Great black-backed gull |
| | Seabird assemblage |
| East Caithness Cliffs SPA | Northern fulmar |
| | Black-legged kittiwake |

| European site | Qualifying feature |
|---|---|
| | Seabird assemblage |
| Shiant Isles SPA | Northern fulmar |
| | Common guillemot |
| | Razorbill |
| | Atlantic puffin |
| | Seabird assemblage |
| Handa SPA | Northern fulmar |
| | Great skua |
| | Black-legged kittiwake |
| | Common guillemot |
| | Razorbill |
| | Seabird assemblage |
| North Caithness Cliffs SPA | Northern fulmar |
| | Black-legged kittiwake |
| | Seabird assemblage |
| St Kilda SPA | Northern fulmar |
| | Manx shearwater |
| | Leach’s storm-petrel |
| | Great skua |
| | Common guillemot |
| | Atlantic puffin |
| | Northern gannet |
| | Seabird assemblage |
| Cape Wrath SPA | Northern fulmar |
| | Black-legged kittiwake |
| | Common guillemot |
| | Razorbill |
| | Seabird assemblage |
| Flannan Isles SPA | Northern fulmar |
| | Leach’s storm-petrel |

| European site | Qualifying feature |
|---|---|
| | Common guillemot |
| | Atlantic puffin |
| | Seabird assemblage |
| Hoy SPA | Red-throated diver |
| | Northern fulmar |
| | Great skua |
| | Seabird assemblage |
| Copinsay SPA | Northern fulmar |
| | Seabird assemblage |
| Sule Skerry and Sule Stack SPA | Leach’s storm-petrel |
| | Northern gannet |
| | Common guillemot |
| | Atlantic puffin |
| | Seabird assemblage |
| Rousay SPA | Northern fulmar |
| | Seabird assemblage |
| North Rona and Sula Sgeir SPA | Northern fulmar |
| | Leach’s storm-petrel |
| | Northern gannet |
| | Common guillemot |
| | Seabird assemblage |
| Calf of Eday SPA | Northern fulmar |
| | Seabird assemblage |
| West Westray SPA | Northern fulmar |
| | Black-legged kittiwake |
| | Seabird assemblage |
| Fair Isle SPA | Northern fulmar |
| | Great skua |
| | Seabird assemblage |
| Sumburgh Head SPA | Northern fulmar |

| European site | Qualifying feature |
|---|---|
| | Seabird assemblage |
| Foula SPA | Northern fulmar |
| | Great skua |
| | Red-throated diver |
| | Atlantic puffin |
| | Seabird assemblage |
| Noss SPA | Northern fulmar |
| | Great skua |
| | Northern gannet |
| | Seabird assemblage |
| Ronas Hill - North Roe and Tingon SPA and Ramsar | Red-throated diver |
| | Great skua |
| Fetlar SPA | Northern fulmar |
| | Great skua |
| | Seabird assemblage |
| Hermaness, Saxa Vord and Valla Field SPA | Northern fulmar |
| | Great skua |
| | Northern gannet |
| | Red-throated diver |
| | Atlantic puffin |
| | Seabird assemblage |
| Ballaugh Curragh Ramsar SPA | Hen harrier |
| Lambay Island SPA | Guillemot |
| | Puffin |
| | Fulmar |
| | Lesser black-backed gull |
| | Kittiwake |
| | Razorbill |
| | Herring gull |
| | Shag |

| European site | Qualifying feature |
|---|---|
| | Cormorant |
| Howth Head Coast SPA | Kittiwake |
| Ireland's Eye SPA | Kittiwake |
| | Razorbill |
| | Cormorant |
| Wicklow Head SPA | Kittiwake |
| Saltee Islands SPA | Puffin |
| | Fulmar |
| | Gannet |
| | Kittiwake |
| | Guillemot |
| | Shag |
| | Cormorant |
| | Razorbill |
| Horn Head to Fanad Head SPA | Fulmar |
| | Kittiwake |
| | Shag |
| | Cormorant |
| West Donegal Coast SPA | Fulmar |
| | Shag |
| | Cormorant |
| Tory Island SPA | Fulmar |
| Cliffs of Moher SPA | Fulmar |
| | Guillemot |
| | Kittiwake |
| | Razorbill |
| Stags of Broad Haven SPA | Leach's petrel |
| Clare Island SPA | Fulmar |
| Duvillaun Islands SPA | Fulmar |
| High Island, Inishshark and Davillaun SPA | Fulmar |

| European site | Qualifying feature |
|---|---|
| Kerry Head SPA | Fulmar |
| Cruagh Island SPA | Manx shearwater |
| Dingle Peninsula SPA | Fulmar |
| Iveragh Peninsula SPA | Fulmar |
| Blasket Islands SPA | Fulmar |
| | Manx shearwater |
| | Puffin |
| | Lesser black-backed gull |
| Deenish Island and Scariff Island SPA | Fulmar |
| | Manx shearwater |
| Puffin Island SPA | Fulmar |
| | Manx shearwater |
| | Puffin |
| The Bull and The Cow Rocks SPA | Gannet |
| Skelligs SPA | Gannet |
| | Manx shearwater |
| | Fulmar |
| | Puffin |
| * Indicates SPA assemblage species | |

## 5.4 Offshore Annex II sites designated for marine mammals

83. The HRA screening exercise (Morecambe Offshore Windfarm Ltd, 2023a; Document Reference 4.10) considered marine mammal sites that met the following criteria:

- The distance between the potential effect of the Project and a European site with marine mammals as a qualifying feature is within the range for which there could be an interaction (for example, the pathway is not too long for significant noise propagation and therefore the European site is within the ZoI for underwater noise effects)
- The distance between the Project and resources on which the qualifying marine mammal feature depends (i.e. an indirect effect acting though prey or access to habitat) is within the potential ZoI
- The likelihood that a foraging area or a migratory route occurs within the ZoI of the proposed Project (applied to mobile interest features when outside the European site)

84. As a result, the following sites have been screened in for further assessment:

- Sites where harbour porpoise is a qualifying feature:
  - North Anglesey Marine SAC
  - North Channel SAC
  - West Wales Marine SAC
  - Rockabill to Dalkey Island SAC
  - Bristol Channel Approaches SAC
- Sites where bottlenose dolphin is a qualifying feature:
  - Pen Llŷn a`r Sarnau SAC
  - Cardigan Bay SAC

85. There were no European sites within the known average foraging ranges for grey seals and harbour seals. However, as a precautionary approach, the nearest sites designated for harbour and grey seals have been screened in for further assessment:

- Pen Llŷn a`r Sarnau SAC (grey seal)
- Cardigan Bay SAC (grey seal)
- Pembrokeshire Marine SAC (grey seal)
- Strangford Lough SAC (harbour seal)

# 6 Offshore Annex I Habitats

## 6.1 Approach to assessment

86. This section provides information to determine the potential for the Project to have an AEoI on sites designated for Annex I benthic habitats.

87. The assessment focused on those features that are present within the ZoI of the Project. For offshore Annex I habitats, the ZoI has been determined by the excursion of the spring tidal ellipse (i.e. the maximum distance to which disturbed sediment may be advected). As detailed in the Screening Report (Morecambe Offshore Windfarm Ltd, 2023a; Document Reference 4.10), this is understood to be around 10km from a central point, hence the ZoI has been conservatively considered at 15km from the Project boundaries.

88. For each site screened in for further assessment, the following has been provided:

- A summary of the subtidal benthic ecology of the habitats relevant for each European site
- An assessment of the potential effects during the construction, operation and maintenance, and decommissioning, and assessment on whether the Project-alone could adversely affect the integrity of screened in European sites in view of their conservation objectives
- An assessment of the potential for in-combination effects alongside the Transmission Assets and assessment on whether the Project-alone or in-combination could adversely affect the integrity of screened in European sites in view of their conservation objectives
- An assessment of the potential for in-combination effects alongside other relevant developments and projects, including the Transmission Assets, and assessment on whether the Project-alone or in-combination could adversely affect the integrity of screened in European sites in view of their conservation objectives

## 6.2 Consultation

89. Consultation on benthic ecology has been undertaken in line with the process set out in **Section 4.2**. The feedback received through the EPP has been considered in preparing the RIAA.

90. **Table 6.1** provides a summary of how the consultation responses received in relation to the HRA Screening Report and draft RIAA have influenced the approach that has been taken.

*Table 6.1 Consultation responses received in relation to the draft RIAA – Annex I habitats*

| Consultee | Date/document | Comment | Project response/where addressed |
|---|---|---|---|
| MMO | 24th October 2022 "MCZ Screening Report and HRA Screening Report" | Cumulative effects have been considered, however in-combination effects (different effects from the Morecambe Bay Array on a single receptor) have not been included. This information should be included. | The different effects from the Project on a single receptor have been considered as ‘interactions’ in **Section 6.4.2**.<br>Note that for HRA, in-combination effects have been defined as the effect of similar impacts from multiple schemes on the same receptor (these have been defined as cumulative effects in EIA terms) |
| | | As with the MCZA [Marine Conservation Zone Assessment], the HRA does not provide justification for the 50km and 15km zones of influence. The MMO advise inclusion of supporting information (e.g., tidal excursion, tidal direction relevant to the Array area) to provide evidence for the inclusion/exclusion zones selected. | Tidal ellipse (zone of influence) information has been added to the HRA screening report as justification for the screening conclusions (see Document Reference 4.10). |
| | | The HRA report does not provide sufficient evidence to support the potential effects in Table 5.1. The MMO would expect further information on why each of the potential effects have been screened in/out. MMO request that this information is included. | Further information on the evidence to support the potential effects has been added to the HRA Screening Report as justification for the screening conclusions (see Document Reference 4.10). |
| | | The HRA report includes the assessment of in-combination effects, as defined by the EU Habitats Directive. However, it is not clear how in-combination effects (as defined in this report) differ from cumulative effects (which have not been included in the report) – or whether these two terms are used interchangeably in this report. In the literature, cumulative effects are defined as ‘the effect of similar impacts from multiple schemes on the same receptor’ (which appears to be the same as defined for in-combination effects in this report), whereas in-combination effects relate to ‘multiple effects of a single | See above. |

| Consultee | Date/document | Comment | Project response/where addressed |
|---|---|---|---|
| | | development on a receptor'. Both in-combination and cumulative effects should be assessed as they differ. | |
| Natural England | 14th September 2022<br>Advice on FLO-MOR-REP-0004 HRA Screening Report Morecambe Offshore Windfarm – Generation Assets | Invasive Non-Native Species (INNS) screened out - Screen in as OWFs are a potential habitat for INNS that could present a pathway to more readily colonise designated sites. | Introduction and spread of INNS have been screened in (Document Reference 4.10) and assessed in **Section 6.4.2**. |
| | | A justification for the size of sediment plume currently assessed is required along with consideration of plumes up to 10km. | Tidal ellipse and ZoI information has been added to the HRA screening report (see Document Reference 4.10). |
| PINS | 2nd August 2022<br>Scoping Opinion on the Scoping Report | Remobilisation of contaminated sediments: The Scoping Report notes that if the benthic sampling demonstrates low levels of contamination, then this matter would be scoped out of further assessment through the EPP. The Inspectorate agrees that if this approach is agreed through the EPP then this matter can be scoped out of further assessment. However, the specific contamination levels recorded through benthic sampling should still be provided as an annex to the ES. | Benthic sampling across the Project windfarm site has indicated low levels of contaminants, all below environmental thresholds (Cefas Action level 1 and the US Environmental Protection Agency Effects Range – Low (ERLs)). Further detail, including recorded contamination levels, has been provided in **Chapter 8 Marine Sediment and Water Quality** of the ES and **Appendix 9.1 Benthic Characterisation Survey**. This impact was scoped out, which has been agreed by Natural England. |
| MMO | 30th May 2023<br>Section 42 comments on the PEIR and draft RIAA | There is possible sediment suspension from bedload higher into the water column due to turbulence around the foot of monopiles. Table 7.4 states that to investigate this is not proportionate to the conceptual EIA method being used. The MMO considers this insufficient justification for the screening out of an impact. If this pathway exists, this could alter the assessment of sediment suspension significance, thereby affecting the assessments of the | The impact of increased SSCs due to the presence of WTGs/OSP(s) has been further investigated and outlined in Section 7.6.3.3 of **Chapter 7 Marine Geology, Oceanography and Physical Processes** of the ES.<br>In summary, there was considerable supporting evidence that found turbid wakes to be caused by the 'upward turbulent |

| Consultee | Date/document | Comment | Project response/where addressed |
|---|---|---|---|
| | | Marine Conservation Zones (MCZ) and Habitats Regulation Assessment (HRA) also. | mixing' of existing SSCs from the lower water column up into the middle and upper water column, and not the result of ongoing local scouring of seabed sediments, as previously thought (Titan, 2012, 2013; Forster, 2018). These 'turbid wakes' were unlikely to be continuously present, particularly following tidal reversal and at stormier times when there would be enhanced mixing of the water column (Vattenfall Wind Power Limited, 2014). As no 'additional' sediment would be added to the water column, average SSCs in the Project windfarm area and beyond would not change and would be well within the range of SSCs seen during storms (up to 300mg/l). Therefore, no impact to water quality is anticipated and it is not assessed in **Chapter 8 Marine Sediment and Water Quality** of the ES. Furthermore, the Irish Sea has been defined as well mixed throughout the year due to tidal mixing (Howarth, 2005) and therefore there was no identified potential for AEoI on any SACs. |
| Natural England | 2nd June 2023<br>Section 42 comments on the PEIR and draft RIAA | Relevant designated features have been screened in and out as appropriate. | Noted, no further action. |

## 6.3 Assessment of effects

91. The HRA Screening Report (Morecambe Offshore Windfarm Ltd, 2023a; Document Reference 4.10) identified the following potential effects that should be taken forward for further assessment in relation to the construction, operation and maintenance and decommissioning phases of the Project:

- Increased SSCs and deposition (the potential disruption to sediment pathways was also assessed)
- Remobilisation of contaminated sediments
- Introduction and spread of INNS
- Risk of deterioration of water quality due to spillages/leakages

92. The embedded mitigation and worst-case scenario presented in **Sections 6.3.1** and **6.3.2** therefore relate to these effects.

### 6.3.1 Embedded mitigation

93. This section outlines the embedded mitigation incorporated into the design of the Project (**Table 6.2**) which was relevant to the assessment for Shell Flat and Lune Deep SAC. Note that this did not include embedded mitigation for direct effects (i.e., within the windfarm site) which have also been included within the EIA but were not relevant to the indirect effects considered here.

*Table 6.2 Embedded mitigation measures relevant to benthic ecology*

| Parameter | Mitigation measures embedded into the design of the Project |
|---|---|
| WTG spacing | A minimum separation distance of up to 1,060m has been defined between adjacent WTGs within the same row and 1,410m between each row, minimising the potential for interaction between adjacent WTGs with respect to marine physical process and consequent effects on benthic communities. |
| Seabed preparation | Micro-siting would be used (for foundations and cable installation) where possible to minimise the requirements for seabed preparation prior to foundation and cable installation. |
| Scour protection | Scour protection is built into the design for each foundation type in consideration and, where installed after the foundation, it would be installed as early as practicable (typically within the same season) after the foundation installation. |
| Cables | Cables would be buried where possible. The cable burial range would be between 0.5m and 3.0m below the seabed (with a target depth of 1.5m where ground conditions allow (recognised industry good practice which would reduce effects of electromagnetic fields (EMF))). A detailed CBRA would also be required to confirm the extent to which cable burial can be achieved. Where it is not reasonably practicable to achieve cable burial, additional cable |

| Parameter | Mitigation measures embedded into the design of the Project |
|---|---|
| | protection may be required. Following industry best-practice the Applicant would seek to minimise the use of cable protection.<br>Cables would be specified to reduce EMF and thermal emissions as per industry standards and best practice, such as the relevant IEC (International Electrotechnical Commission) specifications.<br>To minimise the extent of any unnecessary habitat disturbance, material displaced as a result of cable burial activities would be back-filled, where practicable, in order to promote recovery. |
| Foundations | The selection of appropriate foundation designs and sizes at each WTG and OSP location would be made following pre-construction surveys within the windfarm site. |
| | For piled foundation types, such as monopiles and jackets with pin piles, pile-driving would be used in preference to drilling, where it is practicable to do so (i.e. where ground conditions allow). This would minimise the quantity of sub-surface sediment released into the water column from the installation process. |
| Construction hours | During construction, overnight working practices would be employed offshore so that construction activities could continue 24/7, thereby reducing the overall programme for offshore works and the period in which potential construction related impacts may occur. |
| Biosecurity | Implementation of biosecurity measures in line with international and national regulations and guidance, namely:<br>▪ International Convention for the Prevention of Pollution from Ships (MARPOL), which sets out the requirements for appropriate vessel maintenance<br>▪ The Environmental Damage (Prevention and Remediation) (England) Regulations 2015, which set out a 'polluter pays' principle whereby operators who cause a risk of significant damage to water and biodiversity receptors are responsible for i) preventing damage from occurring; and ii) bearing the costs for full reinstation of the environment (to original condition) in the event of damage occurring<br>▪ The International Convention for the Control and Management of Ships' Ballast Water and Sediments (BWM Convention), which provides an international framework for the control of transfer of potentially invasive species from ballast water<br>These would be listed within the Project Environmental Management Plan (PEMP), an Outline of which is provided as part of the DCO Application (Document Reference 6.2). |
| Decommissioning | An Offshore Decommissioning Programme would be developed post-consent and implemented at the time of decommissioning. |

### 6.3.2 Realistic worst-case scenario

94. The final design of the Project would be confirmed through detailed engineering design studies that would be undertaken post-consent to enable the commencement of construction. To provide a precautionary but robust impact assessment at this stage of the development process, realistic worst-case scenarios have been defined. The realistic worst-case scenario (having the most impact) for each individual impact has been derived from the PDE to ensure that all other design scenarios would have less or the same impact. Further details have been provided in **Chapter 6 EIA Methodology** of the ES. This approach has been common practice for developments of this nature, as set out in PINS Advice Note Nine: Rochdale Envelope (PINS, 2018).

95. The realistic worst-case scenario is presented in **Table 6.3**.

*Table 6.3 Realistic worst-case scenarios for Annex I habitat features of marine SACs*

| Impact | Worst-case scenario | Notes and rationale |
|---|---|---|
| **Construction phase** | | |
| Increased SSCs and subsequent deposition<br><br>Remobilisation of contaminants | **Sediment displaced during seabed preparation for WTGs and OSPs foundations:**<br><br>▪ 35 WTGs with GBS foundations = 455,438$m^3$<br>▪ Two OSPs with GBS foundations = 26,025$m^3$<br><br>Total = **481,463$m^3$** | Seabed preparation (e.g. excavation using a trailing suction hopper dredger (TSHD) or other specialist bed leveller/trencher such as mass flow excavation) may be required. This is a volume of sediment that is disturbed prior to installation of WTG/OSP foundation and involves the removal of sediment from the seabed. The worst-case scenario assumes that sediment would be removed and returned to the water column at the sea surface (e.g. during disposal from a dredger vessel[11]) for WTGs and OSP(s).<br>Given the seabed preparation is the same per foundation for smaller and larger WTGs, the worst-case assumes 35 x smaller WTGs with GBS foundations. GBS foundations are assumed to have a diameter of 65m + 10m disturbance either side. The seabed preparation area would be dredged to a depth of up to 1.5m.<br>The worst-case scenario is for two jack-up visits per WTG/OSP foundation in different positions over the construction period (each jack-up with 6 legs, each with a 250$m^2$ footprint). This equates to a total footprint of 1,500$m^2$ per jack- |

[11] It is possible that seabed preparation would be undertaken by plough and sediment would therefore not be released at the surface, however disposal at the surface has been retained for the worst-case scenario.

| Impact | Worst-case scenario | Notes and rationale |
|---|---|---|
| | | up vessel visit and 3,000m$^2$ over the construction period per WTG/OSP foundation.<br>Drill arisings from drive-drill-drive methodology would result in a lower volume of sediment being disturbed (55,865m$^3$ – based on monopile foundations). |
| | **Sediment displaced during sandwave clearance/levelling for cables:**<br>▪ Inter-array cables = 70,000m$^3$<br>▪ Platform link cables = 10,000m$^3$<br>Total = **80,000m$^3$**<br>**Sediment displaced during cable installation:**<br>▪ Inter-array cables = 472,500m$^3$<br>▪ Platform link cables = 67,500m$^3$<br>Total = **540,000m$^3$** | The worst-case length of inter-array cables is 70km and platform link cables is 10km.<br>The worst-case assumes that 10% of the length of inter-array and platform link cables would require sandwave clearance/levelling. A clearance width of 10m and height of 1m is used. The worst-case assumes sediment would be released at the water surface.<br>The worst-case for cable installation assumes that 50% of inter-array and platform link cables are buried at 3m and 50% length is buried at 1.5m by jetting in a box-shaped trench, with a 3m trench width. |
| | **Cumulative volume of sediment disturbed: 1,101,463m$^3$ (approximately 1.1km$^3$)** | |
| Introduction and colonisation of INNS;<br>Spills and leakages | Maximum number of return trips for vessels per year: **2,583**<br>Maximum number of vessels on site at any time: **37** | The risk of introducing INNS during construction primarily relates to vessel activities, should vessels come from other marine bioregions.<br>The worst-case represents the maximum number of vessels, and it is noted that not all |

| Impact | Worst-case scenario | Notes and rationale |
|---|---|---|
| | | vessels would come from other bioregions and once on site would remain for a period of time. |
| **Operation and maintenance phase** | | |
| Temporary increases in SSCs /sedimentation during operational and maintenance activities (as well as disruption to sedimentary pathways);<br>Remobilisation of contaminants | **Sediment displaced during cable repair/replacement and reburial every year:**<br>▪ Average cable repair or replacement sediment volume = 6,000m$^3$<br>▪ Average cable reburial sediment volume = 3,000m$^3$<br>Total disturbed per year (on average) = **9,000m$^3$**<br>Total over operational period = **315,000m$^3$** | Temporary increases in SSCs could result from periodic jack-up vessel deployment, and cable repair, replacement and reburial activities.<br>The worst-case for cable repair/replacement assumed on average 200m of cable repaired/replaced every year with a 10m disturbance width. Cable reburial assumed on average 100m of cable reburied every year with a 10m disturbance width.<br>The worst-case for sediment volume disturbed assumed both cable repair/replacements and reburial would have a 3m maximum depth for a box-shaped trench.<br>It is noted that the total volume over the 35-year operational period is based on yearly averages and thus assesses for example that there may be no cable repair in one year and then longer lengths of cable repair/replacement and/or reburial in other years.<br>The volume of sediment that could be suspended due to the presence of jack-up vessels has not been calculated but would be a much smaller proportion compared to the quantity generated by construction and decommissioning activities.<br>The maximum area of introduced infrastructure that could cause blockages to sediment |

| Impact | Worst-case scenario | Notes and rationale |
|---|---|---|
| | | pathways has been included in the impact below. |
| Colonisation of infrastructure by INNS;<br>Spills and leakages | ▪ 35 x GBS WTGs with scour protection = 248,080$m^2$<br>▪ Two GBS OSPs with scour protection = 14,176$m^2$<br>▪ Inter-array cables = 91,000$m^2$<br>▪ Platform link cables = 13,000$m^2$<br>▪ Entry to WTGs and OSPs = 45,500$m^2$<br>▪ Inter-array cable/pipeline crossings (9) = 40,050$m^2$<br>▪ Platform link cable/pipeline crossings (6) = 26,700$m^2$<br>▪ Replacement scour protection = 13,950$m^2$<br>▪ Replacement cable protection including crossings and entries to WTGs/OSPs) = 21,625$m^2$<br><br>**Total subsurface infrastructure footprint: 514,081$^2$ (approximately 0.51$km^2$)**<br><br>Maximum number of operation vessels on site at any one time: **3 vessels** during a standard year, **10 vessels** during a heavy maintenance year<br><br>Maximum number of vessel return trips from Project windfarm site to port per year: **384 vessels** during a standard year, **832 vessels** during a heavy maintenance year | The risk of introducing INNS during construction would be primarily related to vessel activities, should vessels come from other marine bioregions. The presence of introduced hard substrate has the potential to encourage colonisation of invasive epifaunal species. |

| Impact | Worst-case scenario | Notes and rationale |
|---|---|---|
| **Decommissioning phase** | | |
| Increases in SSCs and subsequent deposition;<br>Remobilisation of contaminants<br><br>Introduction and colonisation of INNS;<br>Spills and leakages | The decommissioning policy for the Project infrastructure is not yet defined however it is anticipated that structures above the seabed would be removed.<br>The following infrastructure is likely be removed, reused, or recycled where practicable:<br>▪ WTGs and foundations<br>▪ OSP(s) including topsides and foundations<br>The following infrastructure is likely to be decommissioned and could be left *in situ* depending on available information at the time of decommissioning:<br>▪ Inter-array and platform link cables<br>▪ Scour protection<br>▪ Crossings and cable protection<br>Part of the foundations (e.g. some foundation material below the seabed may be left *in situ*) | The detail and scope of the decommissioning works would be determined by the relevant legislation and guidance at the time.<br>Decommissioning arrangements would be detailed in a Decommissioning Programme, which would be drawn up and agreed with the relevant authority, prior to decommissioning.<br>For the purposes of the worst-case scenario, it has been anticipated that the impacts would be comparable to those identified for the construction phase. |

# 6.4 Shell Flat and Lune Deep SAC

## 6.4.1 Description of designation

97. Shell Flat and Lune Deep SAC is located c. 9.5km east of the windfarm site, at its nearest point. The SAC was designated for the following Annex I habitats:

- Sandbanks which are slightly covered by seawater all the time (Shell Flat)[12]

98. The conservation objectives for the SAC are to ensure that, subject to natural change, the integrity of the site is maintained or restored as appropriate, and that the site contributes to achieving the Favourable Conservation Status (FCS) of its Qualifying Features, by maintaining or restoring the:

- Extent and distribution of qualifying natural habitats and habitats of the qualifying species
- Structure and function (including typical species) of qualifying natural habitats
- Structure and function of the habitats of the qualifying species
- Supporting processes on which qualifying natural habitats and the habitats of qualifying species rely
- Populations of each of the qualifying species
- Distribution of qualifying species within the site

### 6.4.1.1 Shell Flat sandbank

99. The Shell Flat sandbank is considered to be an excellent example of the Annex I habitat Sandbanks which are slightly covered by seawater all the time (Joint Nature Conservation committee (JNCC), 2022a). The sandbank runs north east from the southern corner of the site boundary in a crescent to the south west, and forms a continuous structure approximately 15km long from east to west. The area of sandbank habitat within the SAC is 89km², equivalent to 0.52% of the UK total resource, however, it should be noted that the bank extends beyond the SAC boundaries (Natural England, 2021a). The sandbank is an example of a banner bank, which are generally only a few kilometres in

[12] The site is also designed for reefs but have been screened out given the distance of the reef features (within Lune Deep) from the Project windfarm site (as described in the HRA Screening Report (Morecambe Offshore Windfarm Ltd, 2023a; Document Reference 4.1)).

length with an elongated pear-shaped form, located in water depths less than 20m below Chart Datum (CD).

100. The bank is comprised primarily of mud and sand sediments, some silts and clays and areas of coarse sands, and is characterised by its low biodiversity and high biomass (JNCC, 2022a). This makes the bank an important foraging ground for over-wintering birds in the IS, with 50,000+ common scoter (*Melanitta nigra*) feeding on the bank each winter (Kaiser *et al*., 2006). The species found inhabiting the bank are typical of those found in sandy substrates and include the bivalve molluscs *Nucula nitidosa, Pharus legumen, Abra alba* and *Fabulina fabula*, as well as the bristle worms *Magelona johnstoni, Glycera alba* and *Magelona filiformis* (Natural England, 2021a).

101. The sub features present within the sandbank are:

- Subtidal mud - only a minor component of the Shell Flat sandbank; a small area in the southern part of the bank is classified as the European Nature Information System (EUNIS) habitat circalittoral sandy mud (A5.35)
- Subtidal sand - Subtidal sand is the main component of the Shell Flat sandbank. The sediment is sandier in the shallow central area and muddier in the deeper areas. Shell Flat is composed of the *Fabulina fabula* and *Magelona mirabilis* biotope (A5.242) in the fine shallower sediments of the bank, with *Abra alba* and *Nucula nitidosa* biotope (A5.261) occurring in the slightly muddier sediments found on the slopes and in deeper areas of the bank

102. The feature condition assessment of the Shell Flat sandbank reported in 2024 identifies the features in a favourable condition (Natural England, 2024), with all primary targets met. One secondary attribute failed which is related to contaminants and water quality, where the target is to '*reduce aqueous contaminants to levels equating to High Status according to Annex VIII and Good Status according to Annex X of the Water Framework Directive (WFD), avoiding deterioration from existing levels.'* Due to the WFD stringent measures, 100% of all waterbodies failed WFD chemical status in the 2019 classification due to measured/assumed elevated levels of polybrominated diphenyl ether (PBDE) and mercury and its compounds. However, this is a secondary attribute and is not assessed to be adversely affecting this feature (Natural England, 2024).

## 6.4.2 Assessment

### 6.4.2.1 Assessment of potential effects from the Project-alone

#### Impact 1: Increased SSCs and deposition

**Construction phase**

103. During construction activities, there may be a temporary and episodic (limited to the period for each seabed installation activity within the 2.5-year construction phase) increases in SSCs and subsequent re-deposition of disturbed sediment. Increases in SSCs have the potential to affect benthic ecology receptors by blocking feeding apparatus as well as by smothering sessile species upon sediment redeposition.

104. A conceptual evidence-based assessment of the extent and magnitude of increases in SSCs and seabed level changes as result of deposition has been detailed in **Chapter 7 Marine Geology, Oceanography and Physical Processes** of the ES. The same chapter also describes how the outcomes of that conceptual assessment have been supported by a modelled assessment undertaken for the Mona and Morgan Offshore Wind Projects and Awel y Môr (AyM) Offshore Wind Farm. The outcomes of the assessment have been summarised below.

105. Disturbance activities, such as excavation during seabed preparation to create a suitable base for WTG and OSP foundations, and the installation of inter-array and platform link cables, would result in a modest concentration plume advected to a distance of up to 1km along the tidal axis. Beyond this distance any increases in SSCs would become low and indistinguishable from background levels. Coarser (i.e. sand) components of the sediment would fall out of suspension rapidly, forming a mound local only to the release point. Given the sediment in the Project windfarm site has been principally recorded as composed of sand with low mud content, this would not represent a significant alteration in seabed composition. Deposition levels would decrease rapidly with distance from the release point and sediment transport and deposition of finer (i.e. mud) material would occur at a maximum distance of a tidal spring excursion (approximately 10km). This has been based on analysis of ABPmer tidal ellipse data which identified a spring tidal excursion of approximately 10km in an east-west orientation at the windfarm site. Beyond this area there would be, at most, very minor bed level change (a matter of millimetres).

106. Other relatively minor seabed disturbances, namely those from deployment of jack-up vessels/anchors and placement of scour protection and cable protection onto the seabed, would not be expected to cause an increase in

SSCs /deposition to the extent that there would be a discernible impact to benthic ecology receptors beyond the immediate vicinity of the windfarm site.

107. Natural England's 2022 'Advice on Operations' provides advice regarding the installation of turbine foundations and power cables in relation to the sensitivity of the site's designated features (Natural England, 2022a).

108. The sensitivity of the Shell Flat habitats (sublittoral sand and mud) was considered to be 'low' to 'not sensitive' to light deposition (up to 5cm). As discussed above, the deposition from the Project at the range of the SAC was likely to be in the order of, at most, millimetres only.

109. The sensitivity of the Shell Flat habitats to changes in suspended sediment solids (water clarity) was considered to be 'low' to 'not sensitive'. Given the distance of the SAC from installation activities for the Project (c. 9.5km at its closest point), increases in SSCs resulting from installation activities would be indistinguishable from background levels within the SAC.

110. The Project-alone had no AEoI on the Shell Flat and Lune Deep SAC from increased SSCs and deposition during construction.

111. It has been noted that changes to sediment pathways and supporting processes could be influenced by seabed level changes, for example, as a result of seabed preparation. Changes to the physical processes supplying and maintaining sediments, associated with changes to tides and currents from the physical presence of Project infrastructure within the windfarm site, have been assessed below for the operation and maintenance phase.

**Operation and maintenance phase**

112. During the operation and maintenance phase, periodic maintenance activities may include repair to subsea cables and/or foundations which could require limited disturbance of the seabed. During such maintenance activities, small volumes of sediment could be re-suspended. The volumes of sediment disturbed would be lower than those during construction-phase seabed preparation and cable burial works.

113. Sediment disturbance as a result of operation and maintenance phase activities would be expected to cause localised and short-term increases in SSCs at the location of works. Released sediment may then be transported by tidal currents in suspension in the water column before being redeposited back on to the seabed.

114. Benthic biotopes associated with the Annex I sandbank feature have low sensitivity to light deposition and changes to water clarity (Natural England, 2022a). Furthermore, the distance of the SAC from the source of sediment disturbance means that SSCs increases would be indistinguishable from the background and deposition would be, at most, in the order of millimetres.

115. In addition to operation-phase disturbance activities, the potential exists for changes to occur in sediment transport pathways at long distances from the Project windfarm site due to the presence of windfarm infrastructure (i.e. foundations and cable protection).

116. Tidal currents are the main driving force of sediment transport and, as a result, move sediments in an easterly direction. The assessment in **Chapter 7 Marine Geology, Oceanography and Physical Processes** of the ES concluded that during the operation and maintenance phase there would be no significant changes to the broad-scale flow regime or sediment transport pathways from the Project windfarm site and as such no effects to the sediment supply to the Shell Flat and Lune Deep SAC.

117. Benthic biotopes associated with the Annex I sandbank habitat ranged from 'not sensitive' to 'highly sensitive' to changes in water flows (Natural England, 2022a). The benchmark for flow velocity is 0.1m/s to 0.2m/s for more than one year. However, **Chapter 7 Marine Geology, Oceanography and Physical Processes** of the ES predicted that flow speeds would not be affected outside the windfarm site, with changes outside the wake of turbines to be less than ±0.01m/s. The wake signature would dissipate and recover with distance downstream, becoming indistinguishable from baseline conditions within tens to a few hundreds of metres.

118. The Project-alone has no AEoI on the Shell Flat and Lune Deep SAC from increased SSCs and deposition during operation and maintenance. The confidence in the assessment was high and aligned with the detailed assessment presented in **Chapter 9 Benthic Ecology** of the ES.

**Decommissioning phase**

119. Increases in SSCs and sediment deposition from the decommissioning works may arise during the removal of infrastructure and disturbance of seabed from jack-up vessels and anchored vessels. However, the magnitude of any effect is likely to be lower than for construction as, for example, seabed preparation would not be required and cables may be left *in situ*. As a worst-case, the effects of decommissioning activities were considered to be as per the conclusions of the construction-phase assessment.

120. The Project-alone had no AEoI on the Shell Flat and Lune Deep SAC from increased SSCs and deposition during decommissioning. The confidence in the assessment was high and aligned with the detailed assessment presented in **Chapter 9 Benthic Ecology** of the ES.

### Impact 2: Remobilisation of contaminated sediments (all phases)

121. Increases in SSCs and sediment deposition could lead to the remobilisation of contaminated sediments and effects upon the SAC.

122. Grab sampling was undertaken for chemical analysis during benthic characterisation surveys of the Project windfarm site conducted in 2022, as presented in **Chapter 8 Marine Sediment and Water Quality** of the ES. The level of contaminant concentrations within the sediment samples was established through comparison with recognised guidelines and action levels. Cefas Action Levels (ALs) have been widely used for assessing contamination risk in UK marine developments and are available for a range of contaminants. The US Environmental Protection Agency's ERL are quality guidelines used by the Oslo-Paris Convention (OSPAR) and have been defined as the lower tenth percentile of the dataset of concentrations in sediments which were associated with biological effects. If concentrations within the sampled sediment generally did not exceed the lower threshold values (i.e. AL 1 and ERL), then contamination levels were not considered to be of significant concern and were deemed low risk in terms of potential impacts on marine benthic communities.

123. The survey results demonstrated that no samples exceeded either Cefas AL 1 or ERLs, hence the risk of biological effects arising from disturbance of the sediment is low. As contaminant levels were not found to be present at levels whereby effects would arise, this impact (remobilisation of contaminated sediments) was therefore scoped out of the assessment for all phases. The scoping out of this impact has been agreed by Natural England and the MMO (confirmed by email on 28$^{th}$ September 2023) (**Table 6.1**).

### Impact 3: Introduction and spread of INNS

#### Construction phase

124. Should INNS become established within a new habitat they can out-compete native species for space and resources, or may prey on native species, or introduce new pathogens (Roy *et al*., 2012). As such, the introduction and/or spread of INNS during the construction phase could potentially lead to changes in the ecological functionality of benthic communities.

125. As a growing consideration for offshore marine developments in the UK, the primary pathway for the potential introduction of INNS would be from the use of vessels and infrastructure that originated from outside the IS and Northeast Atlantic region, particularly from regions that are ecologically distinct from the Eastern IS. Ship ballast water appears to be the largest single vector for INNS, and bio-fouling communities on ships is also a contributor (Glasby *et al.,* 2007). The pathway for introduction of INNS would be greatest during the construction phase (due to the regularity and volume of construction-related vessel movements). An anticipated 2,583 vessel round trips were expected per year during the construction phase.

126. The embedded measures in place, as set out in **Table 6.2**, would be equally appropriate for minimising the risk of INNS transfer from vessels sourced both locally and from other parts of the UK or further field. With such measures in place, the risk of introduction of INNS from vessel activity would be reduced to as low as reasonably practicable. As such, there was no significant risk to Annex I habitat within the SAC.

127. The Project-alone had no AEoI on the Shell Flat and Lune Deep SAC from the introduction and spread of INNS during construction. The confidence in the assessment was high and aligned with the assessment presented in **Chapter 9 Benthic Ecology** of the ES.

**Operation and maintenance phase**

128. There is a risk that artificial hard substrates including foundations, scour protection and cable protection could act as potential 'stepping stones' or vectors for INNS, thereby facilitating the spread of such species. In total, an area of up to 0.4km$^2$ of new hard substrate may be introduced for the Project.

129. As per the construction phase, the primary pathway for the potential introduction of INNS is from the use of vessels, particularly those that have originated from outside the region. An anticipated 384 round trips between the Project windfarm site and port would be undertaken during a standard year, or 832 round trips during a 'heavy maintenance' year during the operation and maintenance phase of the Project. However, these trips would be largely expected to originate from a port within 50km of the Project windfarm site.

130. The measures set out for the construction phase to control risk of INNS introduction and spread would apply also during the operation and maintenance phase. With such measures in place, the risk of introduction of INNS would be reduced to as low as reasonably practicable. As such, there would be no long-term or significant risk to Annex I habitat in the SAC.

131. Monitoring of INNS colonisation of the Project structures would be taken into consideration when developing post-construction inspection surveys of the hard substrate. Data from monitoring would allow the effects of potential colonisation to be gauged and further control measures put in place, where necessary.

132. The Project-alone would have no AEoI on the Shell Flat and Lune Deep SAC from the introduction and spread of INNS during operation and maintenance. The confidence in the assessment was high and aligned with the assessment presented in **Chapter 9 Benthic Ecology** of the ES.

### Decommissioning phase

133. As with the construction phase, the risk of introduction and/or spread of INNS during the decommissioning phase would primarily be attributed to the use of vessels that originate from outside the region.

134. Quantification of vessel movement during the decommissioning phase is not possible at this stage given that i) vessel capacity/capability may evolve during the lifetime of the Project; and ii) it was unclear at this stage exactly what assets may be left *in situ*. As a worst-case scenario, it has been assumed that all assets would be removed, in which case vessel use is likely to be similar to that predicted for the construction phase (although ground preparation would not be required for example). As with the construction phase, mandated and best-practice biosecurity measures would be implemented; these may be similar to those set out in **Table 6.2**, although the most up-to-date guidance/best-practice available at the time of decommissioning would be considered. As such, there would be no significant risk to benthos either within the Project windfarm site or further afield.

135. The Project-alone would have no AEoI on the Shell Flat and Lune Deep SAC from the introduction and spread of INNS during decommissioning. The confidence in the assessment was high and aligned with the assessment presented in **Chapter 9 Benthic Ecology** of the ES.

### Impact 4: Risk of deterioration of water quality due to spillages/leakages (all phases)

136. Following embedded mitigation included in **Section 6.3.1** and adhering to best practices would reduce this risk as low as reasonably practicable. As such, there would be no significant risk to benthos either within the Project windfarm site or further afield from spillages/leaks.

137. The Project-alone would have no AEoI on the Shell Flat and Lune Deep SAC from the deterioration of water quality due to spillages/leakages during construction, operation and maintenance or decommissioning. The confidence in the assessment was high and aligned with the assessment presented in **Chapter 9 Benthic Ecology** of the ES.

### 6.4.2.2 Potential interactions of Project effects

138. The effects identified and assessed in this section have the potential to interact with each other. The effects of the Project were:

- Increased SSCs and deposition
- Risk of deterioration of water quality due to spillages/leakages

139. The introduction and spread of INNS were not related to any of the other effects, and there was no additional effect from interactions to assess.

140. It has been considered that there would be no greater risk or magnitude of effect from interaction of these effects due to i) the low risk of deterioration of water quality due to spillages/leakages; and ii) the large distance between SAC and source of sediment disturbance (which meant that SSCs increases would be indistinguishable from background levels within the SAC).

#### 6.4.2.3 Project-alone conclusions

141. Considering the assessment against the conservation objectives, **Section 6.4.1**, the Project would have no AEoI on the Shell Flat and Lune Deep SAC. This was largely due to the magnitude of effects, given the separation of the Project to the SAC. The confidence in the assessment was high and aligned with the assessment presented in **Chapter 9 Benthic Ecology** of the ES.

#### 6.4.2.4 In-combination assessment – the Project and Transmission Assets combined

142. A 'combined' assessment has been made with the Transmission Assets[13], for the purpose of an in-combination assessment considering its functional link with the Project.

143. This assessment refers to Shell Flat and Lune Deep SAC, which was the only SAC (for benthic features) screened in for both the Project and the Transmission Assets. For both projects, no AEoI has been concluded.

144. The predicted combined volume of material likely to be disturbed during the construction phase of the Project and the Transmission Assets would be in the region of 13.4 million $m^3$. This includes approximately 1.1 million $m^3$ associated with the Project (see **Table 6.3**) plus c.12.3 million $m^3$ associated with the Transmission Assets (Morgan Offshore Wind Limited and Morecambe Offshore Windfarm Ltd, 2023a).

145. As described in **Section 6.4.2.1**, 'heavy' deposition would only occur within a very short distance of the source of disturbance; at more than 1km distance SSCs increases and deposition levels would be low. As such, areas of interaction between plumes from the Project and Transmission Assets within the SAC may only experience, at most, 'light' deposition (in the order of millimetres). The sensitivity of biotopes associated with Annex I sandbank habitat to 'light' deposition was 'low'.

[13] As the Transmission Assets includes infrastructure associated with both the Project and the Morgan Offshore Wind Project Generation Assets, it should be noted that the combined assessment considers the transmission infrastructure for both the Project and the Morgan Offshore Wind Project Generation Assets.

146. Given the relationship of the Project and the Transmission Assets, site preparation and installation of infrastructure would be phased and SSC increases would be unlikely to occur concurrently. However, should multiple operations be undertaken simultaneously, plumes would be advected on the tide (not towards one another). Activities would be of limited spatial extent and plume interactions would be of a low magnitude and short duration. For both projects, the majority of sedimentation would occur within close proximity of each installation activity; however, given the active sediment transport regime, deposited material would be redistributed across the vicinity. Given the distance of the SAC from both projects the magnitude of any effect would be limited.

147. Both projects would adopt INNS and pollution measures and as such no In-combination effects have been identified.

148. Potential for changes to occur in sediment transport pathways have been identified due to the presence of windfarm infrastructure (i.e. foundations and cable protection) and cable protection for the Transmission Assets. There may be local changes to these processes in the vicinity of cable protection, however, the Shell Flat and Lune Deep SAC is beyond the range of potential changes to the tidal current, wave and sediment transport regimes as a result of the Project. The additive impacts to sediment transport from the Generation Project and the Transmission Assets would not significantly impact sediment transport pathways moving across the IS to the coast and as such no effects to the sediment supply to the Shell Flat and Lune Deep SAC were anticipated.

149. Given the distance of the SAC to the projects any additive effects would be minor and any interaction of sediment plumes and deposition would be localised (i.e. of small spatial extent) and temporary. There would be no adverse in-combination effect on the integrity of the Shell Flat and Lune Deep SAC.

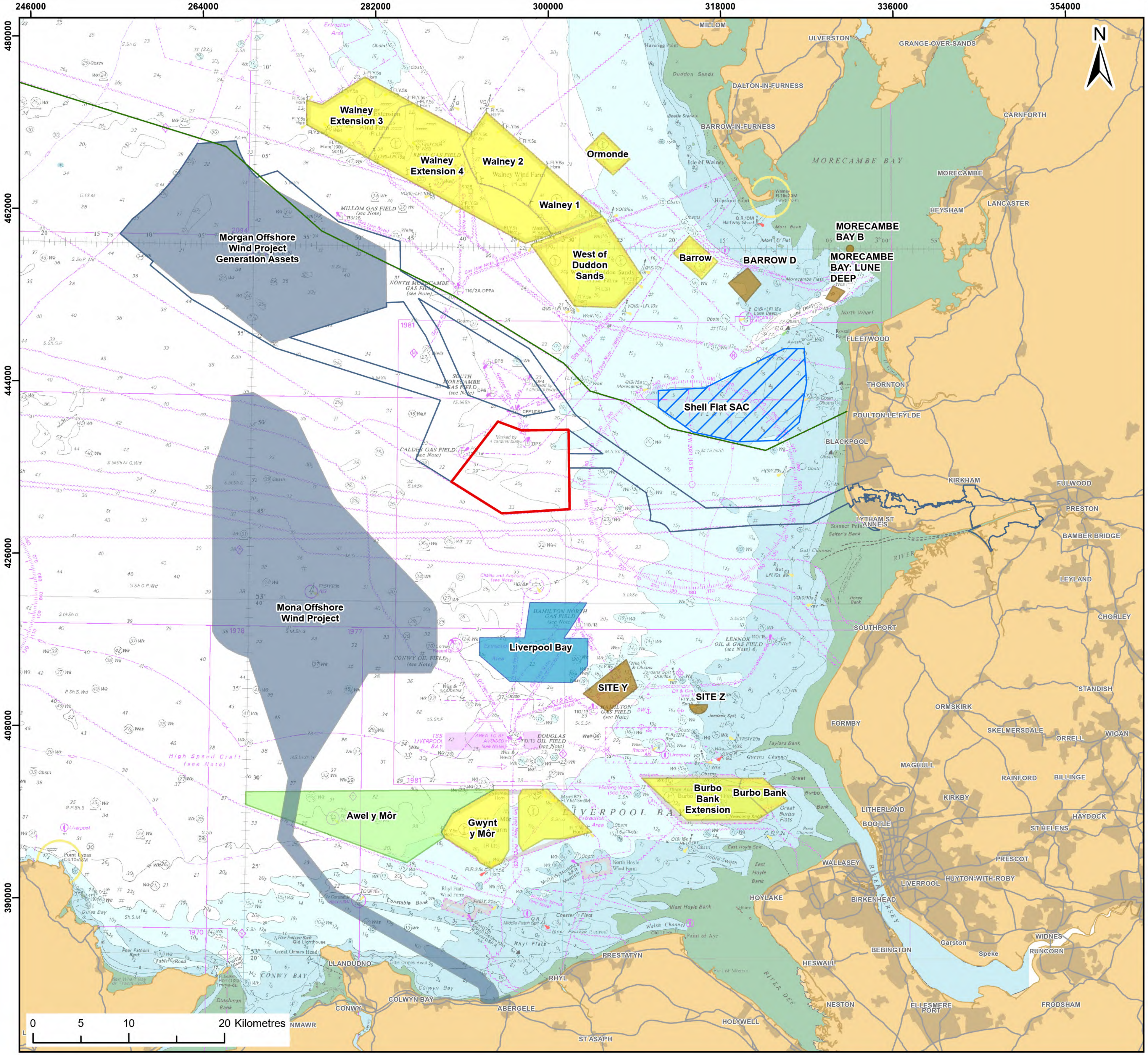
#### 6.4.2.5 In-combination assessment – Other plans and projects

150. The projects in **Table 6.4** and **Figure 6.1** have been identified as having the potential to cause in-combination effects, given there could be an overlap of the ZoI with the SAC or cause incremental effects in the region. Further information of the Project screening has been provided in the HRA Screening Report (Morecambe Offshore Windfarm Ltd, 2023a; Document Reference 4.10).

*Table 6.4 Projects identified as having the potential to cause in-combination effects at Shell Flat and Lune Deep SAC*

| Project/plan | Distance from windfarm site (km) | Distance from Shell Flat and Lune Deep SAC (km) | Description |
|---|---|---|---|
| Morgan and Morecambe Offshore Wind Farms: Transmission Assets | 0 (Adjacent) | 5.7 | Increases in SSCs and presence of physical infrastructure |
| Isle of Man Interconnector (cable protection remedial works) | 4.6 | 0.2 | Increases in SSCs and presence of physical infrastructure |
| Morgan Offshore Wind Project Generation Assets | 16.7 | 29.6 | Increases in SSCs and presence of physical infrastructure |
| Mona Offshore Wind Project | 10.0 | 31.1 | Increases in SSCs and presence of physical infrastructure |
| West of Duddon Sands Offshore Windfarm (maintenance activities) | 12.9 | 9.7 | Increases in SSCs |
| Walney 1,2 and extension Offshore Wind Farms (OWF) (maintenance activities) | 18.8 + | 19.0 + | Increases in SSCs |
| Barrow OWF (maintenance activities) | 21.0 | 7.3 | Increases in SSCs |
| Ormonde OWF (maintenance activities) | 27.0 | 21.5 | Increases in SSCs |
| Gwynt y Môr OWF (maintenance activities) | 28.9 | 40.3 | Increases in SSCs |
| Burbo Bank Extension OWF (maintenance activities) | 29.1 | 35.2 | Increases in SSCs |
| AyM OWF | 28.9 | 42.8 | Increases in SSCs and presence of physical infrastructure |
| Liverpool Bay Aggregate Production Area | 9.5 | 21.0 | Increases in SSCs |
| Disposal sites Y and Z | Site Y: 16.8<br>Site Z: 24.0 | Site Y: 24.8<br>Site Z: 27.7 | Increases in SSCs |
| Barrow D disposal site | 22.7 | 2.5 | Increases in SSCs |
| Morecambe Bay B disposal site | 34.6 | 5.4 | Increases in SSCs |

| Project/plan | Distance from windfarm site (km) | Distance from Shell Flat and Lune Deep SAC (km) | Description |
|---|---|---|---|
| Morecambe Bay Lune Deep disposal site | 30.1 | 0.6 | Increases in SSCs |





Legend:

- Morecambe Offshore Windfarm Site
- Morgan and Morecambe Offshore Wind Farms: Transmission Assets (In Planning)
- Special Areas of Conservation (SAC) screened in for Annex I features
- Isle of Man Interconnector

**Disposal Sites Status**

- Open

**Minerals & Aggregates Site Agreements**

- Production Agreement Area

**Windfarm status**

- Fully commissioned
- Consented
- In Planning



Report:
Morecambe Offshore Windfarm: Generation Assets
Habitat Regulations Report to Inform Appropriate Assessment

Title:
Projects with the potential to cause in combination effects on Annex I features

Figure: 6.1 | Drawing No: PC1165-RHD-ES-OF-DR-Z-0065

| Revision: | Date: | Drawn: | Checked: | Size: | Scale: |
|---|---|---|---|---|---|
| P02 | 15/01/2024 | JH | AS | A3 | 1:400,000 |
| P03 | 09/04/2024 | JH | AS | A3 | 1:400,000 |

Co-ordinate system: WGS 1984 UTM Zone 30N

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### In-combination impact 1: Increased SSCs and deposition

151. There is potential for activities at other developments/projects to result in sediment disturbance leading to advection of sediment plumes, in addition to those that may arise during the Project's construction/operation and maintenance/decommissioning phases. Where sediment plumes interact, there is likely to be a corresponding increase in SSCs (and consequent deposition) at that location exceeding that from an individual project. Should such interaction occur within the boundaries of the SACs, there would be potential for in-combination effects.

152. As discussed in **Chapter 9 Benthic Ecology** of the ES and based on a conceptual evidence-based assessment supported by modelling for AyM Offshore Windfarm and Morgan and Mona Offshore Wind Projects (set out in **Chapter 7 Marine Geology, Oceanography and Physical Processes** of the ES), increases in seabed level at any stage of the Project would be temporary (i.e. deposited fines would be redistributed within a short period of time by hydrodynamic actions) and very localised. At a distance of more than 1km from the point of release, impacts would be of negligible magnitude (in the order of millimetres). The ZoI for the distribution of fine sediments extended to a maximum of 10km, however SSCs would be at background levels.

153. In-combination effects could only realistically occur in the instance that sediment-disturbing activities were taking place at the Project and other developments simultaneously, and sediment plumes from other developments encroached into the 'near field' area (i.e. within 1km) of the Project's activities. Beyond this distance, SSCs would be indistinguishable from background levels. Given that Shell Flat and Lune Deep SAC would be c. 9.5km from the Project windfarm site there would be no risk of in-combination impacts affecting benthic features.

154. Effects assessed during the construction phase would apply during the operation and maintenance phase, given that activities during the operation and maintenance phase would be small, discrete works to specific parts of the site, rather than a site-wide impact.

155. During the decommissioning phase of the Project, it has been predicted that the magnitude and extent of increases in SSCs would be similar to, or less than, those during the construction phase, hence there would similarly be no in-combination effects.

156. The Project in-combination with other projects would have no AEoI on the Shell Flat and Lune Deep SAC during construction, operation and maintenance, or decommissioning.

### In-combination impact 2: Introduction and spread of INNS (all phases)

157. Biosecurity measures would be put in place to prevent the introduction of INNS from the Project. Other plans and projects would also follow best practice guidelines and mitigation measures to reduce the spread of INNS and, as such, this risk was deemed to be as low as reasonably practicable.

158. There would be no adverse in-combination effect on the integrity of the Shell Flat and Lune Deep SAC from introduction and spread of INNS in any phase of the Project.

### In-combination impact 3: Risk of deterioration of water quality due to spillages/leakages

159. Through all phases of the Project, adherence to guidelines and application of best practice measures would result in the risk of spillages/leakages being as low as reasonably possible. As such, this would minimise the risk of in-combination effects should water quality issues arise from other projects. There would be no AEoI on the Shell Flat and Lune Deep SAC.

### In-combination conclusions

160. There would be no adverse effects on the integrity of the Shell Flat and Lune Deep SAC as a result of the Project in-combination with other projects and plans (including the associated Transmission Assets), during the construction, operation and maintenance, or decommissioning phases. The confidence in the assessment was high and aligned with the assessment presented in **Chapter 9 Benthic Ecology** of the ES.

## 6.4.3 Summary

161. There would be no adverse effects on the integrity of the Shell Flat and Lune Deep SAC as a result of the Project, either alone or in-combination with other projects and plans (including the associated Transmission Assets), during the construction, operation and maintenance, or decommissioning phases

# 7 Offshore Annex II sites designated for fish

## 7.1 Approach to assessment

162. This section provides information in order to determine the potential for the Project to have an AEoI on sites designated for Annex II fish species.

163. For each site designated for fish species screened in for further assessment, the following has been provided:

- A summary of the ecology of the fish species relevant for each European site
- An assessment of the potential effects during the construction, operation and maintenance, and decommissioning, and assessment on whether the Project-alone could adversely affect the integrity of screened in European sites in view of their conservation objectives
- An assessment of the potential for in-combination effects alongside the Transmission Assets and assessment on whether the Project-alone or in-combination could adversely affect the integrity of screened in European sites in view of their conservation objectives
- An assessment of the potential for in-combination effects alongside other relevant developments and projects, including the Transmission Assets, and assessment on whether the Project-alone or in-combination could adversely affect the integrity of screened in European sites in view of their conservation objectives

## 7.2 Consultation

164. Consultation on fish ecology has been undertaken in line with the process set out in **Section 4.2**. The feedback received through the EPP has been considered in preparing the RIAA.

165. **Table 7.1** provides a summary of how the consultation responses received in relation to the HRA Screening Report and draft RIAA have influenced the approach that has been taken.

*Table 7.1 Consultation responses received in relation to fish and shellfish ecology*

| Consultee | Date/document | Comment | Project response/where addressed |
|---|---|---|---|
| MMO | 24th October 2022<br>Comments on: Morecambe Offshore Windfarm Generation Assets Marine Conservation Zone (MCZ) Screening Report and HRA Screening Report | The MMO note the Applicant has assigned fish according to the hearing groups described by Popper *et al.,* (2014) for the purpose of the assessment of underwater noise and vibration. However, there is no further information on how the hearing thresholds will be applied in the underwater noise modelling. Please note that the MMO recommend that all underwater modelling is based on a stationary rather than a fleeing receptor for fish, for the reasons outlined below:<br>i. The MMO know that fish will respond to loud noise and vibration, through observed reactions including schooling more closely; moving to the bottom of the water column; swimming away, and; burying in substrate (Popper *et al.,* 2014). However, this is not the same as fleeing, which would require a fish to flee directly away from the source over the distance shown in the modelling. We are not aware of scientific or empirical evidence to support the assumption that fish will flee in this manner.<br>ii. The assumption that a fish will flee from the source of noise is overly simplistic as it overlooks factors such as fish size and mobility, biological drivers, and philopatric behaviour which may cause an animal to remain/return to the area of impact. This is of particular relevance to herring, as they are benthic spawners which spawn in a specific location due to its substrate composition.<br>iii. Eggs and larvae have little to no mobility, which makes them vulnerable to barotrauma and developmental effects. Accordingly, they should also be assessed and modelled as a stationary | The Applicant's approach has been to conservatively treat shellfish, larvae, eggs and all fish as stationary receptors (**Table 7.4**) |

| Consultee | Date/document | Comment | Project response/where addressed |
|---|---|---|---|
| | | receptor, as per the Popper *et al.*, (2014) guidelines." | |
| Natural England | 14th September 2022<br>Advice on draft Screening Report Morecambe Offshore Windfarm – Generation Assets | We agree that a full assessment of the impacts from UXO clearance should be included in the Marine Licence application for UXO clearance. Nonetheless, the potential for UXO clearance (from this project and other projects nearby) to occur at the same time as other impact pathways from this project, and so act in-combination, should be considered. | UXO clearance from the Project and other projects have been considered in the in-combination assessment. The assessment for the Project is only indicative at this stage given a full assessment would be undertaken to support a separate marine licence application for UXO clearance. |
| PINS<br>(ref. 3.4.8) | 2nd August 2022<br>Scoping Opinion | Designated sites: The Scoping Report notes the presence of various designated sites within 30–45km of the windfarm site, but also notes the potential for migratory fish species associated with other designated sites to occur in the windfarm site. The ES should explain how the zone of influence for the Proposed Development has been defined and how this has led to the identification of designated sites which could be affected. | **Section 5.2** describes the screening process for sites related to Annex II fish species, and HRA Screening Report (Morecambe Offshore Windfarm Ltd, 2023a; Document Reference 4.10) lists all sites considered and the ZoI applied. Further information on the ZoI for each receptor group has been provided within the **Chapter 10 Fish and Shellfish Ecology** of the ES (Document Reference 5.1.10) and Marine Conservation Zone Stage 1 Assessment (Document Reference 4.13) supplied with the DCO Application. |
| MMO | 30th May 2023<br>Section 42 comments on the PEIR and draft RIAA | The MMO note that the report does not include the River Ehen SAC and River Eden SAC in Section 10.5.10. The rationale for this is due to both sites being located to the north of the project area, and that fish receptors are "recorded as travelling north when moving from rivers into the sea". At present, this | To clarify, it was only Atlantic salmon smolt that were recorded as travelling northwards in the Irish Sea as they left river systems from both Northern Irish and English Rivers, as outlined in Barry *et al*., (2020) and Green *et* |

| Consultee | Date/document | Comment | Project response/where addressed |
|---|---|---|---|
| | | statement is unsupported within the HRA report and the potential effects to diadromous fish travelling from the south has not been considered. Statements on the directional movements of migratory fishes must be supported with data or references to determine which receptors are screened in/out of further assessment.<br><br>This is particularly important as the River Ehen SAC is designated for Atlantic salmon (*Salmo salar*), which have medium-sensitivity to underwater noise (Popper *et al*., 2014). Similarly, the River Eden SAC is designated for brook lamprey (*Lampetra planeri*), river lamprey (*Lampetra fluviatilis*) and sea lamprey (*Petromyzon marinus*), which are benthic spawners and known to construct nests along riverbeds. As such, these receptors are vulnerable to underwater noise and vibration associated with pile driving activities. The MMO considers that the River Ehen SAC and River Eden SAC should not be scoped out of the HRA. | *al*., (2022). This was consistent with the fact that UK salmon were known to migrate to Norwegian feeding grounds (Malcolm *et al*., 2010). More recent evidence showed a strong preference for Irish Sea smolts to migrate in a north westerly direction, out of the Irish Sea to the North East Atlantic, after exiting their natal rivers (Lilly *et al*., 2023). This evidence has been presented in Section 10.5.8 of **Chapter 10 Fish and Shellfish Ecology** of the ES.<br>The River Eden SAC is located more than 50km away from the Project (straight line distance) and over 100km via sea to the estuary (through the Solway Firth) and is therefore beyond the ZoI for worst-case noise impacts to interfere with spawning lamprey species, which spawn on the riverbed, as noted by the MMO. The Applicant therefore considered there to be no potential for an LSE at the River Eden SAC via impacts to lamprey spawning on the riverbed. Lamprey species (outside of designated sites) have been assessed in the ES as a receptor (see Section 10.5.8 of **Chapter 10 Fish and Shellfish Ecology** of the ES (Document Reference 5.1.10)). |

| Consultee | Date/document | Comment | Project response/where addressed |
|---|---|---|---|
| | | | On a precautionary basis however the River Ehen and River Eden have been screened into the RIAA. |
| NWWT | 22nd May 2023<br>Section 42 comments on the PEIR and draft RIAA | Both species of shad have been omitted from the HRA despite presence in the region. | Response outlined as below. |
| Natural England | 2nd June 2023<br>Section 42 comments on the PEIR and draft RIAA | Both shad species (*Alosa alosa* and *Alosa fallax*) are omitted from the diadromous fish receptor group, despite being present in the region (non-spawning). Given the species is present in the region, either shad should be included within all assessments of impacts on diadromous fish, particularly underwater noise, or a justification for its exclusion provided. | Whilst shad are present in the region and noted to have non-significance presence at a number of SACs, there is no SAC designated for shad within 100km of the Project, thereby ruling out direct effects on these sites. The worst-case noise impact range for temporary behavioural disturbance (breaking up of schools before reforming) is less than 50km. Whilst adult non-spawning shad may be present at the site, there was no way to apportion individuals to any one SAC river population (or non-designated population). The nearest SAC where shad are present as a qualifying feature is the Pembrokeshire Marine/ Sir Benfro Forol SAC at the edge of the Celtic Sea. However, shad species have now been considered in the ES as part of the diadromous fish assemblage (Section 10.5.8 of **Chapter 10 Fish and Shellfish Ecology**). |
| Natural England | | Both shad species (Alosa alosa and Alosa fallax) are omitted from the diadromous fish receptor group, despite being present in the region (non-spawning).<br>Include shad within all assessments of impacts on diadromous fish, particularly underwater noise, or provide a justification for excluding them. The species is regionally present. https://sac.jncc.gov.uk/species/S1103/ | |

| Consultee | Date/document | Comment | Project response/where addressed |
|---|---|---|---|
| Natural England | | Several designated sites from the region are not included in the assessment. However, all the omitted fish designated features have coincidentally been assessed due to their presence within other designated sites which were assessed.<br>Recommendation:<br>Incorporate the following designated site features into the appropriate assessments:<br>Solway Firth MCZ (Smelt)<br>Solway Firth SAC (Sea lamprey, River lamprey)<br>River Ehen SAC (Atlantic Salmon)<br>River Derwent and Bassenthwaite Lake SAC (Atlantic Salmon, Sea lamprey, River lamprey) | The River Ehen (Atlantic Salmon) and River Derwent and Bassenthwaite Lake (Atlantic Salmon, Sea lamprey, River lamprey) SACs have been included and listed in **Chapter 10 Fish and Shellfish Ecology** of the ES. Designated sites beyond 100km have not been listed in the ES, but an assessment of the species listed as part of the Solway Firth MCZ (Smelt), Solway Firth SAC (Sea lamprey, River lamprey) has been considered in the fish assemblages **Chapter 10 Fish and Shellfish Ecology** of the ES.<br>Within this RIAA on a precautionary basis the Solway Firth SAC, River Eden SAC and River Derwent and Bassenthwaite Lake SAC have been included.<br>MCZs have been discussed within the MCZA) (Document Reference 4.13) as part of the DCO Application. |
| Natural Resources Wales (NRW) | 21st May 2023 | Overall, NRW (A) agree with the conclusion of no significant impact to site integrity for diadromous fish features of the following sites: Dee Estuary/ Aber Dyfrwy SAC, River Dee and Bala Lake/ Afon Dyfrwy a Llyn Tegid SAC, Afon Gwyrfai a Llyn Cwellyn SAC and Afon Eden – Cors Goch Trawsfynydd SAC. | Agreement noted, no further action. |

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## 7.3 Assessment of effects

166. The HRA Screening Report (Morecambe Offshore Windfarm Ltd, 2023a; Document Reference 4.10) identified the following potential effects that should be taken forward for further assessment in relation to the construction, operation and maintenance and decommissioning phases of the Project:

- Increased SSCs and deposition
- Temporary or permanent habitat loss
- Remobilisation of contaminated sediments
- Underwater noise and vibration
- Barrier effects
- EMF
- Introduction/removal of hard substrate.

167. The embedded mitigation and worst-case scenario presented in **Sections 7.3.1** and **7.3.2** therefore relate to these effects.

### 7.3.1 Embedded mitigation

168. This section outlines the embedded mitigation incorporated into the design of the Project (presented in **Table 7.2**) relevant to the assessment for Annex II fish species.

*Table 7.2 Embedded mitigation measures relevant to fish ecology*

| Parameter | Mitigation measures embedded into the design of the Project |
|---|---|
| Cables | The cable burial range is between 0.5m and 3.0m below the seabed (with a target depth of 1.5m, where ground conditions allow (recognised industry good practice, which would reduce effects of EMF)). A detailed Cable Burial Risk Assessment (CBRA) would also be required to confirm the extent to which cable burial can be achieved. Where it is not reasonably practicable to achieve cable burial, additional cable protection may be required.<br>Cables would be specified to reduce EMF emissions, as per industry standards and best practice, such as the relevant IEC (International Electrotechnical Commission) specifications.<br>To minimise the extent of any unnecessary habitat disturbance, material displaced as a result of cable burial activities would be back filled, where necessary, in order to promote recovery. |
| Foundation installation | The selection of appropriate foundation designs and sizes at each WTG and OSP location would be made following pre-construction surveys within the windfarm site.<br>A soft start and ramp up protocol for pile driving (if piled foundations are selected) may also allow mobile species to move away from the |

| Parameter | Mitigation measures embedded into the design of the Project |
|---|---|
| | area before the maximum hammer energy with the greatest noise impact area is reached.<br>Any further mitigation beneficial to marine mammals (as outlined in **Chapter 11 Marine Mammals**) could also potentially reduce impacts on fish and shellfish ecology. |
| Construction | During construction, overnight working practices would be employed offshore, so that construction activities could be 24 hours, thus reducing the overall period for potential impacts to fish communities in proximity to the windfarm site.<br>Vessels would avoid deliberate approaching when basking sharks are sighted. Further, vessel management protocols for marine mammals are outlined in **Chapter 11 Marine Mammals**. |
| Decommissioning | An Offshore Decommissioning Programme would be developed post-consent and implemented at the time of decommissioning. |

### 7.3.2 Realistic worst-case scenario

169. The final design of the Project would be confirmed through detailed engineering design studies that would be undertaken post-consent to enable the commencement of construction. To provide a precautionary, but robust impact assessment at this stage of the development process, realistic worst-case scenarios have been defined. The realistic worst-case scenario (having the most impact) for each individual impact was derived from the PDE to ensure that all other design scenarios would have less or the same impact. Further details have been provided in **Chapter 6 EIA Methodology** of the ES. This approach has been common practice for developments of this nature, as set out in PINS Advice Note Nine: Rochdale Envelope (PINS, 2018).

170. The realistic worst-case scenario is presented in **Table 7.3**.

*Table 7.3 Realistic worst-case scenarios for Annex II fish species*

| Impact | Worst-case scenario | Notes and rationale |
|---|---|---|
| **Construction phase** | | |
| Temporary habitat loss/ physical disturbance | **WTG & OSP foundations:**<br>▪ 35 x WTGs with GBS foundations (including jack-up vessel footprint) = 303,625m$^2$<br>▪ Two x OSPs with GBS foundations (including jack-up vessel footprint) = 17,350m$^2$<br>▪ Anchoring for 35 WTGs and two OSPs = 26,640m$^2$<br>Total = **347,615m$^2$** | Given the seabed preparation is the same per foundation for smaller and larger WTGs, the worst-case assumes 35 x smaller WTGs with GBS foundations. GBS foundations are assumed to have a diameter of 65m + 10m disturbance either side.<br>The worst-case scenario is for two jack-up visits per WTG/OSP foundation in different positions over the construction period (each jack-up with 6 legs, each with a 250m$^2$ footprint). This equates to a total footprint of 1,500m$^2$ per jack-up vessel visit and 3,000m$^2$ over the construction period per WTG/OSP foundation.<br>The worst-case scenario is for two anchor positions per foundation (including resetting), with up to 12 anchors per location. Each anchor width is estimated to be 6m, with an approximate seabed footprint of 30m$^2$ per anchor.<br>Scour protection is encompassed within the seabed preparation area and therefore has not been presented. |
| | **Inter-array and platform link cables:**<br>▪ Inter-array cables = 1,750,000m$^2$<br>▪ Platform link cables = 250,000m$^2$<br>Total = **2,000,000m$^2$** | The worst-case scenario for physical disturbance for cables is based on a maximum length of 70km of inter-array cables and 10km of platform link cables, with a 25m wide installation corridor in which cable preparation activities may take place (this encompasses pre-lay activities (e.g. boulder removal), trenching and spoil width). |
| | **Cumulative area of seabed disturbance: 2,347,615m$^2$ (approximately 2.4km$^2$)** | |

| Impact | Worst-case scenario | Notes and rationale |
|---|---|---|
| Increased SSCs and subsequent deposition and remobilisation of contaminants | **Sediment displaced during seabed preparation for WTGs and OSP foundations:**<br>▪ 35 WTGs with GBS foundations = 455,438$m^3$<br>▪ Two OSPs with GBS foundations = 26,025$m^3$<br>Total = **481,463$m^3$** | Seabed preparation (e.g. excavation using a TSHD or other specialist bed leveller/trencher such as mass flow excavation) may be required. This is a volume of sediment that is disturbed prior to installation of WTG/OSP foundations and involves the removal of sediment from the seabed. The worst-case scenario assumes that sediment would be removed and returned to the water column at the sea surface (e.g. during disposal from a dredger vessel[14]) for WTGs and OSPs.<br>Given the seabed preparation area is the same per foundation for the smaller and larger WTGs, the worst-case assumes the larger number of smaller WTGs with GBS foundations, with a diameter of 65m + 10m either side. The seabed preparation area also includes area for two jack-up visits per WTG/OSP foundation in different positions over the construction period. This equates to a total footprint of 1,500$m^2$ per jack-up vessel visit and 3,000$m^2$ over the construction period per WTG/OSP foundation. The seabed preparation area would be dredged to a depth of up to 1.5m.<br>Drill arisings from drive-drill-drive methodology would result in a lower volume of sediment being disturbed (55,865$m^3$ – based on monopile foundations). |

[14] It is possible that seabed preparation would be undertaken by plough and sediment would therefore not be released at the surface, however disposal at the surface has been retained for the worst-case scenario.

| Impact | Worst-case scenario | Notes and rationale |
|---|---|---|
| | **Sediment displaced during sandwave clearance/levelling for cables:**<br>▪ Inter-array cables = 70,000m$^3$<br>▪ Platform link cables = 10,000m$^3$<br>Total = **80,000m$^3$**<br><br>**Sediment displaced during cable installation:**<br>▪ Inter-array cables = 472,500m$^3$<br>▪ Platform link cables = 67,500m$^3$<br>Total = **540,000m$^3$** | The worst-case length of inter-array cables is 70km and platform link cables is 10km.<br>The worst-case assumes that 10% of the length of inter-array and platform link cables would require sandwave clearance/levelling, with a clearance width of 10m and height of 1m.<br>The worst-case assumes sediment would be released at the water surface.<br>The worst-case assumes that 50% of inter-array and platform link cables are buried at 3m and 50% length is buried at 1.5m by jetting in a box-shaped trench, with a 3m trench width. |
| | **Cumulative volume of sediment disturbed: 1,101,463m$^3$ (approximately 1.1km$^2$)** | |
| Underwater noise and vibration impacts to hearing sensitive species during foundation piling | **Largest hammer energy**<br>▪ Diameter of monopiles: 12.0m<br>▪ Maximum monopile penetration depth: 56m<br>▪ Maximum hammer driving energy: 6,600kJ<br>▪ Number of piled foundations: 37<br><br>**Longest duration**<br>▪ Number of pin pile foundations: 148 (each WTG/OSP foundation with four pin piles)<br>▪ Diameter of pin piles: 3.0m<br>▪ Maximum hammer driving energy: 2,500kJ | Larger turbines require a greater pile diameter than smaller turbines and therefore would generate more noise for a given hammer driving energy. This assessment assumed the largest pile diameter (12m) and is therefore conservative.<br>Pin piles are the worst-case scenario in terms of the length of time likely to be taken for installation. See **Appendix 11.1 Underwater Noise Assessment** (Document Reference 5.2.11.1) for underwater noise modelling parameters and scenarios.<br>Cumulative sound exposure levels have been modelled for each piling event under consideration: single monopiles, single pin piles, three sequential monopiles and four pin piles piled sequentially. Four |

| Impact | Worst-case scenario | Notes and rationale |
|---|---|---|
| | ▪ Duration: One pile = four hours 30 minutes duration. Four piles = 18 hours duration (four piles per foundation). Total duration was 666 hours for all WTGs & OSP(s)<br><br>**Highest strike rate**<br>▪ Fastest strike rate: 100 blows per minute.<br>▪ Maximum hammer energy: 6,600kJ<br>▪ Duration: One monopile = three hours 48 minutes duration; one pin pile = three hours 13 minutes. Four pin piles = 12 hours 54 minutes. | sequential pin piles provided the worst-case in terms of cumulative sound exposure levels at this stage. Two scenarios for cumulative sound exposure have been modelled reflecting both the longest duration (with a lower strike rate) and a shorter duration (with a higher strike rate). |
| Underwater noise and vibration impacts to hearing sensitive species due to other activities (seabed preparation, cable installation etc.) | **Seabed clearance**<br>Methods could include: Pre-lay grapnel run, boulder grab, plough, sandwave levelling (pre-sweeping) and dredging<br><br>**Inter-array and platform link cable installation**<br>Continuous noise levels associated with a range of cable laying activities have been considered:<br>▪ Cable laying<br>▪ Suction dredging<br>▪ Trenching<br>▪ Rock placement<br>▪ Vessel noise (large)<br>▪ Vessel noise (medium)<br><br>**Maximum length of cables**<br>▪ Inter-array cables: 70km<br>▪ Platform link cables: 10km | Example source levels from literature have been used to assess continuous noise sources. Underwater noise modelling undertaken for dredging, trenching, cable laying and rock placement was considered the worst-case in terms of underwater noise for construction activities other than piling (see **Appendix 11.1**). |

| Impact | Worst-case scenario | Notes and rationale |
|---|---|---|
| | **Vessels**<br>▪ Maximum number of vessels on site at any one time: **37** | |
| **Operation and maintenance phase** | | |
| Permanent habitat loss<br>Introduction of hard substrate | **Footprint of WTG/OSP foundations and scour protection:**<br>▪ 35 x GBS WTGs with scour protection = 248,080m²<br>▪ Two GBS OSPs with scour protection = 14,176m²<br><br>Total worst-case footprint of WTGs/OSPs = **262,256m²** | The worst-case scenario assumes 35 x WTGs and two x OSP(s) (each with a 65m diameter conical GBS foundation, plus scour protection extending 15m from foundations in all directions). |
| | **Footprint of cable protection:**<br>▪ Inter-array cables = 91,000m²<br>▪ Platform link cables = 13,000m²<br>▪ Entry to WTGs and OSPs = 45,500m²<br><br>Total footprint of cable protection = **149,500m²** | The worst-case is based on 70km of inter-array cables and 10km of platform link cables. Assumed 10% of cable length is unburied due to ground conditions with a 13m cable protection width and 2m height.<br>The worst-case for cable protection for the entry points to WTG/OSP foundations assumes 70 points of entry, each with a length of cable protection of 50m and width at the base of 13m. The seabed footprint of cable protection per entry point is 650m². |
| | **Footprint of cable/pipeline crossings:**<br>▪ Inter-array cable crossings (9) = 40,050m²<br>▪ Platform link cable crossings (6) = 26,700m²<br><br>Total footprint of crossings = **66,750m²** | The worst-case for cable crossings is based on nine cable crossings across inter-array cables and six cable crossings across platform link cables.<br>Each crossing footprint was calculated as 4,450m² (17.8m width at the base, 250m length and 2.8m in height). |
| | **Replacement scour protection material and cable protection:**<br>▪ WTGs/OPSs = 13,950m²<br>▪ Cables = 21,625m² | It has been assumed that up to 10% of the total scour protection material and cable protection installed during construction would be required to be replaced or replenished during the operation and maintenance phase. |

| Impact | Worst-case scenario | Notes and rationale |
|---|---|---|
| | Footprint of replacement of scour protection material/cable protection = **35,575m$^2$** | |
| | **Cumulative footprint: 413,431m$^2$ (approximately 0.4km$^2$)** | |
| Temporary habitat loss/disturbance and increased SSCs and subsequent deposition and remobilisation of contaminants | **Jack-up vessel deployments:**<br>▪ Jack-up vessel footprint every other year = 1,500m$^2$<br><br>**Cable repair/replacement:**<br>▪ Average cable repair/replacement footprint per year = 2,000m$^2$<br>▪ Average cable reburial footprint per year = 1,000m$^2$<br><br>**Anchoring:**<br>▪ Average temporary anchor footprint per year = 720m$^2$<br><br>Total per year (noting jack-ups are only assumed every other year) = **5,220m$^2$**<br>Total over operational period = **155,700m$^2$** | The worst-case scenario for jack-up deployments assumes the use of one jack-up vessel with a seabed footprint of 1,500m$^2$ (up to six legs, each with a footprint of up to 250m$^2$) every other year.<br>The worst-case is based on an average of 200m of cable repaired/replaced every year and an average of 100m of cable reburied every year, with a 10m disturbance width.<br>The worst-case for anchoring is anticipated to be on average one anchoring event per year.<br>It is noted that the total disturbance over the 35-year operational period is based on yearly averages and thus assesses for example that there may be no cable repair in one year and then longer lengths of cable repair/replacement and/or reburial in other years. |
| | **Sediment displaced during cable repair/replacement and reburial every year:**<br><br>▪ Average cable repair or replacement sediment volume = 6,000m$^3$<br>▪ Average cable reburial sediment volume = 3,000m$^3$<br><br>Total disturbed per year (on average) = **9,000m$^3$**<br>Total over operational period = **315,000m$^3$** | Temporary increases in SSCs would result from periodic jack-up vessel deployment, and cable repair, replacement and reburial activities.<br>The worst-case assumes on average 200m of cables would be repaired/replaced every year, with a 10m disturbance width and 3m maximum depth for a box-shaped trench.<br>The worst-case assumes up to 100m of cable would be reburied every year, with a 10m disturbance width and 3m maximum depth for a box-shaped trench. |

| Impact | Worst-case scenario | Notes and rationale |
|---|---|---|
| | | It is noted that the total volume over the 35-year operational period is based on yearly averages and thus assesses for example that there may be no cable repair in one year and then longer lengths of cable repair/replacement and/or reburial in other years.<br>The volume of sediment that could be suspended due to the presence of jack-up vessels has not been calculated but would be a much smaller proportion compared to the quantity generated by construction and decommissioning activities. |
| Underwater noise and vibration | The following impacts were relevant to the worst-case scenario for fish and shellfish ecology:<br><br>**Underwater noise from operational turbines:**<br>▪ WTG parameters (e.g. size and number) as outlined above and underwater noise parameters described in **Appendix 11.1**<br>▪ Operational life of windfarm = 35 years<br><br>**Underwater noise from maintenance activities (cable repair, replacement and reburial and cable protection works):**<br>▪ Average length of inter-array/platform link cable repair/replacement every year = up to 200m<br>▪ Average length of inter-array/platform link cable reburial every year = up to 100m<br>**Underwater noise from vessels:**<br>▪ Types of vessels: cable laying and burial, rock placement, support vessels, crew transfer vessels | Underwater noise modelling undertaken for operational turbines, dredging, trenching, cable laying and rock placement can be found in **Appendix 11.1** of the ES.<br>Vessel assessments based on worst-case scenario for maximum number of vessels on site at any one-time and maximum number of return vessel trips during operation and maintenance, and construction period. Operation and maintenance port(s) are still to be determined. |

| Impact | Worst-case scenario | Notes and rationale |
|---|---|---|
| | ▪ Maximum number of vessels on site at any one time = up to three vessels during a standard year and up to 10 vessels on a 'heavy maintenance' year (every five years)<br>▪ Maximum annual number of operation and maintenance vessel return trips to port = 384 during a standard year and up to 832 vessels on a 'heavy maintenance' year | |
| EMF | **Platform link and inter-array cables**<br>▪ Burial range 0.5-3.0m with a target burial depth of 1.5m<br>▪ Inter-array cable operating voltage of up to 132kV AC and 275kV for a platform link cable<br>▪ 70km of inter-array and 10km of platform link cables | The maximum length of cables would result in the greatest potential for EMF-related effects.<br>It should be noted that where cables were unable to be buried, they would instead be protected which would afford a degree of attenuation of EMF. |
| Barrier effects | As per above (EMF, noise and SSCs) | |
| **Decommissioning phase** | | |
| As per construction and removal of hard substrates | The decommissioning policy for the Project infrastructure has not yet been defined however it is anticipated that structures above the seabed would be removed.<br>The following infrastructure is likely be removed, reused, or recycled where practicable:<br>▪ WTGs and foundations<br>▪ OSP(s) including topsides and foundations.<br>The following infrastructure is likely to be decommissioned and could be left *in situ* depending on available information at the time of decommissioning:<br>▪ Inter array and platform link cables<br>▪ Scour protection<br>▪ Crossings and cable protection<br>▪ Part of the foundations (e.g. some foundation material below the seabed may be left *in situ*) | The detail and scope of the decommissioning works would be determined by the relevant legislation and guidance at the time.<br>Decommissioning arrangements would be detailed in a Decommissioning Programme, which would be drawn up and agreed with the relevant authority, prior to decommissioning.<br>For the purposes of the worst-case scenario, it has been anticipated that the impacts would be comparable to those identified for the construction phase. |

## 7.4 Dee Estuary/Aber Dyfrdwy SAC

### 7.4.1 Description of designation

171. This section relates to Annex II fish species designated for Dee Estuary/Aber Dyfrdwy SAC. The Dee Estuary/Aber Dyfrdwy SAC is one of the largest estuaries in the UK at 158km$^2$ and its designation protects a number of habitats and species. This site crosses the border between England and Wales and supports significant populations of river lamprey *Lampetra fluviatilis* and sea lamprey *Petromyzon marinus.* The intertidal area has been recorded as being dominated by mudflats and sandflats with the remainder being largely saltmarsh. At low water spring tides, over 90% of the estuary dries out. The extensive intertidal flats of the Dee Estuary form the fifth largest such area within an estuary in the UK (NRW, 2018a).

#### 7.4.1.1 Qualifying features

172. The site was designated for the following Annex II fish species:

- River lamprey
- Sea lamprey

#### 7.4.1.2 Conservation objectives

173. The conservation objectives of the SAC are to ensure that the integrity of the site is maintained or restored as appropriate, and ensure that the site contributes to achieving the FCS of its Qualifying Features, by maintaining or restoring the:

- Extent and distribution of qualifying natural habitats and habitats of qualifying species
- Structure and function (including typical species) of qualifying natural habitats
- Structure and function of the habitats of qualifying species
- Supporting processes on which qualifying natural habitats and the habitats of qualifying species rely
- Populations of qualifying species
- Distribution of qualifying species within the site (Natural England, 2018a)

#### 7.4.1.3 Condition assessment

174. NRW conducted a condition assessment for the species and habitats protected under this SAC to provide an indicative condition of the feature at

this site at the time of assessment. For river lamprey, the assessment determined that the freshwater population variables were in favourable condition and the marine habitat was unfavourable. For sea lamprey, the classification for both marine and freshwater population variables were determined to be unfavourable (NRW, 2018a).

## 7.4.2 Assessment

### 7.4.2.1 Assessment of potential effects of the Project-alone

#### Impact 1: Increased SSCs and deposition (all phases)

175. During construction, and to a lesser degree operation and maintenance and decommissioning activities, there may be a temporary increase in SSCs and deposition which may have an impact on sea lamprey or river lamprey migrating from the Dee Estuary/Aber Dyfrdwy SAC.

176. Suspended sediment has the potential to impair respiratory functions and disrupt migration/spawning activity. Sediment deposition could affect the quality of spawning and nursery habitats, especially if it changes the characteristics of the existing seabed sediments.

177. The Project windfarm site was predominantly composed of sand and fine sand. Based on the sediment sizes present, finer suspended sediment was expected to exist as a passive plume extending to a maximum of one spring tidal ellipse (10km), with other sediments settling quickly in proximity to its release, within a few hundred metres and up to around a kilometre away from the construction activity (**Chapter 7 Marine Geology, Oceanography and Physical Processes**). At a distance of 42km, there was no direct pathway of effect to fish within the SAC or to any supporting habitat.

178. Given that river lamprey are restricted to coastal waters, there was therefore no pathway for effects upon them (Elliot *et al.,* 2021). Sea lamprey are more widely distributed and have been found within shallow coastal waters and deep offshore waters (Maitland, 2003). Adult sea lamprey could potentially cross the Project windfarm site (and 10km ZoI impacted by increased SSCs) during migration to or from freshwater. During this time, they could be exposed to an increased water column sediment loading for a limited period of time, associated with each disturbance activity. However, the increased sediment loading would be short-term and localised in nature. Suspended sediments would largely form a passive plume with minimal (millimetres) deposition (beyond the immediate vicinity of works) across the 10km tidal excursion. The highest levels of suspended sediments would cover a much smaller area (around 1km from release) and, as discussed in **Section 6.4.2** and in **Chapter 9 Benthic Ecology** of the ES, beyond this distance suspended sediments would be low, becoming indistinguishable from background levels. Therefore,

the likelihood of fish encountering an area of increased water column sediment loading would be low. Furthermore, lamprey have been known to tolerate silty, turbid and poor light conditions (Behrmann-Godel and Eckmann, 2003; Hansen *et al*., 2016; Christoffersen *et al*., 2018).

179. The Project-alone would have no adverse effect on integrity for Dee Estuary/Aber Dyfrdwy SAC due to increases in SSCs and deposition during construction, operation and maintenance or decommissioning. The confidence in the assessment was high and aligned with the assessment presented in **Chapter 10 Fish and Shellfish Ecology** of the ES.

### Impact 2: Temporary or permanent habitat loss (all phases)

180. Given the distance of the Project windfarm site from the SAC there would be no direct habitat loss within the SAC.

181. River lamprey at sea typically occupy inshore or estuarine habitat (Elliot *et al.,* 2021). Therefore, there was no pathway for effect from any habitat loss from the windfarm site. Although sea lamprey could be present within the windfarm site, there was no habitat type within the Project windfarm site that was particularly important to sea lamprey or that was not common across the region. In addition, as sea lamprey have high levels of mobility they would be capable of navigating away from any temporary physical disturbance/habitat loss caused by construction, operation and maintenance or decommissioning activities.

182. The Project-alone would have no adverse effect on integrity of the Dee Estuary/ Aber Dyfrdwy SAC due to temporary or permanent habitat loss during construction, operation and maintenance or decommissioning. The confidence in the assessment was high and aligned with the assessment presented in **Chapter 10 Fish and Shellfish Ecology** of the ES.

### Impact 3: Remobilisation of contaminated sediments (all phases)

183. As described in **Section 6.4.2**, this impact (the remobilisation of contaminated sediments) was scoped out of the assessment for all phases.

### Impact 4: Underwater noise and vibration (all phases)

184. By listening to the sounds around them, fish obtain substantial information about their environment and use sound to communicate (Popper *et al.,* 2019; Popper and Hawkins, 2019), as described in **Chapter 10 Fish and Shellfish Ecology** of the ES. Each species has differing sensitivity to noise and therefore the potential impact of noise on fish varies. Anthropogenic sounds can be so intense that they result in death or mortal injury. Lower sound levels may result in temporary hearing impairment, physiological changes (including

stress effects), changes in behaviour or the masking of biologically important sounds (Popper and Hawkins, 2019; Kastelein *et al.,* 2017).

185. Relatively few experiments on the hearing of different fish species have been carried out under suitable acoustic conditions, and there is valid data for only a few species that provide actual thresholds of effect (Popper and Hawkins, 2019). Recent papers on the effects of underwater noise on fish and shellfish species have highlighted the lack of clear evidence to support setting thresholds for impacts on fish and shellfish receptors (Hawkins and Popper, 2016; Popper *et al.*, 2014). These have pointed to some of the shortcomings of impact assessments, including the use of broad criteria for injury and behavioural effects based on limited studies. The effects of particle motion are not well understood but are considered to be more important for many fish and species than sound pressure, which has been the main consideration in noise impact assessments to date (Popper and Hawkins, 2018).

186. The most recent and relevant guidelines for the purposes of this assessment is the Acoustical Society of America Sound Exposure Guidelines for Fishes and Sea Turtles (Popper *et al*., 2014). These guidelines provided directions and recommendations for setting criteria (including injury and behavioural criteria) for fish. The Popper *et al.,* (2014) guidelines broadly grouped fish into the following four categories based on their anatomy and the available information on hearing of other fish species with comparable anatomies:

- Group 1: Fishes lacking swim bladders that are sensitive only to sound particle motion and show sensitivity to a narrow band of frequencies (includes flatfishes and elasmobranchs)
- Group 2: Fishes with a swim bladder where the organ does not appear to play a role in hearing. These fish are sensitive only to particle motion and show sensitivity to a narrow band of frequencies (includes salmonids and some tuna)
- Group 3: Fishes with swim bladders that are close, but not intimately connected to the ear. These fishes are sensitive to both particle motion and sound pressure and show a more extended frequency range than groups 1 and 2, extending to about 500Hz (includes gadoids and eels)
- Group 4: Fishes that have special structures mechanically linking the swim bladder to the ear. These fishes are sensitive primarily to sound pressure, although they also detect particle motion. These species have a wider frequency range, extending to several kHz and generally show higher sensitivity to sound pressure than fishes in Groups 1, 2 and 3 (includes clupeids such as herring, sprat and shads)

187. Lamprey species lack specialist hearing structures and are considered to have low noise sensitivity (Popper, 2005) (Group 1).

188. Underwater noise modelling has been carried out for the Project considering construction and operational noise. The largest impact range during construction would result from piling activities. Full details of the modelling have been provided in **Appendix 11.1** and **Chapter 10 Fish and Shellfish Ecology** (Document Reference 5.1.10) of the ES. The largest impact ranges from piling for fish have been shown in **Table 7.4**. Impact ranges during operation and maintenance were less than 50m around each WTG. Further modelling results from vessel activities and other activities have been provided in the noise modelling report (see **Appendix 11.1**).

*Table 7.4 Single piling and sequential piling within a 24-hour period underwater noise modelling results for both a single monopile and four sequential pin piles with maximum hammer energies, for the worst-case modelling location only (using a stationary animal model) for the Dee Estuary/Aber Dyfrdwy SAC*

| Fish group | Species included | Impact criteria | Potential impact | Impact areas and ranges<br>Monopile (maximum hammer energy 6,600kJ) (Sound Exposure Level Cumulative Exposure (SELcum) relates to three sequential monopiles within 24 hours) | | | | Pin pile (maximum hammer energy 2,500kJ) ($SEL_{cum}$ relates to four sequential pin piles within 24-hours) | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | | Area | Max | Min | Mean | Area | Max | Min | Mean |
| Fish group 1: no swim bladder (particle motion detection) | River lamprey and sea lamprey | >213 dB unweighted Sound Pressure Level $(SPL)_{peak}$ | Mortality and potential mortal injury | 0.05km$^2$ | 130m | 130m | 130m | 0.03km$^2$ | 100m | 100m | 100m |
| | | >219 dB unweighted $SEL_{cum}$ [stationary] | Mortality and potential mortal injury | 11km$^2$ | 2km | 1.9km | 1.9km | 5.9km$^2$ | 1.4km | 1.4km | 1.4km |
| | | >216 dB unweighted $SEL_{cum}$ [stationary] | Recoverable injury | 25km$^2$ | 2.9km | 2.8km | 2.8km | 14km$^2$ | 2.1km | 2.1km | 2.1km |
| | | >186 dB unweighted $SEL_{cum}$ [stationary] | Temporary Threshold Shift (TTS) | 2400km$^2$ | 33km | 20km | 27km | 1900km$^2$ | 30km | 19km | 25km |

189. River lamprey typically remain within estuarine environments during their adult life stages (Maitland, 2003) and therefore would not interact with noisy activities from the Project, given that the windfarm site is 42km from the SAC and the maximum range of effect was 33km. It has also been considered that there would be no barrier to migration along the coast.

190. Sea lamprey are more widely distributed and have been found within shallow coastal waters and deep offshore waters (Maitland, 2003). As for river lamprey, impact ranges for injury and behavioural effects would not reach the SAC, which is over 42km from the windfarm site. Only sea lamprey individuals outside the SAC could interact with any impact and therefore noise levels generated during construction of the Project would not affect spawning activity. Sea lamprey are not thought to specifically migrate back to their natal rivers (Bergstedt and Seelye 1995; Waldman *et al*., 2008); instead, they are thought to return to rivers within the region, navigating primarily by detection of larval pheromones to identify suitable rivers (i.e. those with pre-existing larvae) (reviewed in Hansen *et al*., 2016). This flexibility in homing behaviour, combined with the low sensitivity of this species to underwater noise suggested that noise effects upon sea lamprey individuals outside the SAC would be minimal.

191. Given that noise ranges for operation and maintenance would be highly localised and decommissioning would not require activities such as piling, the effects for these phases would be lower than for construction.

192. The Project-alone would have no AEoI on the Dee Estuary/Aber Dyfrdwy SAC due to underwater noise effects during construction, operation and maintenance or decommissioning. The confidence in the assessment was high and aligned with the assessment presented in **Chapter 10 Fish and Shellfish Ecology** of the ES.

### Impact 5: Barrier effects (all phases)

193. Barrier effects could result from noise, suspended sediments and the physical presence of infrastructure.

194. Only sea lamprey have the potential to be present in the windfarm site, there is no pathway for effects upon river lamprey or sea lamprey in coastal waters.

195. The noise impacts on sea lamprey would be intermittent, and, given the flexibility in homing behaviour and low sensitivity to noise of sea lamprey individuals, noise effects would not present a barrier to migration for fish moving through the wider IS. Suspended sediments and the introduction of hard substrate would also be localised in the context of the species distribution.

196. The Project-alone would have no AEoI on the Dee Estuary/Aber Dyfrdwy SAC due to barrier effects during construction, operation and maintenance or decommissioning. The confidence in the assessment was high and aligned with the assessment presented in **Chapter 10 Fish and Shellfish Ecology** of the ES.

### Impact 6: EMF (operation and maintenance phase)

197. EMF could have the potential to interfere with the navigation of sensitive migratory and pelagic species by affecting the speed and/or course of their movements through the windfarm site, causing subsequent potential issues if they were not able to reach spawning, nursery or feeding grounds. Lamprey possess ampullary electroreceptors, used to survey their surroundings for prey or predators. Most EMF exposure would be expected to be short, in the order of minutes, whilst these highly mobile species were moving through the windfarm site. The area around the cable where EMF would be elevated would be small (less than 10m based on Taormina *et al.,* (2020)). Given the distance of the Project windfarm site from the SAC there would be no direct EMF effects within the SAC. Given that river lamprey are restricted to coastal waters, there was therefore no pathway for effects upon them.

198. The area around the cable where EMF would be elevated would represent a very small fraction of the available habitat for sea lamprey, which may travel multiple kilometres per day and are less likely to swim close to the seafloor (Snyder *et al*., 2019). Effects on sea lamprey (if present within the windfarm site) from EMF were expected to be minimal.

199. The Project-alone would have no AEoI on the Dee Estuary/Aber Dyfrdwy SAC due to EMF effects during construction, operation and maintenance or decommissioning. The confidence in the assessment was high and aligned with the assessment presented in **Chapter 10 Fish and Shellfish Ecology** of the ES.

### Impact 7: Introduction/removal of hard substrate (all phases)

200. There would be no introduction or removal of hard substrate within the SAC. Given that river lamprey are restricted to coastal waters, there would be no pathway for effects upon them.

201. The area of hard substrate introduced within the Project windfarm site would be a worst-case of $0.4km^2$. The area of introduced hard substrate would represent a very small fraction of the available habitat for sea lamprey, which may travel multiple kilometres per day and are less likely to swim close to the seafloor (Snyder *et al.*, 2019). Effects on sea lamprey (if present within the windfarm site) from introduced/removal hard substrate were expected to be minimal.

202. The Project-alone would have no AEoI on the Dee Estuary/Aber Dyfrdwy SAC due to introduction/removal of hard substrate during construction, operation and maintenance or decommissioning. The confidence in the assessment was high and was based on the assessment presented in **Chapter 10 Fish and Shellfish Ecology** of the ES.

## Potential interactions of Project effects

203. The effects identified and assessed in this section have the potential to interact with each other. The effects of the Project were:

- Increased SSCs and deposition
- Temporary or permanent habitat loss
- Remobilisation of contaminated sediments
- Underwater noise and vibration
- Barrier effects
- EMF
- Introduction/removal of hard substrate.

204. There would be no introduction or removal of hard substrate, or temporary or permanent habitat loss within the SAC and therefore there would be no potential for these effects to interact with other effects. Given the low levels of contaminants, there would also be no interaction with other effects.

205. There were potential interactions between increased SSCs and deposition, underwater noise and vibration and barrier effects with other effects.

### Construction phase

206. Underwater noise impacts would be greatest in spatial extent for foundation piling, but these would occur only during a short part of the construction phase, therefore there would be limited potential for interaction with habitat disturbance from seabed preparation, installation of cables etc. and associated effects (increased SSCs). The effects resulting from habitat disturbance would be localised, temporary and episodic with limited potential for interaction (i.e. causing increased barrier effects). The potential for noise to cause barrier effects has already been captured in the barrier effect assessment in **Paragraphs 193 – 196**. It was therefore considered that these impacts would not interact to change the significance level overall.

### Operation and maintenance phase

207. Disturbance to or loss of habitat would be confined to the immediate footprint of the infrastructure/activities. The magnitude of effect was, in all cases, low to negligible. Temporary habitat loss or disturbance during the operation and

maintenance phase would be additional to the permanent habitat loss due to infrastructure footprint, however, this would remain a localised and temporary effect with low to negligible magnitude in the context of the broadscale habitat in the IS. EMF and noise effects would also be locally confined and again the magnitude of effect was low to negligible and related to largely the same spatial footprint. The potential for noise and EMF to cause barrier effects has already been considered in the standalone barrier effect assessment in **Paragraphs 193 – 196**. It was therefore considered that none of these impacts would interact to increase the significance level overall.

**Decommissioning phase**

208. It has been anticipated that the decommissioning impacts would be similar in nature to those of construction.

### Project-alone conclusion

209. Considering the assessment against the conservation objectives, **Section 7.4.1.2**, the Project would have no AEoI on the Dee Estuary/Aber Dyfrdwy SAC. This was largely due to the magnitude of effects, given the separation of the Project to the SAC. The confidence in the assessment was high and aligned with the assessment presented in **Chapter 10 Fish and Shellfish Ecology** of the ES.

## 7.4.2.2 In-combination assessment – the Project and Transmission Assets combined

210. A ‘combined’ assessment has been made with the Transmission Assets[15], for the purpose of an in-combination assessment considering its functional link with the Project.

211. This section provides assessment of impact interactions and additive effects for the Dee Estuary/Aber Dyfrdwy SAC which has been screened in for both the Project and Transmission Assets.

### In-combination impact 1: Increased SSCs and deposition (all phases)

212. The predicted combined volume of material likely to be disturbed during the construction phase of the Project and the Transmission Assets would be in the region of 13.4 million m$^3$. This includes approximately 1.1 million m$^3$ associated with the Project (see **Table 7.3**) plus c.12.3 million m$^3$ associated

[15] As the Transmission Assets includes infrastructure associated with both the Project and the Morgan Offshore Wind Project Generation Assets, it should be noted that the combined assessment considers the transmission infrastructure for both the Project and the Morgan Offshore Wind Project Generation Assets.

with the Transmission Assets (Morgan Offshore Wind Limited and Morecambe Offshore Windfarm Ltd, 2023a).

213. As described in **Section 7.4.2.1**, 'heavy' deposition would only occur within a very short distance of the source of disturbance; at more than 1km distance SSC increases and deposition levels would be low, becoming indistinguishable from background within a maximum ZoI of 10km. Therefore, the likelihood of fish encountering an area of increased water column sediment loading would be low. Furthermore, sea lamprey are known to tolerate silty turbid and poor light conditions (Behrmann-Godel and Eckmann, 2003; Hansen *et al*., 2016; Christoffersen *et al*., 2018).

214. Given the relationship between the Project and the Transmission Assets, site preparation and installation of infrastructure would be phased and SSC increases would be unlikely to occur concurrently. However, should multiple operations be undertaken simultaneously, plumes would be advected on the tide (not towards one another). Activities would be of limited spatial extent and plume interactions of a low magnitude and short duration. For both projects, the majority of sedimentation would occur within close proximity of each installation activity; however, given the active sediment transport regime, deposited material would be redistributed. Given the distance of the SAC from both projects and the context of localised effects across the IS, the magnitude of any effect would be limited.

215. As any interaction of sediment plumes and deposition would be localised (i.e. of small spatial extent) and temporary, there would be no adverse in-combination effect on the integrity of the Dee Estuary/Aber Dyfrdwy SAC.

### In-combination impact 2: Temporary or permanent habitat loss (all phases)

216. The combined temporary habitat loss/disturbance from the Project and the Transmission Assets during the construction phase (when temporary loss would be greatest) would equate to c.46.8km$^2$. This includes the c.2.3km$^2$ associated with the Project (**Table 7.3**), plus c.44.5km$^2$ associated with the Transmission Assets. The cumulative temporary habitat loss/disturbance footprint from the Project and the Transmission Assets during the operation and maintenance phase would equate to c.11.0km$^2$. This includes the c.0.2km$^2$ associated with the Project (**Table 7.3**) plus 10.9km$^2$ associated with the Transmission Assets (Morgan Offshore Wind Limited and Morecambe Offshore Windfarm Ltd, 2023a).

217. The combined long term/permanent presence of physical infrastructure from the Project and the Transmission Assets during the operation and maintenance phase (leading to a change in habitat type and loss of soft sediment) would equate to c.1.9km$^2$. This includes the c.0.4km$^2$ associated with the Project (**Table 7.3**), plus c.1.5km$^2$ associated with the Transmission

Assets (Morgan Offshore Wind Limited and Morecambe Offshore Windfarm Ltd, 2023a). However, no habitat loss would occur within the SAC.

218. For effects on a migratory Annex II fish outside of its associated SAC to occur, the Transmission Assets would also need to interact with habitat suitable for that species in a detrimental way. Suitable habitat that would be present in the Project windfarm site is also ubiquitous across the wider region, and changes of this scale from soft to hard substrate would not impact on the ability of a fish to migrate through the region to and from the SAC.

219. The Project in-combination with the Transmission Assets would have no AEoI on the Dee Estuary/Aber Dyfrdwy SAC due to temporary or permanent habitat loss (all phases).

#### In-combination impact 3: Remobilisation of contaminated sediments (all phases)

220. As described in **Section 6.4.2**, this impact (remobilisation of contaminated sediments) was scoped out of the assessment for all phases.

#### In-combination impact 4: Underwater noise and vibration (all phases)

221. The key components of the Transmission Assets that require piling comprise of four OSPs at Morgan, two OSPs at Morecambe, and the Morgan offshore booster station. UXO clearance for both projects may also be required.

222. The construction phase of the Transmission Assets may have temporal and spatial overlap with the Project in terms of sound associated with piling and UXO clearance, potentially resulting in a cumulative impact. The assessment of sound impacts associated with piling for the Project-alone has been presented above (**Section 7.4.2.1**), with a low magnitude identified based on a range of technical specifications and sound modelling outputs. There would be the potential for piling to occur concurrently at the Project and the Morgan offshore booster substation and Morgan OSP(s).

223. Sound modelling for the Transmission Assets indicated similar patterns as those for the Project, with injury and mortality from sound produced within the Transmission Assets for a single monopile (maximum hammer energy of 5,500kJ to ranges of up to 755m for Group 1 fish (the group relevant to lamprey species), if modelled as stationary receptors (Morgan Offshore Wind Limited and Morecambe Offshore Windfarm Ltd, 2023a). See **Section 7.4.2.1** for an explanation of fish sound sensitivity groups. Recoverable injury distances were calculated to reach out to up to 4,340m for Group 2 stationary receptors with similar patterns for all other groups of fish, in comparison to the worst-case 7.1km modelled for a single monopile for the Project (see **Appendix 11.1**).

224. Overall, the short piling duration expected for the Transmission Assets would only represent a very short-term increase in the ensonified area when considered cumulatively with planned piling at the Project.

225. The construction phase of the Transmission Assets may also have temporal overlap with the Project in terms of UXO clearance, potentially resulting in a cumulative impact with construction activities. The assessment for UXO clearance for the Transmission Assets has determined a low magnitude for the impact, and based on modelling, found similar mortality and potential mortal injury ranges for high order detonations of explosive quantities of 1.2kg to 907kg with ranges up to 590m (Morgan Offshore Wind Limited and Morecambe Offshore Windfarm Ltd, 2023a), with the Project finding equivalent maximum impact ranges of up to 710m.

226. As noted for the Project-alone assessment, there would be a short term, intermittent nature of impact, which would remain true with the addition of the Transmission Assets.

227. In this context, the additional piling and UXO clearance from the Transmission Assets for a short duration did not alter the findings of the Project-alone assessment.

228. Given that noise ranges for operation and maintenance were highly localised and decommissioning would not require activities such as piling, the effects for these phases would be lower than for construction.

229. The Project in-combination with the Transmission Assets would have no AEoI on the Dee Estuary/Aber Dyfrdwy SAC due to underwater noise effects during construction, operation and maintenance or decommissioning.

### In-combination impact 5: Barrier effects (all phases)

230. As detailed in **Section 7.4.2**, migrating lamprey have a low sensitivity to noise impacts and, due to the distance from the SAC to the noise source at the windfarm site, no in-combination effects on lamprey were identified from barrier effects during construction In-combination with Transmission Assets. The increase above background noise levels expected during operation would be very small and localised in nature and it was considered that in-combination effects from operational noise would not occur beyond Project-alone effects. No in-combination effects with Transmission Assets have been identified in relation to SSCs or introduced substrate given the short term and transient effects, spatial spread of the projects, and the fact that sediments would be moved by tides in an easterly direction, with low potential for plumes to interact.

231. Given the distance of the SAC from both projects, and the context of effects across the IS, the magnitude of any effect was limited and thus no barrier effects have been identified.

232. The Project in-combination with the Transmission Assets would have no AEoI on the Dee Estuary/Aber Dyfrdwy SAC due to barrier effects during construction, operation and maintenance or decommissioning.

### In-combination impact 6: EMF (operation and maintenance phase)

233. Given that river lamprey are restricted to coastal waters, there was no pathway for in-combination EMF effects upon them (i.e. there was no pathway for Project-alone effect, see **Section 7.4.2**). Given the distance of the Project windfarm site from the SAC, there would be no pathway for direct EMF effects within the SAC.

234. As EMF effects would be highly localised to within 10m of cabling (based on Taormina *et al.,* (2020)) there would be no spatial overlap in effects given the distances between the Project and Transmission Assets (see **Table 7.5**). The area around the cable where EMF would be elevated would represent a very small fraction of the available habitat for sea lamprey even if multiple cables were encountered by an individual on any one day. Therefore, effects on sea lamprey from EMF were expected to be minimal.

235. While effects from the Project and Transmission Assets would be additive, diadromous species such as lamprey are highly mobile and were considered to be capable of changing course during migration between natal rivers and the open sea. Any impact of EMF from subsea electrical cabling would be localised in context with the wider IS region and would not result in any barriers to migration to and from the SAC.

236. The Project in-combination with Transmission Assets would have no AEoI on the Dee Estuary/Aber Dyfrdwy SAC due to EMF effects during construction, operation and maintenance or decommissioning.

### In-combination impact 7: Introduction/removal of hard substrate

237. The area of hard substrate introduced within the Project windfarm site would be a worst-case of 0.4km$^2$. This hard substrate could be colonised by encrusting organisms, thereby forming hard substrate-associated biological communities (including the aggregation of fish species, which would feed on the encrusting organisms). The hard substrate would remain in place for the lifetime of the Project and, therefore, the creation of any hard substrate habitat has been assessed as a permanent effect. Subsea infrastructure and cable protection associated with the Transmission Assets would cause similar permanent introductions of hard substrate, and the changes in biological communities that would be associated with the additional hard substrate. In

this way, there was the potential for incremental effects as more hard substrate was added to the region. The Transmission Assets would contribute an additional 1.5km$^2$ of hard substrate.

238. Given the highly localised effects associated with introduced hard substrate habitat (see **Section 7.4.2.1**), the small areas affected and the distance of the projects from the SAC the impact of introduced (and removal of) hard substrate for the Project and the Transmission Assets was limited.

239. The Project in-combination with other plans and projects would have no AEoI on the Dee Estuary/Aber Dyfrdwy SAC due to introduction/removal of hard substrates during construction, operation and maintenance or decommissioning.

#### 7.4.2.3 Assessment of potential effects of the Project in-combination with other plans and projects

240. The projects in **Table 7.5** and **Figure 7.1** have been identified as having the potential to cause in-combination effects given there could be overlap with the ZoI for noise and suspended sediments or incremental effects in the region from habitat loss, introduction/removal of hard substrate and EMF effects.

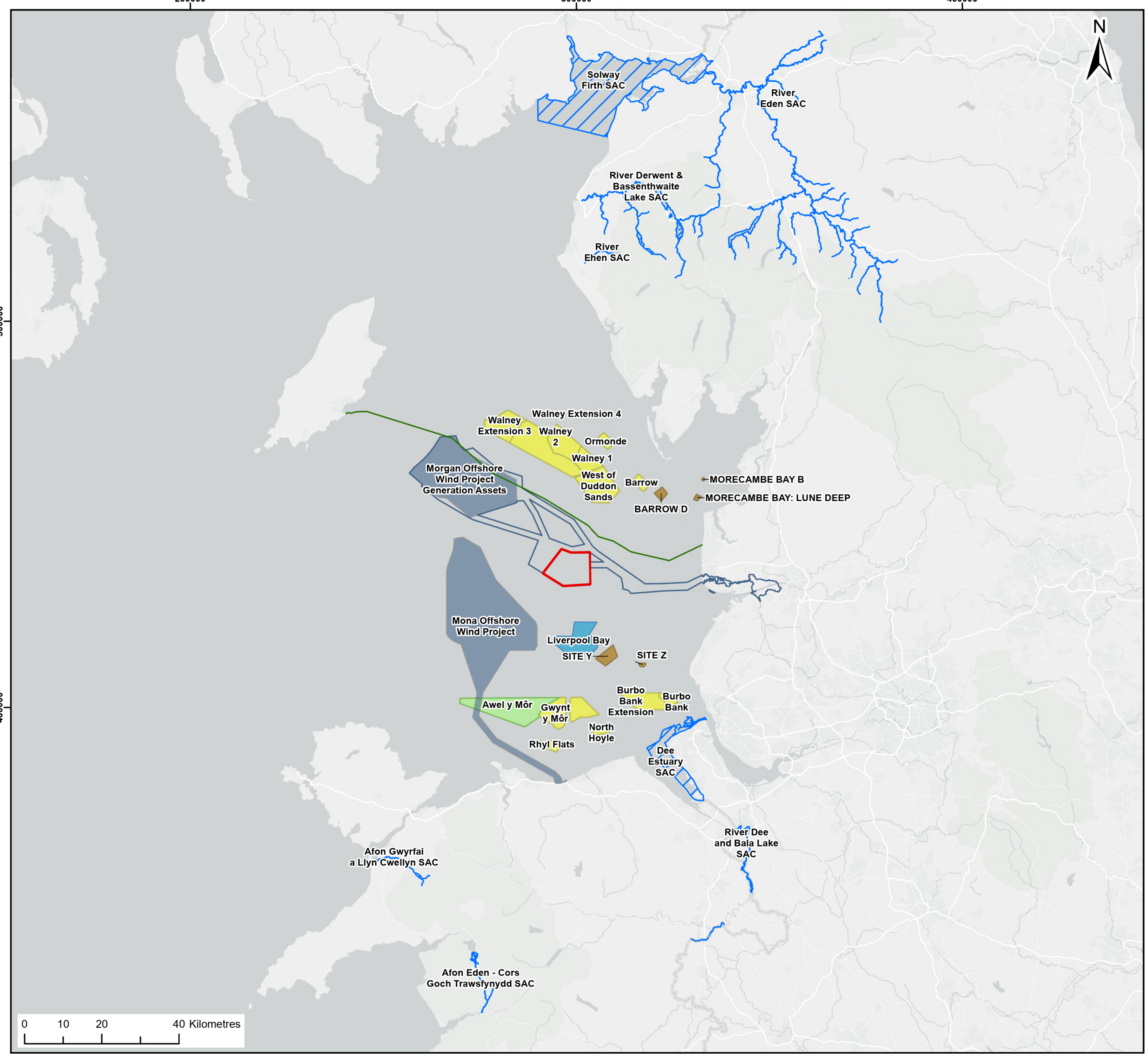
241. For noise and barrier effects, projects with piling and UXO clearance (both also potentially required for the Project) activity that could occur at the same time and within the impact ranges of the Project have been identified as:

- Morgan and Morecambe OWFs: Transmission Assets
- Morgan Offshore Wind Project Generation Assets
- Mona Offshore Wind Project
- AyM OWF

*Table 7.5 Projects identified as having the potential to cause in-combination effects at Dee Estuary/Aber Dyfrdwy SAC*

| Project/plan | Distance from windfarm site (km) | Distance from Dee Estuary/ Aber Dyfrdwy SAC (km) | Description |
|---|---|---|---|
| Morgan and Morecambe Offshore Wind Farms: Transmission Assets | 0 (Adjacent) | 32.4 | Increases in SSCs, presence of physical infrastructure, underwater noise |
| Isle of Man Interconnector (cable protection remedial works) | 4.6 | 40.7 | Increases in SSCs and presence of physical infrastructure |
| Morgan Offshore Wind Project Generation Assets | 16.7 | 70.1 | Increases in SSCs, presence of physical |

| Project/plan | Distance from windfarm site (km) | Distance from Dee Estuary/ Aber Dyfrdwy SAC (km) | Description |
|---|---|---|---|
| | | | infrastructure, underwater noise |
| Mona Offshore Wind Project | 10.0 | 13.1 | Increases in SSCs, presence of physical infrastructure, underwater noise |
| West of Duddon Sands Offshore Windfarm | 12.9 | 59.0 | Increases in SSCs |
| Walney 1,2 and extensions Offshore Wind Farms (OWF) (maintenance activities) | 18.8 | 68.4 | Increases in SSCs |
| Barrow OWF (maintenance activities) | 21.0 | 59.4 | Increases in SSCs |
| Ormonde OWF (maintenance activities) | 27.0 | 72.2 | Increases in SSCs |
| Gwynt y Môr OWF (maintenance activities) | 28.9 | 15.1 | Increases in SSCs |
| Burbo Bank Extension OWF (maintenance activities) | 29.1 | 4.5 | Increases in SSCs |
| Rhyl Flats OWF (maintenance activities) | 40.0 | 14.0 | Increases in SSCs |
| North Hoyle OWF (maintenance activities) | 36.3 | 7.0 | Increases in SSCs |
| AyM OWF | 28.9 | 20.9 | Increases in SSCs, presence of physical infrastructure, underwater noise |
| Liverpool Bay Aggregate Production Area | 9.7 | 27.1 | Increases in SSCs |
| Disposal sites Y and Z | Site Y: 16.8<br>Site Z: 23.9 | Site Y: 22.3<br>Site Z: 17.0 | Increases in SSCs |
| Barrow D disposal site | 22.7 | 56.4 | Increases in SSCs |
| Morecambe Bay B disposal site | 34.6 | 63.1 | Increases in SSCs |
| Morecambe Bay Lune Deep disposal site | 30.1 | 56.0 | Increases in SSCs |





Legend:

- Morecambe Offshore Windfarm Site
- Morgan and Morecambe Offshore Wind Farms: Transmission Assets (In Planning)
- Special Areas of Conservation (SAC) screened for Annex II fish species
- Isle of Man Interconnector

**Disposal Sites Status**

- Open

**Minerals & Aggregates Site Agreements**

- Production Agreement Area

**Windfarm status**

- Fully commissioned
- Consented
- In Planning



Report: Morecambe Offshore Windfarm: Generation Assets Habitat Regulations Report to Inform Appropriate Assessment

Title: Projects with the potential to cause in combination effects on Annex II fish species

Figure: 7.1 | Drawing No: PC1165-RHD-ES-OF-DR-Z-0067

| Revision: | Date: | Drawn: | Checked: | Size: | Scale: |
|---|---|---|---|---|---|
| P02 | 15/01/2024 | JH | AS | A3 | 1:1,000,000 |
| P03 | 24/04/2024 | JH | SB | A3 | 1:1,000,000 |

Co-ordinate system: WGS 1984 UTM Zone 30N

MORECAMBE
Royal HaskoningDHV
Enhancing Society Together

### In-combination impact 1: Increased SSCs and deposition (all phases)

242. As detailed in **Section 6.4.2**, the ZoI for increases in SSCs for the Project during the construction phases (the phase during which the greatest amount of suspended sediment would be produced) was 10km (approximately the spring tidal excursion in an east-west orientation). The direction of travel of sediment plumes of other projects would be dictated by the directionality of the currents at the time of the works associated with those projects. This means that sediment plumes from nearby projects (if occurring at the same time as construction of the Project) would likely travel in a parallel direction to sediment plumes from the Project.

243. For sediment plumes from multiple projects to interact, the projects would need to be within 10km of the Project windfarm site with works occurring simultaneously, this includes the Transmission Assets, Mona Offshore Wind Project, Isle of Man Interconnector (cable protection remedial works), as well as the Liverpool Bay aggregate production area. However, it was only within the nearfield (maximum of 1km) where suspended levels were expected to be distinguishable beyond background levels. Given the distance of the SAC at over 42km from the site, there was no potential for suspended sediment plumes to coalesce within the SAC and therefore no potential for direct in-combination effects. Given that river lamprey are restricted to coastal waters there was no pathway for in-combination effects upon them.

244. In the case of sea lamprey, the Transmission Assets have the potential for overlap of the highest suspended sediments in the nearfield but effects would be limited in temporal and spatial extent (assuming construction between projects was simultaneous). Therefore, the likelihood of sea lamprey encountering an area of increased water column sediment loading would be low. Furthermore, sea lamprey are known to tolerate silty turbid and poor light conditions (Behrmann-Godel and Eckmann, 2003; Hansen *et al.,* 2016; Christoffersen *et al.,* 2018).

245. All other plans and projects were outwith 1km (suspended sediments would have reduced rapidly after this distance), and as such, in-combination effects would be unlikely to occur.

246. Given that the amount of suspended sediment that would be produced would be highest during construction, the effects for operation and maintenance and decommissioning would be lower than for construction.

247. The Project in-combination with other plans and projects would have no AEoI on the Dee Estuary/Aber Dyfrdwy SAC due to increases in SSCs and deposition during construction, operation and maintenance or decommissioning.

### In-combination impact 2: Temporary or permanent habitat loss (all phases)

248. In terms of temporary habitat loss during construction, the habitat types found within the Project windfarm site have a high recoverability, and the temporary habitat disturbance associated with this Project and other projects identified in **Table 7.5** was negligible in the context of wider disturbance in the region from e.g. mobile fishing.

249. In terms of permanent habitat loss, there was the potential for incremental additional effects resulting from the loss of habitat due to the construction of other planned OWFs in the region. Morgan Offshore Wind Project, Transmission Assets, Mona Offshore Wind Project and AyM Offshore Wind Farm are all planned to be constructed in the region and would therefore cause additional permanent habitat loss.

250. The habitat lost in the Project windfarm site would be of negligible importance to migrating sea lamprey. Any habitat losses from the other projects identified in **Table 7.5** would also relate to habitat which was also of negligible importance to sea lamprey. The Project in-combination with other plans and projects would have no AEoI on the Dee Estuary/Aber Dyfrdwy SAC during construction, operation and maintenance or decommissioning.

### In-combination impact 3: Remobilisation of contaminated sediments (all phases)

251. As described in **Section 6.4.2,** this impact (remobilisation of contaminated sediments) was scoped out of the assessment for all phases.

### In-combination impact 4: Underwater noise and vibration (all phases)

252. There would be potential for piling and UXO clearance during construction of the Project and other windfarm projects, namely Morgan Offshore Wind Project, Transmission Assets, Mona Offshore Wind Project and AyM Offshore Wind Farm to result in in-combination effects on fish.

253. The largest potential in-combination effects would be the result of either spatial or temporal effects resulting from concurrent or sequential piling, and UXO clearance at different OWFs, or a combination of both.

254. As identified in **Appendix 11.1** of the ES, the worst-case range for mortality, and potential mortal injury, from a high order UXO detonation for the Project was 710m. In reality, the use of a high order detonation would be unlikely and only be used as a last resort, with low order deflagration of UXO preferred, with greatly reduced noise as a result. It was not expected that UXO clearance from the Project would be undertaken at the same time as piling for the Project, however UXO clearance from other sites was possible. With impact in the

order of that modelled for the Project and the fact that a blast would last for a very short duration, no in-combination effect was identified.

255. Project-alone piling effects have been outlined in **Section 7.4.2.1**. Similar noise ranges have been identified for the Transmission Assets, Mona Offshore Wind Project, Morgan Offshore Wind Project and AyM OWF.

256. River lamprey at sea typically occupy in inshore or estuarine habitat (Elliot *et al.,* 2021) and were not likely to be present in the ZoI of the Project. For sea lamprey, given their low sensitivity to noise (Hawkins and Johnstone, 1978; Popper, 2005), any noise-induced behavioural effects during migration were expected to be highly temporary and not detrimental to migration. For this reason, whilst similar temporary behavioural effects could arise from piling associated with other projects, these other impacts were also considered to be temporary and not detrimental to the migration as a whole. The closest piling activity to the SAC was at AyM OWF, and given the distance of that project from the mouth of the River Dee, no Project-alone adverse effects on integrity have been identified for the Project (Awel y Môr Offshore Wind Farm Ltd., 2022). Given the distance of the Project from the SAC (42km), and the distances of Mona and Morgan Offshore Wind Projects and the Transmission Assets no in-combination effects directly upon the SAC were identified.

257. Given that noise ranges for operation and maintenance would be highly localised and decommissioning would not require activities such as piling, the effects for these phases would be lower than for construction.

258. The Project in-combination with other plans and projects would have no AEoI on the Dee Estuary/Aber Dyfrdwy SAC due to underwater noise effects during construction, operation and maintenance or decommissioning.

### In-combination impact 5: Barrier effects (all phases)

259. Barrier effects could result from noise, suspended sediments and the physical presence of infrastructure from multiple windfarm projects within the IS.

260. Only sea lamprey have the potential to be present in the windfarm site. There was no pathway for effect upon river lamprey or sea lamprey in coastal waters away from the windfarm site.

261. While modelled noise contours (during construction) extended over tens of kilometres for each of the projects, effects would be intermittent, and, given the flexibility in homing behaviour and low sensitivity to noise of sea lamprey individuals, noise effects would present minimal risk of disruption to migration. Suspended sediments and the introduction of hard substrate would also be localised in the context of the species distribution. Furthermore, lamprey are known to tolerate silty turbid and poor light conditions (Behrmann-Godel and Eckmann, 2003; Hansen *et al*., 2016; Christoffersen *et al*., 2018).

262. The increase above background noise levels expected during operation for all projects would be very small and localised in nature and it was considered that in-combination effects from operational noise would not occur beyond Project-alone effects.

263. The Project in-combination with other plans and projects would have no AEoI on the Dee Estuary/Aber Dyfrdwy SAC due to barrier effects during construction, operation and maintenance or decommissioning.

### In-combination impact 6: EMF (operation and maintenance phase)

264. Given that river lamprey are restricted to coastal waters, there was no pathway for in-combination EMF effects upon them (i.e. there was no pathway for Project-alone effect, see **Section 7.4.2**). Given the distance of the Project windfarm site from the SAC, there would be no pathway for direct EMF effects within the SAC.

265. EMF effects from multiple projects would be additive across the region however, as EMF effects would be highly localised to within 10m of cabling (Taormina *et al.*, 2020), there would be no spatial overlap in effects given the distances between projects, and the distance to the SAC (see **Table 7.5**). The area around the cable where EMF would be elevated represented a very small fraction of the available habitat for sea lamprey even if multiple cables were encountered by an individual on any one day. Therefore, effects on sea lamprey from EMF were expected to be minimal.

266. The Project in-combination with other plans and projects would have no AEoI on the Dee Estuary/Aber Dyfrdwy SAC due to EMF effects during construction, operation and maintenance or decommissioning.

### In-combination impact 7: Introduction/removal of hard substrate

267. The area of hard substrate introduced within the Project windfarm site would be a worst-case of $0.4km^2$. The hard substrate may remain in place for the lifetime of the project and therefore the creation of any hard substrate habitat has been assessed as a permanent effect. The area of introduced hard substrate would represent a very small fraction of the available habitat for sea lamprey, which may travel multiple kilometres per day and are less likely to swim close to the seafloor (Snyder *et al*., 2019). Other windfarms constructed in the region, in additional to existing activities, would have similar scale effects, which would be additive.

268. Given the highly localised effects associated with introduction/removal of hard substrate habitat, the distance between the Project windfarm site and other projects (and the distance of projects from the SAC) and the wider available habitat, the in-combination impact of introduced hard substrate on populations

of migrating fish was not anticipated to be significantly greater than the effects of the Project-alone.

269. The Project in-combination with other plans and projects would have no AEoI on the Dee Estuary/Aber Dyfrdwy SAC due to introduction/removal of hard substrates during construction, operation and maintenance or decommissioning.

#### In-combination conclusion

270. Considering the assessment against the conservation objectives (**Section 7.4.1.2**), the Project-alone and In-combination would have no AEoI on the Dee Estuary/Aber Dyfrdwy SAC. This was largely due to the magnitude of effects, given the separation of the Project from the site. The confidence in the assessment was high, as per Project-alone.

#### 7.4.2.4 Summary

271. The Project, alone and in-combination with other plans and projects, would have no AEoI on the Dee Estuary/Aber Dyfrdwy SAC during construction, operation and maintenance or decommissioning.

## 7.5 River Dee and Bala Lake/Afon Dyfrdwy a Llyn Tegid SAC

### 7.5.1 Description of designation

272. The site represents an area of 11.5km$^2$ and extends from the western extremity of Llyn Tegid covering the lake and its banks to its outfall into the River Dee. It then takes in the river and its banks downstream to where it joins the Dee Estuary Site of Special Scientific Interest (SSSI). Parts of the Rivers Dee and Ceiriog lie within both Wales and England.

#### 7.5.1.1 Qualifying features

273. The site was assessed for the following Annex II migratory fish species:

- Atlantic salmon *Salmo salar.*

274. Annex II species present as a qualifying feature, but not a primary reason for site selection

- River lamprey
- Sea lamprey

#### 7.5.1.2 Conservation objectives

275. The conservation objectives of the SAC are to ensure that the integrity of the site is maintained or restored as appropriate and ensure that the site contributes to achieving the FCS of its Qualifying Features. The parameters defined in the vision for the watercourse that must be met are:

- The SAC feature populations will be stable or increasing over the long term
- The natural range of the features in the SAC is neither being reduced nor is likely to be reduced for the foreseeable future
- There will be no reduction in the area or quality of habitat for the feature populations in the SAC on a long-term basis
- All known, controllable factors, affecting the achievement of these conditions are under control (many factors may be unknown or beyond human control) (JNCC, 2022b).

#### 7.5.1.3 Condition assessment

276. The conservation status of features of the SAC were assessed by NRW for the development of the core management plan. For Atlantic salmon the assessment determined that conditions were unfavourable based on population estimates, water quality and levels of environmental disturbance. For sea lamprey, condition was assessed as unfavourable un-classified, based on low numbers of ammocoetes recorded. The condition for river lamprey was assessed as unfavourable un-classified due to the low numbers recorded (NRW, 2021).

### 7.5.2 Assessment

#### 7.5.2.1 Assessment of potential effects of the Project-alone

277. Given that the SAC is up river of the Dee Estuary/Aber Dyfrdwy SAC (assessed in **Section 7.4.2**), the assessment for lamprey species in relation to the Dee Estuary/Aber Dyfrdwy SAC was considered to be applicable. Therefore, with regard to the sea lamprey and river lamprey features the Project would have no AEoI on the River Dee and Bala Lake/Afon Dyfrdwy a Llyn Tegid SAC during construction, operation and maintenance or decommissioning.

278. As such, from this point onwards, this section considers only Atlantic salmon.

### Impact 1: Increased SSCs and deposition (all phases)

279. During construction, and to a lesser degree operation and maintenance and decommissioning activities, there may be a temporary increase in SSCs and deposition which may interact with Atlantic salmon migrating from the River Dee and Bala Lake/Afon Dyfrdwy a Llyn Tegid SAC.

280. Suspended sediment has the potential to impair respiratory or reproductive functions, including the disruption of migration/spawning activity. Sediment deposition, especially if it changes the characteristics of the existing seabed sediments, could affect the quality of spawning and nursery habitats.

281. Whilst limited information exists on the impacts of suspended sediment on Atlantic salmon, salmon species are known to successfully migrate through estuaries that have naturally high suspended sediment levels to enter rivers and increased turbidity may lead to lower rates of predation (Gillson *et al.,* 2022). As these species are all highly mobile and active in the water column above the seabed, there would also be no risk of smothering or burial.

282. With the sediment distribution during any phase of the project extending to a maximum of 10km (see **Section 6.4.2**), at over 60km from the SAC, no fish within the SAC or its supporting habitats would be impacted by the Project. Migrating individuals could feasibly cross the Project windfarm site (and area impacted by increased SSCs) during migration to or from freshwater. During this time, they could be exposed to increased water column sediment loading for a limited period of time, associated with each disturbance activity. The increased sediment loading would be short-term and localised in nature, occurring sequentially with the location of the installation activity.

283. As discussed previously, the highest increases in SSCs would be within 1km of the release point. Therefore, the likelihood of Atlantic salmon encountering an area of increased water column sediment loading was low. Furthermore, as they were highly mobile species, should they encounter an area of suspended sediments, they were capable of moving to avoid the area.

284. The Project-alone would have no AEoI on the River Dee and Bala Lake/Afon Dyfrdwy a Llyn Tegid SAC due to increases in SSCs and deposition during construction, operation and maintenance or decommissioning.

### Impact 2: Temporary or permanent habitat loss (all phases)

285. Given the distance of the Project windfarm site from the SAC there would be no direct habitat loss within the SAC.

286. Although Atlantic salmon could be present at the windfarm site, there was no habitat type within the Project windfarm site that would be particularly important to them or that was not common across the region. In addition, Atlantic salmon have high levels of mobility, they would therefore be capable

of navigating away from any temporary physical disturbance/habitat loss caused by construction, operation and maintenance or decommissioning activities.

287. The Project-alone would have no AEoI on the River Dee and Bala Lake/Afon Dyfrdwy a Llyn Tegid due to temporary or permanent habitat loss during construction, operation and maintenance or decommissioning.

### Impact 3: Remobilisation of contaminated sediments (all phases)

288. As described in **Section 6.4.2**, this impact (remobilisation of contaminated sediments) was scoped out of the assessment for all phases.

### Impact 4: Underwater noise and vibration (all phases)

289. Studies by Hawkins and Johnstone (1978) found salmon showed low sensitivity to noise. Their ability to respond to noise was regarded as poor with a narrow frequency span and a limited ability to discriminate between different noises. The swim bladder does not play a role in the hearing of Atlantic salmon.

290. As a close relative of salmon, sea trout were used as a model to determine the possible implications to salmon during piling operations at Southampton Water in 2003. Nedwell *et al.,* (2008) presented the results from the study conducted simultaneously to the piling operations. Nedwell *et al.,* (2008) found no obvious signs of trauma in any examined fish and no increase in activity or startle response was observed at any range from the piling.

291. Laboratory work on brown trout has shown that repeated sine sweeps (up to 2kHz), and, more relevant to piling, intermittent 140Hz tones did not affect swimming behaviour (Jesus *et al*., 2019). Further, high intensity (114dB above the hearing threshold) low frequency sound at 150Hz had no effect on downstream smolt migration (Knudsen *et al*., 2005). At high intensities, very low frequency infrasound of 10Hz did deter smolt movement (Jesus *et al*., 2019), but the vast majority of sound energy in a pile frequency spectrum was contained at frequencies above 20Hz (Gill *et al*., 2012). Overall, the evidence suggested that changes to salmonid swimming behaviour during migration may occur only in extreme proximity to the piles.

292. Salmon have been assessed as fish species with a swim bladder not involved in hearing (Group 2, (Popper *et al.,* 2014)). Underwater noise impact ranges from modelling from piling are presented in **Table 7.6**. UXO detonation would be further assessed when details of any UXO present in the Project windfarm site are available, however modelling has been provided in **Appendix 11.1** of the ES. Impact ranges during operation and maintenance would be less than 50m around each WTG. Further modelling results from vessel activities and

other operation and maintenance activities have been provided in the noise modelling report (**Appendix 11.1** of the ES).

*Table 7.6 Single piling and sequential piling within a 24-hour period underwater noise modelling results for both a single monopile and four sequential pin piles with maximum hammer energies, for the worst-case modelling location only (using a stationary animal model) for River Dee and Bala Lake/Afon Dyfrdwy a Llyn Tegid SAC*

| Fish group | Species included | Impact criteria | Potential impact | Impact areas and ranges | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | | Monopile (maximum hammer energy 6,600kJ) (SELcum relates to three sequential monopiles within 24 hours) | | | | Pin pile (maximum hammer energy 2,500kJ) ($SEL_{cum}$ relates to four sequential pin piles within 24-hours) | | | |
| | | | | Area | Max | Min | Mean | Area | Max | Min | Mean |
| Fish group 2: swim bladder is not involved in hearing (particle motion detection) | Atlantic salmon | >207 dB unweighted $SPL_{peak}$ | Mortality and potential mortal injury | 0.32km$^2$ | 320m | 320m | 320m | 0.19km$^2$ | 250m | 250m | 250m |
| | | 210 dB unweighted $SEL_{cum}$ [stationary] | Mortality and potential mortal injury | 100km$^2$ | 6km | 5.4km | 5.6km | 60km$^2$ | 4.6km | 4.2km | 4.4km |
| | | 203 dB unweighted $SEL_{cum}$ [stationary] | Recoverable injury | 360km$^2$ | 12km | 9.4km | 11km | 240km$^2$ | 9.6km | 8.0km | 8.8km |
| | | >186 dB unweighted $SEL_{cum}$ [stationary] | TTS | 2400km$^2$ | 33km | 20km | 27km | 1900km$^2$ | 30km | 19km | 25km |

293. Given the impact ranges for noise, it is unlikely that noise levels generated during any phase of the Project would affect feeding and migration behaviours of Atlantic salmon. The impact ranges for injury and behavioural effects would not reach the SAC, which is over 60km from the windfarm site. Only individuals outside the SAC could interact with any impact, and therefore, noise levels generated during construction of the Project would not affect spawning activity. Atlantic salmon typically migrate in coastal waters and interaction with the Project windfarm site and areas within impact ranges for mortality and injury would be low. While impact ranges for behavioural effects would be more wide reaching, effects would be temporally limited and unlikely to affect migratory behaviour.

294. The Project-alone would have no AEoI on the River Dee and Bala Lake/Afon Dyfrdwy a Llyn Tegid SAC due to underwater noise effects during construction, operation and maintenance or decommissioning.

### Impact 5: Barrier effects (all phases)

295. Atlantic salmon have the potential to be present within the range of underwater noise effects, but individuals are considered to have low sensitivity to noise, and noise impact ranges would not reach the SAC or surrounding coastal waters. Therefore, it was considered that noise effects would not present a barrier to migration for fish moving through the wider IS.

296. Suspended sediments and the introduction of hard substrate would also be localised in the context of the species distribution.

297. The Project-alone would have no AEoI on the River Dee and Bala Lake/Afon Dyfrdwy a Llyn Tegid SAC due to barrier effects during construction, operation and maintenance or decommissioning.

### Impact 6: EMF (operation and maintenance phase)

298. EMF has the potential to interfere with the navigation of sensitive migratory and pelagic species by affecting the speed and/or course of their movements through the windfarm site, causing subsequent potential issues if they were not able to reach spawning, nursery or feeding grounds. Studies conducted by Marine Scotland Science (Armstrong *et al.,* 2016) and Walker (2001) found no evidence of unusual behaviour in Atlantic salmon associated with EMFs produced by cables.

299. Most EMF exposures would be expected to be short, in the order of minutes, whilst these highly mobile species would be moving through the windfarm site. The area around the cable where EMF would be elevated would be small (less

than 10m[16]). Given the distance of the Project windfarm site from the SAC, there would be no direct EMF effects within the SAC.

300. The area around the cable where EMF would be elevated represented a very small fraction of the available habitat for Atlantic salmon outwith the SAC. Effects on Atlantic salmon (if present within the windfarm site) from EMF would be expected to be minimal.

301. The Project-alone would have no AEoI on the River Dee and Bala Lake/Afon Dyfrdwy a Llyn Tegid SAC due to EMF effects during construction, operation and maintenance or decommissioning.

### Impact 7: Introduction/removal of hard substrate (all phases)

302. There would be no introduction or removal of hard substrate within the SAC.

303. The area of hard substrate introduced within the Project windfarm site would be a worst-case of $0.4km^2$. The area of introduced hard substrate would represent a very small fraction of the available habitat available to migrating Atlantic salmon. Any introduced hard substrate would not create a significant amount of habitat that could impact migrating Atlantic salmon.

304. The Project-alone would have no AEoI on the River Dee and Bala Lake/Afon Dyfrdwy a Llyn Tegid SAC due to introduction/removal of hard substrate during construction, operation and maintenance or decommissioning.

### Project-alone conclusion

305. Considering the assessment against the conservation objectives, **Section 7.5.1.2**, the Project would have no AEoI on the River Dee and Bala Lake/Afon Dyfrdwy a Llyn Tegid SAC. This is largely due to the magnitude of effects, given the separation of the Project to the site. The confidence in the assessment was high and aligned with the assessment presented in **Chapter 10 Fish and Shellfish Ecology** of the ES.

### Potential interactions of Project effects

306. Interactions of Project effects would be as per those outlined in **Section 7.4.2.1**. It was therefore considered that none of these impacts would interact to increase the significance level overall.

[16] Based on Taormina *et al.,* (2020) analysis of export and interconnector cables.

#### 7.5.2.2 In-combination assessment – the Project and Transmission Assets combined

307. A 'combined' assessment has been made with the Transmission Assets[17], for the purpose of an in-combination assessment considering its functional link with the Project.

308. This section provides assessment of impact interactions and additive effects for the River Dee and Bala Lake/Afon Dyfrdwy a Llyn Tegid SAC which was screened in for both the Project and Transmission Assets.

#### In-combination impact 1: Increased SSCs and deposition (all phases)

309. Whilst limited information exists on the impacts of suspended sediment on Atlantic salmon, salmon species have been known to successfully migrate through estuaries that have naturally high suspended sediment levels to enter rivers and increased turbidity may lead to lower rates of predation (Gillson *et al*., 2022). As these species are all highly mobile and active in the water column above the seabed, there would also be no risk of smothering or burial.

310. The predicted combined volume of material likely to be disturbed during the construction phase of the Project and the Transmission Assets would be in the region of 13.4 million $m^3$. This includes approximately 1.1 million $m^3$ associated with the Project (see **Table 7.3**) plus c.12.3 million $m^3$ associated with the Transmission Assets (Morgan Offshore Wind Limited and Morecambe Offshore Windfarm Ltd, 2023a).

311. As described in **Section 7.4.2.1**, 'heavy' deposition would only occur within a very short distance of the source of disturbance; at more than 1km distance SSC increases and deposition levels would be low, becoming indistinguishable from background within a maximum ZoI of 10km. Therefore, the likelihood of fish encountering an area of increased water column sediment loading would be low.

312. Given the relationship of the Project and the Transmission Assets, site preparation and installation of infrastructure would be phased and SSC increases are unlikely to occur concurrently. However, should multiple operations be undertaken simultaneously, plumes would be advected on the tide (not towards one another). Activities would be of limited spatial extent and plume interactions of a low magnitude and short duration. For both projects, the majority of sedimentation would occur within close proximity of each installation activity; however, given the active sediment transport regime,

[17] As the Transmission Assets includes infrastructure associated with both the Project and the Morgan Offshore Wind Project Generation Assets, it should be noted that the combined assessment considers the transmission infrastructure for both the Project and the Morgan Offshore Wind Project Generation Assets.

deposited material would be redistributed. Given the distance of the SAC from both projects, and the context of localised effects across the IS the magnitude of any effect would be limited.

313. As any interaction of sediment plumes and deposition would be localised (i.e. of small spatial extent) and temporary, there would be no adverse in-combination effect on the integrity of the River Dee and Bala Lake/Afon Dyfrdwy a Llyn Tegid SAC.

### In-combination impact 2: Temporary or permanent habitat loss (all phases)

314. The assessment for Atlantic salmon reflected that detailed for Lamprey, see **Section 7.4.2**. In summary, changes from soft to hard substrate (from both the Project and Transmission Assets infrastructure) would not impact on the ability of a fish to migrate through the region to and from the SAC.

315. The Project in-combination with Transmission Assets would have no AEoI on the River Dee and Bala Lake/Afon Dyfrdwy a Llyn Tegid SAC due to temporary or permanent habitat loss (all phases).

### In-combination impact 3: Remobilisation of contaminated sediments (all phases)

316. As described in **Section 6.4.2**, this impact (remobilisation of contaminated sediments) was scoped out of the assessment for all phases.

### In-combination impact 4: Underwater noise and vibration (all phases)

317. The key components of the Transmission Assets that would require piling comprise up to four OSPs at Morgan, up to two OSPs at Morecambe, and the Morgan offshore booster station. UXO clearance for both projects may also be required.

318. The construction phase of the Transmission Assets may have temporal and spatial overlap with the Project in terms of sound associated with piling and UXO clearence, potentially resulting in a cumulative impact. The assessment of sound impacts associated with piling for the Project-alone has been presented above (**Section 7.4.2.1**), with a low magnitude identified based on a range of technical specifications and sound modelling outputs. There would be the potential for piling to occur concurrently at the Project and the Morgan offshore booster substation and Morgan OSP(s).

319. Sound modelling for the Transmission Assets indicated similar patterns as those for the Project, with injury and mortality from sound produced within the Transmission Assets for a single monopile (maximum hammer energy of 5,500kJ to ranges of up to 2,020m for Group 2 fish (the Atlantic salmon hearing group), if modelled as stationary receptors (Morgan Offshore Wind Limited and Morecambe Offshore Windfarm Ltd, 2023a). See **Section 7.4.2.1** for an

explanation of fish sound sensitivity groups. Recoverable injury distances were calculated to reach out to up to 4,340m for Group 2 stationary receptors with similar patterns for all other groups of fish, in comparison to the worst-case 7.1km modelled for a single monopile for the Project (see **Appendix 11.1**).

320. Overall, the short piling duration expected for the Transmission Assets would only represent a very short-term increase in the ensonified area when considered cumulatively with planned piling at the Project.

321. The construction phase of the Transmission Assets may also have temporal overlap with the Project in terms of UXO clearance, potentially resulting in a cumulative impact with construction activities. Similar to the Project, the Transmission Assets have developed a list of UXO threat items based on expert opinion, assessing a higher maximum potential explosive quantity of 907kg within their study area. The assessment for UXO clearance for the Transmission Assets determined a low magnitude for the impact, and based on modelling, found similar mortality and potential mortal injury ranges for high order detonations of explosive quantities of 1.2kg to 907kg with ranges up to 590m (Morgan Offshore Wind Limited and Morecambe Offshore Windfarm Ltd, 2023a), with the Project finding equivalent maximum impact ranges of up to 710m.

322. As noted for the Project-alone assessment, there would be a short term, intermittent impact, which remains consistent with the addition of the Transmission Assets.

323. In this context, the additional piling and UXO clearance from the Transmission Assets for a short duration did not alter the findings of the Project-alone assessment.

324. Given that noise ranges for operation and maintenance would be highly localised and decommissioning would not require activities such as piling, the effects for these phases would be lower than for construction.

325. The Project in-combination with Transmission Assets would have no AEoI on the River Dee and Bala Lake/Afon Dyfrdwy a Llyn Tegid SAC due to underwater noise effects during construction, operation and maintenance or decommissioning.

### In-combination impact 5: Barrier effects (all phases)

326. As discussed for the Project-alone (**Paragraphs 295** to **297**) Atlantic salmon would have the potential to be present within the range of underwater noise effects, but individuals were considered to have low sensitivity to noise, and noise impact ranges would not reach the SAC or surrounding coastal waters. Therefore, it was considered that noise effects would not present a barrier to

migration for fish moving through the wider IS. Addition of the Transmission Assets would only represent a very short-term increase in the ensonified area.

327. As discussed above (**Paragraph 313**) any interaction of sediment plumes and deposition would be localised (i.e. of small spatial extent) and temporary and the introduction of hard substrate would also be localised in the context of the species distribution.

328. The Project in-combination with the Transmission Assets would have no AEoI on the River Dee and Bala Lake/Afon Dyfrdwy a Llyn Tegid SAC due to barrier effects during construction, operation and maintenance or decommissioning.

### In-combination impact 6: EMF (operation and maintenance phase)

329. Given the distance of the Project and Transmission Assets from the SAC there would be no pathway for direct EMF effects within the SAC.

330. As EMF effects would be highly localised to within 10m of cabling (Taormina *et al.*, 2020), there would be no spatial overlap in effects given the distances between the Project and Transmission Assets (see **Table 7.5**). The area around the cable where EMF would be elevated represents a very small fraction of the available habitat for Atlantic salmon, even if multiple cables were encountered by an individual on any one day. Therefore, effects on Atlantic salmon from EMF were expected to be minimal.

331. The Project in-combination with Transmission Assets would have no AEoI on the River Dee and Bala Lake/Afon Dyfrdwy a Llyn Tegid SAC due to EMF effects during construction, operation and maintenance or decommissioning.

### In-combination impact 7: Introduction/removal of hard substrate

332. As discussed for the Project-alone (**Paragraphs 302** to **304**) there would be no introduction or removal of hard substrate within the SAC, and this would remain the case In-combination with the Transmission Assets.

333. The area of hard substrate introduced within the Project windfarm site would be a worst-case of 0.4km$^2$. The Transmission Assets would contribute an additional 1.5km$^2$ of hard substrate.

334. Given the highly localised effects associated with introduced hard substrate habitat (see **Section 7.4.2.1**), the small areas affected and the distance of the projects from the SAC the impact of introduced (and removal of) hard substrate for the Project and the Transmission Assets would be limited.

335. Introduced hard substrate is likely to be colonised by encrusting organisms. This hard substrate-associated biological community may in turn attract predators to feed on the encrusting organisms. This change in community

could feasibly alter predator-prey dynamics to the benefit or detriment of migratory fish species which associate with the new hard substrate.

336. The area of introduced hard substrate represents a very small fraction of the available habitat available to migrating Atlantic salmon. Any introduced hard substrate would not create a significant amount of hard substrate habitat (and associated biological communities) that could impact migrating Atlantic salmon.

337. The Project in-combination with other plans and projects would have no AEoI on the River Dee and Bala Lake/Afon Dyfrdwy a Llyn Tegid SAC due to introduction/removal of hard substrates during construction, operation and maintenance or decommissioning.

#### 7.5.2.3 Assessment of potential effects of the Project in-combination with other plans and projects

338. Given the orientation of the SAC upstream of the Dee Estuary, the same activities, plans and projects have been considered (**Table 7.5** and **Figure 7.1**).

#### In-combination impact 1: Increased SSCs and deposition

339. As detailed in **Section 6.4.2**, the ZoI for increases in SSCs for the Project during the construction phase (the phase during which the greatest amount of suspended sediment would be produced) is 10km (approximately the spring tidal excursion in an east-west orientation). The direction of travel of sediment plumes from other projects would be dictated by the directionality of the currents at the time of the works associated with those projects. This means that sediment plumes from nearby projects (if occurring at the same time as construction of the Project) would likely travel in a parallel direction to sediment plumes from the Project.

340. For sediment plumes from multiple projects to interact, the projects would need to be within 10km of the Project windfarm site with works occurring simultaneously, this includes the Transmission Assets, Mona Offshore Wind Project, Isle of Man Interconnector (cable protection remedial works) as well as the Liverpool Bay aggregate production area. However, it was only within the nearfield (maximum of 1km) where suspended levels were expected to be distinguishable beyond background levels. Given the distance of the SAC at over 60km from the site, there would be no potential for suspended sediment plumes to coalesce within the SAC and therefore no potential for direct in-combination effects.

341. The Transmission Assets would have the potential for overlap of the highest suspended sediments in the near-field but effects would be limited in temporal and spatial extent (assuming that construction was simultaneous). Therefore,

the likelihood of fish encountering an area of increased water column sediment loading is low. Furthermore, salmon species have been known to successfully migrate through estuaries that have naturally high suspended sediment levels to enter rivers and increased turbidity may lead to lower rates of predation (Gillson *et al*., 2022).

342. All other plans and projects would be outwith 1km (suspended sediments would have reduced rapidly after this distance), and, as such, in-combination effects would be unlikely to occur.

343. Given that the amount of suspended sediment produced would be highest during construction, the effects for operation and maintenance and decommissioning would be lower than for construction.

344. The Project, in-combination with other plans and projects, would have no AEoI on the River Dee and Bala Lake/Afon Dyfrdwy a Llyn Tegid SAC due to increases in SSCs and deposition during construction, operation and maintenance or decommissioning.

### In-combination impact 2: Temporary or permanent habitat loss

345. In terms of temporary habitat loss during construction, the habitat types found within the Project windfarm site have a high recoverability, and the temporary habitat disturbance associated with this Project and other projects identified in **Table 7.5** would be negligible in the context of wider disturbance in the region from, for example, mobile fishing.

346. In terms of permanent habitat loss, there would be the potential for incremental additional effects resulting from the loss of habitat due to the construction of other planned OWFs in the region. Morgan Offshore Wind Project, Transmission Assets, Mona Offshore Wind Project and AyM OWF are all planned to be constructed in the region and would therefore cause additional permanent habitat loss.

347. There are no habitat types within other planned OWFs in the region that are of particular importance to Atlantic salmon or that are not common across the region. In addition, Atlantic salmon have high levels of mobility, they would therefore be capable of navigating away from any temporary physical disturbance/habitat loss caused by construction, operation and maintenance or decommissioning activities.

348. Given the localised effects associated habitat loss, the distance between the Project windfarm site and other projects which (and the distance of projects from the SAC) (**Table 7.5**) and the wider availability of un-modified habitat, the in-combination impact of introduced hard substrate on populations of migrating Atlantic salmon is not anticipated to be significantly greater than the effects of the Project-alone.

349. The Project, in-combination with other plans and projects, would have no AEoI on the River Dee and Bala Lake/Afon Dyfrdwy a Llyn Tegid SAC during construction, operation and maintenance or decommissioning.

### In-combination impact 3: Remobilisation of contaminated sediments

350. As described in **Section 6.4.2**, this impact (remobilisation of contaminated sediments) was scoped out of the assessment for all phases.

### In-combination impact 4: Underwater noise and vibration

351. There is potential for piling and UXO clearance during construction of the Project and other windfarm projects, namely Morgan Offshore Wind Project, Transmission Assets, Mona Offshore Wind Project and AyM OWF to result in in-combination effects on fish.

352. The largest potential in-combination effects would be the result of either spatial or temporal effects resulting from concurrent or sequential piling, and UXO clearance at different OWFs, or a combination of both.

353. As identified in **Appendix 11.1** of the ES, the worst-case range for mortality, and potential mortal injury, from a high order UXO detonation was 710m. In reality, the use of a high order detonation would be unlikely and would only be used as a last resort, with low order deflagration of UXO preferred, with greatly reduced noise as a result. It was not expected that UXO clearance from the Project would be undertaken at the same time as piling for the Project, however UXO clearance from other sites would be possible. With impact ranges in the order of that modelled for the Project and the fact that a blast would last for a very short duration, no in-combination effect was identified.

354. Project-alone piling effects have been outlined in **Section 7.5.2.1**.

355. Similar noise ranges have been identified for the Transmission Assets, Mona Offshore Wind Project, Morgan Offshore Wind Project and AyM OWF.

356. For Atlantic salmon, given their relatively low sensitivity to noise (Hawkins and Johnstone, 1978; Popper, 2005), any noise-induced behavioural effects would not be expected to be detrimental to migration. For this reason, whilst similar temporary behavioural effects could arise from piling associated with other projects to Atlantic salmon before or after passing through the windfarm site, these other impacts were also considered to be temporary and not detrimental to the migration activities as a whole. The closest piling activity to the SAC would be at AyM OWF and given the distance of that project from the River Dee and Bala Lake/Afon Dyfrdwy a Llyn Tegid SAC (>20 km at its nearest point), no Project-alone AEoI was identified (Awel y Môr Offshore Wind Farm Ltd., 2022a). Given the distance of the Project from the SAC (60km), and the distances from the SAC of Mona and Morgan OWFs and the Morgan and

Morecambe OWFs Transmission Assets, no in-combination effects directly upon the SAC were identified.

357. The Project, in-combination with other plans and projects, would have no AEoI on the River Dee and Bala Lake/Afon Dyfrdwy a Llyn Tegid SAC.

### In-combination impact 5: Barrier effects (all phases)

358. Barrier effects could result from noise, suspended sediments and the physical presence of infrastructure from multiple projects within the IS.

359. While noise contours would extend over tens of kilometres for each of the windfarm projects (during construction), effects would be intermittent. As noted above, migrating Atlantic salmon have a low sensitivity to noise and effects would present minimal risk of disruption to migration across the IS. Atlantic salmon typically migrate in coastal waters and interaction with the Project windfarm site and other projects and areas within impact ranges for mortality and injury would be low. While impact ranges for behavioural effects would be more wide reaching, effects would be temporally limited and unlikely to affect migratory behaviour.

360. The increase above background noise levels expected during operation for all projects would be very small and localised in nature. It was therefore considered that in-combination effects from operational noise would not occur beyond Project-alone effects.

361. Suspended sediments and the introduction of hard substrate would also be localised in the context of the species distribution. Furthermore, Atlantic salmon have been known to successfully migrate through estuaries that have naturally high suspended sediment levels to enter rivers and increased turbidity may lead to lower rates of predation (Gillson *et al.,* 2022). The separation between projects, and the westerly direction of tidal currents, also means limited effects have been considered in relation to suspended sediments and the introduction of hard substrate as a physical barrier.

362. The Project, in-combination with other plans and projects, would have no AEoI on the River Dee and Bala Lake/Afon Dyfrdwy a Llyn Tegid SAC from barrier effects during construction, operation and maintenance or decommissioning.

### In-combination impact 6: EMF (operation and maintenance phase)

363. Given the distance of the windfarm site, and other projects, from the SAC there would be no pathway for direct EMF effects within the SAC.

364. As EMF effects would be highly localised to within 10m of cabling (Taormina *et al.*, 2020) there would be no spatial overlap in effects given the distances between projects (see **Table 7.5**). The area around the cable where EMF would be elevated represented a very small fraction of the available habitat

for Atlantic salmon, even if multiple cables were encountered by an individual on any one day. Therefore, effects on Atlantic salmon from EMF were expected to be minimal.

365. The Project, in-combination with other plans and projects, would have no AEoI on the River Dee and Bala Lake/Afon Dyfrdwy a Llyn Tegid SAC from EMF effects during construction, operation and maintenance or decommissioning.

### In-combination impact 7: Introduction/removal of hard substrate (all phases)

366. The area of hard substrate introduced within the Project windfarm site would be a worst-case of $0.4km^2$. The hard substrate would remain in place for the lifetime of the project and therefore the creation of any hard substrate habitat has been assessed as a permanent effect. The area of introduced hard substrate would represent a very small fraction of the available habitat. Other windfarms constructed in the region would have similar scale effects which would be additive.

367. Given the highly localised effects associated with introduction/removal of hard substrate habitat, the distance between the Project windfarm site and other projects (and the distance of projects from the SAC) and the wider available habitat, the in-combination impact of introduced hard substrate on populations of migrating fish was not anticipated to be significantly greater than the effects of the Project-alone.

368. The Project, in-combination with other plans and projects, would have no AEoI on the River Dee and Bala Lake/Afon Dyfrdwy a Llyn Tegid SAC due to introduction/removal of hard substrates during construction, operation and maintenance or decommissioning.

### In-combination conclusion

369. Considering the assessment relative to the conservation objectives, **Section 7.5.1.2**, the Project-alone and In-combination would have no AEoI on the River Dee and Bala Lake/Afon Dyfrdwy a Llyn Tegid SAC. This is largely due to the magnitide of effects, given the separation of the Project to the site. The confidence in the assessment was high, as per Project-alone.

### 7.5.2.4 Summary

370. The Project, alone and in-combination with other plans and projects, would have no AEoI on the River Dee and Bala Lake/Afon Dyfrdwy a Llyn Tegid SAC during construction, operation and maintenance or decommissioning. This conclusion relates to lamprey (as assessed in detail in **Section 7.4.2**) and Atlantic salmon (assessed above). The confidence in this assessment was as per Project-alone.

# 7.6 Afon Gwyrfai a Llyn Cwellyn SAC

## 7.6.1 Description of designation

371. The Afon Gwyrfai a Llyn Cwellyn SAC covers an area of 1.1km$^2$ and was designated for the population of Atlantic salmon within the site. The SAC is representative of the small montane rivers in this region. It contains a largely unexploited salmon population with a characteristically late spawning run. Environment Agency electrofishing data has indicated the presence of healthy juvenile populations downstream of Llyn Cwellyn (NRW, 2022a).

### 7.6.1.1 Qualifying feature

372. The site is designated for Annex II species Atlantic salmon.

### 7.6.1.2 Conservation objectives

373. The conservation objectives of the SAC are to ensure that the integrity of the site is maintained or restored as appropriate, and ensure that the site contributes to achieving the FCS of its Qualifying Features.

### 7.6.1.3 Condition assessment

374. The conservation status of the features of the SAC were assessed by NRW to develop the core management plan. The assessment determined that the condition was unfavourable for Atlantic salmon.

## 7.6.2 Assessment

375. The site is over 80km (to the river mouth) from the windfarm site, at a greater distance than the River Dee and Bala Lake/Afon Dyfrdwy a Llyn Tegid SAC. Given their distance from the Project, effects have been considered to be similar or less than assessed for River Dee and Bala Lake/Afon Dyfrdwy a Llyn Tegid SAC for both Project-alone and in-combination effects. Therefore, the detailed assessment provided in **Section 7.5** has not been repeated for this site and the conclusions for the River Dee and Bala Lake/Afon Dyfrdwy a Llyn Tegid SAC have been applied.

376. The Project, alone, in-combination with the Transmission Assets, and in-combination with other plans and projects, would have no AEoI on the Afon Gwyrfai a Llyn Cwellyn SAC.

# 7.7 Afon Eden - Cors Goch Trawsfynydd SAC

## 7.7.1 Description of designation

377. The Afon Eden/River Eden - Cors Goch Trawsfynydd SAC represents a relatively unmodified river, mainly upland in character, of approximately 10km in length. The SAC encompasses an area of 2.8km$^2$. The watershed begins just south of Llyn Trawsfynydd, within an area of gently sloping and poorly drained land. Atlantic salmon are known to migrate into the catchment to spawn and develop through their juvenile stages in the river, and have been present in numbers that reflect a healthy and sustainable population supported by well distributed good-quality habitat (NRW, 2022b).

### 7.7.1.1 Qualifying feature

378. The site is designated for Annex II species Atlantic salmon.

### 7.7.1.2 Conservation objectives

379. The conservation objectives of the SAC are to ensure that the integrity of the site is maintained or restored as appropriate, and ensure that the site contributes to achieving the FCS of its Qualifying Features.

### 7.7.1.3 Condition assessment

380. The conservation status of features of the SAC were assessed by NRW for developing the core management plan. The status of Atlantic salmon was determined to be unfavourable, which was due to concern over physical barriers to the adult run existing within the river system (NRW, 2022b).

## 7.7.2 Assessment

381. The site is over 90km (to the river mouth) from the windfarm site, at a greater distance than the River Dee and Bala Lake/Afon Dyfrdwy a Llyn Tegid SAC. Given their distance from the Project, effects were considered to be similar or less than assessed for River Dee and Bala Lake/Afon Dyfrdwy a Llyn Tegid SAC for both Project-alone and in-combination effects. Therefore, the detailed assessment provided in **Section 7.5** has not been repeated for this site and the conclusions for the River Dee and Bala Lake/Afon Dyfrdwy a Llyn Tegid SAC have been applied.

382. The Project, alone, in-combination with the Transmission Assets, and in-combination with other plans and projects, would have no AEoI on the Afon Eden - Cors Goch Trawsfynydd SAC.

# 7.8 River Ehen SAC

## 7.8.1 Description of designation

383. The River Ehen is an oligotrophic river in west Cumbria, spanning the Cumbria High Fells and West Cumbria Coastal Plain National Character Areas. Over half of the upper portion of this site is either within or on the boundary of the Lake District National Park.

### 7.8.1.1 Qualifying feature

384. The site is designated for Annex II species Atlantic salmon, river lamprey, and sea lamprey.

### 7.8.1.2 Conservation objectives

385. The conservation objectives of the SAC are to ensure that the integrity of the site is maintained or restored as appropriate, and ensure that the site contributes to achieving the Favourable Conservation Status of its Qualifying Features, by maintaining or restoring:

- The extent and distribution of the habitats of qualifying species
- The structure and function of the habitats of qualifying species
- The supporting processes on which the habitats of qualifying species rely
- The populations of qualifying species
- The distribution of qualifying species within the site

### 7.8.1.3 Condition assessment

386. The conservation status of all features was assessed unfavourable by Natural England in their Supplementary Advice on Conservation Objectives (SACO) in 2022.

## 7.8.2 Assessment

387. The site is over 70km (to the mouth of the river) from the windfarm site, at a greater distance than the River Dee and Bala Lake/Afon Dyfrdwy a Llyn Tegid SAC, and Dee Estuary/Aber Dyfrdwy SAC. Given their distance from the Project, effects were considered to be similar or less than assessed for River Dee and Bala Lake/Afon Dyfrdwy a Llyn Tegid SAC and Dee Estuary/Aber Dyfrdwy SAC for both Project-alone and in-combination effects. Therefore, the detailed assessments provided for these sites in **Sections 7.4** and **7.5** have not been repeated for this site and the conclusions for the River Dee and Bala

Lake/Afon Dyfrdwy a Llyn Tegid SAC, and Dee Estuary/Aber Dyfrdwy SAC have been applied.

388. The Project, alone and in-combination with other plans and projects (including the Transmission Asets), would have no AEoI on the River Ehen SAC.

## 7.9 River Derwent and Bassenthwaite Lake SAC

### 7.9.1 Description of designation

389. The Derwent is a large nutrient poor (oligotrophic) river system within the West Cumbria Coastal Plain and the Cumbria High Fells National Character Area, with high water quality and a natural channel. There is a natural succession of plant communities from source to mouth, reflecting a slight increase in nutrient status downstream. The Derwent flows through two lakes (Derwent Water and Bassenthwaite), as does its major tributary the Cocker (Buttermere and Crummock Water). These lakes have a hydrological buffering effect which helps stabilise the flow regimes.

#### 7.9.1.1 Qualifying feature

390. The site is designated for Annex II species Atlantic salmon, river lamprey, brook lamprey and sea lamprey.

#### 7.9.1.2 Conservation objectives

391. The conservation objectives of the SAC are to ensure that the integrity of the site is maintained or restored as appropriate, and ensure that the site contributes to achieving the FCS of its Qualifying Features.

#### 7.9.1.3 Condition assessment

392. Unknown at time of designation and not updated at the time of assessment.

### 7.9.2 Assessment

393. The site is over 70km (to the mouth of the river) from the windfarm site, at a greater distance that the River Dee and Bala Lake/Afon Dyfrdwy a Llyn Tegid SAC and Dee Estuary/Aber Dyfrdwy SAC. Given their distance from the Project, effects were considered to be similar or less than assessed for the River Dee and Bala Lake/Afon Dyfrdwy a Llyn Tegid SAC and Dee Estuary/Aber Dyfrdwy SAC for both Project-alone and in-combination effects. Therefore, the detailed assessment provided in **Sections 7.4** and **7.5** has not been repeated for this site and the conclusions for the River Dee and Bala Lake/Afon Dyfrdwy a Llyn Tegid SAC and Dee Estuary/Aber Dyfrdwy SAC have been applied.

394. The Project, alone, in-combination with the Transmission Assets, and in-combination with other plans and projects, would have no AEoI on the River Derwent and Bassenthwaite Lake SAC.

## 7.10 River Eden SAC

### 7.10.1 Description of designation

395. The River Eden is an outstanding floristically rich, northern river on sandstone and hard limestone. Situated within multiple National Character Areas including, Cumbria High Fells, Orton Fells, North Pennines, Solway Basin, Border Moors and Forests, Tyne Gap and Hadrian's Wall and the Yorkshire Dales, the catchment includes headwaters running off the Yorkshire Dales, the North Pennines and the eastern fells of the Lake District and the major standing water body of Ullswater and it flows north to discharge into the Solway Estuary.

#### 7.10.1.1 Qualifying feature

396. The site is designated for Annex II species Atlantic salmon, river lamprey and sea lamprey.

#### 7.10.1.2 Conservation objectives

397. The conservation objectives of the SAC are to ensure that the integrity of the site is maintained or restored as appropriate, and ensure that the site contributes to achieving the Favourable Conservation Status of its Qualifying Features, by maintaining or restoring:

- The extent and distribution of qualifying natural habitats and habitats of qualifying species
- The structure and function (including typical species) of qualifying natural habitats
- The structure and function of the habitats of qualifying species
- The supporting processes on which qualifying natural habitats and the habitats of qualifying species rely
- The populations of qualifying species
- The distribution of qualifying species within the site

#### 7.10.1.3 The distribution of qualifying species within the site Condition assessment

398. The conservation status of all features was assessed unfavourable by Natural England in their SACO in 2022.

### 7.10.2 Assessment

399. The site is over 100km (to the mouth of the river) from the windfarm site, at a greater distance than the River Dee and Bala Lake/Afon Dyfrdwy a Llyn Tegid SAC, Dee Estuary/Aber Dyfrdwy SAC, River Derwent and Bassenthwaite Lake SAC and River Ehen SAC. Given their distance from the Project, effects were considered to be similar or less than assessed for River Dee and Bala Lake/Afon Dyfrdwy a Llyn Tegid SAC and Dee Estuary/Aber Dyfrdwy SAC for both Project-alone and in-combination effects. Therefore, the detailed assessments provided for these sites in **Sections 7.4** and **7.5** have not been repeated for this site and the conclusions for the River Dee and Bala Lake/Afon Dyfrdwy a Llyn Tegid SAC, Dee Estuary/Aber Dyfrdwy SAC and River Derwent and Bassenthwaite Lake SAC have been applied.

400. The Project, alone and in-combination with other plans and projects (including the Transmission Asets), would have no AEoI on the River Eden SAC.

## 7.11 Solway Firth SAC

### 7.11.1 Description of designation

401. The Solway Firth was a large shallow complex estuary formed by a variety of historical physical influences including glaciation, river erosion, sea level change and geological barriers from hard rock outcrops.

#### 7.11.1.1 Qualifying feature

402. The site is designated for Annex II species river lamprey and sea lamprey.

#### 7.11.1.2 Conservation objectives

403. The conservation objectives of the SAC are to ensure that the integrity of the site is maintained or restored as appropriate, and ensure that the site contributes to achieving the FCS of its Qualifying Features.

#### 7.11.1.3 Condition assessment

404. Unknown at time of designation and not updated since.

### 7.11.2 Assessment

405. The site is over 100km from the windfarm site. This is a greater distance than the Dee Estuary/Aber Dyfrdwy SAC and Dee Estuary/Aber Dyfrdwy SAC. Given their distance from the Project, effects were considered to be similar or less than assessed for the Dee Estuary/Aber Dyfrdwy SAC and Dee Estuary/Aber Dyfrdwy SAC for both Project-alone and in-combination effects.

Therefore, the detailed assessment provided in **Sections 7.4** and **7.5** has not been repeated for this site and the conclusions for the Dee Estuary/Aber Dyfrdwy SAC and Dee Estuary/Aber Dyfrdwy SAC have been applied.

406. The Project, alone, in-combination with the Transmission Assets, and in-combination with other plans and projects, would have no AEoI on the Solway Firth SAC.

## 7.12 Summary

407. A summary of the assessment is provided in **Table 7.7**. Given the distance of the windfarm site from the SACs and the coast, there would be no direct effects upon any site and effects on migrating fish would not result in any AEoI. Considering the assessment in light of the conservation objectives, **Sections 7.4.1.2, 7.5.1.2, 7.6.1.2, 7.7.1.2, 7.8.1.2, 7.9.1.2** and **7.10.1.2,** no AEoI on any European site has been identified, either alone or in-combination (including with the associated Transmission Assets).

*Table 7.7 Summary of potential effects upon Annex II migratory fish*

| Summary of potential effects | Qualifying feature | Potential effects | Assessment of effects, alone and in-combination |
|---|---|---|---|
| Dee Estuary/ Aber Dyfrdwy SAC | Sea lamprey | ▪ Increased SSCs and deposition<br>▪ Temporary or permanent habitat loss<br>▪ Underwater noise and vibration<br>▪ Barrier effects<br>▪ EMF<br>▪ Introduction/removal of hard substrate | No adverse effect on site integrity |
| | River lamprey | | No adverse effect on site integrity |
| River Dee and Bala Lake/ Afon Dyfrdwy a Llyn Tegid | Atlantic salmon | ▪ Increased SSCs and deposition<br>▪ Temporary or permanent habitat loss<br>▪ Underwater noise and vibration<br>▪ Barrier effects<br>▪ EMF<br>▪ Introduction/removal of hard substrate | No adverse effect on site integrity |
| | Sea lamprey | | No adverse effect on site integrity |
| | River lamprey | | No adverse effect on site integrity |
| Afon Gwyrfai a Llyn Cwellyn SAC | Atlantic salmon | ▪ Increased SSCs and deposition<br>▪ Temporary or permanent habitat loss<br>▪ Underwater noise and vibration<br>▪ Barrier effects<br>▪ EMF<br>▪ Introduction/removal of hard substrate | No adverse effect on site integrity |
| Afon Eden - Cors Goch Trawsfynydd SAC | Atlantic salmon | ▪ Increased SSCs and deposition<br>▪ Temporary or permanent habitat loss<br>▪ Underwater noise and vibration | No adverse effect on site integrity |

| Summary of potential effects | Qualifying feature | Potential effects | Assessment of effects, alone and in-combination |
|---|---|---|---|
| | | ▪ Barrier effects<br>▪ EMF<br>▪ Introduction/removal of hard substrate | |
| River Eden SAC | Atlantic salmon | ▪ Increased SSCs and deposition<br>▪ Temporary or permanent habitat loss<br>▪ Underwater noise and vibration<br>▪ Barrier effects<br>▪ EMF<br>▪ Introduction/removal of hard substrate | No adverse effect on site integrity |
| | Sea lamprey | | No adverse effect on site integrity |
| | River lamprey | | No adverse effect on site integrity |
| River Ehen SAC | Atlantic salmon | ▪ Increased SSCs and deposition<br>▪ Temporary or permanent habitat loss<br>▪ Underwater noise and vibration<br>▪ Barrier effects<br>▪ EMF<br>▪ Introduction/removal of hard substrate | No adverse effect on site integrity |
| | Sea lamprey | | No adverse effect on site integrity |
| | River lamprey | | No adverse effect on site integrity |
| Solway Firth SAC | Sea lamprey | ▪ Increased SSCs and deposition<br>▪ Temporary or permanent habitat loss<br>▪ Underwater noise and vibration<br>▪ Barrier effects<br>▪ EMF<br>▪ Introduction/removal of hard substrate | No adverse effect on site integrity |
| | River lamprey | | No adverse effect on site integrity |

| Summary of potential effects | Qualifying feature | Potential effects | Assessment of effects, alone and in-combination |
|---|---|---|---|
| River Derwent and Bassenthwaite Lake SAC | Atlantic salmon | ▪ Increased SSCs and deposition<br>▪ Temporary or permanent habitat loss<br>▪ Underwater noise and vibration<br>▪ Barrier effects<br>▪ EMF<br>▪ Introduction/removal of hard substrate | No adverse effect on site integrity |
| | Sea lamprey | | No adverse effect on site integrity |
| | River lamprey | | No adverse effect on site integrity |

# 8 Offshore ornithology (Birds Directive Annex 1 and migratory species)

## 8.1 Approach to assessment

408. The following sections present the assessment of effects on the SPAs and ornithological features that have been screened into the appropriate assessment, as identified in **Section 5.3** and **Table 5.2**.

409. For each European site considered in this RIAA (where LSE cannot be ruled out for one or more qualifying features) a site description is provided. Depending on the information available, this may include information taken from the citation for the site, its conservation objectives, supplementary advice on the conservation objectives, conservation advice, site condition monitoring or other baseline offshore ornithology information.

410. For each qualifying feature of a European site screened into the Appropriate Assessment, the following information is provided:

- The condition of the designated population, including any relevant data on population trends
- A summary of the ecology of the species as relevant to the assessment, and a review of the key evidence in support of functional linkage between the Project and the population
- An assessment of the potential effects of the Project-alone on the qualifying feature including a conclusion of whether or not an adverse effect on integrity of the site can be excluded
- An assessment of potential effects on the qualifying feature when considering the Project in-combination with other relevant projects and including a conclusion of whether or not an adverse effect on integrity of the site can be excluded

411. Where predicted effects (either in Project-alone or in-combination scenarios) equate to an increase of greater than 1% of baseline mortality of the relevant population, then an adverse effect on integrity cannot be ruled out, and further consideration is required e.g. through population modelling, to determine the significance of the mortality for the population in question. This is the approach recommended by Parker *et al.,* (2022). Professional judgement has been employed to consider which features are included within the in-combination assessment, as appropriate. Generally, where the background mortality is predicted to increase by less than 0.1% and/or apportioned mortality is significantly below one individual, it has been assumed that changes would be undetectable against natural variation, and no contribution by the Project to in-combination effects has been assumed. However, the assessment of each

feature has been addressed on a case-by-case basis, and appropriate justification of the approach taken explained accordingly within each feature assessment.

412. Quantitative information from other relevant projects within the area of search has been used to inform the in-combination assessment, where this is available. The projects considered within the in-combination assessment are the same as within the EIA cumulative assessment, as set out in **Chapter 12 Offshore Ornithology** of the ES. It should be noted that no quantitative information is available for some of the older (e.g. pre-2010) OWF projects. In Natural England and NRW's response to the PEIR submission (refer to **Section 8.2**), the consultees requested that quantification of historic projects was presented in the ES and RIAA. Natural England subsequently provided advice on their preferred approach to 'gap filling' for historic projects in October 2023. This was reviewed and a response was submitted to Natural England (and also distributed to NRW) in January 2024. Further information on the approach taken in response to NE and NRW comments is provided below.

413. For the majority of historic projects, population estimates (for displacement assessment) or collision risk estimates have been derived for the ES, using the approach set out in the response sent to Natural England (and also distributed to NRW) and in **Chapter 12 Offshore Ornithology** (Document Reference 5.1.12). This has included the recalculation of collision risk using the most recently agreed avoidance rates recommended by Natural England (Natural England, 2022b). However, for many historic projects, little or no information was available for rates of species apportionment to individual SPAs. Where published apportioned values were available from project assessment reports, these have been used in the in-combination assessment. For the majority of projects where no apportioning information was available, EIA values were apportioned using available rates from nearby projects (including the Project, Mona and Morgan PEIRs and Awel y Môr OWF and White Cross OWF ESs; refer to **Appendix 12.1 Offshore Ornithology Technical Report** of the ES; Document Reference 5.2.12.1). Where quantitative data are available for projects considered in the in-combination assessment, there is also significant inconsistency between projects on the availability and presentation of seasonal values used for species in the assessment, and for that reason only annual values have been considered within the RIAA. Where seasonal data were unavailable (or unclear), a weighted average apportioning rate was applied, using a suitable nearby proxy project. Weighting for each season was undertaken based on the proportion of months within the year for each season (as defined by Furness, 2015), and assuming that estimated total annual population estimates were evenly distributed across the year. An example of an annual apportioning calculation is presented in **Table 8.1**. Whilst it is recognised that this approach

has limitations, it is considered the most appropriate method to generate meaningful and consistent in-combination values across as many historic projects as possible. For a small number of historic projects it has not been possible to obtain data from published information. Where this was the case, the in-combination assessment has considered available qualitative information and provided commentary on the potential effects on the conclusions of the assessment. Where relevant, the in-combination has also considered the conclusions of the Round 4 offshore wind leasing HRA (NIRAS, 2021), and provided commentary on alignment of the Project RIAA conclusions.

*Table 8.1 Example annual apportioning calculation (razorbill at Lambay Island SPA, using Morecambe Project apportioning values)*

| Season | Breeding | Autumn | Winter | Spring | Annual |
|---|---|---|---|---|---|
| **Apportioning value[1]** | 24.74% | 1.20% | 0.70% | 1.20% | |
| **Months[2]** | Apr-Jul | Aug-Oct | Nov-Dec | Jan-Mar | |
| **Proportion of months[3]** | 0.33 | 0.25 | 0.17 | 0.25 | |
| **Weighted value[4]** | 8.25% | 0.30% | 0.12% | 0.30% | **8.96%** |

[1] Taken from the in-combination project apportioning, where available, or a suitable proxy project.

[2] From Furness (2015). Where seasons overlap, breeding season takes priority.

[3] Number of months in each season as a proportion of the year (or all relevant months in the case of Manx shearwater).

[4] Product of seasonal apportioning value and proportion of months. The seasonal values are summed to produce the annual weighted mean.

414. In consideration of the Transmission Assets, for which a separate DCO Application is being sought, a combined assessment is presented in **Chapter 12 Offshore Ornithology** of the ES (Document Reference 5.1.12) to consider the cumulative effects of the Project with the Transmission Assets. The approach to the in-combination assessment differs from the cumulative assessment presented in **Chapter 12 Offshore Ornithology**, as for the majority of features the impacts from the Transmission Assets are not relevant to the in-combination assessment. This is due to the fact there is no collision risk associated with the Transmission Assets, and displacement effects are largely short term and localised over the construction period, which is assessed as part of the Transmission Assets draft information to support an Appropriate Assessment (ISAA; Morgan Offshore Wind Limited and Morecambe Offshore Windfarm Ltd, 2023b). Therefore, a separate combined assessment of the Project and Transmission Assets is not presented as part of the in-combination assessment for each feature assessed. However, the

potential effects of the Project and Transmission Assets are discussed in relation to two features (red-throated diver and common scoter from Liverpool Bay SPA).

415. The outbreak of Highly Pathogenic Avian Influenza (HPAI) affecting UK seabird populations during 2022 and 2023 is also noted (Natural England, 2022c). At this stage the medium and long-term effects on seabird populations, including SPA colonies, are not known. A review of the potential effects of HPAI (as far as they are understood at this stage) is presented in **Chapter 12 Offshore Ornithology**. This concluded that, while there is uncertainty on the medium and long-term effects of HPAI, it is considered unlikely that these would interact significantly with the impacts from offshore wind development, and therefore the conclusions of the EIA and HRA would not be affected.

## 8.2 Consultation

416. Consultation with regard to offshore ornithology has been undertaken in line with the process set out in **Section 4.2.** The feedback received through the EPP has been considered in preparing the RIAA.

417. **Table 8.2** provides a summary of how the consultation responses received in relation to the draft HRA Screening Report and the draft RIAA, as well as through the EPP process, have influenced the approach that has been taken.

*Table 8.2 Consultation responses received in relation to the RIAA (offshore ornithology) and how these have been addressed*

| Consultee | Date/document | Comment | Response/where addressed |
|---|---|---|---|
| Natural England | Advice on draft Habitats Regulations Assessment (HRA) Screening Report Morecambe Offshore Windfarm – Generation Assets<br>14th September 2022 | In-combination assessment for Liverpool Bay SPA. The Preesall gas storage project has permission to construct a brine outfall in the SPA. An assessment of the effect of the project on Liverpool Bay SPA was not included as part of the DCO as the SPA was extended after the permission was granted (the outfall only now falls within a newly extended part of the SPA). This project hasn't been subject to a review of consents and so this impact remains unaccounted for. The impact will likely cause a displacement of RTD from a relatively small portion of the SPA through construction traffic and then through mortality of prey species during operation (about 5 years). The project has been permitted for many years however and whether it actually is ever constructed remains in doubt. | As no quantifiable effect of this development has been identified, this has been considered but not been included in the RIAA. |
| Natural England | Advice on draft HRA Screening Report Morecambe Offshore Windfarm – Generation Assets 14th September 2022 | Table 8.2: All unidentified birds are pooled. Recommend assign unidentified birds wherever possible. E.g., 'diver sp', 'large gull sp' etc. | Unidentified birds have been apportioned to species in the abundance and density estimates used in the RIAA. Refer to **Appendix 12.1**. |

| Consultee | Date/document | Comment | Response/where addressed |
|---|---|---|---|
| Natural England | Advice on draft HRA Screening Report Morecambe Offshore Windfarm – Generation Assets<br>14th September 2022 | Table 8.3: For future assessment species-specific seasonality will need to be agreed and consideration given to the conservation advice for relevant SPAs. Recommend agree seasonality prior to undertaking any displacement assessment or CRM. | Relevant seasons for the displacement and collision risk modelling (CRM) assessments were presented as part of the ETG meeting on 16th November 2022. No comments were received on seasonality, and seasonal values have been applied accordingly in the RIAA. |
| Natural England | Advice on draft HRA Screening Report Morecambe Offshore Windfarm – Generation Assets<br>14th September 2022 | Paragraph 205: NE Phase 3 best practice suggests that further to using the mean max +1SD foraging range to identify colony connectivity, colony-specific maximum foraging ranges should also be cross checked to ensure no potentially relevant colonies are missed in screening. | Relevant colony-specific tracking studies and maximum foraging ranges have been referenced in the RIAA. The Applicant is unaware of any colony-specific studies that would affect the conclusions to the RIAA as presented. |
| Natural England | Advice on draft HRA Screening Report Morecambe Offshore Windfarm – Generation Assets<br>14th September 2022 | Recommend cross check max foraging ranges of colonies. Clearly define cases where Uds [sic] have been used to assess likely origins of particular species. | |

| Consultee | Date/document | Comment | Response/where addressed |
|---|---|---|---|
| Natural England | Advice on draft HRA Screening Report Morecambe Offshore Windfarm – Generation Assets<br>14th September 2022 | Paragraph 215: This paragraph is unclear. It should be noted that indirect effects are poorly understood. The foraging range of breeding birds if of no relevance to, e.g. wintering birds within an SPA, within which habitat constraint may cause significant aggregation. Recommend to explain how indirect effects have been considered. | As the windfarm site is located outside of designated sites, no indirect effects on e.g. habitat or prey that support qualifying features are predicted. Such effects are possible as part of Transmission Assets that pass through Liverpool Bay SPA, but these would be considered as part of the separate DCO process for these elements. |
| Natural England | ETG 1meeting<br>25th May 2022 | For cumulative assessment, Natural England wishes to use consented (as opposed to as-built) layouts [of existing operational windfarms], together with relevant post-construction monitoring. | It is confirmed that consented values have been used for the cumulative and in-combination assessments. |
| Natural England | ETG 1 meeting<br>25th May 2022 | Natural England will provide graduated displacement rates for red-throated diver to 10km from the offshore windfarm, to be used for the displacement analysis. | Displacement rates have been received from Natural England and applied to the assessment for red-throated diver at Liverpool Bay SPA set out in **Section 8.4.2.1.** It is noted that there were insufficient data (i.e. too few birds were present within the survey area) across the 24 months of survey to undertake model-based density estimates (e.g. using MRSea (Marine Renewables Strategic environmental assessment) tool) for this assessment. |

| Consultee | Date/document | Comment | Response/where addressed |
|---|---|---|---|
| Royal Society for the Protection of Birds (RSPB) | ETG 2 meeting<br>7th September 2022 | RSPB does not support use of 70% macro-avoidance for gannet for the CRM, as recommended by Natural England. | Values including and excluding the 70% macro-avoidance (MA) have been provided in the collision risk assessment within **Chapter 12 Offshore Ornithology** of the ES. The assessment presented for gannet features within the RIAA assumed 70% macro-avoidance, but commentary has also been provided to confirm whether the applied macro-avoidance would affect the conclusions of the assessment where relevant. |
| RSPB and Natural England | ETG 2 meeting<br>7th September 2022 | For the apportioning of birds to colonies, Natural England/RSPB recommend use of site-specific information (e.g. from tracking studies) where possible. | Noted. This information has been reviewed and incorporated into the RIAA where available/appropriate. |
| RSPB | ETG 2 meeting<br>7th September 2022 | RSPB noted the potential effects of avian flu on the assessment. | The recently issued preliminary guidance on Highly Pathogenic Avian Influenza (HPAI) is noted (Natural England, 2022c). A review of the potential impacts from HPAI is provided in **Chapter 12 Offshore Ornithology** of the ES. **Section 8.1** of the RIAA confirms that it is not considered that HPAI would affect the conclusions of the assessment. |

| Consultee | Date/document | Comment | Response/where addressed |
|---|---|---|---|
| Natural England | ETG 3 meeting<br>16th November 2022 | Natural England clarified that for red-throated diver, potential increase in background mortality is not the impact Natural England is concerned with. The effective loss of habitat within Special Protection Areas (SPAs) due to displacement is the issue (i.e. habitat loss rather than mortality). | **Section 8.4.2.1** includes an assessment of both mortality and effective area of displacement for the red-throated diver feature of Liverpool Bay SPA. |
| RSPB | ETG 4 meeting<br>7th September 2023 | RSPB noted that Bowland Fells lesser black-backed gull was not included in the draft RIAA. Tracking data of lesser black-backed gulls from Bowland Fells SPA represent only a small sub-sample and research has shown significant variation in foraging behaviour between individual lesser black-backed gulls. There are also potential changes that could occur during the project lifespan. | Impacts on lesser black-backed gulls associated with Bowland Fells SPA are considered in **Section 8.11.3.3.** |
| Isle of Man (IoM) Government | ETG 4 meeting<br>7th September 2023 | IoM Government noted that the island supports one Ramsar site, which should be included in the HRA. In relation to other IoM designated sites, for other projects a separate report has been produced. | An assessment in respect of Ballaugh Curragh Ramsar site is included in **Section 8.64**. Other designated sites are considered in **Chapter 12 Offshore Ornithology** of the ES. |
| Natural England | ETG 5 meeting<br>12th October 2023 | Natural England confirmed delay in work to address gaps in data for historical projects. Proposed draft approach (agreed between Natural England and Natural Resources Wales (NRW)) was circulated shortly before meeting. NE suggested gap filling could be shared between Morecambe, Mona and Morgan to reduce burden and risk of discrepancies. | The proposed approach has been discussed with the developers of the Mona and Morgan Offshore Wind Projects, and the approach to the cumulative and in-combination assessments agreed between the three projects is set out in **Section 8.1**. |

| Consultee | Date/document | Comment | Response/where addressed |
|---|---|---|---|
| Natural England | ETG 5 meeting<br>12th October 2023 | Natural England agreed with the Applicant's approach to apportion SPA populations using the NatureScot tool. The preferred method is to use the Offshore Renewables Joint Industry Programme (ORJIP) AppSaS tool, but it was acknowledged that this was very unlikely to be available in time for submission. | Apportioning using the NatureScot tool has been undertaken in the RIAA. |
| Natural England | ETG 5 meeting<br>12th October 2023 | Natural England welcomed the consideration of Manx shearwater under construction disturbance and displacement, and recommended use of 50% of operational effects for the construction phase disturbance and displacement effects. | Construction impacts on Manx shearwater have been assessed using the advised approach for all SPAs where this species was screened into the appropriate assessment. |
| Natural England | ETG 6 meeting<br>15th January 2024 | Natural England welcomed presentation of lesser black-backed gull apportioning data in two ways (assuming birds are from coastal colonies only, or from both coastal and inland). Natural England also requested colony information used in the apportioning is appended. | Noted. Apportioning information is included in **Appendix 12.1** of the ES. |
| Natural England (ref E1) | Section 42 Consultation Response<br>2nd June 2023 | The minimum rotor clearance above sea level at PEIR is 22m. Natural England highlight that increasing the minimum rotor clearance would reduce collision risk estimates generated by the project and request that the Applicant explore the feasibility of achieving greater clearance. | The minimum rotor clearance above sea level (air gap) has been increased to 25m above HAT (approximately 35m above LAT (Lowest Astronomical Tide)) for the DCO submission, and is used for the assessment presented in the RIAA. |

| Consultee | Date/document | Comment | Response/where addressed |
|---|---|---|---|
| Natural England (ref E5) | Section 42 Consultation Response<br>2nd June 2023 | Natural England notes the forthcoming publication of "Densities of qualifying species within Liverpool Bay / Bae Lerpwl SPA: 2015 to 2020" which will provide up to date density estimates for red-throated diver, common scoter and the waterbird assemblage within the original SPA bound ary. | The publication (HiDef 2023) has been considered in the Liverpool Bay SPA assessment in **Section 8.4**. |
| Natural England (ref E10) | Section 42 Consultation Response<br>2nd June 2023 | Manx shearwater has been screened out of assessment for disturbance and displacement during construction in the PEIR. There is no specific justification for this decision. Natural England note that the relative species abundance in the study area is high and there is low confidence in the (low) sensitivity to OWF disturbance and displacement estimate. | Manx shearwater is generally considered to have a low susceptibility to disturbance and displacement, particularly during wind farm construction, based on previous studies e.g. Bradbury *et al* (2014). However, on a precautionary basis, Manx shearwater have been included in the assessment of construction displacement for all SPAs where this species was screened into the RIAA. |
| Natural England (ref E21) | Section 42 Consultation Response<br>2nd June 2023 | The cumulative (and in-combination) assessments do not factor in impacts from a number of other projects due to a lack of data. Unknown impacts have been treated as zero which will inevitably underestimate impacts, potentially significantly. A qualitative assessment is mentioned for consideration of some projects, but this process is not detailed, or the results fully presented. Natural England consider this approach to be unacceptable, and hence consider it inappropriate to comment on the potential significance of cumulative (or in-combination) presented in the PEIR submission. | The in-combination assessment presented in the RIAA has been updated and has taken into account 'unknown' historic projects, in accordance with the approach set out in **Section 8.1**, which addresses the concerns and comments provided by Natural England and others. Refer also to response to Natural England comments at ETG 5 above. |

| Consultee | Date/document | Comment | Response/where addressed |
|---|---|---|---|
| Natural England (ref E22) | Section 42 Consultation Response<br>2nd June 2023 | Breeding season apportioning has been undertaken using the NatureScot apportioning tool. Natural England retain some concerns regarding the current limitations of this approach and the apportioning values generated. However, updates to the method are being progressed through the ORJIP AppSaS project that we hope will address these concerns. | The ORJIP AppSaS tool has not been made available in time for the DCO submission. Apportioning to SPA populations in the RIAA has therefore been undertaken using the NatureScot apportioning tool, which has been agreed with Natural England. Refer also to response to Natural England comments at ETG 5 above. |
| Natural England (ref E23) | Section 42 Consultation Response<br>2nd June 2023 | The use of a 100km buffer to screen sites for migratory non-seabirds is not a standard approach, though we recognise the need to identify a proportionate set of SPAs for a more detailed assessment. | The approach undertaken is considered appropriate to screen sites for migratory non-seabirds; Natural England subsequently agreed that this approach was acceptable (meeting 25th September 2023). |
| Natural England (ref E24) | Section 42 Consultation Response<br>2nd June 2023 | Natural England note that for seabirds in the non-breeding season potential connectivity has been assumed for SPA populations that contribute >1% of the BDMPS population. Whilst not in a position to confirm wider applicability of this method at this stage, Natural England considers it broadly appropriate for this particular project. | Noted. |
| Natural England (ref E26) | Section 42 Consultation Response<br>2nd June 2023 | Error in the figure given for common scoter abundance in paragraph 1.333 of the draft RIAA. | Common scoter abundance estimates within the RIAA have been checked and updated based on the full 24 months of baseline data. |

| Consultee | Date/document | Comment | Response/where addressed |
|---|---|---|---|
| Natural England (ref E27) | Section 42 Consultation Response<br>2nd June 2023 | Breeding season apportioning in the draft RIAA has been undertaken using the NatureScot apportioning tool. Natural England retain some concerns regarding the current limitations of this approach. However, an updated method is being progressed through the ORJIP AppSaS project that we hope will address these concerns. | The ORJIP AppSaS tool has not been made available in time for the DCO submission. Apportioning to SPA populations in the RIAA has therefore been undertaken using the NatureScot apportioning tool, which has been agreed with Natural England (ETG 5; see above).<br>Refer to **Appendix 12.1** of the ES for further information on the apportioning approach. |
| Natural England (ref E28) | Section 42 Consultation Response<br>2nd June 2023 | Natural England consider the calculation of an 'effective displacement area' for red-throated diver to be fundamentally flawed and misleading. There is no logical way to proportionally reduce the area of effective habitat loss by the expected level of displacement. The displaced proportion of the population cannot use any of the area, i.e., displacement is occurring over the full extent of the area. Birds that are not displaced are likely (but not necessarily) dispersed over the entire area. Ultimately, calculating a (reduced) area of effect in this way risks underestimating the % of the SPA that is subject to displacement effects. | The Applicant does not agree that application of the displacement gradient to the effective area of displacement is without merit. It is established that the displacement effect will diminish as distance from the windfarm increases, and therefore it is logical to conclude that the effective area would also be reduced. It is acknowledged that the application of the Natural England gradient is a proxy, but it should be noted that the total (uncorrected) values have also been presented in **Section 8.4.2.1**, to enable Natural England to consider both values. |
| | | Natural England consider that it is appropriate to take into account the original SPA boundary when calculating the area of red-throated diver supporting habitat within the SPA that could be affected by the project, though given red-throated diver are likely to | Displacement values for both the original and updated SPA boundary are presented in **Section 8.4.2.1**. |

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| | | be present beyond the original boundary, albeit in lower densities, there is merit in presenting displacement values that include as well as exclude those parts of the SPA that fall beyond the original boundary. | |
| Natural England (ref E29) | Section 42 Consultation Response<br>2nd June 2023 | The in-combination assessment in the draft RIAA suggests a 60% increase in baseline mortality for non-breeding lesser black-backed gull at Morecambe Bay and Duddon Estuary SPA yet concludes that an adverse effect is unlikely. NE accepts that the mortality estimate is likely to be precautionary, and the apportioning of impacts may be problematic. However, we highlight the obvious need for thorough investigation into this impact, including through PVA.<br><br>Tracking studies are used to evidence that the apportioning undertaken is not appropriate for the consideration of impacts. Natural England consider this suggests an alternative approach to apportioning should be investigated. | Project-alone and in-combination assessments in **Section 8.5.2.2** have been updated with the full 24 months of baseline survey data and include presentation of PVA for this feature. The NatureScot apportioning tool has been used for this species, although the Applicant maintains that tracking studies indicate that few birds from the SPA are likely to occur at the windfarm site during the breeding season. However, this assumption does not affect the apportioning approach. The Applicant has also presented data using two different apportioning approaches, assuming birds present at the windfarm site are likely to originate from just coastal colonies, or from coastal and inland colonies. |
| Natural England (ref E30) | Section 42 Consultation Response<br>2nd June 2023 | Awel-Y-Mor is not considered in-combination as impacts would not lead to a detectable increase in lesser-black backed gull mortality of the SPA population. Natural England advise that all impacts should be scoped into the in-combination assessment. I.e. impacts that do not result in >1% increases of baseline mortality should still be considered. | In-combination collision mortality for lesser black-backed gulls has been updated and includes data from Awel y Môr, as presented in **Sections 8.5.2.2** and **8.6.3.2**. |

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| Natural England (ref E31) | Section 42 Consultation Response<br>2nd June 2023 | NE does not agree that the results of the tracking study carried out by Clewley *et al.,* (2020) comprise sufficient evidence to conclude that the birds identified in the study area are unlikely to originate from the Morecambe Bay and Duddon Estuary SPA, and therefore dismiss potential significant impacts. The study covered the period from 2016-2019 so there is no overlap with the aerial surveys carried out for the project. During that time connectivity with existing wind farms was found for >50% of the birds from the South Walney colony surveyed. The authors of the study noted that lesser black-backed gulls are more likely to forage offshore when rearing chicks. The study coincided with a period of very poor productivity at the South Walney colony. Productivity has since improved; hence more offshore foraging may be occurring. Note there is also an error in the text whereby Clewley *et al.,* (2021) is cited rather than Clewley *et al.,* (2020). | The assessment presented in **Section 8.5.2.2** includes data that assumes birds are apportioned to Morecambe and Duddon Bay Estuary. However, the Clewley *et al.,* (2020) data do indicate that this may result in an overestimate of the effects on this feature. |
| Natural England (ref E32) | Section 42 Consultation Response<br>2nd June 2023 | Hodbarrow is to the Northeast of the windfarm site. Therefore, it is entirely possible that breeding Sandwich terns from the Morecambe Bay and Duddon Estuary SPA pass through the windfarm site on migration to reach known post-breeding roost sites on the North Wales coast via a relatively direct route. | The assessment of effects on Sandwich tern from the Morecambe Bay and Duddon Estuary SPA has been updated and is presented in **Section 8.5.2.4** |
| Natural Resources Wales (ref 48) | Section 42 Consultation Response<br>2nd June 2023 | Once the full 24 months of data have been included, the project alone and in-combination assessments should be revisited to account for the complete baseline survey data and any updates to cumulative and in-combination totals. NRW (A) advise that where predicted impacts equate to >1% of baseline mortality of the relevant population, further consideration Is | Project-alone and in-combination assessments in the RIAA have been updated with the full 24 months of baseline survey data. PVA has been undertaken where predicted impacts equate to >1% of the baseline mortality of an SPA population. |

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| | | required through Population Viability Analysis (PVA) modelling. | |
| Natural Resources Wales (ref 53) | Section 42 Consultation Response<br>2nd June 2023 | There has been no consideration given to construction vessel routes. NRW (A) advise that some indication should be given as to the port where construction vessels are likely to sail from and note that routes through the Liverpool Bay SPA should follow best practice protocols (including adhering to existing routes wherever possible) to minimise disturbance to red-throated diver and common scoter. This is also relevant for HRA, particularly for Liverpool Bay SPA. | The final selection of the port(s) facilities required to service the Project have not yet been determined, however it is assumed the construction port will be in the UK and the operational port will be within 50km of the windfarm site and that vessels would pass through the Liverpool Bay SPA. Embedded mitigation includes restricting vessel movements where possible to existing navigation routes, and best practice vessel management; refer to **Section 8.3.1**. |
| Natural Resources Wales (ref 55) | Section 42 Consultation Response<br>21st May 2023 | As with construction displacement, no consideration of operation and maintenance vessel routes has been given. Again, some indication should be given as to the port where operation and maintenance vessels are likely to sail from and NRW (A) routes through the Liverpool Bay SPA should follow best practice protocols to minimise disturbance to red-throated diver and common scoter. This is also relevant for HRA. | |
| Natural Resources Wales (ref 59) | Section 42 Consultation Response<br>21st May 2023 | NRW (A) do not consider it appropriate to base the cumulative, and hence also in-combination, assessments on so many unknowns for impacts from many of the relevant other projects. Whilst these historic projects may not have undertaken quantitative assessments, or assessments using current approaches, estimates will need to be generated for these unknown projects in order to undertake meaningful assessments. NRW (A) suggest this should be explored collaboratively through the relevant EWG. These discussions could also cover potential issues over different avoidance rates, collision model options etc. used by other projects | The in-combination assessment presented in the RIAA has been updated and has taken into account 'unknown' historic projects, in accordance with the approach set out in **Section 8.1**, which addresses the concerns and comments provided by NRW and others. |

| Consultee | Date/document | Comment | Response/where addressed |
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| | | where there are data available. As a result, NRW (A) have not made any comments on the overall level of cumulative (or in-combination) impacts or their significance. | |
| Natural Resources Wales (ref 65) | Section 42 Consultation Response<br>21st May 2023 | The Morecambe HRA screening and Stage 2 RIAA have been based on only 12 months of digital aerial survey data. Although NRW (A) note that a further 12 months have been collected, they are not presented and analysed for review in the PEIR and associated HRA documents. Until the full data set is available, NRW (A) are not in a position to agree to any conclusions as there isn't adequate survey data to screen out sites and/or species. At present NRW (A)consider that all Welsh sites (SPAs/Ramsar's/SSSIs) designated for seabirds and wintering estuarine birds should be screened in. | Project-alone and in-combination assessments in the RIAA have been updated with the full 24 months of baseline survey data. |
| Natural Resources Wales (ref 66) | Section 42 Consultation Response<br>21st May 2023 | Section 8.4.1 Seabirds non-breeding, Paragraph 214: For seabirds in the non-breeding season, potential connectivity has been assumed for Special Protected Area (SPA) populations that contribute >1% of the Biologically Defined Minimum Population Scales (DMPS) population. NRW (A) notes that this is not a standard approach and whilst it may seem broadly appropriate for this project, NRW (A) suggest that at this stage the applicability of the approach is | Noted. The approach to determining connectivity with SPAs and to screen sites for migratory non-seabirds has been discussed and agreed with Natural England (meeting 25th September 2023). |

| Consultee | Date/document | Comment | Response/where addressed |
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| | | discussed further through the relevant Expert Working Group (EWG). | The screening of the great cormorant feature of Ynys Seiriol/Puffin Island SPA has been checked and it is confirmed that this feature is screened in and assessed within the RIAA set out in **Section 8.13**. |
| | | Section 8.4.2 Migratory birds other than seabirds, Paragraph 216: A 100 km buffer has been used to screen SPAs/Ramsar's for migratory non-seabirds. NRW (A) advise that this is not a standard approach. NRW (A) recognise the need to identify a proportionate set of SPAs for a more detailed assessment and hence recommend that the merits of this approach be discussed further through the EWG. | |
| | | Appendix 2 screening outcome for UK SPA and Ramsar Sites with ornithology qualifying features: Ynys Seiriol / Puffin Island SPA, Great cormorant: NRW (A) query the conclusion of significance of effect for this site and feature to be no LSE (screened out). This is because the justification column states, "Project beyond the published foraging range (mean max +1SD), therefore no connectivity during the breeding season. Screened in for non-breeding season effects as species was recorded during baseline surveys, and >1% of birds within the BDMPS region during this period will originate from this population." NRW (A) advise that the screening of this site and feature is checked. | |

| Consultee | Date/document | Comment | Response/where addressed |
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| Natural Resources Wales (ref 67) | Section 42 Consultation Response<br>21st May 2023 | NRW (A) note that the assessments for a number of the Welsh designated sites are incomplete (e.g. Anglesey Terns SPA; Skomer, Skokholm and seas of Pembrokeshire (SSSP) SPA). This is because not all of the qualifying features that the HRA Screening Report has concluded to be screened in for LSE have been considered. NRW (A) Advise that once the full 24 months of data are available and the sites and features screened in for LSE have been reviewed, the RIAA should be reviewed and updated, and all relevant qualifying features of sites screened in should be assessed. NRW (A) are therefore unable to make any conclusive judgements as to levels of impact and significance of effect at this stage. | It is confirmed that the RIAA has been reviewed based on the full 24 months of aerial survey data. All sites screened into the assessment (i.e. where LSE was identified) are assessed in RIAA. |
| Natural Resources Wales (ref 69) | Section 42 Consultation Response<br>21st May 2023 | Consideration should be given to NRW (A) advice on the EIA methodologies above (e.g. regarding disturbance/displacement assessments and cumulative assessments) as these are also relevant for RIAA assessments for the project alone and in-combination. In addition, NRW (A) notes the following regarding the approaches taken for the assessments included for Welsh designated sites in the draft RIAA: | Noted; NRW comments have been considered as appropriate throughout the RIAA. See additional responses below. |
| | | With reference to Liverpool Bay SPA red-throated diver, Paragraph 1.319, NRW (A) notes that there was insufficient data to assess graduated displacement over 10 km buffer (as was advised by NE). This should be reviewed for analysis of the full data set once the 24 months of data are available. NRW (A) also highlight the potential to consider other relevant data sources if the projects survey data proves insufficient (e.g. Seabird Sensitivity and Mapping Tool, SeaMaST) | It has been confirmed that there was insufficient data (due to number of birds identified) from the 24 months of survey data to enable model-based density estimates for red-throated diver to be calculated. It was therefore agreed with Natural England during ETGs that a weighted average displacement rate is calculated, using the displacement |

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| | | | gradient provided by Natural England. This is the same approach used in the draft PEIR and is considered to provide a suitable (and precautionary) level of assessment. |
| | | Liverpool Bay SPA red-throated diver (paragraphs 1.320, 1.322 & Table 8.6): NRW (A) does not agree with the calculation of an 'effective displacement area' as there is no logical way to proportionally reduce the area of effective habitat loss by the expected level of displacement. The displaced proportion of the red-throated diver population cannot use any of the area – displacement occurs over the full extent of the area. Birds that are not displaced are likely (but not necessarily) dispersed over the entire area. Ultimately, the approach taken appears to incorrectly downplay the % of the SPA that is subject to displacement effects. NRW (A) consider that variable displacement rate should be applied to abundance figures and not to the area of effective habitat loss. Therefore, for the submission, NRW (A) advise that the area of effect within the SPA is calculated for both the original and extended SPA boundaries, without reducing the area proportionally according to the level of displacement of red-throated diver expected to occur. | The Applicant does not agree that application of the displacement gradient to the effective area of displacement is without merit. It is established that the displacement effect will diminish as distance from the windfarm increases, and therefore it is logical to conclude that the effective area would also be reduced. It is acknowledged that the application of a linear displacement gradient is a proxy, but it should be noted that the total (uncorrected) values (i.e. without the application of the gradient) have also been presented for comparison in **Section 8.4.2.1**, to enable NRW to consider both values. Red-throated diver displacement values for both the original and updated SPA boundary are presented in the RIAA. |
| | | NRW (A) also advise that the area of the SPA subject to displacement for red-throated diver is considered in-combination with other plans and projects. | It is confirmed that the area of displacement for red-throated diver is considered within the in-combination assessment within the RIAA. |

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| | | With reference to Section 8.8 Glannau Aberdaron ac Ynys Enlli/ Aberdaron Coast and Bardsey Island SPA & SSSP SPA Manx shearwater, no evidence has been provided in the draft RIAA to support the assertion that 50% displacement for Manx shearwater can be considered realistic and NRW (A) note that there is currently no evidence for any particular range of displacement rates (1-10%, 50%, 30-70% or any other) for this species from offshore wind farms. Therefore, NRW (A) suggest that once the full dataset has been analysed, the whole apportioned annual matrices are provided for these sites and that these indicate where 1% of baseline mortality of the relevant colonies is exceeded. NRW (A) would then suggest that any further approach to the assessment is discussed collaboratively through the EWG. NRW (A) also recommend that following this, the appropriate impact figures for the Morecambe generation assets project to take through to the in-combination assessments for Manx shearwater at these sites is discussed through the EWG. | Manx shearwater are generally considered to have a low susceptibility to disturbance and displacement, based on previous studies (e.g Bradbury *et al* (2014)). A rate of 50% is therefore considered suitably precautionary; however, the assessment considers a range of displacement and mortality values (i.e. 30-70% and 1-10% respectively), and the full range is available (within the accompanying technical appendix to **Chapter 12 Offshore Ornithology** of the ES) should NRW require this in order to consider its position. |
| | | Furthermore, no consideration has been given to potential impacts of lighting during any phase on Manx shearwater at these sites. Deakin at al., (2022) notes that a higher level of disturbance to shearwaters and petrels may occur during the construction phase, when activity, noise and light levels may be greatest. | The new Marine Scotland report on OWF lighting impacts on Manx shearwater (Deakin *et al* 2022) has been considered in the ES; refer to **Chapter 12 Offshore Ornithology** of the ES, and the conclusions of this referenced in the RIAA. Overall, it is considered that lighting is not likely to significantly affect Manx shearwaters, and that any such impacts would not affect the conclusions of the assessment. |

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| | | Apportionment of impacts to colonies in the non-breeding season(s): It appears that the number of adult birds at colonies (e.g. SSSP SPA Manx shearwater Section 8.9.2.1 and Grassholm SPA gannet, Section 1.572) used in the non-breeding season(s) apportionment are not those from the Tables in Appendix A of Furness (2015) and are updated colony figures. However, the respective non-breeding season(s) BDMPS total figures used in the calculations have not bee193amsarted to account for new colony data and use those presented in the tables in Appendix A (Furness, 2015). NRW (A) do not consider this to be appropriate as updating the SPA colonies figures presented in the tables in Appendix A of Furness (2015) with more recent figures is not recommended, unless there is evidence to suggest that the colony in question has increased or decreased significantly relative to other colonies. | It is confirmed that the approach to apportioning outside of the breeding season has been updated in the RIAA in accordance with NRW's advice. |
| | | As an example, the proportion of SSSP SPA adult Manx shearwaters present at the Morecambe site during the migration seasons should be calculated using the information in Table 13 of Furness (2015) and calculated as: During the migration seasons for the UK western waters and Channel BDMPS, the number of SSSP SPA adult birds is 700,000 whilst the total number of Manx shearwaters of all ages across the BDMPS is 1,580,895 birds. Therefore, the proportion of SSSP SPA adult birds across the BDMPS during the migration seasons can be calculated as 44.3% (and not 57.6% as presented in Paragraph 1.549). | See response above. |

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| | | Taking the same approach for Grassholm SPA gannets, NRW (A) advise the proportions of Grassholm SPA adult gannets present at the Morecambe site during the autumn and spring should be 14.4% and 11.9% respectively (rather than the 13.19% and 10.88% as presented in Section 1.572). | Apportioning for all species has been updated in accordance with NRW advice. |
| | | In-combination assessments: In addition to NRW (A) comments above regarding data for existing projects to include in assessments, the in-combination assessment of impacts from other plans and projects should include all plans/projects located within foraging range of the colony in question in the breeding season and for the non-breeding season(s) should include impacts from a wider range of projects, i.e. all those located within the relevant non-breeding season BDMPS in Furness (2015). NRW (A) advise that all impacts should be scoped into the in-combination assessments, i.e. impacts that do not result in >1% increases of baseline mortality should still be considered–- project alone impacts considered to be negligible should not be. | The in-combination assessment for all species has been updated. The assessment includes all relevant projects located within the relevant BDMPS (i.e. in most cases the UK Western Waters), as agreed with Natural England. The approach is considered sufficient and appropriate to account for likely in-combination effects. It is confirmed that all projects (including those less than 1% increase in background mortality) have been considered within in-combination assessments. |
| Isle of Man (IoM) Government | Section 42 Consultation Response<br>2nd June 2023 | There is one designated Ramsar Site (Ballaugh Curragh) and potential further Ramsar sites have been identified in a report to the Overseas Territories Conservation Forum (https://www.ukotcf.org.uk/conventions194amsarr-2/). A nuanced discussion of conservation value has been provided and it is hoped that the Isle of Man status of site designations, being different from the UK, can be accounted for, without Manx site statuses skewing down the perceived conservation value of any species within the analyses (as non-SPA sites). | Impacts on Ballaugh Curragh Ramsar site are considered in **Section 8.64**. |

| Consultee | Date/document | Comment | Response/where addressed |
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| RSPB | Section 42 Consultation Response<br>5th June 2023 | We also have concerns with breeding Lesser Black-backed Gull, despite the low frequency of occurrence during the reported survey work. This is because, with the exception of the Ribble and Alt Estuary SPA colony, the main Irish Sea breeding colonies (at Bowland Fells SPA and Morecambe Bay and Duddon Estuary SPA) require restoration to a favourable conservation status and the implications of this needs careful consideration via the Expert Working Groups. | Impact on SPA lesser black-backed gull colonies, including Morecambe Bay and Duddon Estuary SPA and Bowland Fells SPA (**Section 8.11**), have been fully considered in the RIAA. |
| RSPB | Section 42 Consultation Response<br>5th June 2023 | Additionally, we are surprised that the Bowland Fells SPA, Large gull super colony was not mentioned within your documents as a recent paper published by the RSPB and Natural England as part of the Life on The Edge (LOTE) project stated that the 'Bowland Fells may be the largest lesser black-backed gull colony in the world', as previously mentioned, and despite its apparent size, the colony is still considered in recovery from the impact of decades of licenced culling. | Impact on SPA lesser black-backed gull colonies, including Bowland Fells SPA, have been fully considered in **Section 8.11**. |

## 8.3 Assessment of potential effects

### 8.3.1 Embedded mitigation

418. This embedded mitigation for the Project relevant to the ornithological assessment is provided in **Table 8.3**.

*Table 8.3 Embedded mitigation measures relevant to offshore ornithology*

| Parameter | Mitigation measures embedded into the design of the Project |
|---|---|
| Site location | Location was selected as part of the Round Four site selection process undertaken by The Crown Estates. It is located outside of areas designated for their importance to bird populations. |
| Air gap | The Project design has an air gap (minimum rotor clearance above sea level) of 25m above HAT (approximately 35m above LAT).<br>At PEIR the air gap was 22m above HAT which was set at a value greater than the minimum required for shipping and navigation safety to reduce the potential collision risk for offshore ornithology receptors. Between PEIR and the production of the ES, the air gap has been further increased to 25m above HAT in response to consultation feedback, providing further reduction of potential collision risk for offshore ornithology receptors. |
| Best practice protocol for minimising disturbance to red-throated diver and common scoter | Potential impacts on red-throated diver and common scoter during construction, operation and maintenance, and decommissioning works would be mitigated through:<br>▪ Restricting vessel movements where possible to existing navigation routes (where the densities of red-throated diver and common scoter are typically relatively low)<br>▪ As far as possible maintaining direct transit routes (to minimise transit distances through areas used by red-throated diver)<br>▪ Where it is necessary to go outside of established navigational routes, avoid rafting birds either en-route to the windfarm site from port and/or within the windfarm site (dependent on location) and where possible avoid disturbance to areas with consistently high bird densities<br>▪ Avoidance of over-revving of engines (to minimise noise disturbance)<br>▪ Briefing of vessel crew on the purpose and implications of these vessel management practices (through, for example, tool-box talks and issuing of 'Best Practice' guidance)<br>The Project Team would make construction and maintenance vessel operators aware of the importance of these species and the associated mitigation measures through tool-box talks. |

### 8.3.2 Realistic worst-case scenario

419. The realistic worst-case scenarios for the offshore ornithology assessment are summarised in **Table 8.4**, and have been presented in accordance with Natural England guidance (Parker *et al.*, 2022). These are based on the Project parameters described in relevant chapters of the ES, including **Chapter 5 Project Description**, which provides further details regarding specific activities and their durations, and **Chapter 6 EIA Methodology**. The assessed parameters are considered to be the worst-case in respect of ornithology receptors, comprising the highest number of smallest turbines.

*Table 8.4 Realistic worst-case scenarios for offshore ornithology*[18]

| Parameter | Values |
|---|---|
| Latitude (decimal degrees) | 53.8 |
| Area of OWF ($km^2$) | 87 |
| Area of OWF + 2km buffer ($km^2$) | 174 |
| Area of OWF + 4km buffer ($km^2$) | 285 |
| Area of OWF + 10km buffer ($km^2$) | 651 |
| Width of OWF (km)[19] | 10.52 |
| Length of operational period (years) | 35 |
| Number of turbines | 35 |
| Number of blades | 3 |
| Maximum blade width (m) | 6.45 |
| Average blade pitch at mean predicted wind speed (degrees) | 6 |
| Rotor radius (m) | 130 |
| Average rotation speed at mean predicted wind speed (rpm) | 7.64 |
| Hub height relative to HAT (m) | 155 |
| Tidal offset (m) | 4.82 |

[18] Presented in format requested by Natural England

[19] The width is calculated as the diameter of a circle with the same area as the offshore windfarm site (for the Project 86.79$km^2$).

## 8.4 Liverpool Bay SPA

420. Liverpool Bay SPA is directly adjacent to the eastern boundary of the windfarm site.

### 8.4.1 Description of designation

421. Liverpool Bay SPA runs as a broad arc from Morecambe Bay to the east coast of Anglesey. It covers an area of c. 2,528km$^2$, classified for the protection of red-throated diver, common scoter and little gull during the non-breeding season, as well as a waterbird assemblage, and foraging areas for little tern and common tern breeding within coastal SPAs.

422. The seabed of the SPA contains a wide range of mobile sediments. Sand is the most common substrate, with a concentrated area of gravelly sand located off the Mersey Estuary. Tidal currents within the Bay are generally weak and do not exceed 2 m/sec. This in conjunction with an extended tidal range of 6–8 m facilitates deposition of sediments and encourages mud and sand belts to accumulate.

423. Natural England's Site Improvement Plan (SIP) for the SPA (2015) identified the key pressures and threats to the qualifying features of the site as:

- Commercial fisheries (removal of prey fish species used by qualifying bird species, damage to seabed, entanglement of birds in nets and disturbance to birds)
- Shipping and transport corridors (disturbance to qualifying bird species, particularly outside established corridors)
- Recreational fishing (disturbance to birds from recreational vessels, primarily in the nearshore zone)
- Aggregate dredging (damage to seabed)
- Siltation (change to cease deposition of dredged material from Mersey Estuary within the SPA may result in habitat improvement in the SPA)
- Water pollution (risk of oil spills or other pollution from shipping and industry)

424. Liverpool Bay SPA was originally designated for two species (red-throated diver and common scoter) and covered a smaller area to the east, approximately 7km from the current windfarm site boundary. The extension area (which adjoins the windfarm site) was designated in 2017, for its non-breeding little gull population, and also for breeding little tern (nearshore areas away from the windfarm site). The areas immediately adjacent to the windfarm site, therefore, are considered primarily to be of importance for little gull.

### Conservation objectives

425. The SPA's conservation objectives are to ensure that the integrity of the site is maintained or restored as appropriate, and ensure that the site contributes to achieving the aims of the Wild Birds Directive, by maintaining or restoring:

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

## 8.4.2 Assessment

426. The qualifying features of Liverpool Bay SPA screened into the Appropriate Assessment are red-throated diver (non-breeding), common scoter (non-breeding), little gull (non-breeding) and common tern (breeding).

### 8.4.2.1 Red-throated diver

### Status

427. Non-breeding red-throated diver is a qualifying feature of the SPA. Liverpool Bay supports the third largest aggregation for this species in UK offshore waters. Prior to revision of the SPA boundary in 2017, the population comprised 1,171 birds (Lawson *et al*., 2016), which was 6.89% of the GB population. This population estimate was based on visual aerial surveys undertaken between 2004 and 2011, which identified a peak mean abundance of 1,171 individuals. Subsequently, digital aerial surveys of the SPA have been undertaken as part of monitoring of the Burbo Bank Extension OWF (HiDef, 2020). Surveys covered the period between 2011 and 2020, and the resultant monitoring report concluded that, while there was annual variation between population estimates, there was no evidence of an overall change in population size during this period. Surveys of the original SPA boundary covering the period 2015-2020 are documented in the Natural England commissioned report 440 (HiDef, 2023). These surveys primarily covered the peak winter period (January and February), with mean monthly abundance estimates of between 372 and 2,073 birds, and a mean peak count of 1,800 birds over that period. It is likely that a small number of birds also occurred within the SPA extension area, and therefore this estimate is likely to underestimate the population size for the full the SPA area (i.e. original plus extension) by a small amount. The change in recorded population size for the SPA between 2011 and 2020 (i.e. from 1,171 birds to 1,800) indicates that the

population may have increased during this period, although there is some uncertainty given that survey methods differed between the two periods. Nonetheless, these results do indicate that there is no evidence of population decline. The Conservation Advice Package for Liverpool Bay SPA (Natural England *et al*., 2022) confirmed that 1,800 birds is the population size used for the purposes of informing the conservation objectives, with a target to ‘Maintain the size of the non-breeding population at a level which is at or above 1,800 individual'. 1,800 birds has therefore been assumed to be the reference population for the current assessment.

428. The Conservation Advice Package for Liverpool Bay SPA (Natural England *et al*., 2022) identified that disturbance and displacement are key threats to the wintering red-throated diver population, both from shipping and offshore windfarms, but acknowledged that such effects were already occurring at the time the SPA was designated.

429. Based on an SPA population of 1,800 birds, and an annual baseline mortality rate (all age classes) of 0.233 (derived from Horswill and Robinson, 2015; refer to **Chapter 12 Offshore Ornithology** of the ES), 419 birds from the SPA population would be expected to die each year.

430. The digital aerial surveys undertaken for the Project (see **Appendix 12.1** and **Appendix 12.2 Aerial Survey Two Year Report March 2021 to February 2023** of the ES (Document Reference 5.2.12.2)) recorded low numbers of red-throated divers during months within the species’ autumn migration, wintering, and spring migration seasons (as defined by Furness, 2015). Birds were recorded predominantly outside the windfarm site, in the eastern part of the 10km survey buffer, i.e. within Liverpool Bay SPA (refer to **Appendix 12.2** of the ES; noting that the survey buffer only extended to 10km on the northern and eastern side of the windfarm site where it abutted the SPA – see **Figure 8.1**). Birds occurred within the windfarm site + 10km buffer area in April, November and December 2021; February, March, May, November and December 2022; and February 2023. Birds occurred within the windfarm site in December 2021, March 2022 and December 2022 only. Over the two years of survey, the estimated peak population sizes within the windfarm site, windfarm site + 4km and windfarm site + 10km buffers were five individuals (March 2022), 13 individuals (December 2021) and 64 individuals (March 2022) respectively.

### Functional linkage and seasonal apportionment of potential effects

431. The Liverpool Bay SPA boundary was selected to include important marine areas for this qualifying feature. All red-throated divers within the SPA are assumed to belong to the SPA population, i.e. 100% of birds are apportioned to the SPA. Liverpool Bay SPA formerly covered a smaller area, approximately 7km to the east of the current boundary at its closest point (i.e. not adjoining

the windfarm site – see **Figure 8.1**), and at that time the boundary was defined by the distribution of non-breeding common scoter and red-throated diver (Natural England *et al*., 2016). The recent SPA boundary extension was defined on the basis of the occurrence and distribution of other qualifying features, primarily little gull but with small extensions also on the landward edge of the SPA around little tern and common tern breeding areas. Therefore, it is the original boundary which encompasses those areas that has been identified as being of importance for the non-breeding red-throated diver population. Accordingly, the areas adjacent to the windfarm site are not considered to be of high importance for red-throated diver. Nevertheless, all red-throated divers recorded within the SPA boundary are considered to form part of the designated population.

432. Operational displacement effects on red-throated diver can occur at considerable distances from OWFs (e.g. APEM 2021; Dorsch *et al*., 2020; Mendel *et al*., 2019; Vilela *et al*., 2020, Webb *et al*., 2017). As a result, Natural England have advised that assessments for OWFs within 10km of a European site designated for non-breeding red-throated diver are required to consider the potential impacts on red-throated divers within that SPA (UK SNCBs, 2022).

### Potential effects on the qualifying feature

433. The red-throated diver qualifying feature of the Liverpool Bay SPA has been screened into the Appropriate Assessment due to the potential risk of disturbance and displacement to the SPA population during the construction/decommissioning and operation and maintenance phases of the Project.

### Construction and decommissioning phase disturbance/displacement

#### Project-alone

434. Estimation of construction-phase disturbance and displacement has been undertaken assuming 50% of the operational phase effect, to a distance of 4km from the windfarm site (as construction and decommissioning activities are assumed to affect birds to a distance of ≤4km); i.e. a displacement rate of 50% and mortality range of 1-10% for displaced birds, applied to birds within 4km of the windfarm. This is set out in **Chapter 12 Offshore Ornithology** of the ES. Literature indicates that the majority of red-throated divers will flush from approaching vessels at a distance of 1km or less (Bellebaum *et al*., 2006; Jarrett *et al*., 2018; Topping and Petersen, 2011). Fliessbach *et al.,* (2019) indicated similar flushing distances, stating that 95% of red-throated divers observed during their study elicited an escape response when approached by a vessel, with a mean escape (flushing) distance of 750m (standard deviation (SD) 437m) and a maximum escape distance of 1,700m. Unidentified diver

species were recorded flushing at distances of 2km from the survey vessel. On the basis of this information, it is therefore considered that displacement at a maximum of 4km from construction activities is considered to be appropriately precautionary. Further precaution is inherent in the assessment approach, as it assumes an even distribution of birds within the 4km buffer, whereas the Project survey data (refer to **Appendix 12.2** of the ES) demonstrate that densities were lowest in those areas adjacent to the windfarm site, and the effect will diminish in more distant areas where densities are higher. Furthermore, the assessment included areas within the 4km buffer that are outside the SPA, and therefore this would result in a slight overestimation of the number of SPA birds that are assumed to be displaced.

435. The available evidence regarding red-throated diver displacement by operational OWFs suggests that there will be little or no impact on adult survival as a result of displacement, and that any impact would probably be undetectable at the population level. No evidence has been identified which supports the upper range of the potential mortality effects for birds displaced from OWFs, currently advised by Natural England (i.e. up to 10%). A review of the available evidence (MacArthur Green, 2019a) indicates that a mortality rate of 1% is considered to be appropriately precautionary. It is assumed that these conclusions can also be applied to birds displaced by construction activity, particularly given that construction effects are temporary.

436. The review 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 budget (for example by spending a higher proportion of the time foraging rather than resting in order to compensate for an increase in energetic costs).

437. From the range of 1-10% mortality advised by Natural England, MacArthur Green (2019a) considered that a 1% mortality rate for displaced birds is an appropriately 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 background mortality rate is estimated at

16% per annum, which will include mortality from existing anthropogenic sources of disturbance and displacement such as shipping traffic. Thus, it seems biologically implausible that displacement due to OWF activities would add substantially to the existing mortality rate of this species.

438. The MacArthur Green (2019a) review is supported by more recent studies. For example, at the Outer Thames Estuary SPA there was no evidence that the population had decreased as a result of OWF development following notification of the SPA in 2010 (Natural England, 2021b). Long-term studies of red-throated (and black-throated) divers in the German North Sea found no changes in the overall population size during spring migration over the period 2001-2021, despite the construction of 20 OWFs (Vilela *et al.,* 2021, 2022). Although the divers changed their distribution, away from the OWFs, the population size remained stable, suggesting no or minimal consequences for displaced birds. A study by Thompson *et al.,* (2023) combined time-depth recorder (TDR) and global location sensor (GLS) tag 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 nonbreeding 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 energetic costs of displacement from OWFs (if any), although this is likely to be constrained by factors such as available daylight and food availability.

439. The displacement assessment for red-throated divers within 4km of the windfarm during the non-breeding season is presented in **Chapter 12 Offshore Ornithology** of the ES. On the assumption that all red-throated divers within 4km of the windfarm are birds from the SPA (which is precautionary, as a small proportion of birds recorded during surveys occurred outside of the SPA boundary), a maximum of 20 (0-55) birds could be affected by displacement across all non-breeding seasons. Assuming a displacement rate of 50%, <1 bird (0.98 (0.0-2.76)) would be predicted to die at 10% mortality. Using a 1% mortality rate (which is considered to be sufficiently precautionary – see above), 0.10 (0.00-0.28) birds would be expected to die. Assuming an SPA population of 1,800 birds and background mortality of 0.233 (all age classes), 419 birds from the SPA population would be expected to die each year. The addition of 0.1 birds would increase the annual mortality rate

by 0.02%. This magnitude of increase in mortality would not materially alter the background mortality of the population and would be undetectable.

440. In addition to the effects resulting from vessel activity and construction works within the windfarm site itself, there is also the potential that construction vessels could cause displacement of red-throated divers within the SPA during transit between port(s) and the windfarm site. Although transit routes are not known as the port(s) selection has not been made, the assessment assumes that vessels would transit though the SPA, and that embedded mitigation to minimise such impacts would be implemented; refer to **Table 8.3**. This would include, for example, adherence to existing navigation routes as far as possible, which would mean that little or no additional disturbance effect would occur. As details of the transit routes used by construction vessels are unknown, it is not possible to quantitatively assess the potential effect of these activities. However, given the review of evidence for mortality rates of displaced birds (MacArthur Green, 2019a), and embedded mitigation measures (**Table 8.3**), it is predicted that the mortality rate of displaced birds would be very small. It is therefore concluded that any impacts will be small, there would be no adverse effect on the integrity of the SPA.

441. **Accordingly, no significant effects on red-throated diver are predicted during the construction phase, and it is concluded that there is no potential for the Project-alone to have an adverse effect on the integrity of Liverpool Bay SPA**.

442. The confidence in the assessment is medium. Firstly, the evidence used to set the displacement rates presented in **Chapter 12 Offshore Ornithology** of the ES and **Appendix 12.1** of the ES is of high applicability and quality. As set out above, there is good evidence to suggest that 1% mortality for displaced birds is suitably precautionary. However, uncertainty remains around the effects of displacement on this species.

### In-combination

443. No in-combination effects in respect of red-throated diver are predicted during the construction or decommissioning phases of the Project. This is because Project effects are temporary and reversible, and it is unlikely that there would be significant temporal and/or spatial overlap with other plans or projects. There is the potential that temporal overlap could occur with construction activities associated with Morgan and Morecambe OWFs Transmission Assets, and Morgan and Mona Offshore Wind Projects. However, it is assumed that these projects would be required to implement similar best practice construction methods to minimise any potential effects. The Morgan and Morecambe OWFs Transmission Assets Information to Support Appropriate Assessment (ISAA; Morgan Offshore Wind Limited and Morecambe Offshore Windfarm Ltd, 2023b) estimated an annual mortality of

0.08 red-throated divers from Liverpool SPA during construction and decommissioning. The respective Mona (Mona Offshore Wind Limited, 2023) and Morgan (Morgan Offshore Wind Limited, 2023) ISAAs predicted no measurable mortality for this feature. Even if the works overlapped, therefore, in-combination mortality is unlikely to be more than 0.2 birds (assuming a 1% mortality rate for displaced birds), which would increase background mortality by less than 0.1%. Such an increase would be undetectable against background variation. **It is therefore concluded that there is no potential for the Project to have an adverse effect on the integrity of Liverpool Bay SPA, either alone or in-combination with other plans or projects.**

### Operation and maintenance phase disturbance/displacement/barrier effects

#### Project-alone

444. In accordance with Natural England guidance (SNCBs 2022), it is considered that there is the potential that disturbance, displacement and barrier effects could affect red-throated divers present in areas of the SPA within 10km of the windfarm site.

445. Operational displacement is defined as ‘a reduced number of birds occurring within or immediately adjacent to an offshore windfarm’ (Furness *et al.,* 2013) and involves birds present in the air and on the water (UK SNCBs 2017). Birds that do not intend to utilise an OWF site 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 an OWF site, are subject to barrier effects (UK SNCBs 2017). For the purposes of assessment of birds present in an OWF site during a given season, 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 site, even when tracking data are available. Therefore, in this assessment the effects of displacement and barrier effects on non-breeding red-throated diver are considered together.

446. The assessment assumes that a proportion of the birds recorded during baseline surveys would be subject to displacement from the windfarm site and buffer area, and that a proportion of displaced birds would die as a result of displacement. The proportion of red-throated divers displaced is based on evidence from empirical studies of red-throated diver responses to OWFs; further background on this is provided below. There is no robust empirical evidence to predict the number of displaced divers which might die so the assessment considers a range of 1-10% mortality, based on advice from Natural England, and identifies what is considered to be the most likely proportion based on expert judgement of what is considered to be biologically plausible.

447. 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 0-2km to 20km from the array areas of an OWF. These studies reported that 55-100% (mean of 86% based on 8 studies) of birds were displaced within the array area of an OWF, and provided evidence that the proportion of red-throated divers displaced declined with distance from the OWF with, for example, displacement rates reducing to 12.6% at a distance of 11.5km from the London Array (APEM 2021). Unsurprisingly, the evidence for declining rates of displacement with increasing distance from OWFs derives mainly from those studies which considered effects over more extensive distances from OWFs.

448. Based on this summary of the available studies, SNCBs (2022) advise that a displacement buffer of at least 10km should be used for impact assessments where a plan or project is within 10km of an SPA designated for non-breeding red-throated diver.

449. It is unknown why red-throated divers show such large displacement distances from OWFs. It has been suggested that these might reflect distances moved away from OWFs to alternative areas of preferred habitat (McGregor *et al.,* 2022), rather than avoidance of extensive areas around OWFs per-se, which could result in variation in displacement distances between areas and in different directions from a given OWF. Mendel *et al.,* (2019) commented that displacement may not be a result of visual cues (a bird sitting on the sea surface may not be able to see a wind farm array at a distance of 10km); whilst OWFs may enhance mixing in the water column with ecosystem effects manifesting 10-20km from the OWF, which is of a scale similar to red-throated diver displacement distances identified in some studies. However, the potential mechanisms for such an effect are not clear, nor the reasons why they might affect red-throated divers, but appear not to affect other seabird species over such large distances.

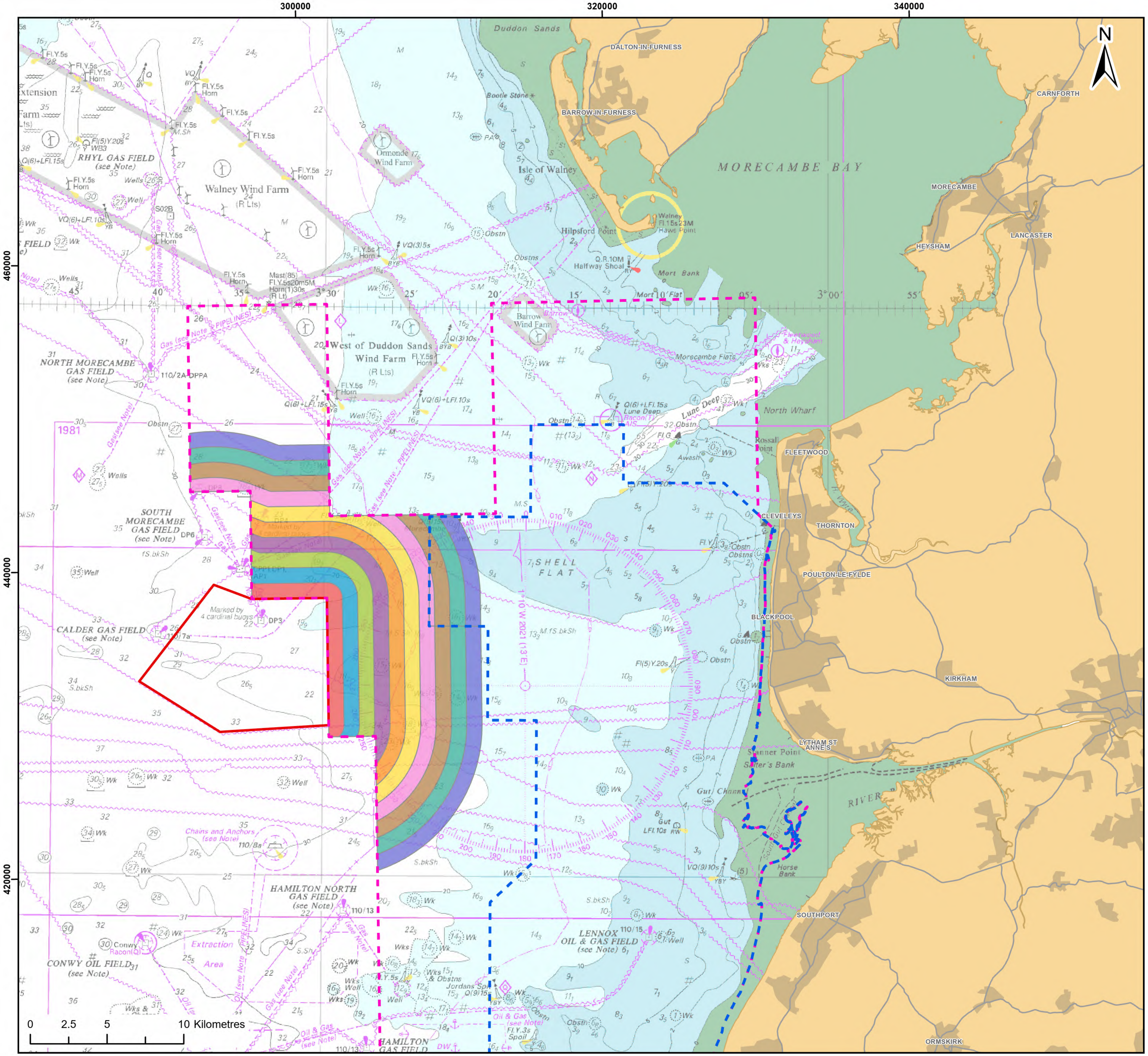
450. While OWFs and other anthropogenic activities in the marine environment have demonstrable displacement effects on red-throated divers, it is unclear how these might interact with other drivers of the non-breeding season distribution of this species offshore, of which habitat and prey availability must be of primary importance. The post-construction monitoring study at the London Array (which compared densities and distribution between the pre- and post-construction periods) found that prior to construction of the OWF, there was a pattern of diver density increasing with distance from the array area up to 9km and then decreasing (APEM, 2021). This suggests that preferred habitat for divers across the whole study area was outside the array

area footprint, and that the displacement effects from the OWF should be considered in the context of an existing gradient in density for the species.

451. While studies consistently show avoidance of OWFs by red-throated divers, with no evidence for habituation, divers are sometimes recorded within and close to OWFs, suggesting a strong avoidance reaction might not always be triggered. For example, Vilela *et al.,* (2022) refer to large numbers (estimated to be 100+ birds from Figure A-1 of Vilela *et al.,* 2022) of divers within about 5km of an OWF in the German Bight during a survey in March 2021, the first time in their long-term study that such high numbers had been observed close to an OWF. Post-construction surveys of red-throated divers at Burbo Bank extension OWF in Liverpool Bay, found particularly high numbers of red-throated divers within the array area and 4km buffer in March 2020; this survey coincided with the beginning of UK lockdowns due to coronavirus, and it was speculated that reduced shipping traffic may have led to increased numbers of red-throated divers (Humphries, 2020).

452. As set out in **Paragraphs 435** and **438** above, evidence presented by MacArthur Green (2019a), Thompson *et al.,* (2023), Vilela *et al.,* (2020 and 2021) suggests that there will be little or no impact on adult survival as a result of displacement, and that any impact would probably be undetectable at the population level. No evidence has been identified which supports the upper range of the potential mortality effects for birds displaced from OWFs, currently advised by Natural England (i.e. up to 10%). Based on this evidence, a mortality rate of 1% is therefore considered to be appropriately precautionary.

453. Natural England advised during the ETG process (refer to **Table 8.2**) that for the appropriate assessment for the Liverpool Bay SPA, a linear displacement gradient of 1km increments should be applied from 0-10km from the windfarm boundary where this overlaps with the SPA (**Table 8.5** and **Figure 8.1**). The data used to inform the gradient was from a range of OWF sites in English waters, namely Gunfleet Sands, Kentish Flats, Lincs, Lynn & Inner Dowsing and London Array, together with a gradient for the effects of OWFs on the distribution of non-breeding red-throated diver calculated for Natural England from the German Bight data in Vilela *et al.,* (2020).





Legend:

- Morecambe Offshore Windfarm Site
- Liverpool Bay SPA
- Liverpool Bay SPA boundary at original designation

**Red-throated diver 1km displacement bands**

- 0-1km
- 1-2km
- 2-3km
- 3-4km
- 4-5km
- 5-6km
- 6-7km
- 7-8km
- 8-9km
- 9-10km



Report: Morecambe Offshore Windfarm: Generation Assets Habitat Regulations Report to Inform Appropriate Assessment

Title: Red-throated diver displacement bands used in the displacement estimate

| Figure: 8.1 | Drawing No: PC1165-RHD-ES-OF-DR-Z-0061 | | | | |
|---|---|---|---|---|---|
| Revision: | Date: | Drawn: | Checked: | Size: | Scale: |
| P01 | 19/04/2024 | JH | SB | A3 | 1:250,000 |

Co-ordinate system: WGS 1984 UTM Zone 30N

MORECAMBE

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*Table 8.5 Natural England displacement gradient for red-throated diver*

| Buffer region (km) | Displacement rate |
|---|---|
| Within OWF | 100% |
| 0-1 km | 80% |
| 1-2km | 74% |
| 2-3km | 68% |
| 3-4km | 63% |
| 4-5km | 57% |
| 5-6km | 51% |
| 6-7km | 46% |
| 7-8km | 40% |
| 8-9km | 34% |
| 9-10km | 29% |

454. Due to the low numbers of red-throated diver recorded within the survey area, it was agreed with Natural England that there were insufficient data to enable model-based density estimates to be calculated. Therefore, it has not been possible to estimate the density and abundance of red-throated diver for each of the 1km bands in the area of overlap between the 10km buffer of the windfarm site and the SPA (**Figure 8.1**). Following discussions with Natural England, it was agreed for the DCO submission that a simplified approach should be used, where a weighted average displacement rate (taking into account the relative area of each of the 1km bands where they overlap with the SPA) is applied to the overall mean abundance (and 95% CIs (Confidence Intervals)) of red-throated diver. The weighted average displacement rate was applied to the design-based mean abundance of red-throated diver within the overall survey area (see **Table 8.7** and **Table 8.8**). This approach is considered precautionary for the following reasons:

- The approach assumes that the whole estimated population (including the small number of birds recorded outside of the SPA) are present within the overlap between the buffer area and the SPA, and that birds are distributed evenly within the buffer area. This will result in an overestimate of displacement effect, as surveys indicated (as expected) that more birds were present in the more distant 1km bands (i.e. within the SPA area originally designated for red-throated diver, but where the displacement effect is smaller), and very few birds were present in the areas closest to the windfarm site where the predicted displacement effect is greatest (refer to Figures 71-74 of **Appendix 12.2** of the ES for the locations of birds recorded during surveys).

- Results have been presented assuming a mortality rate of 1% and 10%. As discussed in **Paragraphs 435 – 438**, 1% is considered a more realistic value (with higher mortality rates considered to be biologically implausible), but it is considered that even the 1% value is precautionary and may overestimate the actual mortality effect.
- The assessment includes small numbers of birds recorded outside of the wintering period. It is likely that these are passage birds not associated with the SPA population, and this will therefore lead to an overestimate in the effect.

455. In addition to the estimate of displacement and mortality for red-throated diver, the effective area of the SPA which would be subject to displacement as a percentage of the SPA has also been calculated. This was derived as the product of the Natural England displacement gradient and the area of each of the 1km bands as a proportion of the total SPA area. Again, this approach is considered precautionary, given the lower densities (and therefore assumed lower habitat suitability) of the areas closer to the windfarm site where the displacement rates are predicted to be highest (noting that such areas also lie outside the area of the SPA which was actually designated on the basis of the occurrence and distribution of red-throated diver).

456. **Table 8.7** presents the results of the precautionary potential displacement and mortality estimates for the 10km buffer in relation to the entirety of the Liverpool Bay SPA (current boundary). The total mean number of birds potentially affected in each season is 5.2 (autumn), 21.1 (spring), 4.1 (winter) and 4.2 (breeding; noting that these are unlikely to be associated with the SPA population); these represent 0.29%, 1.17%, 0.23% and 0.23% of the SPA population (1,800 individuals) respectively. Predicted annual mortality due to displacement, assuming the more plausible (but still precautionary) 1% mortality rate, is 0.35 (95% CI: 0.00-1.11), representing a net increase in background mortality of 0.08% (95% CI: 0.00-0.26%). This is based on the SPA population of 1,800 birds and a background annual mortality rate of 0.233 (419 birds per annum).

457. **Table 8.8** presents the same calculation, based on the original (pre-2017) Liverpool Bay SPA boundary. This is considered most relevant as it is focussed on the area of the SPA that was designated on the basis of red-throated diver and which is of highest importance for this feature (and which, as expected, held the majority of birds during baseline surveys). For a 1% mortality rate, it was estimated that 0.02 (95% CI: 0.00-0.07) birds would be predicted to die per annum as a result of displacement, representing a net increase in background mortality of 0.01% (95% CI: 0.00-0.02%).

458. **Table 8.9** presents the results of an assessment to estimate the effective area of the SPA which would be subject to displacement. This estimates that a maximum of 9.07% of the SPA would be affected by the Project (this would be 1.24% if only effects on the original (i.e. pre-2017) SPA boundary are considered – **Table 8.10**). However, taking into account the diminishing effect of the windfarm as distance from the windfarm array increases, the effective area (applying the same gradient as for the mortality calculation) would be 4.63% (and 0.43% if based upon the original SPA boundary – **Table 8.10**). As a means of assessing the extent to which the area of the SPA would be affected by the Project, it is noted that the former figure is preferred by Natural England, while the latter is considered by the Applicant to be more appropriate. This is because the former approach takes no account of the diminishing scale of the potential effect with increasing distance from the windfarm site and the Applicant considers that this leads to a potentially misleading overestimate of the scale of the predicted effect. By contrast, the latter approach incorporates the effect of distance within the calculated metric.

459. Review of information presented by HiDef for Liverpool Bay SPA (2023) confirms that concentrations of red-throated occurred predominantly in areas closest to the coast, with very low densities (effectively zero in most surveys) occurring within the area potentially impacted by the Project (i.e. the overlap between the project buffer and the original SPA boundary as shown in **Figure 8.1**, and as shown in Figures 9 and 10 of the HiDef (2023) report). This indicates that areas potentially impacted by the Project are rarely used by red-throated diver, and given that these are also relatively distant from the windfarm site (>7km), it is considered that significant effects on the abundance and distribution of this species within this area are very unlikely.

460. The Project-alone assessment does not take into account the effects from existing windfarms in the area, but as demonstrated for the in-combination assessment below, some areas which are within the area of overlap between the SPA and the Project's 10km buffer are already potentially affected by existing projects. Therefore, the actual effect of the Project will be less than predicted above because (on the basis of the underpinning assumptions of the assessment approach) such areas are already subject to reduced red-throated densities as a consequence of windfarm displacement.

461. **Table 8.3** sets out embedded mitigation measures that would be implemented to reduce potential impacts on red-throated divers, both during the construction and operation/maintenance phases of the Project. Such measures would be agreed with Natural England and included in relevant construction and operation/maintenance management plans.

462. On the basis of the above, it is concluded that the Project-alone would not affect the conservation objectives of Liverpool Bay SPA as set out in **Table 8.6**. This confirms that there would be no significant effects on red-throated

diver during the operation and maintenance phase, and consequently **no adverse effect on the integrity of Liverpool Bay SPA**.

*Table 8.6 Red-throated diver: Summary of Project-alone effects on conservation objectives of Liverpool Bay SPA*

| Conservation objective | Potential effect | Adverse effect on integrity? |
|---|---|---|
| Extent and distribution of the habitats of the qualifying features | No impacts on the extent and distribution of supporting habitats predicted | No |
| Structure and function of the habitats of the qualifying features | No impacts on structure and function of habitats predicted | No |
| Supporting processes on which the habitats of the qualifying features rely | No impacts on supporting processes predicted | No |
| Population of each of the qualifying features | Increase in background mortality predicted to be significantly below 1% and therefore undetectable against background variation | No |
| Distribution of the qualifying features within the site | Potentially impacted areas within the SPA support very low densities of red-throated diver, and are distant (>7km) from the windfarm site. No significant changes to the distribution of red-throated diver within the SPA are therefore predicted | No |

463. The confidence in the assessment is medium. Firstly, the evidence used to set the displacement rates presented in **Chapter 12 Offshore Ornithology** of the ES and **Appendix 12.1** of the ES is of high applicability and quality. As set out above, there is good evidence to suggest that 1% mortality for displaced birds is suitably precautionary. However, uncertainty remains around the effects of displacement on this species.

*Table 8.7 Seasonal and annual displacement and mortality estimates for red-throated diver within overlap between the survey area and the Liverpool Bay SPA (current SPA boundary)*

| | | | | Autumn migration No. of displaced birds | | | Spring migration No. of displaced birds | | | Winter No. of displaced birds | | | Breeding No. of displaced birds | | | Annual No. of displaced birds | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Buffer** | **Area ($km^2$)** | **% of overlap area** | **Displacement rate** | **LCL** | **Mean** | **UCL** | **LCL** | **Mean** | **UCL** | **LCL** | **Mean** | **UCL** | **LCL** | **Mean** | **UCL** | **LCL** | **Mean** | **UCL** |
| **Mean peak seasonal abundance estimates (whole survey area)** | | | | **0** | **10** | **24** | **0** | **41** | **115** | **0** | **8** | **43** | **0** | **8** | **35** | **-** | **-** | **-** |
| **0-1 km** | 14.65 | 6% | 80% | 0.00 | 0.52 | 1.22 | 0.00 | 2.12 | 5.90 | 0.00 | 0.42 | 2.20 | 0.00 | 0.42 | 1.79 | 0.00 | 3.47 | 11.11 |
| **1-2km** | 16.37 | 7% | 74% | 0.00 | 0.54 | 1.26 | 0.00 | 2.19 | 6.10 | 0.00 | 0.43 | 2.27 | 0.00 | 0.44 | 1.85 | 0.00 | 3.59 | 11.48 |
| **2-3km** | 17.93 | 8% | 68% | 0.00 | 0.54 | 1.27 | 0.00 | 2.20 | 6.13 | 0.00 | 0.43 | 2.29 | 0.00 | 0.44 | 1.86 | 0.00 | 3.61 | 11.55 |
| **3-4km** | 20.47 | 9% | 63% | 0.00 | 0.57 | 1.34 | 0.00 | 2.33 | 6.49 | 0.00 | 0.46 | 2.42 | 0.00 | 0.46 | 1.97 | 0.00 | 3.82 | 12.22 |
| **4-5km** | 24.02 | 10% | 57% | 0.00 | 0.61 | 1.43 | 0.00 | 2.47 | 6.89 | 0.00 | 0.49 | 2.57 | 0.00 | 0.49 | 2.09 | 0.00 | 4.06 | 12.98 |
| **5-6km** | 26.27 | 11% | 51% | 0.00 | 0.59 | 1.40 | 0.00 | 2.42 | 6.74 | 0.00 | 0.48 | 2.52 | 0.00 | 0.48 | 2.05 | 0.00 | 3.97 | 12.70 |
| **6-7km** | 28.92 | 13% | 46% | 0.00 | 0.59 | 1.39 | 0.00 | 2.40 | 6.69 | 0.00 | 0.47 | 2.50 | 0.00 | 0.48 | 2.03 | 0.00 | 3.94 | 12.61 |
| **7-8km** | 26.80 | 12% | 40% | 0.00 | 0.48 | 1.12 | 0.00 | 1.94 | 5.39 | 0.00 | 0.38 | 2.01 | 0.00 | 0.39 | 1.64 | 0.00 | 3.18 | 10.16 |
| **8-9km** | 26.26 | 11% | 34% | 0.00 | 0.40 | 0.93 | 0.00 | 1.61 | 4.49 | 0.00 | 0.32 | 1.68 | 0.00 | 0.32 | 1.36 | 0.00 | 2.65 | 8.46 |
| **9-10km** | 27.52 | 12% | 29% | 0.00 | 0.35 | 0.83 | 0.00 | 1.44 | 4.02 | 0.00 | 0.28 | 1.50 | 0.00 | 0.29 | 1.22 | 0.00 | 2.36 | 7.56 |
| **Total area** | 229.22 | | **Total birds displaced** | 0.00 | 5.19 | 12.18 | 0.00 | 21.12 | 58.85 | 0.00 | 4.15 | 21.95 | 0.00 | 4.20 | 17.86 | 0.00 | 34.66 | 110.83 |
| **Total mortality** | | | **1% mortality** | 0.00 | 0.05 | 0.12 | 0.00 | 0.21 | 0.59 | 0.00 | 0.04 | 0.22 | 0.00 | 0.04 | 0.18 | 0.00 | 0.35 | 1.11 |
| | | | **10% mortality** | 0.00 | 0.52 | 1.22 | 0.00 | 2.11 | 5.88 | 0.00 | 0.41 | 2.20 | 0.00 | 0.42 | 1.79 | 0.00 | 3.47 | 11.08 |
| **Increase in background mortality** | | | **1% mortality** | 0.00% | 0.01% | 0.03% | 0.00% | 0.05% | 0.14% | 0.00% | 0.01% | 0.05% | 0.00% | 0.01% | 0.04% | **0.00%** | **0.08%** | **0.26%** |
| | | | **10% mortality** | 0.00% | 0.12% | 0.29% | 0.00% | 0.50% | 1.40% | 0.00% | 0.10% | 0.52% | 0.00% | 0.10% | 0.43% | **0.00%** | **0.83%** | **2.64%** |

Note: Table presents seasonal and annual estimates of red-throated diver displacement for each 1km band from 0-10km (column 1). The area of each band has been calculated, where this overlaps with the SPA (column 2; see **Figure 8.1**), and the relative area (percentage of the total SPA overlap; 229.22$km^2$) of each band then calculated (column 3). The abundance estimates for each season (row in pale blue) have then been used to estimate the number of birds displaced; this is the product of the relative area (column 3), the Natural England displacement rate (column 4) and the abundance estimate, calculated for mean and 95% CIs. For each season, the number of birds displaced for each 1km band has been summed to provide an estimate for the total number of birds displaced, by season and annually. Mortality estimates have been calculated assuming 1% and 10% mortality, and these values used to estimate the predicted increase in background mortality (assuming SPA population of 1,800 individuals and background annual mortality rate of 0.233). LCL = Lower Confidence Limit, UCL = Upper Confidence Limit

*Table 8.8 Seasonal and annual displacement and mortality estimates for red-throated diver within overlap between the survey area and the Liverpool Bay SPA (pre-2017 SPA boundary)*

| | | | | Autumn migration No. of displaced birds | | | Spring migration No. of displaced birds | | | Winter No. of displaced birds | | | Breeding No. of displaced birds | | | Annual No. of displaced birds | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Buffer** | **Area ($km^2$)** | **% of overlap area** | **Displacement rate** | **LCL** | **Mean** | **UCL** | **LCL** | **Mean** | **UCL** | **LCL** | **Mean** | **UCL** | **LCL** | **Mean** | **UCL** | **LCL** | **Mean** | **UCL** |
| **Peak seasonal abundance estimates** | | | | 0 | 10 | 24 | 0 | 41 | 115 | 0 | 8 | 43 | 0 | 8 | 35 | - | - | - |
| **0-1 km** | 0.00 | 0% | 80% | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| **1-2km** | 0.00 | 0% | 74% | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| **2-3km** | 0.00 | 0% | 68% | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| **3-4km** | 0.00 | 0% | 63% | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| **4-5km** | 0.00 | 0% | 57% | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| **5-6km** | 0.00 | 0% | 51% | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| **6-7km** | 1.09 | 0% | 46% | 0.00 | 0.02 | 0.05 | 0.00 | 0.09 | 0.25 | 0.00 | 0.02 | 0.09 | 0.00 | 0.02 | 0.08 | 0.00 | 0.15 | 0.47 |
| **7-8km** | 5.30 | 2% | 40% | 0.00 | 0.09 | 0.22 | 0.00 | 0.38 | 1.07 | 0.00 | 0.08 | 0.40 | 0.00 | 0.08 | 0.32 | 0.00 | 0.63 | 2.01 |
| **8-9km** | 7.26 | 3% | 34% | 0.00 | 0.11 | 0.26 | 0.00 | 0.45 | 1.24 | 0.00 | 0.09 | 0.46 | 0.00 | 0.09 | 0.38 | 0.00 | 0.73 | 2.34 |
| **9-10km** | 7.47 | 3% | 29% | 0.00 | 0.10 | 0.23 | 0.00 | 0.39 | 1.09 | 0.00 | 0.08 | 0.41 | 0.00 | 0.08 | 0.33 | 0.00 | 0.64 | 2.05 |
| **Total area** | 21.12 | | **Total birds displaced** | 0.00 | 0.32 | 0.76 | 0.00 | 1.31 | 3.65 | 0.00 | 0.26 | 1.36 | 0.00 | 0.26 | 1.11 | 0.00 | 2.15 | 6.88 |
| **Total mortality** | | | **1% mortality** | 0.00 | 0.00 | 0.01 | 0.00 | 0.01 | 0.04 | 0.00 | 0.00 | 0.01 | 0.00 | 0.00 | 0.01 | 0.00 | 0.02 | 0.07 |
| | | | **10% mortality** | 0.00 | 0.03 | 0.08 | 0.00 | 0.13 | 0.37 | 0.00 | 0.03 | 0.14 | 0.00 | 0.03 | 0.11 | 0.00 | 0.22 | 0.69 |
| **Increase in background mortality** | | | **1% mortality** | 0.00% | 0.00% | 0.00% | 0.00% | 0.00% | 0.01% | 0.00% | 0.00% | 0.00% | 0.00% | 0.00% | 0.00% | 0.00% | 0.01% | 0.02% |
| | | | **10% mortality** | 0.00% | 0.01% | 0.02% | 0.00% | 0.03% | 0.09% | 0.00% | 0.01% | 0.03% | 0.00% | 0.01% | 0.03% | 0.00% | 0.05% | 0.16% |

Note: Table presents seasonal and annual estimates of red-throated diver displacement for each 1km band from 0-10km (column 1). The area of each band has been calculated, where this overlaps with the SPA (column 2; see **Figure 8.1**), and the relative area (percentage of the total SPA overlap; 21.12$km^2$) of each band then calculated (column 3). The abundance estimates for each season (row in pale blue) have then been used to estimate the number of birds displaced; this is the product of the relative area (column 3), the Natural England displacement rate (column 4) and the abundance estimate, calculated for mean and 95% CIs. For each season, the number of birds displaced for each 1km band has been summed to provide an estimate for the total number of birds displaced, by season and annually. Mortality estimates have been calculated assuming 1% and 10% mortality, and these values used to estimate the predicted increase in background mortality (assuming SPA population of 1,800 individuals and background annual mortality rate of 0.233). LCL = Lower Confidence Limit, UCL = Upper Confidence Limit

*Table 8.9 Estimate of effective area of the SPA which would be subject to displacement within Liverpool Bay SPA (current SPA boundary)*

| Buffer | Area ($km^2$) | Effect gradient | Effective area of SPA subject to displacement ($km^2$) |
|---|---|---|---|
| **0-1 km** | 14.65 | 80% | 11.72 |
| **1-2km** | 16.37 | 74% | 12.12 |
| **2-3km** | 17.93 | 68% | 12.19 |
| **3-4km** | 20.47 | 63% | 12.90 |
| **4-5km** | 24.02 | 57% | 13.69 |
| **5-6km** | 26.27 | 51% | 13.40 |
| **6-7km** | 28.92 | 46% | 13.30 |
| **7-8km** | 26.80 | 40% | 10.72 |
| **8-9km** | 26.26 | 34% | 8.93 |
| **9-10km** | 27.52 | 29% | 7.98 |
| **Total** | 229.22 | | 116.95 |
| **Percentage of SPA**[1] | **9.07%** | | **4.63%** |
| [1] Assumes SPA area of 2527.58$km^2$ | | | |

*Table 8.10 Estimate of effective area of the SPA which would be subject to displacement within Liverpool Bay SPA (pre-2017 SPA boundary)*

| Buffer | Area ($km^2$) | Effect gradient | Effective area of SPA subject to displacement ($km^2$) |
|---|---|---|---|
| **0-1 km** | 0.00 | 80% | 0.00 |
| **1-2km** | 0.00 | 74% | 0.00 |
| **2-3km** | 0.00 | 68% | 0.00 |
| **3-4km** | 0.00 | 63% | 0.00 |
| **4-5km** | 0.00 | 57% | 0.00 |
| **5-6km** | 0.00 | 51% | 0.00 |
| **6-7km** | 1.09 | 46% | 0.50 |
| **7-8km** | 5.30 | 40% | 2.12 |
| **8-9km** | 7.26 | 34% | 2.47 |
| **9-10km** | 7.47 | 29% | 2.17 |

| Buffer | Area (km$^2$) | Effect gradient | Effective area of SPA subject to displacement (km$^2$) |
|---|---|---|---|
| **Total** | 21.12 | | 7.25 |
| **Percentage of SPA[1] (boundary at designation)** | **1.24%** | | **0.43%** |
| **Percentage of SPA[2] (current boundary)** | **0.84%** | | **0.29%** |
| [1] Assumes SPA area of 1702.93km$^2$ <br> [2] Assumes SPA area of 2527.58km$^2$ | | | |

**In-combination**

464. On the basis of the conclusions of the Project-alone assessment (i.e. very low predicted red-throated diver mortality, and no impact on the distribution of the species within the SPA), there would be no discernible contribution of the Project to in-combination effects. **Accordingly, no in-combination assessment is required for this feature.** Notwithstanding this conclusion, an in-combination assessment is presented below, to provide context to the Project-alone assessment.

465. For the operation and maintenance phase, the assessment of in-combination disturbance, displacement and barrier effects considers both mortality and effective area of displacement for other relevant OWF projects within 10km of Liverpool Bay SPA in-combination with the Project. The first stage in the assessment involved project screening to identify projects that are relevant to the SPA, as follows:

- Where a project was operational prior to designation of the SPA in 2010, this has been excluded from the in-combination assessment as it is considered that any impacts arising from these projects were accounted for at the time of designation. Of the projects considered in the wider in-combination assessment (refer to projects considered in the cumulative assessment in **Chapter 12 Offshore Ornithology** of the ES), Rhyl Flats (2009) and Burbo Bank (2007) OWFs have therefore been excluded from the in-combination assessment. As set out in **Chapter 12 Offshore Ornithology**, Barrow and North Hoyle OWFs have also been excluded from the cumulative and in-combination assessments as the consent for these historic projects will not overlap with the Project.
- The assessment has considered both the original SPA boundary (which was determined on the basis of red-throated diver distribution) and the revised (2017) SPA boundary (which was designated primarily for little

gull and not on the basis of red-throated diver distribution – see above). Those OWF projects that were operational prior to the extension of the SPA boundary (West of Duddon Sands (2014) and Gwynt y Môr (2015)) have been excluded from the in-combination assessment in respect of the SPA extension area, as it was considered that any impacts arising from these projects on this area have been accounted for at the time of designation.

- The assessment has also considered the overlap of OWFs (and 10km buffer around each) with the original and extended SPA boundaries. As discussed above, the SPA extension area was designated primarily for little gull, and therefore any impacts on this area are considered less relevant in respect of red-throated diver.

466. Given that Morgan and Mona Offshore Wind Projects are both greater than 10km from the SPA, no displacement is predicted due to the presence/operation of the windfarms themselves, i.e. any potential disturbance and displacement effects would instead be due to vessel traffic (assuming these vessels will transit the SPA). It is assumed that embedded mitigation measures similar to those set out in **Table 8.3** for operation and maintenance for the Project would also be implemented by Morgan and Mona Offshore Wind Projects, and accordingly no significant disturbance and displacement effects associated with vessel traffic (either alone or in-combination) would occur. Similarly, the Morgan and Morecambe OWFs Transmission Assets would not contribute any measurable effect during the operation and maintenance phase. The Morgan and Morecambe OWFs Transmission Assets Draft ISAA (2023) estimated a maximum increase in background mortality for red-throated diver of 0.07 birds / 0.04%. This was acknowledged as a significant overestimate, as it was based on a relatively large search area, rather than the actual area that potential structures would occupy. The assessment concluded that the 'level of mortality is considered to be precautionary and falls below any perceptible threshold of significance that could be considered In-combination with any other projects'. Accordingly, the Morgan and Morecambe OWFs Transmission Assets are not considered further within the in-combination assessment.

467. Considering the project screening approach described above, **Table 8.11** sets out the OWFs that have been considered in the in-combination assessment. In summary, Burbo Bank Extension, West of Duddon Sands, Gwynt y Môr and Awel y Môr OWFs are relevant to the in-combination assessment in relation to the original and extended SPA boundaries, while Walney 1&2 and Walney 3&4 OWFs are primarily relevant to the SPA extension area, as buffers from these projects only overlap with the SPA extension. All other projects are screened out of the in-combination assessment for the red-throated diver feature.

*Table 8.11 OWF projects considered in the in-combination assessment for Liverpool Bay SPA red-throated diver*

| Project name | Within 10km of original (2010) SPA boundary | Operational prior to designation (2010) | Within 10km of extended (2017) SPA boundary | Operational prior to extension (2017) | Included in assessment |
|---|---|---|---|---|---|
| Burbo Bank Extension | Yes | No | Yes | No | Yes |
| Burbo Bank | Yes | Yes | - | - | No |
| Walney 1&2 | No | - | Yes | No | Yes* |
| Walney 3&4 | No | - | Yes | No | Yes* |
| West of Duddon Sands | Yes | No | Yes | Yes | Yes |
| Ormonde | No | - | Yes | Yes | No |
| Awel y Môr | Yes | No | Yes | No | Yes |
| Gwynt y Môr | Yes | No | Yes | Yes | Yes |
| Rhyl Flats | Yes | Yes | - | - | No |
| Mona | No | - | No | - | No |
| Morgan Generation Assets | No | - | No | - | No |

* Relevant to SPA extension boundary only

Note: Cells shaded red indicate a feature that would screen out that project from assessment, and green cells indicate a feature that would screen in. The screening has considered both the original and current SPA boundaries. For example, for Ormonde, the project is beyond the 10km buffer for the original SPA boundary, so would not contribute to the assessment for that area. It is located within 10km of the boundary extension, but was operational at the time of designation of the extension. Overall, therefore, Ormonde is excluded from the assessment.

468. **Table 8.13** sets out the relevant population estimates for each of the projects considered for the in-combination assessment, together with predicted mortality (assuming 10% and evidence-based 1% mortality of displaced birds, as set out in **Paragraphs 435** and **438** above, and noting that 1% is analogous with the approach used by the consented Awel y Môr project (Awel y Môr Offshore Wind Farm Ltd, 2022).

469. Limited population data were available for historic projects. For Burbo Bank Extension, Awel y Môr and Gwynt y Môr OWFs the relevant populations have been taken from data presented in the Awel y Môr RIAA (Awel y Môr Offshore Wind Farm Ltd, 2022). For West of Duddon Sands, the population has been calculated from density data presented in the Natural England commissioned report (HiDef, 2023). The latter is the basis of the current population estimate (1,800 birds) used in the conservation objectives for the SPA, and also provides an estimation of mean density across the whole SPA (1.06 birds/km$^2$).

470. The gradient applied to the Project-alone assessment (**Table 8.5**) has been used to estimate the number of birds likely to be impacted in 1km bands around the relevant windfarm (for those parts of the 1km bands that overlap with the SPA). It is noted that for West of Duddon Sands OWF, the use of the mean density estimates is likely to overestimate the number of impacted birds as the buffer is primarily located within the SPA extension area (where densities of red-throated diver are predicted to be low) and HiDef (2023) also indicates that densities within the remaining buffer areas that overlap with the original SPA boundary were low.

471. The approach used to estimate the abundance of red-throated diver within each applicable windfarm considered within the in-combination assessment is set out in **Table 8.12**.

*Table 8.12 Approach to estimating red-throated diver population for projects considered in the in-combination assessment for Liverpool Bay SPA*

| OWF project name | Population estimation approach |
|---|---|
| Burbo Bank Extension | Re-estimated from density data presented in Burbo Bank Extension (0.48 birds/km$^2$, taken from Table 8 of NIRAS, 2013) and using gradient to estimate number of potentially impacted birds within the original SPA boundary (120 birds total). |
| Walney 1 & 2 | The areas potentially impacted by this project (i.e. within 10km) are already affected by West of Duddon Sands OWF, which is closer to the SPA. Therefore, no additional birds (zero) are included in the in-combination assessment. It is noted that no part of this site lies within 10km of the original SPA boundary, where the densities of red-throated divers are expected to be highest. |

| OWF project name | Population estimation approach |
|---|---|
| Walney 3 & 4 | The areas potentially impacted by this project (i.e. within 10km) are already affected by West of Duddon Sands OWF, which is closer to the SPA. Therefore, no additional birds (zero) are included in the in-combination assessment. It is noted that no part of this site lies within 10km of the original SPA boundary, where the densities of red-throated divers are expected to be highest. |
| West of Duddon Sands | Population estimate calculated from mean density estimated from data presented in HiDef (2023) (1.06 birds/km$^2$), with the displacement rate gradient in Table 8.5 used to estimate the effective number of impacted birds. As the windfarm was operational prior to the extension of the SPA, the effects will apply only to the original SPA boundary. This gives an estimated 31.9 birds at risk of displacement. |
| Awel y Môr | Estimate re-calculated from Awel y Môr RIAA (Awel y Môr Offshore Wind Farm Ltd, 2022). This estimated 195 birds would occur within areas potentially impacted by the OWF (windfarm+8km), an equivalent density of 0.94 birds/km$^2$. This density and displacement gradient has been used to estimate the total number of birds at risk of displacement for windfarm +10km, 133 birds. |
| Gwynt y Môr | Estimate from Awel y Môr RIAA; 35 individuals (Awel y Môr Offshore Wind Farm Ltd, 2022). As the windfarm was operational prior to the extension of the SPA, the effects will apply only to the original SPA boundary |

*Table 8.13 Liverpool Bay SPA Red-throated diver – in-combination population estimates and predicted mortality (assuming 10 % and 1% mortality of displaced birds) due to disturbance and displacement*

| OWF project name | Population estimate | Predicted annual mortality (10%) | Predicted annual mortality (1%) |
|---|---|---|---|
| Burbo Bank Extension | 120 | 12.00 | 1.20 |
| Walney 1 & 2 | 0 | 0.00 | 0.00 |
| Walney 3 & 4 | 0 | 0.00 | 0.00 |
| West of Duddon Sands | 32 | 3.19 | 0.32 |
| Awel y Môr | 133 | 13.30 | 1.33 |
| Gwynt y Môr | 35 | 3.50 | 0.35 |
| **The Project** | 35 | 3.47 | 0.35 |
| **Total** | **355** | **35.45** | **3.55** |

472. Assuming a realistic, evidence-based mortality rate of 1% for displaced birds and based on the Liverpool Bay SPA non-breeding population of 1,800 birds and a background mortality of 0.233 (419 birds per annum), an increase in mortality of 3.55 birds would increase background mortality by 0.85%. This is below the 1% threshold where a detectable effect on the SPA population could occur. It is noted that the assessment included a number of layers of precaution (e.g. as set out in **Paragraph 454**), which provides further confidence in the conclusions to the assessment.

473. The assessment has also considered the effective area of the SPA that would be impacted by OWF projects in-combination. For each applicable windfarm project, the total overlap of the windfarm site and 10km buffer with the current and former Liverpool Bay SPA boundaries has been calculated. In addition, areas have been calculated for the 1km bands from 1-10km around each windfarm site to enable the effective in-combination area of potential displacement to be calculated, using the gradient values presented in **Table 8.5**. Where there was overlap between buffers from different windfarms, the calculation did not double-count the area of overlap, with the higher gradient value applied to each overlap area to ensure that these were not underestimated.

474. The areas of overlap of the SPA with the 1km bands of the 10km buffers for each of the OWFs contributing to the in-combination assessment are presented in **Table 8.15** (current SPA boundary) and **Table 8.16** (original SPA boundary) and shown on **Figure 8.2**.

475. As explained in **Paragraph 457**, the assessment against the original SPA boundary is considered most insightful, as this represents the core areas for red-throated diver within the SPA. Nonetheless, data for both the original and extended (current) SPA are presented.

476. Based on the current SPA boundary, 53.29% of the SPA would be impacted in-combination, of which the Project contributes 8.75%. Applying the gradient to these values resulted in an effective impacted area of 30.46% of the SPA, of which the Project contributes 4.52%.

477. Based on the original (pre-2017) SPA boundary (for which red-throated diver was designated), the gross impacted area would be 42.55% of the original SPA of which the Project would contribute 1.06%. If the gradient is applied, the total impacted area would be 23.50%, of which the Project contributes 0.37%. It is noted that the percentage of the original SPA that would be impacted by the Project is less when considered in-combination than for the Project-alone, reflecting the impact of existing projects (particularly West of Duddon Sands) on the area potentially affected by the Project.

478. As the original SPA boundary represents the core habitat for red-throated diver, it is considered that these latter values provide the most appropriate

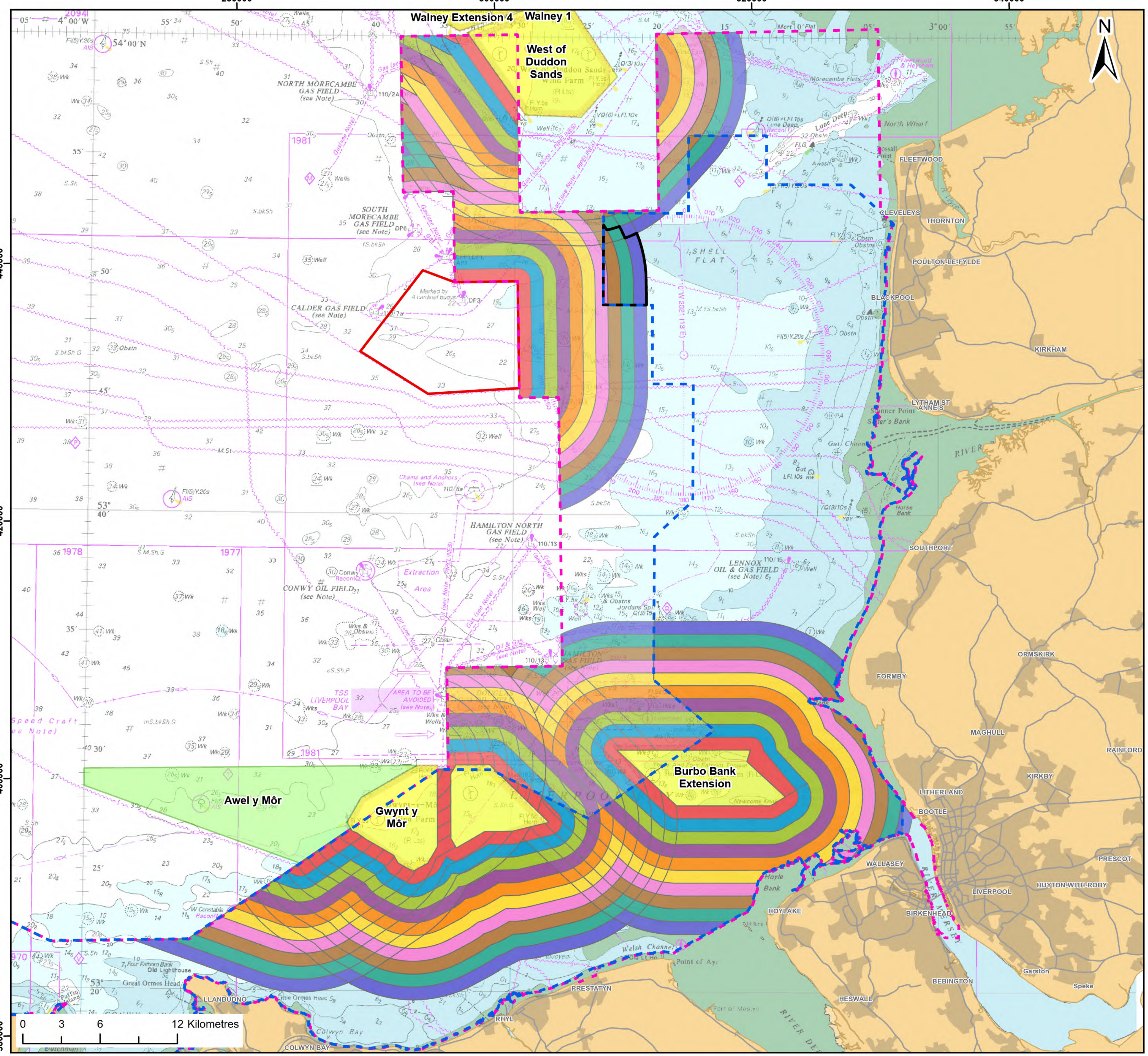
measure of the effect on this feature. Given the precautionary nature of the Project assessment and taking into account the potential effects of operational and consented offshore windfarms, the relative contribution of the Project to the in-combination value (whether or not the gradient is applied, i.e. 0.37% or 1.06% of the original SPA area) is considered inconsequential to the overall in-combination assessment. As set out in **Paragraph 459**, the area within the original SPA boundary that is potentially impacted by the Project (and is not already potentially impacted by existing projects, as shown on **Figure 8.2**) supports very low numbers of red-throated diver (as evidenced by HiDef, 2023). It is therefore considered that the Project is very unlikely to result in additional measurable change to the abundance or distribution of red-throated diver within the SPA.

479. It is noted that in the HRA of the Awel y Môr OWF project (DESNZ, 2023a), the Secretary of State (SoS) concluded that an adverse effect on the integrity on the red-throated diver feature of the SPA from the Awel y Môr project in-combination with other projects could be excluded. This confirms that the SoS's position was that the areas already potentially impacted by Awel y Môr OWF and other existing projects (as shown on **Figure 8.2**) would not result in an adverse effect on the integrity of the SPA. As the contribution of the Project, together with the Mona and Morgan Offshore Wind Projects, would have no measurable additional effect on the distribution of red-throated divers within the SPA, and that the predicted in-combination mortality is below the threshold likely to be detectable against background variation, it is therefore considered unlikely that the SoS would reach a materially different conclusion in this regard.

480. **It is therefore concluded that the Project (alone and in-combination) would not affect the conservation objectives of Liverpool Bay SPA as set out in Table 8.14. This confirms that there would be no adverse effect on the integrity of Liverpool Bay SPA, when considering the Project in-combination with other plans or projects.**

*Table 8.14 Red-throated diver: Summary of in-combination effects on conservation objectives of Liverpool Bay SPA*

| Conservation objective | Potential effect | Adverse effect on integrity? |
|---|---|---|
| Extent and distribution of the habitats of the qualifying features | SoS has agreed in their assessment of the recently consented Awel y Môr OWF project that there would be no adverse effect on site integrity, when considered in-combination. The Project would make no measurable change to habitats of importance to the SPA red-throated diver population, and therefore it is unlikely that this conclusion would change. | No |
| Structure and function of the habitats of the qualifying features | | No |
| Supporting processes on which the habitats of the qualifying features rely | | No |
| Population of each of the qualifying features | Increase in background mortality predicted to be below 1% and therefore undetectable against background variation | No |
| Distribution of the qualifying features within the site | SoS has agreed in their assessment of the recently consented Awel y Môr OWF project that there would be no adverse effect on site integrity, when considered in-combination. The Project would have no measurable additional effect on the distribution of red-throated divers within the SPA, and therefore it is unlikely that this conclusion would change. | No |




Legend:

- Morecambe Offshore Windfarm Site
- Liverpool Bay Special Protection Area (SPA) boundary at original designation
- Liverpool Bay Special Protection Area (SPA)
- Area within original Special Protection Area (SPA) boundary potentially impacted by Morecambe Project only

**Wind farm status**

- Fully commissioned
- Consented

**Red Throated Diver Distance Buffer**

- 0-1km
- 1-2km
- 2-3km
- 3-4km
- 4-5km
- 5-6km
- 6-7km
- 7-8km
- 8-9km
- 9-10km



Report:
Morecambe Offshore Windfarm: Generation Assets
Habitat Regulations Report to Inform Appropriate Assessment

Title:
Red-throated diver - buffer areas used in the in-combination assessment

Figure: 8.2 | Drawing No: PC1165-RHD-ES-OF-DR-Z-0118

| Revision: | Date: | Drawn: | Checked: | Size: | Scale: |
|---|---|---|---|---|---|
| P01 | 21/02/2024 | JH | GC | A3 | 1:300,000 |
| P02 | 22/04/2024 | JH | SB | A3 | 1:300,000 |

Co-ordinate system: WGS 1984 UTM Zone 30N

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*Table 8.15 In-combination estimate of effective area of the SPA which would be subject to displacement within Liverpool Bay SPA (current boundary)*

| Buffer | Effect gradient | Total area excluding the Project (km²) | Total area – Project only (km²) | Total area in-combination (km²) | Effective area of SPA subject to displacement excluding the Project (km²) | Effective area of SPA subject to displacement - Project only (km²) | Total effective area of SPA subject to displacement in-combination (km²) |
|---|---|---|---|---|---|---|---|
| **Within SPA** | 100% | 108.03 | 0.00 | 108.03 | 108.03 | 0.00 | 108.03 |
| **0-1 km** | 80% | 83.64 | 14.65 | 98.29 | 66.91 | 11.72 | 78.63 |
| **1-2km** | 74% | 87.87 | 16.37 | 104.24 | 65.02 | 12.12 | 77.14 |
| **2-3km** | 68% | 100.27 | 17.93 | 118.19 | 68.18 | 12.19 | 80.37 |
| **3-4km** | 63% | 114.16 | 20.48 | 134.64 | 71.92 | 12.90 | 84.82 |
| **4-5km** | 57% | 114.92 | 24.03 | 138.94 | 65.50 | 13.70 | 79.20 |
| **5-6km** | 51% | 112.04 | 26.23 | 138.27 | 57.14 | 13.38 | 70.52 |
| **6-7km** | 46% | 112.96 | 27.82 | 140.78 | 51.96 | 12.80 | 64.76 |
| **7-8km** | 40% | 106.48 | 26.43 | 132.91 | 42.59 | 10.57 | 53.17 |
| **8-9km** | 34% | 94.32 | 23.74 | 118.06 | 32.07 | 8.07 | 40.14 |
| **9-10km** | 29% | 90.93 | 23.60 | 114.52 | 26.37 | 6.84 | 33.21 |
| **Total** | | **1125.61** | **221.27** | **1346.88** | **655.70** | **114.28** | **769.98** |
| **Percentage of SPA[1]** | | **44.53%** | **8.75%** | **53.29%** | **25.94%** | **4.52%** | **30.46%** |

[1] Assumes SPA area of 2527.58km²

*Table 8.16 In-combination estimate of effective area of the SPA which would be subject to displacement within Liverpool Bay SPA (original pre-2017 boundary)*

| Buffer | Effect gradient | Total area excluding the Project ($km^2$) | Total area – Project only ($km^2$) | Total area in-combination ($km^2$) | Effective area of SPA subject to displacement excluding the Project ($km^2$) | Effective area of SPA subject to displacement - Project only ($km^2$) | Total effective area of SPA subject to displacement in-combination ($km^2$) |
|---|---|---|---|---|---|---|---|
| **Within SPA** | 100% | 39.64 | 0.00 | 39.64 | 39.64 | 0.00 | 39.64 |
| **0-1 km** | 80% | 58.79 | 0.00 | 58.79 | 47.03 | 0.00 | 47.03 |
| **1-2km** | 74% | 53.47 | 0.00 | 53.47 | 39.56 | 0.00 | 39.56 |
| **2-3km** | 68% | 59.62 | 0.00 | 59.62 | 40.54 | 0.00 | 40.54 |
| **3-4km** | 63% | 67.58 | 0.00 | 67.58 | 42.58 | 0.00 | 42.58 |
| **4-5km** | 57% | 73.29 | 0.00 | 73.29 | 41.78 | 0.00 | 41.78 |
| **5-6km** | 51% | 71.50 | 0.00 | 71.50 | 36.46 | 0.00 | 36.46 |
| **6-7km** | 46% | 71.68 | 1.09 | 72.77 | 32.97 | 0.50 | 33.47 |
| **7-8km** | 40% | 84.31 | 5.30 | 89.61 | 33.72 | 2.12 | 35.84 |
| **8-9km** | 34% | 56.68 | 6.13 | 62.81 | 19.27 | 2.08 | 21.36 |
| **9-10km** | 29% | 70.09 | 5.47 | 75.56 | 20.33 | 1.59 | 21.91 |
| **Total** | | **706.64** | **17.99** | **724.64** | **393.88** | **6.29** | **400.18** |
| **Percentage of original SPA[1]** | | **41.50%** | **1.06%** | **42.55%** | **23.13%** | **0.37%** | **23.50%** |

[1] Assumes area of 1702.93$km^2$

#### 8.4.2.2 Common scoter

##### Status

481. Non-breeding common scoter is a qualifying species of the SPA. Liverpool Bay supports the largest aggregation for this species in the UK. At designation, the population comprised 56,679 birds (Lawson *et al.*, 2016), which was 10.31% of the NW European population. Population estimates were based on visual aerial surveys undertaken between 2004 and 2011. Surveys of the original SPA boundary covering the period 2015-2020 are documented in Natural England commissioned report 440 (HiDef, 2023). These surveys primarily covered the peak winter period (January and February), with mean monthly abundance estimates of between 78,797 and 202,224 birds, and a mean peak count of 141,801 birds over that period. It is likely that a small number of birds also occurred within the SPA extension area, and therefore this estimate is likely to be an underestimate. The Conservation Advice Package for Liverpool Bay SPA (Natural England *et al*., 2022) confirmed that 141,801 birds was the population used for the purposes of the conservation objectives, with an abundance target to 'Maintain the size of the non-breeding population at a level which is at or above 141,801 individuals'. This has therefore been assumed to be the reference population for the assessment.

482. Based on the SPA population, and a baseline mortality rate (all age classes) of 0.238 (derived from Horswill and Robinson, 2015; refer to **Chapter 12 Offshore Ornithology** of the ES), 33,749 birds from the SPA population would be expected to die each year.

483. The digital aerial surveys undertaken for the Project (refer to **Appendix 12.1** of the ES and **Appendix 12.2**) recorded very low numbers of common scoter, with the majority of birds recorded outside of the windfarm site, in the eastern part of the 10km survey buffer, i.e. within Liverpool Bay SPA. Very few birds were recorded within the windfarm site + 4km buffer, with an estimated peak mean of 43 birds (0-131) recorded in the non-breeding period.

##### Functional linkage and seasonal apportionment of potential effects

484. No common scoters were recorded within the windfarm site during surveys. Low numbers of this species were recorded within the 4km buffer located within the boundary of Liverpool Bay SPA. The assessment therefore assumes that 100% of common scoters present belong to the Liverpool Bay SPA population.

##### Potential effects on the qualifying feature

485. The common scoter qualifying feature of the Liverpool Bay SPA has been screened into the assessment due to the potential risk of displacement and

barrier effects during the construction, and operation and maintenance, and decommissioning phases of the Project.

486. Common scoter are highly susceptible to disturbance from boat and helicopter traffic (Garthe and Hüppop, 2004), showing disturbance behaviours at distances of over 1km from boats (Kaiser *et al*., 2006; Schwemmer *et al.*, 2011). Fliessbach *et al.,* (2019) found that 81% of common scoters showed escape behaviour in response to ship traffic, and that escape distance for individual birds was higher than other species, with an average distance of 1,600m. This response was reduced to 1,015m when birds were in a flock. There is less evidence regarding their displacement behaviour from the permanent infrastructure associated with OWFs, with Dierschke *et al.,* (2016) claiming that common scoters only weakly avoid OWFs themselves, with the majority of displacement the result of avoidance of boat and helicopter traffic associated with maintenance of OWFs.

### Construction and decommissioning phase disturbance/displacement/barrier effects

*Project-alone*

487. Common scoters were only recorded within the 2km buffer on two occasions during surveys. Therefore, it is considered unlikely that significant disturbance, displacement or barrier impacts would occur during the construction and decommissioning phases of the Project. Nonetheless, a precautionary assessment of disturbance/displacement/barrier effects has been undertaken, assuming 50% of the operational phase effect, to a distance of 4km from the windfarm site; i.e. a displacement rate of 45%-50% and mortality range of 1-10% for displaced birds. This approach is considered precautionary, for the reasons set out in **Paragraph 494**.

488. The assessment concludes that 43 birds (0-131) would be displaced during the winter period, of which 2 birds (0-7) would be predicted to die at 10% mortality. Using a realistic 1% mortality rate, 0.2 (0.0-0.6) birds would be expected to die. The addition of 0.2 birds would result in no measurable increase in the annual mortality rate (i.e. 0.00%). This magnitude of increase in mortality would not materially alter the background mortality of the population and would be undetectable.

489. There is the potential that construction vessels could cause displacement of common scoters within the SPA during transit between port(s) and the windfarm site. Although transit routes are not known, given the port(s) to service the Project have not been selected, it has been assumed there would be transit through the SPA, and embedded mitigation has been defined to minimise such impacts; refer to **Table 8.3**. As details of the transit routes used by construction vessels are unknown it is not possible to quantitatively assess the potential effect of these activities. However, given the expected frequency

of vessel transits relative to existing vessel traffic in the area (particularly during the winter months, where activity would be limited by sea conditions), and embedded mitigation measures, it is predicted that the mortality rate of displaced birds would be very small. It is therefore concluded that impacts will be small, and insufficient to represent an adverse effect on the integrity of the SPA.

490. Accordingly, no significant effects on common scoter are predicted during the construction and decommissioning phases, and **it is concluded that there would be no adverse effect on the integrity of Liverpool Bay SPA**.

491. The confidence in the assessment is medium. Firstly, the evidence used to set the displacement rates presented in **Chapter 12 Offshore Ornithology** of the ES and **Appendix 12.1** of the ES is of high applicability and quality. Whilst there was limited available evidence to inform mortality rates, those selected are considered to be sufficiently precautionary based on expert opinion. However, uncertainty remains around the effects of displacement on this species.

*In-combination*

492. No in-combination effects in respect of common scoter are predicted during the construction or decommissioning phases of the Project. As Project effects are temporary and reversible, it is unlikely that there would be temporal and/or spatial overlap with other plans or projects. Furthermore, any applicable projects would be expected to have similar best practice construction methods to minimise any potential effects. The draft ISAAs for the Morgan and Morecambe OWFs Transmission Assets (Morgan Offshore Wind Limited and Morecambe Offshore Windfarm Ltd, 2023b), Mona Offshore Wind Project (Mona Offshore Wind Limited, 2023) and Morgan Offshore Wind Project Generation Assets (Morgan Offshore Wind Limited, 2023) predicted no measurable mortality for this feature and even if the works overlapped, there would be no measurable increase in common scoter mortality. **It is therefore concluded that there is no potential for the Project to have an adverse effect on the integrity of Liverpool Bay SPA, either alone or in-combination with other plans or projects**.

**Operation and maintenance phase disturbance / displacement / barrier effects**

*Project-alone*

493. Displacement effects for common scoter for the Project were assessed during the non-breeding period, based on a peak mean population of 43 individual birds, calculated for the windfarm site and a 4km buffer, in line with recommendations within the SNCB guidance (SNCB 2017). The population estimate assumes that all birds recorded were located within the SPA boundary, which, as set out in **Paragraph 483**, appeared to be the case, but

in any event this provides additional precaution to the assessment. The displacement matrices used to calculate potential impacts are presented in **Appendix 12.1** of the ES, and summarised in **Table 8.17**. The application of a 90-100% displacement rate (refer to **Paragraph 494**) to all birds within the 4km buffer to determine the total number of birds subject to displacement is precautionary. In reality the proportion of birds displaced is likely to reduce with distance from the windfarm site.

494. Due to the limited evidence available, a displacement rate of 90-100% and mortality rate of 1-10% has been presented. Given that 10% would represent a rate approximately half of the expected 'natural' annual mortality, this rate is considered very unlikely. Accordingly, a 1% mortality rate is considered to be most appropriate, with the upper end of this range likely to be precautionary.

*Table 8.17 Common scoter – predicted operation and maintenance phase displacement and mortality from Liverpool Bay SPA*

| Mean peak abundance estimate type | Mean peak abundance estimate | SPA population | Annual mortality range[1] | Annual baseline mortality increase range[2] |
|---|---|---|---|---|
| Upper 95% CI | 131 | 141,801 | 1-13 | 0.00-0.04% |
| Mean | 43 | 141,801 | 0-4 | 0.00-0.01% |
| Lower 95% CI | 0 | 141,801 | 0-0 | 0.00-0.00% |

[1] Assumes displacement rates of 90-100% and mortality rates of 1-10%
[2] Background mortality rate of 23.8% (Horswill and Robinson, 2015)

495. In addition to disturbance and displacement arising from the windfarm site, there is also the potential that service vessels could cause displacement of common scoters within the SPA during transit between port and the windfarm site. The risk of such effects would be similar to the construction phase (refer to **Paragraph 489**), and would be subject to equivalent embedded mitigation (**Table 8.3**). Accordingly, **it is concluded that the mortality of displaced birds would be very small and would not add significantly to the effect arising from displacement around the windfarm site.**

496. Using realistic but still precautionary values (i.e. mean density and 1% mortality), there would be no measurable increase in common scoter mortality (**Table 8.17**). However, even the maximum (precautionary) values (upper 95% CI density and 10% mortality) would only result in an annual increase in mortality of 0.04%. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. Accordingly, no significant effects on common scoter are predicted during the operation and maintenance

phase, and **it is concluded that there would be no adverse effect on the integrity of Liverpool Bay SPA.**

497. The confidence in the assessment is medium. Firstly, the evidence used to set the displacement rates presented in **Chapter 12 Offshore Ornithology** of the ES and **Appendix 12.1** of the ES submitted alongside this RIAA is of high applicability and quality. Whilst there is limited available evidence to inform mortality rates, those selected are considered to be sufficiently precautionary based on expert opinion. However, uncertainty remains around the effects of displacement on this species.

*In-combination*

498. For the operation and maintenance phase, in-combination disturbance and displacement mortality values have been calculated for the RIAA of the recently consented Awel y Môr OWF application (Awel y Môr Offshore Wind Farm Ltd, 2022). Given the proximity of Awel y Môr OWF, and its location adjacent to the southern section of Liverpool Bay SPA, the in-combination approach is also considered suitable for the Project in-combination assessment. As for red-throated diver, effects from Rhyl Flats and Burbo Bank OWFs have not been included in the assessment as these projects were operational prior to designation of the SPA and therefore any impacts arising from these projects would have been accounted for at the time of designation. OWF projects to the north of the SPA (including West of Duddon Sands, and Walney 1 to 4) are considered too distant (i.e. >4km) from areas where common scoter are likely to occur to have any measurable effect on this species and have therefore also been excluded from the in-combination assessment.

499. Given that Morgan and Mona Offshore Wind Projects are both greater than 4km from the SPA, no displacement is predicted due to the presence/operation of the windfarms themselves (and hence this impact pathway was not screened in to assessment within the respective draft ISAAs for these projects (Morgan Offshore Wind Limited, 2023; Mona Offshore Wind Limited, 2023)), and any potential disturbance and displacement effects would be due to vessel traffic (assuming these vessels would transit the SPA). It is assumed that embedded best practice vessel management mitigation measures similar to those set out in **Table 8.3** for operation and maintenance for the Project would also be implemented by Morgan and Mona Offshore Wind Projects, and accordingly no significant disturbance and displacement effects (either alone or in-combination) would occur. This was confirmed in the draft ISAAs for the two projects (2023), which state that the projects would include *‘The development of and adherence to an EMP* [Environmental Management Plan] *which will include measures to minimise disturbance to seabirds, in particular red-throated diver and common scoter’.*

500. **Table 8.18** sets out the predicted non-breeding mortality as a result of disturbance and displacement for relevant projects, where data are available.

*Table 8.18 Common scoter – predicted in-combination disturbance and displacement mortality from Liverpool Bay SPA*

| OWF project name | Population estimate subject to displacement | Predicted annual mortality[1] |
|---|---|---|
| Morgan and Morecambe Offshore Wind Farms: Transmission Assets | 484 | 4-48 |
| Burbo Bank Extension | 40 | 0.4-4.0 |
| Awel y Môr | 31 | 0.3-3.1 |
| Gwynt y Môr | None presented | 'no significant effect' |
| The Project | 43 | 0.4-4.3 |
| **Total** | **598** | **5-60** |

[1] Assumes displacement of 90-100% and mortality of 1-10% of displaced birds. The lower value (90% displacement and 1% mortality) is considered the most realistic scenario.

501. Based on the Liverpool Bay SPA non-breeding population of 141,801 birds and a background mortality of 0.238 (33,748 birds per annum), an increase in mortality of maximum 60 birds would increase background mortality by 0.18%.

502. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. Accordingly, no significant effects on common scoter are predicted during the operation and maintenance phase. It is noted that no data are available for the in-combination assessment for the Gwynt y Môr OWF project. However, even under an unrealistic worst-case scenario (i.e. assuming 100% displacement and 10% mortality), in order to reach a 1% increase in mortality, annual mortality would need to increase by 3,315 birds, which would be the equivalent of a population of 33,150 common scoter subject to displacement at the Gwynt y Môr OWF site. Given that the Gwynt y Môr OWF is located away from areas known to support the higher densities of this species, such a scenario is considered extremely unlikely.

503. **It is therefore concluded that there would be no adverse effect on the integrity of Liverpool Bay SPA, when considering the Project in-combination with other plans or projects.**

#### 8.4.2.3 Little gull

##### Status

504. Non-breeding little gull is a qualifying species of the SPA. Liverpool Bay supports the second largest marine aggregation for this species in the UK; the designated population comprises 319 birds (Lawson *et al.*, 2016). Population estimates were based on aerial surveys undertaken between 2004 and 2011. Data presented for little gull within the Natural England commissioned report 440 (HiDef, 2023) covered only the original SPA boundary and did not include the additional SPA area that was designated primarily for this species in 2017. Therefore, these data do not provide a meaningful update of the estimated SPA population. No site-specific conservation advice for this species has been published by Natural England.

505. Based on the SPA population, and a baseline mortality rate (all age classes) of 0.200 (derived from Horswill and Robinson, 2015), 64 birds from the SPA population would be expected to die each year under baseline conditions.

506. The population and migration patterns of little gull are poorly understood and there is no agreed Great Britain population value for this species. Lawson *et al.,* (2016) acknowledged that the population estimates for Liverpool Bay SPA (which were based on visual aerial surveys undertaken between 2001 and 2011) are likely to underestimate the actual population present in the area given the transient nature of the passage population and difficulties of distinguishing this from other small gull species. Surveys between 2015 and 2020 (HiDef, 2023) recorded a peak population estimate of 286 little gulls within the original (pre-2017) SPA boundary and as densities of this species would be expected to be significantly higher within the SPA extension area, this also indicates that the SPA population is underestimated. It is also noted that unidentified 'small gulls' (likely to be little or black-headed gulls) were not considered within the population estimates from the 2015-2020 surveys, indicating a further source of underestimation of the total population.

507. It is considered likely that the SPA population comprises a much larger pool of birds present within the Irish Sea and the wider North Atlantic which circulate through the SPA site during the winter and migratory periods. Surveys of the western Irish Sea undertaken in 2016 estimated that 1,539 little gulls were present within this area (Jessopp *et al*., 2018). It is also noted that the numbers of little gulls in UK waters appears to have recently increased (Natural England, 2012), with a 410% increase in population recorded between 1995/6 and 2020/21 (Austin *et al*., 2023). As there was no agreed biogeographic population value for this species (e.g. little gull is not included in Furness, 2015) the assessment below also includes a comparison against the European Union winter population (European Commission, 2022). This is considered reasonable given the generally northern European breeding

distribution of this species, suggesting that a significant proportion of this population is likely to pass through waters around the UK and Ireland during passage to wider wintering grounds in the North Atlantic (meaning that the birds present within the SPA at any one point in time are likely to be a small fraction of those that actually use the SPA and comprise the SPA population). However, it is also noted that the value for the estimate European Union population (maximum 10,200) may represent an underestimate of the true population, with an estimate of 100,000 little gulls recorded in the Dutch North Sea during spring migration in recent years (Fijn *et al*., 2022).

### Functional linkage and seasonal apportionment of potential effects

508. The area within the Liverpool Bay SPA boundary contains habitats that are considered to represent important marine areas for little gull during the non-breeding season. Although Lawson *et al.,* (2016) stated that the boundary of the SPA was closely aligned with the concentration of little gulls in the Irish Sea, it seems likely that, although outside the SPA, the small number of birds recorded within the windfarm site are associated with the SPA population. The numerical assessment therefore assumes that 100% of little gulls present at the windfarm site belong to the Liverpool Bay SPA population. However, it is considered that this approach may be unduly precautionary, as discussed in **Paragraph 511** below.

### Potential effects on the qualifying feature

509. The little gull qualifying feature of the Liverpool Bay SPA has been screened into the Appropriate Assessment due to the potential risk of collision during the operation and maintenance phase of the Project.

#### Operation and maintenance phase collision risk

*Project-alone*

510. Collision risk predictions for little gull (mean values with upper and lower 95% CIs) are shown in **Table 8.19**, with collision estimates presented by month. A summary of the annual outputs and the corresponding increase in the annual baseline mortality rate is presented in **Table 8.20**. Outputs are based on Option 2 of the stochastic collision risk model (sCRM), avoidance rates of 0.993 and generic flight height distributions (“Corrigendum” 2014; Johnston *et al*., 2014), in accordance with Natural England advice. Nocturnal activity was set at 25% of daytime activity. Further information is provided within **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES.

511. The mean annual collision estimate for little gull was 2.9 (0-6.8; **Table 8.20**). If the SPA population estimate is used, the predicted increase in existing mortality levels for the Liverpool SPA population due to the Project is 4.57% (0-10.67%; **Table 8.20**). As discussed in **Paragraphs 506** and **507**, this result

is precautionary as the SPA reference population is considered an underestimate. The estimated mortality increase also assumes that all birds present at the windfarm site form part of the SPA population and while it is very likely that the birds recorded during surveys will also utilise the SPA, it is considered likely that such birds would be in addition to the SPA population count. This is on the assumption that, as described above, little gulls from the SPA are likely to form part of a larger population that would circulate through the wider Irish Sea and beyond. It can be assumed, therefore, either that (1) birds present within the windfarm site do not form part of the SPA population; or (2) that the reference population used to apportion birds to the SPA is substantially larger than the counts used to inform the designation of the SPA. It is therefore considered reasonable to apportion fewer (or even zero) little gulls to the SPA population. Relating the collision estimate to the wider European winter population (which itself is likely to be underestimated) suggests an increase in annual mortality of 0.14% (0-0.33%, based on maximum European population estimate) to 0.26% (0-0.60%, based on minimum European population estimate; see to **Table 8.20**). Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation.

512. For the reasons set out above, no significant effects on little gulls from the SPA are predicted, and **it is concluded that there would be no adverse effect on the integrity of Liverpool Bay SPA.**

513. The confidence in the assessment is medium. The evidence used to define the CRM input parameters presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES alongside this RIAA is of high applicability and quality. Whilst there is uncertainty around some of the input parameters (e.g. avoidance rate), the rates selected are considered to be sufficiently precautionary based on expert opinion to provide confidence that collision rates are not underestimated. However, there is some uncertainty about the extent to which birds present within the windfarm site will be directly associated with the SPA population, which would indicate a likely overestimate of mortality apportioned to the SPA.

*Table 8.19 Little gull – predicted collisions per month, stochastic CRM (Option 2 – NAF 0.25, AR 0.993±0.0003)*

| Month | Jan | Feb | Mar | Apr | May | Jun | Jul | Aug | Sep | Oct | Nov | Dec | Total |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Mean** | 0.328 | 0.647 | 0.044 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0.047 | 1.852 | 2.918 |
| **SD** | 0.362 | 0.722 | 0.076 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0.076 | 1.97 | 2.132 |
| **CV** | 1.103 | 1.117 | 1.723 | n/a | n/a | n/a | n/a | n/a | n/a | n/a | 1.602 | 1.064 | 0.731 |
| **Median** | 0.184 | 0.242 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2.586 |
| **2.5%** | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| **25%** | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0.957 |
| **75%** | 0.641 | 1.277 | 0.089 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0.086 | 3.675 | 4.698 |
| **97.5%** | 1.049 | 2.077 | 0.251 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0.258 | 5.215 | 6.805 |

*Table 8.20 Little gull – Predicted increase in annual baseline collision mortality*

| | Liverpool Bay SPA | EU winter population (min)[1] | EU winter population (max)[1] |
|---|---|---|---|
| Population size | 319 | 5,700 | 10,200 |
| Predicted annual background mortality (20.0%) | 63.8 | 1,140 | 2,040 |
| Predicted Project mortality (no. of birds) (mean and 95% CIs) | 2.9 (0-6.8) | 2.9 (0-6.8) | 2.9 (0-6.8) |
| % Increase in predicted mortality (mean and 95% CIs) | 4.57% (0.00-10.67%) | 0.26% (0.00-0.60%) | 0.14% (0.00-0.33%) |
| [1] 2013-18 population from 'Population status and trends at the EU and Member State levels' https://nature-art12.eionet.europa.eu/article12 | | | |

*In-combination*

514. For all projects considered for the in-combination assessment (refer to Section 12.7.2 in **Chapter 12 Offshore Ornithology** of the ES), no measurable little gull collision risk estimates have been identified, reflecting the absence, or very low numbers, of this species present/expected at the respective project sites. The RIAA for Awel y Môr OWF confirmed that no little gulls were recorded during surveys of the OWF site, therefore concluded that there would be no LSE on this species. For all other OWF projects in the vicinity where a published assessment was available (including Morgan and Mona Offshore Wind Projects, Morgan and Morecambe OWFs Transmission Assets, Burbo Bank Extension, Ormonde, Walney I, II and Extension and West of Duddon Sands), no quantifiable collision risk for this species was identified.

515. **Therefore, it is concluded that there is no additional in-combination risk for the little gull population from Liverpool Bay SPA, i.e. the assessment for the Project-alone (Paragraph 511) is unchanged, and that there would be no adverse effect on the integrity of Liverpool Bay SPA, when considering the Project in-combination with other plans or projects.** This accords with the conclusions of the Round 4 offshore wind leasing HRA (NIRAS, 2021), which stated that '*for little gull the impact from the Round 4 Plan alone is considered to be negligible, and any additional impact from the Round 4 Plan alone would not make an appreciable difference to any in-combination impact'*.

### 8.4.2.4 Common tern

#### Status

516. Liverpool Bay SPA is designated for foraging areas used by common tern from the Mersey Narrows and North Wirral Foreshore SPA during the breeding season, with a population of 180 pairs (360 adult birds). This represented 1.8% of the Great Britain breeding population (Natural England, NRW and JNCC, 2016). The most recent count (2021) from Mersey Narrows and North Wirral Foreshore SPA from the Seabird Monitoring Programme (SMP) database was 208 apparently occupied nests (AON); 416 adult birds. The baseline mortality of this population is 48 breeding adult birds per year based on the published adult mortality rate of 0.117 (Horswill and Robinson, 2015).

#### Functional linkage and seasonal apportionment of potential effects

517. Although the boundary of Liverpool Bay SPA is located within the mean maximum and mean maximum +1SD foraging range of the windfarm site (18km and 27km respectively; Woodward *et al.*, 2019), the breeding colony from which the population is derived is located in the Mersey Estuary, approximately 43km from the boundary of the windfarm site. Modelling of

predicted common tern activity around the colony (Natural England, NRW and JNCC, 2016) confirmed that usage within the SPA would predominantly be limited to an area around the mouth and coastal fringes of the Mersey Estuary. Therefore, no overlap with breeding common terns from the Liverpool Bay SPA population and the windfarm site is predicted. This is reflected in the absence of breeding-season records of this species from within the windfarm site during baseline surveys. Very low densities of this species were recorded within the windfarm site during May and September only which were assumed to be birds on passage. It is considered very unlikely that these individuals would be associated with the Liverpool Bay SPA population, as these were more likely to be birds moving to or from colonies to the north (e.g. in Scotland) or from Ireland. The species is known to follow the coastline during migration (Furness, 2015) and therefore birds from the Mersey Estuary colony would be expected to disperse along the coast (e.g. towards north Wales) rather than north-westwards towards the windfarm site.

518. It is therefore concluded that common terns present at the windfarm site are very unlikely to be associated with the Liverpool Bay SPA population.

## Potential effects on the qualifying feature

### Project-alone

519. As common terns from Liverpool Bay SPA are not considered to occur at the windfarm site, no effects on this feature are predicted as a result of the Project. **It is therefore concluded that there would be no adverse effect on the integrity of Liverpool Bay SPA**.

### Potential effects in-combination with other projects

520. As no effects on common tern are predicted as a result of the Project-alone, there would be no contribution to other plans or projects in-combination. **It is therefore concluded that there would be no adverse effect on the integrity of Liverpool Bay SPA**.

521. The confidence in the assessment is high. The evidence used to inform the assessment is considered to be of high quality and applicability, and is supported by the results of the site baseline surveys.

## 8.5 Morecambe Bay and Duddon Estuary SPA and Ramsar sites

522. Morecambe Bay and Duddon Estuary SPA and Ramsar site is located approximately 26km from the windfarm site.

### 8.5.1 Description of designation

523. The SPA extends between Rossall Point in Lancashire and Drigg Dunes in Cumbria. The site includes the former Morecambe Bay SPA and Ramsar and Duddon Estuary SPA and Ramsar areas, with the SPA extension including the Ravenglass Estuary and intervening coast and the shallow offshore area off the south west Cumbria coast.

524. At over 310km$^2$, Morecambe Bay is the second largest embayment in Britain after The Wash and has four estuaries – the Wyre, Lune, Kent and Leven. It contains the largest continuous area of intertidal mudflats and sandflats in the UK.

525. The parts of the SPA away from the coast are sandy and shallow, mostly less than 15 metres deep. The site is designated for the following species:

- **Non-breeding species:**
  - Whooper swan
  - Little egret
  - Golden plover
  - Bar-tailed godwit
  - Ruff
  - Mediterranean gull
  - Pink-footed goose
  - Shelduck
  - Pintail
  - Lesser black-backed gull
  - Oystercatcher
  - Grey plover
  - Ringed plover
  - Curlew
  - Black-tailed godwit
  - Turnstone
  - Knot
  - Sanderling
  - Dunlin
  - Redshank
- **Breeding species:**
  - Little tern
  - Sandwich tern
  - Common tern
  - Lesser black-backed gull
  - Herring gull
  - Waterbird assemblage
  - Seabird assemblage

#### 8.5.1.1 Conservation objectives

526. The SPA's conservation objectives are to ensure that the integrity of the site is maintained or restored as appropriate, and ensure that the site contributes to achieving the aims of the Wild Birds Directive, by maintaining or restoring;

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

### 8.5.2 Assessment

#### 8.5.2.1 Migratory waterbird qualifying features

#### Status

527. The status of each migratory (non-breeding) waterbird qualifying feature screened into the Appropriate Assessment for this site is presented in **Table 8.21**. This consists of the site population at designation, national population in 2012 (Wright *et al*., 2012) and latest five-year peak mean WeBS count (Frost *et al*., 2021).

#### Functional linkage and seasonal apportionment of potential effects

528. All qualifying features of this designated site have been screened into the Appropriate Assessment due to the risk of potential impacts occurring during the spring and autumn migration seasons. Other than grey plover and dunlin (which were both recorded on one occasion in the first year of surveys only), the qualifying features were not recorded in the aerial survey study area during the baseline surveys undertaken at the windfarm site. However, it is recognised that the qualifying features may pass through the habitat in the windfarm site during migration periods and may have been missed by the surveys.

529. The apportioning of impacts to this designated site was calculated for each qualifying feature by dividing the number of collisions calculated at the national level by the proportion of the national population that were members of the designated site population at citation. The numbers used to define the national populations were the Great Britain populations presented in Wright *et al.,* (2012). Designated site populations were obtained from the SPA citation, or

the Ramsar site population if the SPA citation did not include a population estimate.

### Potential effects on the qualifying features

530. The qualifying features of this designated site have been screened into the Appropriate Assessment due to the potential risk of collision.

531. The magnitude of potential collision impacts have been investigated using the SOSSMAT tool (Wright *et al*., 2012).

### Potential effects on the qualifying feature from the Project-alone

#### Collision risk

532. The estimated annual collision risk for each qualifying feature from this designated site, along with the conclusion of the assessment based on this annual collision rate, is presented in **Table 8.21**. An avoidance rate of 0.980 has been assumed for all species.

533. The number of annual collisions predicted for all qualifying features is very low. It is expected that the increases to existing mortality rates for each qualifying feature due to this impact would be undetectable within the site population. Such impacts would consequently not result in any measurable effect.

534. **It is concluded that the predicted mortality of all qualifying features due to collision at the Project windfarm site would not adversely affect the integrity of the Morecambe Bay and Duddon Estuary SPA and Ramsar site**.

535. Whilst extensive information exists on the responses of waterbirds to onshore OWFs, there is substantial uncertainty regarding waterbird movements at sea. The confidence level assigned to this section of the assessment is therefore medium. However, since such low levels of collision are predicted, an adverse effect on the integrity of the site is considered highly unlikely, even in the unlikely event that impacts have been underestimated.

### Potential effects on the qualifying feature in-combination with other projects

#### Collision risk

536. The migration corridors identified by Wright *et al.,* (2012) indicate that migration activity of all qualifying features from this designated site is widespread across UK waters. Similarly low numbers of birds, and hence collisions, are therefore expected at other OWFs in UK waters. The total collision mortality of non-breeding waterbirds at all UK OWFs is still likely to be small in the context of their respective national populations, and the number of collisions associated with this designated site will be smaller still. It is

expected that the increases to existing mortality rates for each qualifying feature due to this impact would be undetectable within the site population. Such impacts would consequently not result in any measurable effect.

537. **It is concluded that predicted mortality of all qualifying features due to collision at the Project windfarm site, in-combination with other projects, would not adversely affect the integrity of the Morecambe Bay and Duddon Estuary SPA and Ramsar site**.

*Table 8.21 Information to support the Appropriate Assessment for Morecambe Bay and Duddon Estuary SPA and Ramsar site (migratory waterbird qualifying features)*

| Qualifying feature | GB population (Wright *et al.*, 2012) | SPA population (citation / standard data form) | Ramsar site population (citation) | Five-year peak mean 2015/16 - 2019/20 | Apportioning rate | Unapportioned predicted mean annual collisions (avoidance rate 0.980) | Annual collisions apportioned to SPA | Conclusion of adverse effect on site integrity |
|---|---|---|---|---|---|---|---|---|
| Little egret | 4,500 | 134 | N/A | 356 | 3.0% | 0.00 | 0.00 | No adverse effect on site integrity. Numbers of collisions are so small that effects on population would be extremely small. It would not be possible for impacts of this magnitude to have an effect at the site level given the background populations |
| Whooper swan | 11,000 | 113 | N/A | 203 | 1.0% | 0.03 | 0.00 | |
| Pink-footed goose | 360,000 | 15,648 | 2,475 | 25,341 | 4.3% | 0.01 | 0.00 | |
| Shelduck | 61,000 | 5,878 | 6,372 | 5,281 | 9.6% | 0.03 | 0.00 | |
| Pintail | 29,000 | 2,498 | 2,804 | 3,552 | 8.6% | 0.01 | 0.00 | |
| Oystercatcher | 320,000 | 55,888 | 47,572 | 43,016 | 17.5% | 0.23 | 0.04 | |
| Ringed plover | 34,000 | 1,049 | 693 | 1,321 | 3.1% | 0.02 | 0.00 | |
| Golden plover | 400,000 | 3,494 | 4,097 | 4,798 | 0.9% | 0.26 | 0.00 | |
| Grey plover | 43,000 | 1,013 | 1,813 | 1,079 | 2.4% | 0.03 | 0.00 | |
| Knot | 320,000 | 32,739 | 29,426 | 20,681 | 10.2% | 0.20 | 0.02 | |
| Sanderling | 16,000 | 849 | 2,466 | 2,351 | 5.3% | 0.01 | 0.00 | |
| Ruff | 800 | 8 | N/A | 8 | 1.0% | 0.00 | 0.00 | |
| Bar-tailed godwit | 38,000 | 3,046 | 2,611 | 2,828 | 8.0% | 0.04 | 0.00 | |
| Curlew | 140,000 | 12,209 | 13,620 | 11,469 | 8.7% | 0.10 | 0.01 | |
| Redshank | 120,000 | 11,133 | 6,336 | 9,549 | 9.3% | 0.08 | 0.01 | |
| Turnstone | 48,000 | 1,359 | 1,583 | 1,106 | 2.8% | 0.03 | 0.00 | |
| Mediterranean gull | 1,800 | 18 | N/A | 19 | 1.0% | 0.00 | 0.00 | |
| Lesser black-backed gull | 120,000 | 11,133 | N/A | 3,826 | 9.3% | 0.11 | 0.01 | |
| Black-tailed godwit | 43,000 | 2,413 | N/A | 3,726 | 5.6% | 0.03 | 0.00 | |
| Dunlin | 350,000 | 26,982 | 52,671 | 26,566 | 7.7% | 0.38 | 0.03 | |
| Waterbird assemblage | - | 266,751 | 210,668 | 211,555 | - | - | - | No adverse effect on site integrity. Based on the small number of collisions predicted for named qualifying features, no adverse effect on integrity is anticipated |

#### 8.5.2.2 Lesser black-backed gull (breeding)

##### Status

538. Breeding lesser black-backed gull is listed as a qualifying species of this SPA during the breeding and non-breeding seasons. The assessment considered effects on the breeding population (refer to **Table 5.2**).

539. The SPA breeding population has been cited as 22,000 pairs in 1996 (Furness, 2015, Stroud *et al*., 2001), and 9,720 individuals (4,860 pairs) for the period 2011-15 (Natural England, 2016). Furness (2015) detailed a breeding population of 4,987 pairs in 2012. The 2022 count from the SMP database was 530 AON (with AON essentially equivalent to breeding pairs), which would represent 1,060 breeding adults and this has been used as the reference population for the assessment. Natural England (2020a) set a target to '*Restore the size of the [SPA] breeding population to a level which is above 10,000 pairs whilst avoiding deterioration from its current level as indicated by the latest mean peak count or equivalent'*.

540. The decline in population has been partly attributed to mammalian predation at the South Walney breeding colony (North West England Gull Project, 2022), and initial results have indicated that a recently installed predator exclosure fence appeared to have been successful in beginning to reverse this decline (Cumbria Wildlife Trust, 2022). There is also evidence that some birds may have relocated to the disused Barrow Gas Terminal (North West England Gull Project, 2022) where numbers of occupied nests doubled from 329 in 2019 to 680 in 2021 (SMP database), noting that ringed birds from the South Walney colony have been recorded nesting at the gas terminal (North West England Gull Project, 2022). It is understood however that the Barrow Gas Terminal colony has subsequently declined due to mammal predation (Natural England, pers. comm.). Numbers at the nearby Ribble Estuary have also shown an increase, with JNCC (2021) reporting a 69% increase from 4,150 AON in 1998 to 7,022 AON in 2016. JNCC (2021) stated that the causes of lesser black-backed gull declines *'may be due to a decrease in the availability of domestic refuse, reduced discards from fisheries, predation, cannibalism and human disturbance'*.

541. Tracking studies undertaken as part of monitoring of the Walney Extension and Burbo Bank Extension OWFs (Clewley *et al*., 2020) showed that during the breeding season birds from the South Walney colony predominantly foraged within terrestrial areas (landfill, agricultural and urban habitats), with on average, less than 5% of tracked birds' time spent offshore. Foraging distances during the period of the study ranged from 9.3±10.2 (SD) km in 2016 to 14.2±18.4km in 2019. Tracking studies from the Walney colony reported in Langley *et al.,* (2021) showed that lesser black-backed gulls increased

foraging duration and distance in response to the closure of nearby landfill sites. Prior to closure, mean trip length was estimated at 15.0km (95% CIs 12.3-18.2km); this value increased to 23.5km (18.8-29.4km) following landfill closure. It can be concluded therefore that changes in waste management practice have also been a possible contributory factor in the decline of the population at South Walney.

542. Based on the most recent SPA population of breeding adults, and an annual breeding adult baseline mortality rate of 0.115 (Horswill and Robinson, 2015), 122 breeding adults from the SPA population would be expected to die each year.

### Functional linkage and seasonal apportionment of potential effects

543. The windfarm site is situated approximately 31km at its nearest point from South Walney, the breeding location for lesser black-backed gull within the Morecambe Bay and Duddon Estuary SPA. The mean maximum foraging range of lesser black-backed gull is 127km (±109, 1SD) (Woodward *et al*., 2019). The windfarm site is therefore within the mean maximum foraging range of lesser black-backed gulls from the Morecambe Bay and Duddon Estuary SPA. As described in **Paragraph 541**, tracking data during the breeding season indicated that the majority of birds from the South Walney colony foraged in inland areas, with less than 5% of tracked birds' time spent offshore. This information does not mean that breeding adult lesser black-backed gulls from the SPA will not be present at the windfarm site during the breeding season. However, it does suggest that birds from the SPA are likely to make little use of the windfarm site and spend little time there.

544. The windfarm site is located within the mean maximum foraging range of two additional UK SPAs where breeding lesser black-backed gull is a qualifying feature; Ribble and Alt Estuaries SPA (and Ramsar) and Bowland Fells SPA. One further SPA is located within mean maximum +1SD; Ailsa Craig SPA. Ribble and Alt Estuaries SPA is located approximately 27km from the windfarm site, with the most recent count (2021) from the SMP database of 4,489 AON (refer to **Paragraph 625**). Bowland Fells SPA is an inland site, located approximately 53km from the windfarm site with the most recent (2018) count of 14,627 AON (SMP database). Ailsa Craig SPA is located approximately 177km from the windfarm site, with the most recent (2019) count of 189 AON (SMP database). One further transboundary site, Lambay Island SPA, is also located within mean maximum +1SD, approximately 156km from the site, with the most recent (2010) count 476 AON (SMP database).

545. In addition to SPA colonies, the SMP database identified a total of 168 lesser black-backed gull colonies within 236km (i.e. the mean maximum plus 1SD foraging range) of the windfarm site. Considering all colonies (SPA and non-

SPA), there is an estimated total of 72,320 individuals (based on the most recently available post-1999 counts) with potential breeding season connectivity to the windfarm site (as determined by the mean maximum plus 1SD foraging range). It is therefore likely that birds present at the windfarm site during the breeding season may originate from a number of different colonies within the mean maximum +1SD foraging range for this species. Tracking studies on birds from Bowland Fells SPA (Clewley *et al.,* 2017) have found that birds from these colonies utilised terrestrial habitats almost exclusively, with a small proportion of trips to near-shore areas. It is therefore considered unlikely that birds from this SPA will occur at the windfarm site during the breeding season.

546. The NatureScot apportioning tool (NatureScot, 2018) has been used to estimate the proportion of lesser black-backed gulls from each of the relevant SPAs present at the windfarm site during the breeding season (see **Appendix 12.1** of the ES for apportioning outputs). As there is uncertainty regarding the extent to which inland colonies may contribute to the population present in coastal/offshore areas (i.e. at the windfarm site), two estimates for breeding season apportioning have been calculated. The first has used all breeding colonies (both coastal and inland, including Bowland Fells SPA) within mean maximum +1SD foraging range of the windfarm site. The second approach has included only coastal sites (defined as being located within 5km of the coast), which excludes Bowland Fells SPA from the calculation (i.e. assumes that no birds from Bowland Fells SPA occur at the windfarm site during the breeding season).

547. When all lesser black-backed gull colonies are included, it is estimated that 5.35% of birds present at the windfarm site are apportioned to Morecambe Bay and Duddon Estuary SPA during the breeding season. If only coastal sites are included, it is estimated that 9.50% are apportioned to the SPA.

548. In addition, some of the lesser black-backed gulls recorded at the windfarm site during the breeding season will be sub-adult birds. Based on review of raw survey data, 286 lesser black-backed gulls were recorded during the two-year baseline digital aerial surveys. Of these, 177 birds were able to be assigned to an age class, and of these, 126 birds (71.2% of those assigned to an age class) were classified as adults. It is therefore assumed, on a precautionary basis, that 71.2% of lesser black-backed gulls apportioned to the SPA during the full breeding season are breeding adult birds from Morecambe Bay and Duddon Estuary SPA. This estimate is likely to include additional precaution for two reasons. Firstly, lesser black-backed gulls display plumage that is effectively indistinguishable from adult birds by their third winter (Cramp and Simmons, 1983), but typically start breeding in their fifth year (Horswill and Robinson, 2015). Therefore, the proportion of adult (breeding age) birds may be overestimated when based solely on plumage

characteristics. Secondly, it is likely that any adult lesser black-backed present will include a proportion of sabbatical (non-breeding) individuals, so the proportion of breeding adult birds is likely to be further overestimated.

549. In addition to the potential for connectivity during the breeding season, there is also potential for the breeding lesser black-backed gull qualifying feature of the Morecambe Bay and Duddon Estuary SPA to have connectivity with the windfarm site during the non-breeding periods. Thus, during the pre- and post-breeding periods, breeding lesser black-backed gulls from the Morecambe Bay and Duddon Estuary SPA migrate through UK waters, whilst some birds remain in the UK during the winter. The relevant reference population is considered to be the UK Western Waters BDMPS, within which the birds from different breeding colony source populations and of different age classes are assumed to be distributed evenly throughout. This consists of 163,304 individuals during autumn migration (September to October), 41,159 individuals during winter (November to February) and 163,304 individuals during spring migration (March) (Furness, 2015).

550. Furness (2015) estimated that 50% of the Morecambe Bay and Duddon Estuary SPA breeding adults (9,974) are present within the UK Western Waters BDMPS during the autumn and spring migration periods, representing 4,987 birds. During the winter period 20% of the SPA population was estimated to be present, representing 1,995 birds. This is equivalent to 3.05% of the BDMPS population for the autumn and spring periods, and 4.85% during winter. During autumn migration, winter, and spring migration, 3.05%, 4.85%, and 3.05% of impacts are considered to affect birds from the SPA respectively (Furness, 2015).

## Potential effects on the qualifying feature

551. The lesser black-backed gull qualifying feature of the Morecambe Bay and Duddon Estuary SPA has been screened into the assessment due to the potential risk of collision during the operation and maintenance phase.

### Operation and maintenance phase collision risk

#### *Project-alone*

552. Information on collision risk for breeding adult lesser black-backed gulls belonging to the Morecambe Bay and Duddon Estuary SPA population is presented in **Table 8.22**. Collision estimates, calculated using Option 2 of the sCRM (McGregor *et al.*, 2018), are presented by biological season. A summary of the annual outputs and the corresponding increase in the annual baseline mortality rate is also presented. Parameters used in the sCRM, together with the avoidance rate applied, were agreed with Natural England during the ETG process and are described in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES.

553. If all lesser black-backed gull colonies are included in the breeding season apportioning, and based on the mean collision rates, the annual total of breeding adult lesser black backed gulls from the Morecambe Bay and Duddon Estuary SPA at risk of collision due to the Project is 0.13. This would increase the existing mortality of the SPA breeding population by 0.10%. If only coastal sites are used for apportioning, annual mortality would be 0.19, representing an increase of 0.15% to the existing baseline annual mortality.

*Table 8.22 Predicted seasonal and annual collision mortality (Stochastic model Option 2, avoidance rate 0.994 (±0.0004)) for breeding adult lesser black-backed gulls at the windfarm site, apportioned to Morecambe Bay and Duddon Estuary SPA, with corresponding increases to baseline mortality of the population*

| | Breeding Season (all sites apportioned) | Breeding Season (only coastal sites apportioned) | Autumn Migration | Non-breeding/winter | Spring Migration | Annual (all sites apportioned) | Annual (only coastal sites apportioned) |
|---|---|---|---|---|---|---|---|
| Period | Apr-Aug | Apr-Aug | Sep-Oct | Nov-Feb | Mar | Jan-Dec | Jan-Dec |
| Total collisions[1] (mean and 95% CIs) | 1.44 (0.00-4.53) | 1.44 (0.00-4.53) | 1.25 (0.00-5.63) | 0.15 (0.00-0.80) | 0.15 (0.00-0.94) | **2.98 (0.00-11.90)** | **2.98 (0.00-11.90)** |
| % apportioned to the SPA | 5.35% | 9.50% | 3.05% | 4.85% | 3.05% | **-** | **-** |
| Total SPA collisions (mean and 95% CIs) | 0.08 (0.00-0.24) | 0.14 (0.00-0.43) | 0.04 (0.00-0.17) | 0.01 (0.00-0.04) | 0.00 (0.00-0.03) | **0.13 (0.00-0.48)** | **0.19 (0.00-0.67)** |
| Mortality increase[2] (mean and 95% CIs) | 0.06% (0.00-0.20%) | 0.11% (0.00-0.35%) | 0.03% (0.00-0.14%) | 0.01% (0.00-0.03%) | 0.00% (0.00-0.02%) | **0.10% (0.00-0.40%)** | **0.15% (0.00-0.55%)** |

[1] Breeding season collision values reduced to 71.9% of modelled value to reflect proportion of adult birds recorded during site surveys.
[2] Assuming predicted annual SPA mortality of 122 birds (1,060 x 0.115)

554. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that no detectable changes in mortality rates would occur in this population from the mean monthly collision estimates for the Project. This applies for both apportioning approaches considered, and for the upper 95% CI estimate.

555. **It is concluded that predicted lesser black-backed gull mortality due to collision at the Project windfarm site would not adversely affect the integrity of the Morecambe Bay and Duddon Estuary SPA and Ramsar.**

556. The confidence in the assessment is high. The evidence used to define the CRM input parameters presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES is of high applicability and quality. Whilst there is uncertainty around some of the input parameters (e.g. avoidance rate), the rates selected are considered to be sufficiently precautionary based on expert opinion to provide confidence that collision rates are not underestimated.

*In-combination*

557. On the basis of the conclusions of the Project alone assessment (i.e. very low predicted lesser black-backed gull collision mortality, equating to a small fraction of a bird), and for the reasons set out below, **there would be no measurable contribution of the Project to in-combination effects**. **Accordingly, no in-combination assessment is required for this feature**. The conclusion of the Project-alone assessment is therefore unchanged, **i.e. that predicted lesser black-backed gull mortality due to collision at the Project windfarm site would not adversely affect the integrity of the Morecambe Bay and Duddon Estuary SPA and Ramsar.**

558. Notwithstanding this conclusion, **Paragraphs 561** to **566** below present an estimate of in-combination mortality and a Population Viability Analysis (PVA), to provide context to the Project-alone assessment. This information is presented without prejudice to the conclusion above.

559. The reasons that the Project would not contribute to the in-combination effect are as follows:

- Evidence from tracking studies (e.g. in Clewley *et al.*, 2020) suggests that birds from the SPA are unlikely to frequently occur at the windfarm site (or other project sites), and therefore the apportioned project-alone collision values are likely to be a significant overestimate.
- Even if birds from the SPA frequently occur at the windfarm site, the Project contribution to the in-combination total (based on a 'worst-case', assuming that only birds from coastal sites contribute to apportioning) is very small; significantly less than one (0.19) bird per annum and

representing less than 2% of all predicted collisions apportioned to the SPA. Even at this precautionary rate (precautionary for the reasons set out in the Project-alone assessment above), it would be expected that less than one bird from the SPA would die for every five years that the Project was operational.

- There is very strong evidence to suggest that the key causes of decline in the SPA population have been driven by other factors (including predation and changes in land-use), and the very small level of additional mortality attributable to the Project is very likely be inconsequential in this wider context.
- In the absence of the Project, the predicted lesser-black backed gull mortality apportioned to the SPA (10.05 birds; refer to **Table 8.23** below) would increase background mortality by 8.24%; the contribution of the Project (resulting in a total increase in background mortality of 8.40%) is considered inconsequential to the in-combination effect.
- PVA outputs for the in-combination mortality (refer to **Paragraphs 565** to **566** and **Table 8.24** below) confirm that the contribution of the Project would make no measurable difference to the annual growth rate or reduction in population size, taking into account the uncertainties around the PVA outputs (particularly over the 35-year period assessed within the PVA). The reduction in annual growth rate for in-combination mortality would be 1.08% when the Project is excluded, increasing to 1.10% including the Project (**Table 8.24**). The reduction in population size at the end of the 35-year period would be 35.2% in the absence of the Project, or 35.8% including the Project. This difference (i.e. 0.6%) is below a threshold that is likely to be detectable at a population level, and indistinguishable from natural variation.

560. It is noted that during Examination for the Sheringham and Dudgeon Extension Projects (SEP and DEP), Natural England (2023a) concluded that for comparable lesser black-backed gull mortality levels apportioned to Alde-Ore Estuary SPA (mortality of 0.24 birds per annum, equivalent to 0.06% increase in background mortality; Equinor, 2023) there would be *'no measurable contribution from SEP and DEP to in-combination effects'*.

561. The in-combination estimation for lesser black-backed gull mortality from Morecambe Bay and Duddon Estuary SPA due to collision risk has been undertaken in accordance with the approach presented in **Section 8.1** and **Appendix 12.1** of the ES. **Table 8.23** sets out the predicted annual mortality for relevant projects (refer to **Chapter 12 Offshore Ornithology** of the ES). This information is presented without prejudice to the conclusion presented above, i.e. that the Project would make no meaningful contribution to the in-combination effect. The contribution of the Project used for the in-combination

estimation is based on the 'worst-case' set out above, i.e. assuming that only birds from coastal sites are apportioned to the SPA (which predicts a mortality of 0.19 collisions per annum).

*Table 8.23 Lesser black-backed gull – predicted in-combination collision mortality from Morecambe Bay and Duddon Estuary SPA*

| OWF project name | Predicted annual mortality |
|---|---|
| Burbo Bank Extension | 1.05 |
| Ormonde | 1.15 |
| Walney 1 + 2 | 2.97 |
| Walney 3 + 4 | 1.52 |
| West of Duddon Sands | 2.72 |
| Gwynt y Môr | 0.12 |
| Rhyl Flats | 0.02 |
| Robin Rigg | 'Low/negligible' |
| Awel y Môr | 0.00 |
| Erebus | 0.16 |
| Twin Hub | 0.16 |
| Morgan Generation Assets | 0.04 |
| Mona | 0.07 |
| Burbo Bank | 0.05 |
| West of Orkney | 0.00 |
| White Cross | 0.01 |
| **Sub-total excluding the Project** | **10.05** |
| **The Project (worst-case)** | **0.19** |
| **Total** | **10.24** |

562. In accordance with discussions with Natural England through the ETG process, consideration has also been given to potential in-combination effects with lethal control licensing for lesser black-backed gulls. Natural England (2020b) has undertaken HRA of the licensing process, which estimated a total of 3,354 lesser black-backed gulls were killed in England under licence in 2019. The Natural England HRA did not include quantification of gull mortality apportioned to individual SPAs, but concluded no adverse effect on integrity for all SPAs when considered in-combination with other plans and projects (including offshore wind development). On this basis, it is concluded that

licensed lethal control of lesser black-backed gulls would not affect the presented in-combination estimation.

563. Based on the Morecambe Bay and Duddon Estuary SPA breeding population of 1,060 adult birds and a background mortality of 0.115 (122 birds per annum), an increase in mortality of 10.24 birds would increase background mortality by 8.40%.

564. It is noted that for one historic project (Robin Rigg) no collision data for lesser black-backed gull are available. This project assessed the significance of effect on this species as ‘low/negligible’, and it is also the case that the Robin Rigg windfarm is a small development, which (at c.80km from the South Walney Colony) is highly unlikely to have effects on the population. Therefore, the absence of data for this project, it is considered unlikely that this would significantly alter the estimation presented above.

565. As background mortality, based on the estimates presented above, would exceed 1%, a population viability analysis (PVA) for the in-combination estimation has been undertaken. The results of the PVA are summarised in **Table 8.24**, with full details presented in **Appendix 12.1** of the ES. The PVA indicates that there would be a 1.10% reduction in annual population growth rate, and a net 35.8% reduction in population size at the end of the 35-year operational period, compared to the unimpacted scenario. In the absence of the Project, these values would be 1.08% and 35.2% respectively, which, as set out above, are considered well within the bounds of natural variation and therefore indistinguishable from the all-projects scenario.

566. It is noted that other external factors (in particular the ability to control mammalian predation at the SPA colonies) are likely to be the key drivers in determining the future trajectory of the SPA population. The population at the South Walney colony has increased significantly in only a short period (from 106 AON in 2021 to 862 AON in 2023) following installation of a predator-proof fence around part of the colony area. It is expected that ongoing site management (i.e. maintenance of the fence and ongoing implementation of other predator control measures) would be sustained in future years, and that this would contribute to the achievement of the ‘restore’ objective for the lesser black-backed gull population. In this context, it is considered that the predicted in-combination mortality from windfarm projects would have limited effect.

567. As the Project would make no measurable contribution to the in-combination mortality, **it has been concluded that there would be no adverse effect on integrity to Morecambe Bay and Duddon Estuary SPA. Therefore, no conclusion in respect of in-combination effects for the Project is required.**

*Table 8.24 In-combination population viability analysis outputs for lesser black-backed gull at Morecambe and Duddon Estuary SPA*

| Scenario | Predicted mortality | Growth rate | Median CPGR[1] | Median CPS[2] | Reduction in annual growth rate | Reduction in population size |
|---|---|---|---|---|---|---|
| Baseline (unimpacted) | 0 | 1.0080 | 1.0000 | 1.0000 | n/a | n/a |
| In-combination collision mortality including the Project | 10.24 | 0.9968 | 0.9890 | 0.6422 | 1.10% | 35.8% |
| In-combination collision mortality excluding the Project | 10.05 | 0.9971 | 0.9892 | 0.6477 | 1.08% | 35.2% |

[1] CPGR is the counterfactual of annual population growth rate, calculated as the median of the ratio of the annual growth rate of the impacted to un-impacted (or baseline) population, expressed as a proportion.

[2] CPS is the counterfactual of population size, calculated as the median of the ratio of the end-point size of the impacted to un-impacted population size, expressed as a proportion. In this case, the endpoint population size is predicted on the basis of a 35-year operational period.

### 8.5.2.3 Herring gull

#### Status

568. Breeding herring gull is listed as a qualifying species of this SPA.

569. The SPA population was cited as 11,000 pairs in 1996 (Furness, 2015). Furness (2015) proposed a breeding population of 1,734 pairs (3,468 adult birds) in 2012. The most recent count (2022) from the SMP database was 598 AON, or 1,196 breeding adults. Natural England (2020a) set a target to '*Restore the size of the [SPA] breeding population to a level which is above 10,000 pairs whilst avoiding deterioration from its current level as indicated by the latest mean peak count or equivalent*'.

570. There is little published evidence regarding the decline in herring gulls at Morecambe Bay and Duddon Estuary SPA, but this species shares the same breeding site at South Walney as lesser black-backed gull. Given the similar ecology of the two species, it is likely that the same factors (i.e. as set out in **Paragraphs 540** and **541**) will be responsible for the decline of this species.

It is noted that a recent increase in breeding population (Cumbria Wildlife Trust, 2022) has been reported for both herring and lesser black-backed gulls.

571. Based on the most recent SPA population of breeding adults, and an annual breeding adult baseline mortality rate of 0.166 (Horswill and Robinson, 2015), 199 breeding adults from the SPA population would be expected to die each year.

### Functional linkage and seasonal apportionment of potential effects

572. The windfarm site is situated approximately 31km at its nearest point from South Walney, the breeding location for herring gull within the Morecambe Bay and Duddon Estuary SPA. The mean maximum foraging range of herring gull is 59km (±27km) (Woodward *et al*., 2019). The windfarm site is therefore within the mean maximum foraging range of herring gulls from the SPA. Tracking studies undertaken at the South Walney colony (Thaxter *et al.*, 2017) found that breeding birds remained local to the colony area, utilising nearshore/intertidal and terrestrial/urban habitats, with some use of offshore areas recorded only outside of the breeding season. Tracking studies by Clewley *et al.,* (2020) found that tracked birds favoured nearby mussel beds during the breeding season, and that the majority of tracked birds selected intertidal and near-shore habitats over terrestrial and offshore. Foraging distances recorded by Clewley *et al.,* (2020) during the breeding season were 5.4±9.6km in 2014 and 5.4±6.7km in 2015; i.e. substantially below the values in Woodward *et al*., (2019). Therefore, it is considered very unlikely that a significant number of birds from the Morecambe Bay and Duddon Estuary SPA breeding population will be present at the windfarm site during the breeding season.

573. The windfarm site is not located within the mean maximum foraging range (+1SD) of other SPAs where herring gull is a qualifying feature. The SMP database identified a total of 52 herring gull colonies within mean maximum +1SD of the windfarm site, with a total count of 14,300 adult birds (based on the most recently available post-1999 count and including Morecambe Bay and Duddon Estuary SPA). The largest of these is at the Ribble Estuary where 855 AON (1,710 adults) were recorded in 2021, and a large number of urban sites. It is therefore likely that any birds present at the windfarm site during the breeding season are more likely to originate from the larger colony at the Ribble Estuary (which is a similar distance from the windfarm site), but may originate from a number of different colonies within the foraging range for this species. On this basis, it is likely that few birds present at the site during the breeding season would originate from the SPA population.

574. The NatureScot apportioning tool (NatureScot, 2018) has been used to estimate the proportion of herring gulls from the SPA present at the windfarm site during the breeding season. The apportioning to SPA and non-SPA

colonies is set out in **Appendix 12.1** of the ES. 15.37% of impacts at the windfarm site during the breeding season are apportioned to Morecambe Bay and Duddon Estuary SPA.

575. During the pre and post breeding periods, breeding herring gulls from the Morecambe Bay and Duddon Estuary SPA will undergo local dispersal, generally favouring coastal and urban habitats (Furness, 2015, Cramp and Simmons, 1983). Therefore, it is likely that birds present outside of the breeding period will originate from a number of breeding sites in the vicinity. The relevant reference population is considered to be the UK Western Waters BDMPS. This consists of 173,299 individuals during the non-breeding season (Furness, 2015).

576. Furness (2015) estimated that 80% of the Morecambe Bay and Duddon Estuary SPA breeding adults were present within the UK Western Waters BDMPS during the non-breeding period, which accounted for 2,774 birds. Estimates of the proportion of herring gulls present at the windfarm site which originate from the Morecambe Bay and Duddon Estuary SPA during the non-breeding season (and therefore the proportion of predicted mortalities from the SPA population) are based on the SPA population present (i.e. 2,774 adults) as a proportion of the UK Western Waters BDMPS during the non-breeding season. During this period 1.60% of impacts are considered to affect birds from the SPA (Furness, 2015).

### Potential effects on the qualifying feature

577. The herring gull qualifying feature of the Morecambe Bay and Duddon Estuary SPA has been screened into the assessment due to the potential risk of collision.

#### Operation and maintenance phase collision risk

*Project-alone*

578. Information for collision risk on breeding adult herring gulls belonging to the Morecambe Bay and Duddon Estuary SPA population is presented in **Table 8.13**. Collision estimates, calculated using the sCRM, are presented by biological season. A summary of the annual outputs and the corresponding increase in the annual baseline mortality rate is also presented. Parameters used in the sCRM were agreed with Natural England during the ETG process and are described in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES.

579. Based on the mean collision rates, the annual total of breeding adult herring gulls from the Morecambe Bay and Duddon Estuary SPA at risk of collision as a result of the Project is 0.17. This would result in no detectable increase (0.09%) in the existing mortality of the SPA breeding population.

*Table 8.25 Predicted seasonal and annual collision mortality (Stochastic model Option 2, avoidance rate 0.994 (±0.0004)) for breeding adult herring gulls at the windfarm site, apportioned to Morecambe Bay and Duddon Estuary SPA, with corresponding increases to baseline mortality of the population*

| | Breeding Season | Autumn Migration | Non-breeding/winter | Spring Migration | Annual |
|---|---|---|---|---|---|
| Period | Mar-Aug | - | Nov-Feb | - | Jan-Dec |
| Total collisions (mean and 95% CIs) | 0.85 (0.00-3.71) | - | 2.38 (0.00-9.7) | - | 3.23 (0.13-13.41) |
| % apportioned to the SPA | 15.37% | - | 1.60% | - | - |
| Total SPA collisions (mean and 95% CIs) | 0.13 (0.00-0.57) | - | 0.04 (0.00-0.16) | - | 0.17 (0.00-0.72) |
| **Mortality increase**[1] (**mean** and 95% CIs) | **0.07% (0.00-0.29%)** | - | **0.02%** (0.00-0.08%) | **-** | **0.09%** (0.00-0.37%) |
| [1] Breeding season collision values reduced to 94.1% of modelled value to reflect proportion of adult birds recorded during surveys.<br>[1] Assuming predicted annual SPA adult mortality of 199 birds (1,196 x 0.166) | | | | | |

580. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that no detectable changes in mortality rates would occur on this population for the mean monthly collision estimates for the Project, or for the upper 95% CI collision estimate.

581. **It is concluded that predicted herring gull mortality due to collision at the windfarm site would not adversely affect the integrity of the Morecambe Bay and Duddon Estuary SPA and Ramsar.**

582. The confidence in the assessment is high. The evidence used to define the CRM input parameters presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES is of high applicability and quality. Whilst there is uncertainty around some of the input parameters (e.g. avoidance rate), the rates selected are considered to be sufficiently precautionary based on expert opinion to provide confidence that collision rates are not underestimated.

*In-combination*

583. As no measurable effects on herring gull are predicted as a result of the Project-alone, there would be no material contribution to the effects of other plans or projects in-combination. **It is therefore concluded that is no potential for the Project-alone or in-combination to have an adverse effect on the integrity of the Morecambe Bay and Duddon Estuary SPA and Ramsar**.

#### 8.5.2.4 Sandwich tern

##### Status

584. Breeding Sandwich tern is listed as a qualifying species of this SPA.

585. The SPA population has been cited as 1,608 individuals (804 pairs) covering the period 1988-1992, which represented 5.7% of the Great Britain population. However, the citation notes that the peak mean population for the period 2010-14 was 47 pairs. Furness (2015) gave the breeding population as two pairs or four breeding adult birds. Three breeding locations have been identified within the SPA, but for two of these sites (Foulney Island and South Walney) the SMP database had zero counts in recent years. For the third site (Hodbarrow) the most recent count in the SMP database was 805 AON in 2019, the equivalent of 1,610 breeding adults. Natural England (2020a) set a target to '*Restore the size of the [SPA] breeding population to a level which is above 804 pairs whilst avoiding deterioration from its current level as indicated by the latest mean peak count or equivalent'*.

586. Based on the most recent SPA population of breeding adults, and an annual breeding adult baseline mortality rate of 0.102 (Horswill and Robinson, 2015), 164 breeding adults from the SPA population would be expected to die each year.

## Functional linkage and seasonal apportionment of potential effects

587. The Hodbarrow colony is located approximately 50km from the windfarm site at its closest point. The mean maximum foraging range of Sandwich tern is 34.3km (±23.2km) (Woodward *et al.*, 2019). The windfarm site is therefore located outside of the typical foraging range of this species. Modelling of predicted use of waters around the Duddon Estuary by Sandwich terns also indicated that birds will be predominantly restricted to nearshore and coastal areas around the colony (Wilson *et al.*, 2014). It is therefore considered unlikely that this species would occur at the windfarm site during the breeding season; this is confirmed by the absence of records of this species from baseline aerial surveys of the windfarm site. Accordingly, no breeding season impacts are apportioned to this SPA.

588. Outside the breeding season, breeding Sandwich terns are assumed to range widely and to mix with birds of all ages from breeding colonies in the UK and further afield. The relevant background population is considered to be the UK Western waters BDMPS, consisting of 10,761 individuals during autumn migration (July to September), and spring migration (March to May) (Furness, 2015).

589. Estimates of the proportion of Sandwich terns present at the windfarm site during the autumn and spring migration seasons which originate from the Morecambe Bay and Duddon Estuary SPA site are based on the SPA population as a proportion of the UK Western waters BDMPS (Furness, 2015). During both autumn and spring migration seasons, breeding adult Sandwich terns from Morecambe Bay and Duddon Estuary SPA make up 0.04% of the total BDMPS population. The same percentage of impacts are therefore attributable to birds from this SPA during these times of year.

## Potential effects on the qualifying feature from the Project-alone

590. The Sandwich tern qualifying feature of the Morecambe Bay and Duddon Estuary SPA has been screened into the assessment due to the potential risk of collision.

### Operation and maintenance phase collision risk

*Project-alone*

591. Information for collision risk on breeding adult Sandwich terns belonging to the Morecambe Bay and Duddon Estuary SPA population is presented in **Table 8.13**. Collision estimates, calculated using the sCRM, are presented by biological season. A summary of the annual outputs and the corresponding increase in the annual baseline mortality rate is also presented. Parameters used in the sCRM were agreed with Natural England during the ETG process

and are described in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES.

592. Based on the mean collision rates, the annual total of breeding adult Sandwich terns from the Morecambe Bay and Duddon Estuary SPA at risk of collision as a result of the Project is 0.00. This would result in no detectable increase (0.00%) in the existing mortality of the SPA breeding population.

*Table 8.26 Predicted seasonal and annual collision mortality (Stochastic model Option 2, avoidance rate 0.991 (±0.0004)) for breeding adult Sandwich terns at the windfarm site, apportioned to Morecambe Bay and Duddon Estuary SPA, with corresponding increases to baseline mortality of the population*

| | **Breeding Season** | **Autumn Migration** | **Non-breeding/winter** | **Spring Migration** | **Annual** |
|---|---|---|---|---|---|
| **Period** | **Apr-Aug** | **Sep** | **Oct-Feb** | **Mar** | **Jan-Dec** |
| Total collisions (mean and 95% CIs) | 0.00 (0.00-0.00) | 0.33 (0.02-1.07) | 0.00 (0.00-0.00) | 0.00 (0.00-0.00) | 0.33 (0.02-1.07) |
| % apportioned to the SPA | 0.0% | 0.04% | 0.0% | 0.04% | - |
| Total SPA collisions (mean and 95% CIs) | 0.00 (0.00-0.00) | 0.00 (0.00-0.01) | 0.00 (0.00-0.00) | 0.00 (0.00-0.00) | 0.00 (0.00-0.00) |
| **Mortality increase**[1] (**mean** and 95% CIs) | **0.00%** (0.00-0.00%) | **0.00%** (0.00-0.06%) | **0.00%** (0.00-0.00%) | **0.00%** (0.00-0.00%) | **0.00%** (0.00-0.00%) |

[1] Assuming predicted annual SPA adult mortality of 164 birds (1,610 x 0.102)

593. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that no detectable changes in mortality rates would occur on this population for the mean monthly collision estimates for the Project, or for the upper 95% CI collision estimate.

594. **It is concluded that predicted Sandwich tern mortality due to collision at the windfarm site would not adversely affect the integrity of the Morecambe Bay and Duddon Estuary SPA.**

595. The confidence in the assessment is high. The evidence used to define the CRM input parameters presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES is of high applicability and quality. Whilst there is uncertainty around some of the input parameters (e.g. avoidance rate), the rates selected are considered to be sufficiently precautionary based on expert opinion to provide confidence that collision rates are not underestimated.

*In-combination*

596. As no measurable effects on Sandwich tern are predicted as a result of the Project-alone, there would be no material contribution to the effects of other plans or projects in-combination. **It is therefore concluded that is no potential for the Project-alone or in-combination to have an adverse effect on the integrity of the Morecambe Bay and Duddon Estuary SPA**.

#### 8.5.2.5 Common tern

##### Status

597. Breeding common tern is listed as a qualifying species of Morecambe Bay and Duddon Estuary SPA. The SPA population was cited as 570 individuals (285 pairs) in 1991, which represented 2% of the Great Britain population (Natural England, 2016). The citation notes that the peak mean population for the period 2010 - 2014 was 47 pairs (Natural England, 2016) representing a decline of 83.5% since 1991. Common tern has been recorded breeding in the SPA at Hodbarrow and Foulney Island (Natural England, 2020a); the SMP database indicated that there were 48 AON at Hodbarrow in 2019 (most recent available data), the equivalent of 96 breeding adults, and none at Foulney Island (JNCC, 2023a).

598. Based on the most recent SPA population of breeding adults, and an annual adult baseline mortality rate of 0.117 (1 – 0.883; Horswill and Robinson, 2015), 11 breeding adult common terns from the SPA population would be expected to die each year.

##### Functional linkage and seasonal apportionment of potential effects

599. The mean maximum breeding season foraging range of common tern is 18.0km (±8.9km) and the maximum foraging range is 30km (Woodward *et al.,*

2019). The Hodbarrow colony is located approximately 50km from the Project at its closest point, therefore the windfarm site is beyond the maximum foraging range of breeding common terns from the SPA. This is reflected in the absence of breeding-season records of this species from the windfarm site during baseline surveys. Very low densities of this species were recorded within the windfarm site during May and September only which are assumed to be birds on passage. Accordingly, no breeding season impacts are apportioned to this SPA.

600. Outside the breeding season breeding common terns are assumed to range widely and to mix with birds of all ages from breeding colonies in the UK and further afield. The relevant background population is considered to be the UK Western waters BDMPS, consisting of 64,659 individuals during autumn migration (late July to early September), and spring migration (April to May) (Furness, 2015).

601. Estimates of the proportion of common terns present at the windfarm site during the autumn and spring migration seasons which originate from the Morecambe Bay and Duddon Estuary SPA site are based on the SPA population as a proportion of the UK Western waters BDMPS (Furness, 2015). Furness (2015) did not define a SPA population and therefore the closest pre-publication value (i.e. 47 pairs or 94 individuals) has been used. During both autumn and spring migration seasons, breeding adult common terns from Morecambe Bay and Duddon Estuary SPA make up 0.15% of the total BDMPS population. The same percentage of impacts are therefore attributable to birds from this SPA during these times of year.

### Potential effects on the qualifying feature from the Project-alone

602. The common tern qualifying feature of the Morecambe Bay and Duddon Estuary SPA has been screened into the assessment due to the potential risk of collision.

#### Operation and maintenance phase collision risk

*Project-alone*

603. Information for collision risk on breeding adult common terns belonging to the Morecambe Bay and Duddon Estuary SPA population is presented in **Table 8.27**. Collision estimates, calculated using the sCRM, are presented by biological season. A summary of the annual outputs and the corresponding increase in the annual baseline mortality rate is also presented. Parameters used in the sCRM were agreed with Natural England during the ETG process and are described in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES.

604. Based on the mean collision rates, the annual total of breeding adult common terns from the Morecambe Bay and Duddon Estuary SPA at risk of collision as a result of the Project is 0.00. This would result in no detectable increase (<0.01%) in the existing mortality of the SPA breeding population.

*Table 8.27 Predicted seasonal and annual collision mortality (Stochastic model Option 2, avoidance rate 0.991 (±0.0004)) for breeding adult common terns at the windfarm site, apportioned to Morecambe Bay and Duddon Estuary SPA, with corresponding increases to baseline mortality of the population*

| | Breeding Season | Autumn Migration | Non-breeding/winter | Spring Migration | Annual |
|---|---|---|---|---|---|
| Period | Jun-Jul | Aug-Sep | Oct-Mar | Apr-May | Jan-Dec |
| Total collisions (mean and 95% CIs) | 0.00 (0.00-0.00) | 0.14 (0.01-0.37) | 0.00 (0.00-0.00) | 0.08 (0.00-0.22) | 0.22 (0.01-0.60) |
| % apportioned to the SPA | 0.0% | 0.15% | 0.0% | 0.15% | - |
| Total SPA collisions (mean and 95% CIs) | 0.00 (0.00-0.00) | 0.00 (0.00-0.00) | 0.00 (0.00-0.00) | 0.00 (0.00-0.00) | 0.00 (0.00-0.00) |
| **Mortality increase**[2] (**mean** and 95% CIs) | **0.00%** (0.00-0.00%) | **0.00%** (0.00-0.00%) | **0.00%** (0.00-0.00%) | **0.00%** (0.00-0.00%) | **0.00%** (0.00-0.01%) |

[1] May overlaps breeding and spring migration period, has been included in migration period as birds present at the windfarm site are considered most likely to be migrants.

[2] Assuming predicted annual SPA adult mortality of 11 birds (96 x 0.117)

605. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that no detectable changes in mortality rates would occur on this population for the mean monthly collision estimates for the Project, or for the upper 95% CI collision estimate.

606. **It is concluded that predicted common tern mortality due to collision at the windfarm site would not adversely affect the integrity of the Morecambe Bay and Duddon Estuary SPA and Ramsar.**

607. The confidence in the assessment is high. The evidence used to define the CRM input parameters presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES is of high applicability and quality. Whilst there is uncertainty around some of the input parameters (e.g. avoidance rate), the rates selected are considered to be sufficiently precautionary based on expert opinion to provide confidence that collision rates are not underestimated.

*In-combination*

608. As no measurable effects on common tern are predicted as a result of the Project-alone, there would be no material contribution to the effects of other plans or projects in-combination. **It is therefore concluded that is no potential for the Project-alone or in-combination to have an adverse effect on the integrity of the Morecambe Bay and Duddon Estuary SPA and Ramsar**.

## 8.6 Ribble and Alt Estuaries SPA and Ramsar site

609. The Ribble and Alt Estuaries SPA and Ramsar site is located approximately 27km from the windfarm site.

### 8.6.1 Description of designation

610. The Ribble and Alt Estuaries SPA lies on the coast of Lancashire and Sefton in northwest England. The SPA and Ramsar site encompasses all or parts of Ribble Estuary SSSI and Sefton Coast SSSI. It comprises two estuaries, of which the Ribble is by far the larger, together with an extensive area of sandy foreshore along the Sefton Coast, and forms part of the chain of west coast SPAs that fringe the Irish Sea. There is considerable interchange in the movements of birds between this site and Morecambe Bay, Mersey Estuary, Dee Estuary and Martin Mere. A large proportion of the SPA is within the Ribble Estuary National Nature Reserve.

611. The SPA consists of extensive areas of sand and mudflats and, particularly in the Ribble, large areas of saltmarsh. There are also areas of coastal grazing marsh. The intertidal flats are rich in invertebrates on which waders and some wildfowl feed. The highest densities of feeding birds are on the muddier substrates of the Ribble, though sandy shores throughout are also used. Saltmarshes and coastal grazing marshes support high densities of wildfowl and these, together with intertidal sand and mudflats throughout, are used as high tide roosts. The site supports internationally important populations of waterbirds in winter, including swans, geese, ducks and waders. It is also of major importance during migration periods, especially for wader populations moving along the west coast of Britain. The larger expanses of saltmarsh and areas of coastal grazing marsh support breeding birds, including large concentrations of gulls and terns. These seabirds feed both offshore and inland, outside the SPA. Several species of waterfowl (notably Pink-footed Goose) utilise feeding areas on agricultural land outside the SPA boundary.

### 8.6.2 Conservation objectives

612. The SPA's conservation objectives are to ensure that, subject to natural change, the integrity of the site is maintained or restored as appropriate, and that the site contributes to achieving the aims of the Wild Birds Directive, by maintaining or restoring:

- The extent and distribution of the habitats of the qualifying features
- The structure and function of the habitats of the qualifying features
- The supporting processes on which the habitats of the qualifying features rely

- The populations of each of the qualifying features
- The distribution of qualifying features within the site

## 8.6.3 Assessment

### 8.6.3.1 Migratory waterbird qualifying features

#### Status

613. The status of each migratory waterbird qualifying feature screened into the Appropriate Assessment for this site is presented in **Table 5.2**. This consists of the site population at designation, national population in 2012 (Wright *et al*., 2012) and the latest five-year peak mean WeBS count (Frost *et al*., 2021).

#### Functional linkage and seasonal apportionment of potential effects

614. All qualifying features of this designated site have been screened into the Appropriate Assessment due to the risk of potential impacts occurring during the spring and autumn migration seasons. Other than grey plover and dunlin (which were both recorded on one occasion in the first year of surveys only), the qualifying features were not recorded in the aerial survey study area during the baseline surveys undertaken at the windfarm site. However, it is recognised that the qualifying features may pass through the habitat in the windfarm site during migration periods, and may have been missed by the surveys.

615. The apportioning of impacts to this designated site was calculated for each qualifying feature by dividing the number of collisions calculated at the national level by the proportion of the national population that were members of the designated site population at citation. The numbers used to define the national populations were the Great Britain populations presented in Wright *et al.,* (2012). Designated site populations were obtained from the SPA citation, or the Ramsar site population if the SPA citation did not include a population estimate.

#### Potential effects on the qualifying features

616. The qualifying features of this designated site have been screened into the Appropriate Assessment due to the potential risk of collision.

617. The magnitudes of potential collision impacts have been investigated using the SOSSMAT tool (Wright *et al.,* 2012).

## Potential effects on the qualifying feature from the Project-alone

### Collision risk

618. The estimated annual collision risk for of each qualifying feature from this designated site, along with the conclusion of the assessment based on this annual collision rate, is presented in **Table 8.28**. An avoidance rate of 0.980 has been assumed for all species.

619. The number of annual collisions predicted for all qualifying features is very low. It is expected that the increases to existing mortality rates for each qualifying feature due to this impact would be undetectable within the site population. Such impacts would consequently not result in any measurable effect.

620. **It is concluded that the predicted mortality of all qualifying features due to collision at the Project windfarm site would not adversely affect the integrity of the Ribble and Alt Estuaries SPA and Ramsar site.**

621. Whilst extensive information exists on the responses of waterbirds to onshore OWFs, there is substantial uncertainty regarding waterbird movements at sea. The confidence level assigned to this section of the assessment is therefore medium. However, since such low levels of collision are predicted, an adverse effect on the integrity of the site is considered highly unlikely, even in the unlikely event that impacts have been underestimated.

## Potential effects on the qualifying feature in-combination with other projects

### Collision risk

622. The migration corridors identified by Wright *et al.,* (2012) indicate that migration activity of all qualifying features from this designated site is widespread across UK waters. Similarly low numbers of birds, and hence collisions, are therefore expected at other OWFs in UK waters. The total collision mortality of non-breeding waterbirds at all UK OWFs is still likely to be small in the context of their respective national populations, and the number of collisions associated with this designated site will be smaller still. It is expected that the increases to existing mortality rates for each qualifying feature due to this impact would be undetectable within the site population. Such impacts would consequently not result in any measurable effect.

623. **It is concluded that predicted mortality of all qualifying features due to collision at the Project windfarm site, in-combination with other projects, would not adversely affect the integrity of the Ribble and Alt Estuaries SPA and Ramsar site**.

*Table 8.28 Information to support the Appropriate Assessment for Ribble and Alt Estuaries SPA and Ramsar site (migratory waterbird qualifying features)*

| Qualifying feature | GB population (Wright *et al.*, 2012) | SPA population (citation / standard data form) | Ramsar site population (citation) | Five-year peak mean 2015/16 - 2019/20 | Apportioning rate | Unapportioned predicted mean annual collisions (avoidance rate 0.980) | Annual collisions apportioned to SPA | Conclusion of adverse effect on site integrity |
|---|---|---|---|---|---|---|---|---|
| Bewick's swan | 7,000 | 276 | 230 | 3 | 3.9% | 0.00 | 0.00 | No adverse effect on site integrity. Numbers of collisions so small that effects on population would be negligible. It would not be possible for impacts of this magnitude to have an effect at the site level given the background populations |
| Whooper swan | 11,000 | 182 | 211 | 292 | 1.7% | 0.03 | 0.00 | |
| Pink-footed goose | 360,000 | 11,764 | 6,552 | 38,974 | 3.3% | 0.01 | 0.00 | |
| Shelduck | 61,000 | 4,925 | 2,944 | 4,311 | 8.1% | 0.03 | 0.00 | |
| Wigeon | 440,000 | 85,259 | 69,841 | 51,865 | 19.4% | 0.23 | 0.04 | |
| Teal | 210,000 | 7,157 | 5,107 | 6,354 | 3.4% | 0.08 | 0.00 | |
| Pintail | 29,000 | 2,731 | 1,497 | 1,182 | 9.4% | 0.01 | 0.00 | |
| Oystercatcher | 320,000 | 18,535 | 18,926 | 17,837 | 5.8% | 0.23 | 0.01 | |
| Ringed plover | 34,000 | 1,657 | 3,761 | 1,950 | 4.9% | 0.01 | 0.00 | |
| Golden plover | 400,000 | 3,598 | 3,588 | 2,721 | 0.9% | 0.26 | 0.00 | |
| Grey plover | 43,000 | 9,355 | 11,021 | 2,910 | 21.8% | 0.03 | 0.01 | |
| Knot | 320,000 | 68,922 | 42,692 | 50,798 | 21.5% | 0.20 | 0.04 | |
| Sanderling | 16,000 | 2,882 | 7,401 | 8,465 | 18.0% | 0.01 | 0.00 | |
| Bar-tailed godwit | 38,000 | 20,086 | 13,935 | 7,393 | 52.9% | 0.04 | 0.02 | |
| Black-tailed godwit | 43,000 | 1,273 | 3,323 | 4,493 | 3.0% | 0.03 | 0.00 | |
| Redshank | 120,000 | 2,505 | 4,465 | 2,450 | 2.1% | 0.08 | 0.00 | |
| Dunlin | 350,000 | 39,376 | 38,196 | 35,386 | 11.3% | 0.38 | 0.04 | |
| Ruff (breeding) | 800 | 2 (b) | 60 (nb) | 31 (nb) | 0.3% | 0.00 | 0.00 | |
| Waterbird assemblage | - | 323,861 | 222,038 | 266,577 | - | - | - | No adverse effect on site integrity. Based on the small number of collisions predicted for named qualifying features, no adverse effect on integrity is anticipated |

### 8.6.3.2 Lesser black-backed gull

#### Status

624. Breeding lesser black-backed gull is listed as a qualifying species of this SPA.

625. The SPA population was cited as 1,800 pairs in 1993 (Furness, 2015, Stroud *et al*., 2016). Furness (2015) proposed a breeding population of 8,267 pairs in 2012. The most recent count (2021) was 4,489 AON (equivalent to breeding pairs), or 8,978 breeding adults (JNCC, 2022a). Natural England (2020c) have set a target to '*Maintain the size of the [SPA] breeding population at a level which is above 8097 pairs, whilst avoiding deterioration from its current level as indicated by the latest mean peak count or equivalent*'.

626. Based on the most recent SPA population of breeding adults, and an annual breeding adult baseline mortality rate of 0.115 (Horswill and Robinson, 2015), 1,032 breeding adults from the SPA population would be expected to die each year.

#### Functional linkage and seasonal apportionment of potential effects

627. The windfarm site is situated approximately 37km at its nearest point from the breeding location for lesser black-backed gull on the Ribble Estuary within the SPA. The mean maximum foraging range of lesser black-backed gull is 127km (±109, 1SD) (Woodward *et al*., 2019). The windfarm site is therefore within the mean maximum foraging range of lesser black-backed gulls from the Ribble and Alt Estuaries SPA.

628. A tracking study undertaken at the colony during the breeding season (Scragg *et al*., 2016) indicated that the majority of birds from the Ribble Estuary colony foraged in inland areas, with only a 'handful' of trips recorded offshore, and all of these restricted to nearshore areas (i.e. <20km from the coast). Areas predominantly used by lesser black-backed gulls included the Mersey Estuary, urban areas, fields, landfill sites and ground workings, as well as the intertidal mudflats and saltmarsh of the Ribble Estuary itself. While this information does not mean that breeding adult lesser black-backed gulls from the SPA will not be present at the windfarm site during the breeding season, it does suggest that birds from the SPA are likely to make little use of the windfarm site and spend little time there.

629. The windfarm site is located within the mean maximum foraging range +1SD of a total of 168 lesser black-backed gull colonies (both SPA and non-SPA), with a total count of 72,320 individuals. See **Section 8.5.2.2** for a review of these other SPAs and colonies. It is therefore likely that birds present at the windfarm site during the breeding season may originate from a number of

different colonies within the mean maximum +1SD foraging range for this species.

630. In addition, some of the lesser black-backed gulls recorded at the windfarm site during the breeding season would be sub-adult birds. Based on review of raw survey data, 286 lesser black-backed gulls were recorded during the two-year baseline digital aerial surveys. Of these, 177 birds were able to be assigned to an age class, and of these, 126 birds (71.2% of those assigned to an age class) were classified as adults. It is therefore assumed, on a precautionary basis, that 71.2% of lesser black-backed gulls apportioned to the SPA during the full breeding season are breeding adult birds from Ribble and Alt Estuaries SPA. This estimate is likely to include additional precaution for two reasons. Firstly, lesser black-backed gulls display plumage that is effectively indistinguishable from adult birds by their third winter (Cramp and Simmons, 1983), but typically start breeding in their fifth year (Horswill and Robinson, 2015). Therefore, the proportion of adult (breeding age) birds may be overestimated when based solely on plumage characteristics. Secondly, it is likely that any adult lesser black-backed present will include a proportion of sabbatical (non-breeding) individuals, so the proportion of breeding adult birds is likely to be further overestimated.

631. The NatureScot apportioning tool (NatureScot, 2018) has been used to estimate the proportion of lesser black-backed gulls from each of the relevant SPAs present at the windfarm site during the breeding season (see **Appendix 12.1** of the ES for apportioning outputs). As there was uncertainty regarding the extent to which inland colonies may contribute to the population present in coastal/offshore areas (i.e. at the windfarm site), two estimates for breeding season apportioning have been calculated. The first has used all breeding colonies (both coastal and inland, including Bowland Fells SPA) within mean maximum +1SD foraging range of the windfarm site. The second approach has included only coastal sites (defined as being located within 5km of the coast), which excludes Bowland Fells SPA from the calculation (i.e. assumes that no birds from Bowland Fells SPA occur at the windfarm site during the breeding season).

632. When all lesser black-backed gull colonies are included, it is estimated that 34.36% of birds present at the windfarm site are apportioned to Ribble and Alt Estuaries SPA during the breeding season. If only coastal sites are included, it is estimated that 60.94% are apportioned to the SPA.

633. In addition to the potential for connectivity during the breeding season, there is also potential for the breeding lesser black-backed gull qualifying feature of the Ribble and Alt Estuaries SPA to have connectivity with the windfarm site during the non-breeding periods. Thus, during the pre and post breeding periods, breeding lesser black-backed gulls from the Ribble and Alt Estuaries SPA migrate through UK waters, whilst some birds remain in the UK during

the winter. The relevant reference population is considered to be the UK Western Waters BDMPS, within which the birds from different breeding colony source populations and of different age classes are assumed to be distributed evenly throughout. This consists of 163,304 individuals during autumn migration (September to October), 41,159 individuals during winter (November to February) and 163,304 individuals during spring migration (March) (Furness, 2015).

634. Furness (2015) estimated that 50% of the Ribble and Alt Estuaries SPA breeding adults (16,534) were present within the UK Western Waters BDMPS during the autumn and spring migration periods, representing 8,267 birds. During the winter period 20% of the population was estimated to be present, representing 3,307 birds. This is equivalent to 5.06% of the BDMPS population for the autumn and spring periods, and 8.03% during winter. During autumn migration, winter, and spring migration, 5.06%, 8.03%, and 5.06% of impacts are considered to affect birds from the SPA respectively (Furness, 2015).

## Potential effects on the qualifying feature

635. The lesser black-backed gull qualifying feature of the Ribble and Alt Estuaries SPA has been screened into the assessment due to the potential risk of collision.

### Operation and maintenance phase collision risk

*Project-alone*

636. Information to support the Appropriate Assessment for collision risk on breeding adult lesser black-backed gulls belonging to the Ribble and Alt Estuaries SPA population is presented in **Table 8.29**. Collision estimates, calculated using Option 2 the sCRM (McGregor *et al.*, 2018), are presented by biological season. A summary of the annual outputs and the corresponding increase in the annual baseline mortality rate is also presented. Parameters used in the sCRM, together with the avoidance rate applied, were agreed with Natural England during the ETG process and are described in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES.

637. If all lesser black-backed gull colonies are included in the breeding season apportioning, and based on the mean collision rates, the annual total of breeding adult lesser black backed gulls from the Ribble and Alt Estuaries SPA at risk of collision due to the Project is 0.58. This would increase the existing mortality of the SPA breeding population by 0.06%. If only coastal sites are used for apportioning, annual mortality would be 0.96, representing an increase of 0.09% to the existing baseline annual mortality.

*Table 8.29 Predicted seasonal and annual collision mortality (Stochastic model Option 2, avoidance rate 0.994 (±0.0004)) for breeding adult lesser black-backed gulls at the windfarm site, apportioned to Ribble and Alt Estuaries SPA, with corresponding increases to baseline mortality of the population*

| | Breeding Season (all sites apportioned) | Breeding Season (only coastal sites apportioned) | Autumn Migration | Non-breeding/winter | Spring Migration | Annual (all sites apportioned) | Annual (only coastal sites apportioned) |
|---|---|---|---|---|---|---|---|
| **Period** | **Apr-Aug** | **Apr-Aug** | **Sep-Oct** | **Nov-Feb** | **Mar** | **Jan-Dec** | **Jan-Dec** |
| Total collisions[1] (mean and 95% CIs) | 1.44 (0.00-4.53) | 1.44 (0.00-4.53) | 1.25 (0.00-5.63) | 0.15 (0.00-0.80) | 0.15 (0.00-0.94) | **2.98 (0.00-11.90)** | **2.98 (0.00-11.90)** |
| % apportioned to the SPA | 34.36% | 60.94% | 5.06% | 8.03% | 5.06% | **-** | **-** |
| Total SPA collisions (mean and 95% CIs) | 0.49 (0.00-1.56) | 0.88 (0.00-2.76) | 0.06 (0.00-0.28) | 0.01 (0.00-0.06) | 0.01 (0.00-0.05) | **0.58 (0.00-1.95)** | **0.96 (0.00-3.16)** |
| Mortality increase[2] (mean and 95% CIs) | 0.05% (0.00-0.15%) | 0.09% (0.00-0.27%) | 0.01% (0.00-0.03%) | 0.00% (0.00-0.01%) | 0.00% (0.00-0.00%) | **0.06% (0.00-0.19%)** | **0.09% (0.00-0.31%)** |

[1] Breeding season collision values reduced to 71.9% of modelled value to reflect proportion of adult birds recorded site surveys.

[2] Assuming predicted annual SPA mortality of 1032 birds (8,978 x 0.115)

638. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that no detectable changes in mortality rates would occur in this population from the mean monthly collision estimates for the Project, or for the upper 95% CI collision estimate.

639. For the Project-alone, **it is concluded that predicted lesser black-backed gull mortality due to collision at the windfarm site would not adversely affect the integrity of the Ribble and Alt Estuaries SPA and Ramsar.**

640. The confidence in the assessment is high. The evidence used to define the CRM input parameters presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES submitted alongside this RIAA is of high applicability and quality. Whilst there is uncertainty around some of the input parameters (e.g. avoidance rate), the rates selected are considered to be sufficiently precautionary based on expert opinion to provide confidence that collision rates are not underestimated.

*In-combination*

641. On the basis of the conclusions of the Project-alone assessment (i.e. very low predicted lesser black-backed gull collision mortality, equating to less than one bird), and for the reasons set out below, **there would be no measurable contribution of the Project to in-combination effects**. **Accordingly, no in-combination assessment is required for this feature**. The conclusion of the Project-alone assessment is therefore unchanged, **i.e. that predicted lesser black-backed gull mortality due to collision at the Project windfarm site would not adversely affect the integrity of the Ribble and Alt Estuaries SPA and Ramsar.**

642. Notwithstanding this conclusion, **Paragraphs 645** to **649** below present an estimate of in-combination mortality and a Population Viability Analysis (PVA), to provide context to the Project-alone assessment. This information is presented without prejudice to the conclusion above.

643. The reasons that the Project would not contribute to the in-combination effect are as follows:

- Evidence from tracking studies (e.g. in Scragg *et al*., 2016) suggests that birds from the SPA are unlikely to frequently occur at the windfarm site (or other project sites), and therefore the apportioned project-alone collision values are likely to be a significant overestimate.
- Even if birds from the SPA frequently occur at the windfarm site, the Project contribution to the in-combination total (based on a 'worst-case', assuming that only birds from coastal sites contribute to apportioning) is very small; less than one (0.96) bird per annum and representing less than 2.5% of all predicted collisions apportioned to the SPA. Even at this precautionary rate (precautionary for the reasons set out in the Project-

alone assessment above), it would be expected that less than one bird from the SPA would die in each year that the Project was operational.

- There is very strong evidence to suggest that the key causes of decline in the SPA population have been driven by other factors (including predation and changes in land-use), and the very small level of additional mortality attributable to the Project is very likely be inconsequential in this wider context.
- In the absence of the Project, the predicted lesser-black backed gull mortality apportioned to the SPA (38.84 birds; refer to **Table 8.30** below) would increase background mortality by 3.76%; the contribution of the Project (resulting in a total increase in background mortality of 3.86%) is considered inconsequential to the in-combination effect.
- PVA outputs for the in-combination mortality (refer to **Paragraph 649** and **Table 8.31** below) confirm that the contribution of the Project would make no measurable difference to the annual growth rate or reduction in population size, taking into account the uncertainties around the PVA outputs (particularly over the 35-year period assessed within the PVA). The reduction in annual growth rate for in-combination mortality would be 0.49% when the Project is excluded, increasing to 0.51% including the Project (**Table 8.31**). The reduction in population size at the end of the 35-year period would be 17.9% in the absence of the Project, or 18.3% including the Project. This difference (i.e. 0.4%) is below a threshold that is likely to be detectable at a population level, and indistinguishable from natural variation.

644. It is noted that during Examination for the Sheringham and Dudgeon Extension Projects (SEP and DEP), Natural England (2023a) concluded that for comparable lesser black-backed gull mortality levels apportioned to Alde-Ore Estuary SPA (mortality of 0.24 birds per annum, equivalent to 0.06% increase in background mortality; Equinor, 2023) there would be *'no measurable contribution from SEP and DEP to in-combination effects'*.

645. The in-combination estimation for lesser black-backed gull mortality from the Ribble and Alt Estuaries SPA due to collision risk has been undertaken in accordance with the approach presented in **Section 8.1** and **Appendix 12.1** of the ES. **Table 8.30** sets out the predicted annual mortality for relevant projects (refer to **Chapter 12 Offshore Ornithology** of the ES). This information is presented without prejudice to the conclusion presented above, i.e. that the Project would make no meaningful contribution to the in-combination effect. The contribution of the Project used for the in-combination estimation is based on the 'worst-case' set out above, i.e. assuming that only birds from coastal sites are apportioned to the SPA (which predicts a mortality of 0.96 birds per annum).

*Table 8.30 Lesser black-backed gull – predicted in-combination collision mortality from Ribble and Alt Estuaries SPA*

| OWF project name | Predicted annual mortality |
|---|---|
| Burbo Bank Extension | 1.73 |
| Ormonde | 4.87 |
| Walney 1 + 2 | 12.60 |
| Walney 3 + 4 | 6.45 |
| West of Duddon Sands | 11.54 |
| Gwynt y Môr | 0.20 |
| Rhyl Flats | 0.04 |
| Robin Rigg | 'low/negligible' |
| Awel y Môr | 0.05 |
| Erebus | 0.27 |
| Twin Hub | 0.27 |
| Morgan Generation Assets | 0.18 |
| Mona | 0.56 |
| Burbo Bank | 0.08 |
| West of Orkney | 0.00 |
| White Cross | 0.02 |
| **Sub-total excluding the Project** | 38.84 |
| **Project (worst-case)** | 0.96 |
| **Total** | 39.80 |

646. In accordance with discussions with Natural England through the ETG process, consideration has also been given to potential in-combination effects with lethal control licensing for lesser black-backed gulls. Natural England (2020b) has undertaken HRA of the licensing process which estimated a total of 3,354 lesser black-backed gulls were killed in England under licence in 2019. The Natural England HRA did not include quantification of gull mortality apportioned to individual SPAs, but concluded no adverse effect on integrity for all SPAs when considered in-combination with other plans and projects (including offshore wind development). On this basis, it is concluded that licensed lethal control of lesser black-backed gulls would not affect the presented in-combination estimation.

647. Based on the Ribble and Alt Estuaries SPA breeding population of 8,978 adult birds and a background mortality of 0.115 (1,032 birds per annum), an increase in mortality of 39.8 birds would increase background mortality by 3.86%.

648. It is noted that for one historic project (Robin Rigg) no collision data for lesser black-backed gull were available. This project assessed the significance of effect on the species as 'low/negligible', and it is also the case that the Robin Rigg windfarm is a small development, which (at c.115km from the SPA boundary) is highly unlikely to have effects on the population. Therefore, given the absence of data for this project, it is considered unlikely that this would significantly alter the estimation presented above.

649. As background mortality, based on the estimates presented above, would exceed 1%, a population viability analysis (PVA) for the in-combination estimation has been undertaken. The results of the PVA are summarised in **Table 8.31**, with full details presented in **Appendix 12.1** of the ES. The PVA indicates that there would be a 0.51% reduction in annual population growth rate, and a net 18.3% reduction in population size at the end of the 35-year operational period, compared to the unimpacted scenario. In the absence of the Project, these values would be 0.49% and 17.9% respectively, which, as set out above, are considered well within the bounds of natural variation and therefore indistinguishable from the all-projects scenario.

650. As the Project would make no measurable contribution to the in-combination mortality**, it has been concluded that there would be no adverse effect on integrity to Ribble and Alt Estuaries SPA. Therefore, no conclusion in respect of in-combination effects for the Project is required.**

*Table 8.31 In-combination population viability analysis outputs for lesser black-backed gull at Ribble and Alt Estuaries SPA*

| Scenario | Predicted mortality | Growth rate | Median CPGR[1] | Median CPS[2] | Reduction in annual growth rate | Reduction in population size |
|---|---|---|---|---|---|---|
| Baseline (unimpacted) | 0 | 1.0080 | 1.0000 | 1.0000 | n/a | n/a |
| In-combination collision mortality including the Project | 39.8 | 1.0029 | 0.9949 | 0.8168 | 0.51% | 18.3% |

| Scenario | Predicted mortality | Growth rate | Median CPGR[1] | Median CPS[2] | Reduction in annual growth rate | Reduction in population size |
|---|---|---|---|---|---|---|
| In-combination collision mortality excluding the Project | 38.8 | 1.0030 | 0.9951 | 0.8210 | 0.49% | 17.9% |

[1] CPGR is the counterfactual of annual population growth rate, calculated as the median of the ratio of the annual growth rate of the impacted to un-impacted (or baseline) population, expressed as a proportion.

[2] CPS is the counterfactual of population size, calculated as the median of the ratio of the end-point size of the impacted to un-impacted population size, expressed as a proportion. In this case, the endpoint population size is predicted on the basis of a 35-year operational period.

### 8.6.3.3 Common tern

#### Status

651. Breeding common tern is listed as a qualifying feature of Ribble and Alt Estuaries SPA. The SPA population was cited as 182 pairs (364 individuals) in 1996, which represented 1.5% of the Great Britain population (English Nature, 2002). SMP data indicated a decline from 182 pairs in 1996 to 111 pairs in 2008; during 2015 only two birds and no breeding pairs were observed (JNCC, 2023a). The most recent available SMP data indicates that six pairs (12 breeding adults) fledged seven juveniles on Hesketh Out Marsh in 2018, and that one pair (two breeding adults) was present (on Banks Marsh) in 2021 (JNCC, 2023a).

652. Based on the most recent SPA population of breeding adults, and an annual adult baseline mortality rate of 0.117 (1 – 0.883; Horswill and Robinson, 2015), two breeding adult common terns from the SPA population would be expected to die each year.

#### Functional linkage and seasonal apportionment of potential effects

653. The mean maximum breeding season foraging range of common tern is 18.0km (±8.9km) and the maximum foraging range is 30km (Woodward *et al*., 2019). The Project is located 27.4km from Ribble and Alt Estuaries SPA, which means the Project is just beyond the mean maximum foraging range +1SD of breeding common terns from the SPA, but within the maximum foraging range. The maximum foraging range is a poor indicator of typical foraging behaviour. It would be expected that few birds or foraging trips will occur at this distance from the colony, and even fewer with any regularity. No

impacts during the breeding season have therefore been apportioned to birds breeding at this colony.

654. Outside the breeding season breeding common terns are assumed to range widely and to mix with birds of all ages from breeding colonies in the UK and further afield. The relevant background population is considered to be the UK Western waters BDMPS, consisting of 64,659 individuals during autumn migration (late July to early September), and spring migration (April to May) (Furness, 2015).

655. Estimates of the proportion of common terns present at the windfarm site during the autumn and spring migration seasons which originate from the Ribble and Alt Estuaries SPA site are based on the SPA population as a proportion of the UK Western waters BDMPS (Furness, 2015). During both autumn and spring migration seasons, breeding adult common terns from the Ribble and Alt Estuaries SPA make up 0.34% of the total BDMPS population. The same percentage of impacts are therefore attributable to birds from this SPA during these times of year.

### Potential effects on the qualifying feature from the Project-alone

656. The common tern qualifying feature of the Ribble and Alt Estuaries SPA has been screened into the assessment due to the potential risk of collision.

#### Operation and maintenance phase collision risk

*Project-alone*

657. Information for collision risk on breeding adult common terns belonging to the Ribble and Alt Estuaries SPA population is presented in **Table 8.32**. Collision estimates, calculated using the sCRM, are presented by biological season. A summary of the annual outputs and the corresponding increase in the annual baseline mortality rate is also presented. Parameters used in the sCRM were agreed with Natural England during the ETG process and are described in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES.

658. Based on the mean collision rates, the annual total of breeding adult common terns from the Ribble and Alt Estuaries SPA at risk of collision as a result of the Project is 0.00. This would result in no detectable increase (0.05%) in the existing mortality of the SPA breeding population.

*Table 8.32 Predicted seasonal and annual collision mortality (Stochastic model Option 2, avoidance rate 0.991 (±0.0004)) for breeding adult common terns at the windfarm site, apportioned to Ribble and Alt Estuaries SPA, with corresponding increases to baseline mortality of the population*

| | **Breeding Season** | **Autumn Migration** | **Non-breeding/winter** | **Spring Migration** | **Annual** |
|---|---|---|---|---|---|
| **Period** | **Jun-Jul** | **Aug-Sep** | **Oct-Mar** | **Apr-May** | **Jan-Dec** |
| Total collisions (mean and 95% Cis) | 0.00 (0.00-0.00) | 0.14 (0.01-0.37) | 0.00 (0.00-0.00) | 0.08 (0.00-0.22) | 0.22 (0.01-0.60) |
| % apportioned to the SPA | 0.0% | 0.34% | 0.0% | 0.34% | - |
| Total SPA collisions (mean and 95% Cis) | 0.00 (0.00-0.00) | 0.00 (0.00-0.00) | 0.00 (0.00-0.00) | 0.00 (0.00-0.00) | 0.00 (0.00-0.00) |
| **Mortality increase**[2] (**mean** and 95% Cis) | **0.00%** (0.00-0.00%) | **0.03%** (0.00-0.08%) | **0.00%** (0.00-0.00%) | **0.02%** (0.00-0.05%) | **0.05%** (0.00-0.13%) |

[1] May overlaps breeding and spring migration period, has been included in migration period as birds present at the windfarm site are considered most likely to be migrants.

[2] Assuming predicted annual SPA adult mortality of 2 birds (14 x 0.117)

659. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that no detectable changes in mortality rates would occur on this population for the mean monthly collision estimates for the Project, or for the upper 95% CI collision estimate.

660. **It is concluded that predicted common tern mortality due to collision at the windfarm site would not adversely affect the integrity of the Ribble and Alt Estuaries SPA and Ramsar.**

661. The confidence in the assessment is high. The evidence used to define the CRM input parameters presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES is of high applicability and quality. Whilst there is uncertainty around some of the input parameters (e.g. avoidance rate), the rates selected are considered to be sufficiently precautionary based on expert opinion to provide confidence that collision rates are not underestimated.

*In-combination*

662. As no measurable effects on common tern are predicted as a result of the Project-alone, there would be no material contribution to the effects of other plans or projects in-combination. **It is therefore concluded that is no potential for the Project-alone or in-combination to have an adverse effect on the integrity of the Ribble and Alt Estuaries SPA and Ramsar**.

## 8.7 Mersey Narrows and North Wirral Foreshore SPA and Ramsar

663. Mersey Narrows and North Wirral Foreshore SPA and Ramsar site is located approximately 42km from the windfarm site.

### 8.7.1 Description of designation

664. Mersey Narrows and North Wirral Foreshore SPA is located on the northwest coast of England at the mouths of the Mersey and Dee estuaries. The site comprises intertidal habitats at Egremont foreshore, artificial lagoons at Seaforth and the extensive intertidal flats at North Wirral Foreshore. Egremont is most important as a feeding habitat for waders at low tide whilst Seaforth is primarily a high tide roost site, as well as a nesting site for terns. North Wirral Foreshore supports large numbers of feeding waders at low tide and also includes important high tide roost sites.

### 8.7.2 Conservation objectives

665. The SPA's conservation objectives are to ensure that the integrity of the site is maintained or restored as appropriate, and that the site contributes to achieving the aims of the Wild Birds Directive, by maintaining or restoring:

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

### 8.7.3 Assessment

666. Six qualifying features of Mersey Narrows and North Wirral Foreshore SPA and Ramsar site have been screened into the Appropriate Assessment (refer to **Table 5.2**): bar-tailed godwit, knot, little gull, common tern (breeding), common tern (non-breeding) and the waterbird assemblage.

#### 8.7.3.1 Migratory waterbird qualifying features

#### Status

667. The status of each migratory waterbird qualifying feature screened into the Appropriate Assessment for this site is presented in **Table 8.33**. This consists

of the site population at designation, national population in 2012 (Wright *et al*., 2012) and latest five-year peak mean WeBS count (Frost *et al*., 2021).

## Functional linkage and seasonal apportionment of potential effects

668. All qualifying features of this designated site have been screened into the Appropriate Assessment due to the risk of potential impacts occurring during the spring and autumn migration seasons. The qualifying features were not recorded in the aerial survey study area during the baseline surveys undertaken at the windfarm site. However, it is recognised that the qualifying features may pass through the habitat in the windfarm site during migration periods and may have been missed by the surveys.

669. The apportioning of impacts to this designated site was calculated for each qualifying feature by dividing the number of collisions calculated at the national level by the proportion of the national population that were members of the designated site population at citation. The numbers used to define the national populations were the Great Britain populations presented in Wright *et al.,* (2012). Designated site populations were obtained from the SPA citation, or the Ramsar site population if the SPA citation did not include a population estimate.

## Potential effects on the qualifying features

670. The qualifying features of this designated site have been screened into the Appropriate Assessment due to the potential risk of collision.

671. The magnitude of potential collision impacts has been investigated using the SOSSMAT tool (Wright *et al*., 2012).

## Potential effects on the qualifying feature from the Project-alone

### Collision risk

672. The estimated annual collision risk for each qualifying feature from this designated site, along with the conclusion of the assessment based on this annual collision rate, is presented in **Table 8.33**. An avoidance rate of 0.980 has been assumed for all species.

673. The number of annual collisions predicted for all qualifying features is very low. It is expected that the increases to existing mortality rates for each qualifying feature due to this impact would be undetectable within the site population. Such impacts would consequently not result in any measurable effect.

674. **It is concluded that the predicted mortality of all qualifying features due to collision at the Project windfarm site would not adversely affect the**

**integrity of the Mersey Narrows and North Wirral Foreshore SPA and Ramsar site.**

675. Whilst extensive information exists on the responses of waterbirds to onshore OWFs, there is substantial uncertainty regarding waterbird movements at sea. The confidence level assigned to this section of the assessment is therefore medium. However, since such low levels of collision are predicted, an adverse effect on the integrity of the site is considered highly unlikely, even in the unlikely event that impacts have been underestimated.

## Potential effects on the qualifying feature in-combination with other projects

### Collision risk

676. The migration corridors identified by Wright *et al.,* (2012) indicated that migration activity of all qualifying features from this designated site would be widespread across UK waters. Similarly low numbers of birds, and hence collisions, are therefore expected at other OWFs in UK waters. The total collision mortality of non-breeding waterbirds at all UK OWFs is still likely to be small in the context of their respective national populations, and the number of collisions associated with this designated site will be smaller still. It is expected that the increases to existing mortality rates for each qualifying feature due to this impact would be undetectable within the site population. Such impacts would consequently not result in any measurable effect.

677. **It is concluded that predicted mortality of all qualifying features due to collision at the Project windfarm site, in-combination with other projects, would not adversely affect the integrity of the Mersey Narrows and North Wirral Foreshore SPA and Ramsar site**.

*Table 8.33 Information to support the Appropriate Assessment for Mersey Narrows and North Wirral Foreshore SPA and Ramsar site (migratory waterbird qualifying features)*

| Qualifying feature | GB population (Wright *et al.*, 2012) | SPA population (citation / standard data form) | Ramsar site population (citation) | Five-year peak mean 2015/16 – 2019/20 | Apportioning rate | Unapportioned predicted mean annual collisions (avoidance rate 0.980) | Annual collisions apportioned to SPA | Conclusion of adverse effect on site integrity |
|---|---|---|---|---|---|---|---|---|
| Bar-tailed godwit | 38,000 | 3,344 | 3,344 | Not available | 8.8% | 0.04 | 0.00 | No adverse effect on site integrity. Numbers of collisions so small that effects on population would be negligible. It would not be possible for impacts of this magnitude to have an effect at the site level given the background populations |
| Knot | 320,000 | 10,655 | 10,655 | Not available | 3.3% | 0.20 | 0.01 | |
| Waterbird assemblage | - | 32,366 | 32,402 | Not available | - | - | - | No adverse effect on site integrity. Based on the small number of collisions predicted for named qualifying features, no adverse effect on integrity is anticipated |

#### 8.7.3.2 Little gull

##### Status

678. Non-breeding little gull is listed as a qualifying feature of Mersey Narrows and North Wirral Foreshore SPA. The SPA population was cited as 213 individuals for the period 2004/05 – 2008/09 (Natural England, 2013). This species has undergone a decline within this SPA and other SPAs in the Liverpool City Region (Ross-Smith *et al*., 2015), however, on a precautionary basis, the citation count is used as the reference population for the assessment.

679. Based on the SPA population of assumed non-breeding adults, and an annual adult baseline mortality rate of 0.200 (1 – 0.800; Horswill and Robinson, 2015), the expected annual mortality of the SPA population would be 43 adults.

##### Functional linkage and seasonal apportionment of potential effects

680. It is assumed that all birds present at the windfarm site are apportioned to the Liverpool Bay SPA population (**Section 8.4.2.3**). Accordingly, no birds from the Mersey Narrows and North Wirral Foreshore SPA are predicted to occur at the windfarm site.

##### Potential effects on the qualifying feature from the Project-alone and in-combination with other projects

681. As no little gulls from the Mersey Narrows and North Wirral Foreshore SPA are predicted to occur at the windfarm site, there would be no impacts to this feature. **It is therefore concluded that there would be no adverse effect on the integrity of the Mersey Narrows and North Wirral Foreshore SPA, either alone or in-combination with other projects**.

#### 8.7.3.3 Common tern

##### Status

682. Common tern is listed as a qualifying feature of Mersey Narrows and North Wirral Foreshore SPA for both breeding and non-breeding populations.

683. The breeding population was cited as 177 pairs, or 354 breeding adults, for the period 2005 – 2009 (Natural England, 2013). The most recent count was 208 pairs (AON), or 416 breeding adults, in 2021 (JNCC, 2023a); this has been used as the reference population for the assessment. Based on the most recent SPA count of breeding adults and an annual baseline mortality rate of 0.117 (1 – 0.883; Horswil and Robinson, 2015), the expected annual mortality of the SPA breeding population would be 49 adults.

684. The non-breeding population was cited as 1,475 individuals for the period 2004 – 2008 (Natural England, 2013).

## Functional linkage and seasonal apportionment of potential effects

685. The mean maximum breeding season foraging range of common tern is 18.0km (±8.9km) and the maximum foraging range is 30km (Woodward *et al*., 2019). The windfarm site is located approximately 42km from Mersey Narrows and North Wirral Foreshore SPA, which means the windfarm site is beyond the maximum foraging range of common terns from the SPA. No impacts from the Project during the breeding season are therefore apportioned to the SPA common tern colony.

686. Outside the breeding season, breeding common terns are assumed to range widely and to mix with birds of all ages from breeding colonies in the UK and further afield. The relevant background population is considered to be the UK Western waters BDMPS, consisting of 64,659 individuals during autumn migration (late July to early September), and spring migration (April to May) (Furness, 2015).

687. Estimates of the proportion of common terns present at the windfarm site during the autumn and spring migration seasons originating from the Mersey Narrows and North Wirral Foreshore SPA site are based on the SPA population as a proportion of the UK Western waters BDMPS (Furness, 2015). Furness (2015) did not define a SPA population, and therefore the closest pre-publication value (i.e. 177 pairs or 354 individuals) has been used. During both autumn and spring migration seasons, breeding adult common terns from Mersey Narrows and North Wirral Foreshore SPA make up 0.55% of the total BDMPS population. The same percentage of impacts are therefore attributable to birds from this SPA during these times of year.

688. The SPA has been designated to protect important foraging areas for its non-breeding population of common terns. It is therefore considered that the designation does not protect birds once outside of the SPA. Therefore, the non-breeding common tern population is not considered within the Appropriate Assessment for this site.

## Potential effects on the qualifying feature from the Project-alone

689. The common tern qualifying feature of the Mersey Narrows and North Wirral Foreshore SPA has been screened into the assessment due to the potential risk of collision.

### Operation and maintenance phase collision risk

#### *Project-alone*

690. Information for collision risk on breeding adult common terns belonging to the Mersey Narrows and North Wirral Foreshore SPA population is presented in **Table 8.34**. Collision estimates, calculated using the sCRM, are presented by biological season. A summary of the annual outputs and the corresponding

increase in the annual baseline mortality rate is also presented. Parameters used in the sCRM were agreed with Natural England during the ETG process and are described in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES.

691. Based on the mean collision rates, the annual total of breeding adult common terns from the Mersey Narrows and North Wirral Foreshore SPA at risk of collision as a result of the Project is 0.00. This would result in no detectable increase (<0.01%) in the existing mortality of the SPA breeding population.

*Table 8.34 Predicted seasonal and annual collision mortality (Stochastic model Option 2, avoidance rate 0.991 (±0.0004)) for breeding adult common terns at the windfarm site, apportioned to Mersey Narrows and North Wirral Foreshore SPA, with corresponding increases to baseline mortality of the population*

| Period | Breeding Season<br>Jun-Jul | Autumn Migration<br>Aug-Sep | Non-breeding/winter<br>Oct-Mar | Spring Migration<br>Apr-May | Annual<br>Jan-Dec |
|---|---|---|---|---|---|
| Total collisions (mean and 95% Cis) | 0.00<br>(0.00-0.00) | 0.14<br>(0.01-0.37) | 0.00<br>(0.00-0.00) | 0.08<br>(0.00-0.22) | 0.22<br>(0.01-0.60) |
| % apportioned to the SPA | 0.0% | 0.55% | 0.0% | 0.55% | - |
| Total SPA collisions (mean and 95% Cis) | 0.00<br>(0.00-0.00) | 0.00<br>(0.00-0.00) | 0.00<br>(0.00-0.00) | 0.00<br>(0.00-0.00) | 0.00<br>(0.00-0.00) |
| **Mortality increase**[2] (**mean** and 95% Cis) | **0.00%**<br>(0.00-0.00%) | **0.00%**<br>(0.00-0.00%) | **0.00%**<br>(0.00-0.00%) | **0.00%**<br>(0.00-0.00%) | **0.00%**<br>(0.00-0.01%) |

[1] May overlaps breeding and spring migration period, has been included in migration period as birds present at the windfarm site are considered most likely to be migrants.

[2] Assuming predicted annual SPA adult mortality of 49 birds (416 x 0.117)

692. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that no detectable changes in mortality rates would occur on this population for the mean monthly collision estimates for the Project, or for the upper 95% CI collision estimate.

693. **It is concluded that predicted common tern mortality due to collision at the windfarm site would not adversely affect the integrity of the Mersey Narrows and North Wirral Foreshore SPA and Ramsar.**

694. The confidence in the assessment is high. The evidence used to define the CRM input parameters presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES is of high applicability and quality. Whilst there is uncertainty around some of the input parameters (e.g. avoidance rate), the rates selected are considered to be sufficiently precautionary based on expert opinion to provide confidence that collision rates are not underestimated.

*In-combination*

695. As no measurable effects on common tern are predicted as a result of the Project-alone, there would be no material contribution to the effects of other plans or projects in-combination. **It is therefore concluded that there is no potential for the Project-alone or in-combination to have an adverse effect on the integrity of the Mersey Narrows and North Wirral Foreshore SPA and Ramsar**.

# 8.8 Martin Mere SPA and Ramsar

696. Martin Mere SPA and Ramsar site is located approximately 43km from the windfarm site.

## 8.8.1 Description of designation

697. Martin Mere is located north of Ormskirk in West Lancashire. It occupies part of a former lake and mire which extended over some 1,300 ha of the Lancashire Coastal Plain during the 17th century that was, until it was drained, the largest body of freshwater in England Currently, the complex site comprises open water, seasonally flooded marsh and damp, neutral hay meadows overlying deep peat. It includes a wildfowl refuge of international importance, with a large and diverse wintering, passage and breeding bird community.

## 8.8.2 Conservation objectives

698. The SPA's conservation objectives are to ensure that the integrity of the site is maintained or restored as appropriate, subject to natural change, and ensure that the site contributes to achieving the aims of the Wild Birds Directive, by maintaining or restoring:

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

## 8.8.3 Assessment

699. Seven migratory waterbird qualifying features of Martin Mere SPA and Ramsar site have been screened into the Appropriate Assessment: Bewick's swan, whooper swan, pink-footed goose, teal, pintail, wigeon, and the waterbird assemblage.

### 8.8.3.1 Migratory waterbird qualifying features

#### Status

700. The status of each migratory waterbird qualifying feature screened into the Appropriate Assessment for this site is presented in **Table 8.35**. This consists

of the site population at designation, national population in 2012 (Wright *et al*., 2012) and latest five-year peak mean WeBS count (Frost *et al*., 2021).

### Functional linkage and seasonal apportionment of potential effects

701. All qualifying features of this designated site have been screened into the Appropriate Assessment due to the risk of potential impacts occurring during the spring and autumn migration seasons. The qualifying features were not recorded in the aerial survey study area during the baseline surveys undertaken at the windfarm site. However, it is recognised that the qualifying features may pass through the habitat in the windfarm site during migration periods, and may have been missed by the surveys.

702. The apportioning of impacts to this designated site was calculated for each qualifying feature by dividing the number of collisions calculated at the national level by the proportion of the national population that were members of the designated site population at citation. The numbers used to define the national populations were the Great Britain populations presented in Wright *et al.,* (2012). Designated site populations were obtained from the SPA citation, or the Ramsar site population if the SPA citation did not include a population estimate.

### Potential effects on the qualifying features

703. The qualifying features of this designated site have been screened into the Appropriate Assessment due to the potential risk of collision.

704. The magnitude of potential collision impacts has been investigated using the SOSSMAT tool (Wright *et al*., 2012).

### Potential effects on the qualifying feature from the Project-alone

#### Collision risk

705. The estimated annual collision risk for of each qualifying feature from this designated site, along with the conclusion of the assessment based on this annual collision rate, is presented in **Table 8.35**. An avoidance rate of 0.980 has been assumed for all species.

706. The number of annual collisions predicted for all qualifying features is very low. It is expected that the increases to existing mortality rates for each qualifying feature due to this impact would be undetectable within the site population. Such impacts would consequently not result in any measurable effect.

707. **It is concluded that the predicted mortality of all qualifying features due to collision at the Project windfarm site would not adversely affect the integrity of the Martin Mere SPA and Ramsar site.**

708. Whilst extensive information exists on the responses of waterbirds to onshore OWFs, there is substantial uncertainty regarding waterbird movements at sea. The confidence level assigned to this section of the assessment is therefore medium. However, since such low levels of collision are predicted, an adverse effect on the integrity of the site is considered highly unlikely, even in the unlikely event that impacts have been underestimated.

## Potential effects on the qualifying feature in-combination with other projects

### Collision risk

709. The migration corridors identified by Wright *et al.,* (2012) indicate that migration activity of all qualifying features from this designated site is widespread across UK waters. Similarly low numbers of birds, and hence collisions, are therefore expected at other OWFs in UK waters. The total collision mortality of non-breeding waterbirds at all UK OWFs is still likely to be small in the context of their respective national populations, and the number of collisions associated with this designated site will be smaller still. It is expected that the increases to existing mortality rates for each qualifying feature due to this impact would be undetectable within the site population. Such impacts would consequently not result in any measurable effect.

710. **It is concluded that predicted mortality of all qualifying features due to collision at the Project windfarm site, in-combination with other projects, would not adversely affect the integrity of the Martin Mere SPA and Ramsar site**.

*Table 8.35 Information to support the Appropriate Assessment for Martin Mere SPA and Ramsar site (migratory waterbird qualifying features)*

| Qualifying feature | GB population (Wright *et al*., 2012) | SPA population (citation/ standard data form) | Ramsar site population (citation) | Five-year peak mean 2015/16 – 2019/20 | Apportioning rate | Unapportioned predicted mean annual collisions (avoidance rate 0.980) | Annual collisions apportioned to SPA | Conclusion of adverse effect on site integrity |
|---|---|---|---|---|---|---|---|---|
| Bewick's swan | 7,000 | 449 | 747 | 0 | 6.4% | 0.00 | 0.00 | No adverse effect on site integrity. Numbers of collisions so small that effects on population would be negligible. It would not be possible for impacts of this magnitude to have an effect at the site level given the background populations |
| Whooper swan | 11,000 | 621 | 513 | 1,248 | 5.6% | 0.03 | 0.00 | |
| Pink-footed goose | 360,000 | 25,779 | 32,967 | 23,526 | 7.2% | 0.01 | 0.00 | |
| Teal | 210,000 | 3,282 | | 2,753 | 1.6% | 0.08 | 0.00 | |
| Pintail | 29,000 | 978 | 1,344 | 427 | 3.4% | 0.01 | 0.00 | |
| Wigeon | 440,000 | 9,062 | 9,606 | 1,775 | 2.1% | 0.23 | 0.00 | |
| Waterbird assemblage | - | 46,196 | 25,827 | - | - | - | - | No adverse effect on site integrity. Based on the small number of collisions predicted for named qualifying features, no adverse effect on integrity is anticipated |

## 8.9 The Dee Estuary SPA and Ramsar

711. The Dee Estuary SPA and Ramsar site is located approximately 44km from the windfarm site.

### 8.9.1 Description of designation

712. The Dee Estuary is a large, funnel-shaped, sheltered estuary on the border between England and Wales, which supports extensive areas of intertidal sand and mudflats and saltmarsh. Where agricultural reclamation has not occurred, the saltmarshes grade into transitional brackish and swamp vegetation on the upper shore. The site also includes the three sandstone islands of Hilbre, with their important cliff vegetation and maritime heathland and grassland. The site is of major importance for waterbirds; during the winter the intertidal flats, saltmarshes and fringing habitats including coastal grazing marsh/fields, provide feeding and roosting sites for internationally important numbers of ducks and waders; in summer the site supports nationally important breeding colonies of two species of tern. The site is also important during migration periods, particularly for wader populations moving along the west coast of Britain and for Sandwich terns post-breeding.

### 8.9.2 Conservation objectives

713. The SPA's conservation objectives are to ensure that the integrity of the site is maintained or restored as appropriate, subject to natural change, and ensure that the site contributes to achieving the aims of the Wild Birds Directive, by maintaining or restoring:

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

### 8.9.3 Assessment

714. Fifteen qualifying features of The Dee Estuary SPA and Ramsar site have been screened into the Appropriate Assessment; refer to **Table 5.2**.

#### 8.9.3.1 Migratory waterbird qualifying features

##### Status

715. The status of each migratory waterbird qualifying feature screened into the Appropriate Assessment for this site is presented in **Table 8.36**. This consists of the site population at designation, national population in 2012 (Wright *et al.,* 2012) and latest five-year peak mean WeBS count (Frost *et al*., 2021).

##### Functional linkage and seasonal apportionment of potential effects

716. All qualifying features of this designated site have been screened into the Appropriate Assessment due to the risk of potential impacts occurring during the spring and autumn migration seasons. Other than grey plover and dunlin (which were both recorded on one occasion in the first year of surveys only), the qualifying features were not recorded in the aerial survey study area during the baseline surveys undertaken at the windfarm site. However, it is recognised that the qualifying features may pass through the habitat in the windfarm site during migration periods, and may have been missed by the surveys.

717. The apportioning of impacts to this designated site was calculated for each qualifying feature by dividing the number of collisions calculated at the national level by the proportion of the national population that were members of the designated site population at citation. The numbers used to define the national populations were the Great Britain populations presented in Wright *et al.,* (2012). Designated site populations were obtained from the SPA citation, or the Ramsar site population if the SPA citation did not include a population estimate.

##### Potential effects on the qualifying features

718. The qualifying features of this designated site have been screened into the Appropriate Assessment due to the potential risk of collision.

719. The magnitudes of potential collision impacts have been investigated using the SOSSMAT tool (Wright *et al*., 2012).

##### Potential effects on the qualifying feature from the Project-alone

**Collision risk**

720. The estimated annual collision risk for of each qualifying feature from this designated site, along with the conclusion of the assessment based on this annual collision rate, is presented in **Table 8.36**. An avoidance rate of 0.980 has been assumed for all species.

721. The number of annual collisions predicted for all qualifying features is very low. It is expected that the increases to existing mortality rates for each qualifying feature due to this impact would be undetectable within the site population. Such impacts would consequently not result in any measurable effect.

722. **It is concluded that the predicted mortality of all qualifying features due to collision at the Project windfarm site would not adversely affect the integrity of the Dee Estuary SPA and Ramsar site.**

723. Whilst extensive information exists on the responses of waterbirds to onshore OWFs, there is substantial uncertainty regarding waterbird movements at sea. The confidence level assigned to this section of the assessment is therefore medium. However, since such low levels of collision are predicted, an adverse effect on the integrity of the site is considered highly unlikely, even in the unlikely event that impacts have been underestimated.

### Potential effects on the qualifying feature in-combination with other projects

#### Collision risk

724. The migration corridors identified by Wright *et al.,* (2012) indicate that migration activity of all qualifying features from this designated site is widespread across UK waters. Similarly low numbers of birds, and hence collisions, are therefore expected at other OWFs in UK waters. The total collision mortality of non-breeding waterbirds at all UK OWFs is still likely to be small in the context of their respective national populations, and the number of collisions associated with this designated site will be smaller still. It is expected that the increases to existing mortality rates for each qualifying feature due to this impact would be undetectable within the site population. Such impacts would consequently not result in any measurable effect.

725. **It is concluded that predicted mortality of all qualifying features due to collision at the Project windfarm site, in-combination with other projects, would not adversely affect the integrity of the Dee Estuary SPA and Ramsar site**.

*Table 8.36 Information to support the Appropriate Assessment for The Dee Estuary SPA and Ramsar site (migratory waterbird qualifying features)*

| Qualifying feature | GB population (Wright *et al*., 2012) | SPA population (citation/ standard data form) | Ramsar site population (citation) | Five-year peak mean 2015/16 – 2019/20 | Apportioning rate | Unapportioned predicted mean annual collisions (avoidance rate 0.980) | Annual collisions apportioned to SPA | Conclusion of adverse effect on site integrity |
|---|---|---|---|---|---|---|---|---|
| Shelduck | 61,000 | 7,725 | 7,725 | 9,062 | 12.7% | 0.03 | 0.00 | No adverse effect on site integrity. Numbers of collisions so small that effects on population would be negligible. It would not be possible for impacts of this magnitude to have an effect at the site level given the background populations |
| Teal | 210,000 | 5,251 | 5,251 | 6,062 | 2.5% | 0.08 | 0.00 | |
| Pintail | 29,000 | 5,407 | 5,407 | 5,355 | 18.6% | 0.01 | 0.00 | |
| Oystercatcher | 320,000 | 22,677 | 22,677 | 23,309 | 7.1% | 0.23 | 0.02 | |
| Grey plover | 43,000 | 1,643 | 1,643 | 910 | 3.8% | 0.01 | 0.03 | |
| Knot | 320,000 | 12,394 | 12,394 | 17,197 | 3.9% | 0.20 | 0.01 | |
| Bar-tailed godwit | 38,000 | 1,150 | 1,150 | 359 | 3.0% | 0.04 | 0.00 | |
| Curlew | 140,000 | 3,899 | 3,899 | 3,553 | 2.8% | 0.10 | 0.00 | |
| Redshank (p) | 120,000 | 8,795 | 8,795 | 9,614 | 7.3% | 0.08 | 0.01 | |
| Redshank (w) | 120,000 | 5,293 | 5,293 | 9,614 | 4.4% | 0.08 | 0.00 | |
| Black-tailed godwit | 43,000 | 1,747 | 1,747 | 6,206 | 4.1% | 0.03 | 0.00 | |
| Dunlin | 350,000 | 27,769 | 27,769 | 16,922 | 7.9% | 0.38 | 0.03 | |
| Waterbird assemblage | - | 120,726 | 120,726 | - | - | - | - | No adverse effect on site integrity. Based on the small number of collisions predicted for named qualifying features, no adverse effect on integrity is anticipated |

#### 8.9.3.2 Sandwich tern

##### Status

726. Sandwich tern is listed as a qualifying feature of The Dee Estuary SPA due to the site's importance for the species post-breeding. The mean autumn passage population at classification was 957 individuals for the period 1995 – 1999 (Natural England, 2014); this is used as the reference population for the assessment.

727. Based on the SPA population and an annual adult baseline mortality rate of 0.102 (1 – 0.898 (annual survival rate); Horswill and Robinson, 2015) the expected annual mortality of the SPA population would be 98 adults.

##### Functional linkage and seasonal apportionment of potential effects

728. The SPA has been designated to protect important foraging areas for its non-breeding population of Sandwich terns. It is therefore considered that the designation does not protect birds once outside of the SPA. Therefore, the non-breeding common tern population is not considered within the Appropriate Assessment for this site.

##### Potential effects on the qualifying feature from the Project-alone and in-combination

729. As no Sandwich terns from the SPA were apportioned to the windfarm site, **it is concluded that there would be no adverse effect on site integrity in respect of the Dee Estuary SPA, either alone or in-combination with other projects.**

#### 8.9.3.3 Common tern

##### Status

730. Breeding common tern is listed as a qualifying feature of The Dee Estuary SPA. The mean SPA population at classification was 392 pairs, or 784 breeding adults, for the period 1995 – 1999 (Natural England, 2019a). Furness (2015) gave a breeding population of 165 pairs, or 330 breeding adults, in 2013. In the absence of more recent SMP data, the 2013 estimate is used as the reference population for the assessment.

731. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.117 (1 – 0.883 (annual survival rate); Horswill and Robinson, 2015), the expected annual mortality of the SPA population would be 39 breeding adults.

## Functional linkage and seasonal apportionment of potential effects

732. The mean maximum breeding season foraging range of common tern is 18.0km (±8.9km) and the maximum foraging range is 30km (Woodward *et al* 2019). The Project is located approximately 44km from the Dee Estuary SPA, which means that the Project is beyond the maximum foraging range of common terns from the SPA. No impacts during the breeding season from the Project are therefore apportioned to common terns breeding at this SPA.

733. Outside the breeding season, breeding common terns are assumed to range widely and to mix with birds of all ages from breeding colonies in the UK and further afield. The relevant background population is considered to be the UK Western waters BDMPS, consisting of 64,659 individuals during autumn migration (late July to early September), and spring migration (April to May) (Furness, 2015).

734. Estimates of the proportion of common terns present at the Project site during the autumn and spring migration seasons which originate from the Dee Estuary SPA site are based on the SPA population as a proportion of the UK Western waters BDMPS (Furness, 2015). During both autumn and spring migration seasons, breeding adult common terns from the Dee Estuary SPA make up 0.51% of the total BDMPS population. The same percentage of impacts are therefore attributable to birds from this SPA during these times of year.

## Potential effects on the qualifying feature from the Project-alone

735. The common tern qualifying feature of the Dee Estuary SPA has been screened into the assessment due to the potential risk of collision.

### Operation and maintenance phase collision risk

*Project-alone*

736. Information for collision risk on breeding adult common terns belonging to the Dee Estuary SPA population is presented in **Table 8.37**. Collision estimates, calculated using the sCRM, are presented by biological season. A summary of the annual outputs and the corresponding increase in the annual baseline mortality rate is also presented. Parameters used in the sCRM were agreed with Natural England during the ETG process and are described in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES.

737. Based on the mean collision rates, the annual total of breeding adult common terns from the Dee Estuary SPA at risk of collision as a result of the Project is 0.00. This would result in no detectable increase (0.00%) in the existing mortality of the SPA breeding population.

*Table 8.37 Predicted seasonal and annual collision mortality (Stochastic model Option 2, avoidance rate 0.991 (±0.0004)) for breeding adult common terns at the windfarm site, apportioned to Dee Estuary SPA, with corresponding increases to baseline mortality of the population*

| Period | Breeding Season Jun-Jul | Autumn Migration Aug-Sep | Non-breeding/winter Oct-Mar | Spring Migration Apr-May | Annual Jan-Dec |
|---|---|---|---|---|---|
| Total collisions (mean and 95% Cis) | 0.00 (0.00-0.00) | 0.14 (0.01-0.37) | 0.00 (0.00-0.00) | 0.08 (0.00-0.22) | 0.22 (0.01-0.60) |
| % apportioned to the SPA | 0.0% | 0.34% | 0.0% | 0.34% | - |
| Total SPA collisions (mean and 95% Cis) | 0.00 (0.00-0.00) | 0.00 (0.00-0.00) | 0.00 (0.00-0.00) | 0.00 (0.00-0.00) | 0.00 (0.00-0.00) |
| **Mortality increase**[2] (**mean** and 95% Cis) | **0.00%** (0.00-0.00%) | **0.00%** (0.00-0.00%) | **0.00%** (0.00-0.00%) | **0.00%** (0.00-0.00%) | **0.00%** (0.00-0.01%) |

[1] May overlaps breeding and spring migration period, has been included in migration period as birds present at the windfarm site are considered most likely to be migrants.

[2] Assuming predicted annual SPA adult mortality of 39 birds (330 x 0.117)

738. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that no detectable changes in mortality rates would occur on this population for the mean monthly collision estimates for the Project, or for the upper 95% CI collision estimate.

739. **It is concluded that predicted common tern mortality due to collision at the windfarm site would not adversely affect the integrity of the Dee Estuary SPA and Ramsar.**

740. The confidence in the assessment is high. The evidence used to define the CRM input parameters presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES is of high applicability and quality. Whilst there is uncertainty around some of the input parameters (e.g. avoidance rate), the rates selected are considered to be sufficiently precautionary based on expert opinion to provide confidence that collision rates are not underestimated.

*In-combination*

741. As no measurable effects on common tern are predicted as a result of the Project-alone, there would be no material contribution to the effects of other plans or projects in-combination. **It is therefore concluded that is no potential for the Project-alone or in-combination to have an adverse effect on the integrity of the Dee Estuary SPA and Ramsar**.

## 8.10 Anglesey Terns/Morwenoliaid Ynys Môn SPA

742. Anglesey Terns / Morwenoliaid Ynys Môn SPA is located approximately 49km from the windfarm site.

### 8.10.1 Description of designation

743. Anglesey Terns/Morwenoliaid Ynys Môn SPA comprises three tern breeding sites on the coast of Anglesey, North Wales, together with marine habitats used by foraging birds from those colonies during the breeding season. The SPA covers an area of 101,931 ha in total, and includes an area off the east coast of Anglesey which overlaps the Liverpool Bay SPA. The SPA is designated for its breeding populations of common tern, Arctic tern, roseate tern and Sandwich tern.

### 8.10.2 Conservation objectives

744. The draft conservation objectives for Anglesey Terns/Morwenoliaid Ynys Môn SPA for 'Feature 1-4: Breeding population Terns' (NRW, 2015) are as follows:

- The number of breeding terns within the SPA is stable or increasing
- The number of chicks successfully fledged in the SPA and beyond is sufficient to help sustain the population
- The range and distribution of terns within the SPA and beyond is not constrained or hindered
- The extent of supporting habitats used by terns is stable or increasing
- Supporting habitats are of sufficient quality to support the requirements of terns
- There are appropriate and sufficient food sources for terns within access of the SPA
- Actions or events likely to impinge on the sustainability of the population are under control

745. Based on the above conservation objectives, the specific relevant target for the Sandwich tern feature of this SPA is that the breeding population of Sandwich tern should be stable or increasing. The site was designated for 460 pairs across the SPA.

## 8.10.3 Assessment

### 8.10.3.1 Sandwich tern

#### Status

746. Sandwich tern is listed as a qualifying species of this SPA.

747. The SPA population was cited as 460 pairs (920 adult birds), which represented 3.3% of the GB population. This species occurs at one of the three tern colonies within the SPA (Cemlyn Bay) the most recent count in the SMP database was 1972 AON in 2020, the equivalent of 3,944 breeding adults.

748. Based on the most recent SPA population of breeding adults, and an annual breeding adult baseline mortality rate of 0.102 (Horswill and Robinson, 2015), 402 breeding adults from the SPA population would be expected to die each year.

**Functional linkage and seasonal apportionment of potential effects**

749. The Cemlyn Bay colony is located approximately 66km from the windfarm site at its closest point. The mean maximum foraging range of Sandwich tern is 34.3km (±23.2km) (Woodward *et al*., 2019). The windfarm site is therefore located outside of the typical foraging range of this species. It is therefore considered unlikely that birds from this colony would occur at the windfarm site during the breeding season; this is confirmed by the absence of records of this species from aerial surveys of the windfarm site.

750. Very low densities of this species were recorded within the windfarm site during September only, which are assumed to be birds on autumn passage. It is considered very unlikely that these individuals would be associated with the Anglesey Terns/Morwenoliaid Ynys Môn SPA population, as birds present at the windfarm site are more likely to be birds moving from colonies to the north (e.g. in Scotland) or from Ireland. The species has been known to follow the coastline during migration (Cramp, 1985), and therefore birds from the Cemlyn Bay colony would be expected to disperse along the coast (e.g. southwards along the Wales coast) rather than north-eastwards towards the windfarm site.

751. It is therefore concluded that common terns present at the windfarm site are very unlikely to be associated with the Anglesey Terns/Morwenoliaid Ynys Môn SPA population.

**Potential effects on the qualifying feature**

752. As Sandwich terns from the Anglesey Terns/Morwenoliaid Ynys Môn SPA are not considered to occur at the windfarm site, no effects on this feature are predicted as a result of the Project. **It is therefore concluded that there**

**would be no adverse effect on the integrity of the Anglesey Terns/Morwenoliaid Ynys Môn SPA**.

753. The confidence in the assessment is high. The evidence used to inform the assessment is considered to be of high quality and applicability and is supported by the results of the baseline surveys.

**Potential effects in-combination with other projects**

754. As no effects on Sandwich tern are predicted as a result of the Project-alone, there would be no contribution to the effects of other plans or projects in-combination. **It is therefore concluded that there would be no adverse effect on the integrity of Anglesey Terns/Morwenoliaid Ynys Môn SPA**.

#### 8.10.3.2 Common tern

#### Status

755. The Anglesey Terns/Morwenoliaid Ynys Môn SPA breeding common tern population was cited as 189 pairs, or 378 breeding adults. Furness (2015) and Stroud *et al.,* (2014) gave a breeding population of 592 pairs, or 1,184 breeding adults, in 2011. The most recent total count from the three colonies (Cemlyn Lagoon, Ynys Feurig and The Skerries) was 517 pairs (AON), or 1,034 breeding adults, in 2019 (JNCC, 2023a); this is used as the reference population for the assessment.

756. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.117 (1 – 0.883; Horswill and Robinson, 2015), the expected annual mortality of the SPA population would be 121 breeding adults.

#### Functional linkage and seasonal apportionment of potential effects

757. The mean maximum breeding season foraging range of common tern is 18.0km (±8.9km) and the maximum foraging range is 30km (Woodward *et al,* 2019). The Project is located approximately 49km from Anglesey Terns/Morwenoliaid Ynys Môn SPA, which means that the Project is beyond the maximum foraging range of common terns from the SPA. No impacts during the breeding season from the Project are therefore apportioned to common terns breeding at this SPA.

758. Outside the breeding season breeding common terns are assumed to range widely and to mix with birds of all ages from breeding colonies in the UK and further afield. The relevant background population is considered to be the UK Western waters BDMPS, consisting of 64,659 individuals during autumn migration (July to September), and spring migration (March to May) (Furness, 2015).

759. Estimates of the proportion of common terns present at the Project site during the autumn and spring migration seasons which originate from the Anglesey Terns/Morwenoliaid Ynys Môn SPA site are based on the SPA population as a proportion of the UK Western waters BDMPS (Furness, 2015). During both autumn and spring migration seasons, breeding adult common terns from the Anglesey Terns/Morwenoliaid Ynys Môn SPA make up 1.83% of the total BDMPS population. The same percentage of impacts are therefore apportioned to birds from this SPA during these times of year.

### Potential effects on the qualifying feature from the Project-alone

760. The common tern qualifying feature of the Anglesey Terns/Morwenoliaid Ynys Môn SPA has been screened into the assessment due to the potential risk of collision.

#### Operation and maintenance phase collision risk

*Project-alone*

761. Information for collision risk on breeding adult common terns belonging to the Anglesey Terns/Morwenoliaid Ynys Môn SPA population is presented in **Table 8.38**. Collision estimates, calculated using the sCRM, are presented by biological season. A summary of the annual outputs and the corresponding increase in the annual baseline mortality rate is also presented. Parameters used in the sCRM were agreed with Natural England during the ETG process and are described in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES.

762. Based on the mean collision rates, the annual total of breeding adult common terns from the Anglesey Terns/Morwenoliaid Ynys Môn SPA at risk of collision as a result of the Project is 0.00. This would result in no detectable increase (0.04%) in the existing mortality of the SPA breeding population.

*Table 8.38 Predicted seasonal and annual collision mortality (Stochastic model Option 2, avoidance rate 0.991 (±0.0004)) for breeding adult common terns at the windfarm site, apportioned to Anglesey Terns/Morwenoliaid Ynys Môn SPA, with corresponding increases to baseline mortality of the population*

| Period | Breeding Season<br>Jun-Jul | Autumn Migration<br>Aug-Sep | Non-breeding/winter<br>Oct-Mar | Spring Migration<br>Apr-May | Annual<br>Jan-Dec |
|---|---|---|---|---|---|
| Total collisions (mean and 95% Cis) | 0.00<br>(0.00-0.00) | 0.14<br>(0.01-0.37) | 0.00<br>(0.00-0.00) | 0.08<br>(0.00-0.22) | 0.22<br>(0.01-0.60) |
| % apportioned to the SPA | 0.0% | 1.83% | 0.0% | 1.83% | - |
| Total SPA collisions (mean and 95% Cis) | 0.00<br>(0.00-0.00) | 0.00<br>(0.00-0.01) | 0.00<br>(0.00-0.00) | 0.00<br>(0.00-0.00) | 0.00<br>(0.00-0.00) |
| **Mortality increase**[2] (**mean** and 95% Cis) | **0.00%**<br>(0.00-0.00%) | **0.02%**<br>(0.00-0.06%) | **0.00%**<br>(0.00-0.00%) | **0.01%**<br>(0.00-0.04%) | **0.04%**<br>(0.00-0.10%) |

[1] May overlaps breeding and spring migration period, has been included in migration period as birds present at the windfarm site are considered most likely to be migrants.

[2] Assuming predicted annual SPA adult mortality of 121 birds (1,034 x 0.117)

763. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that no detectable changes in mortality rates would occur on this population for the mean monthly collision estimates for the Project, or for the upper 95% CI collision estimate.

764. **It is concluded that predicted common tern mortality due to collision at the windfarm site would not adversely affect the integrity of the Anglesey Terns/Morwenoliaid Ynys Môn SPA.**

765. The confidence in the assessment is high. The evidence used to define the CRM input parameters presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES is of high applicability and quality. Whilst there is uncertainty around some of the input parameters (e.g. avoidance rate), the rates selected are considered to be sufficiently precautionary based on expert opinion to provide confidence that collision rates are not underestimated.

*In-combination*

766. As no measurable effects on common tern are predicted as a result of the Project-alone, there would be no material contribution to the effects of other plans or projects in-combination. **It is therefore concluded that is no potential for the Project-alone or in-combination to have an adverse effect on the integrity of the Anglesey Terns/Morwenoliaid Ynys Môn SPA**.

#### 8.10.3.3 Arctic tern

##### Status

767. The Anglesey Terns/Morwenoliaid Ynys Môn SPA breeding Arctic tern population was cited as 1,290 pairs, or 2,580 breeding adults. Furness (2015) and Stroud *et al* (2014) gave a breeding population of 3,620 pairs, or 7,240 breeding adults, in 2013. The most recent total count from the three colonies (Cemlyn Lagoon, Ynys Feurig and The Skerries) was 3,206 pairs (AON), or 6,412 breeding adults, in 2019 (JNCC, 2023a); this is used as the reference population for the assessment.

768. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.163 (1 – 0.837; Horswill and Robinson, 2015), the expected annual mortality of the SPA population would be 1,045 breeding adults.

##### Functional linkage and seasonal apportionment of potential effects

769. The mean maximum foraging range of Arctic tern is 25.7km (±14.8km), and the maximum foraging range is 46km. The Project is located approximately 49km from Anglesey Terns/Morwenoliaid Ynys Môn SPA, which means that the Project is beyond the maximum foraging range of arctic terns from the

SPA. No impacts during the breeding season from the Project are therefore apportioned to arctic terns breeding at this SPA.

770. Outside the breeding season, Arctic tern are assumed to range widely and to mix with birds of all ages from breeding colonies in the UK and further afield. The relevant background population is considered to be the UK Western waters BDMPS, consisting of 71,398 individuals during autumn migration (August to September), and spring migration (April) (Furness, 2015).

771. Estimates of the proportion of Arctic terns present at the Project site during the autumn and spring migration seasons which originate from the Anglesey Terns/Morwenoliaid Ynys Môn SPA site are based on the SPA population as a proportion of the UK Western waters BDMPS (Furness, 2015). During both autumn and spring migration seasons, breeding adult arctic terns from the Anglesey Terns/Morwenoliaid Ynys Môn SPA make up 1.54% of the total BDMPS population. The same percentage of impacts are therefore attributable to birds from this SPA during these times of year.

### Potential effects on the qualifying feature from the Project-alone

772. The Arctic tern qualifying feature of the Anglesey Terns/Morwenoliaid Ynys Môn SPA has been screened into the assessment due to the potential risk of collision.

#### Operation and maintenance phase collision risk

*Project-alone*

773. Information for collision risk on breeding adult Arctic terns belonging to the Anglesey Terns/Morwenoliaid Ynys Môn SPA population is presented in **Table 8.39**. Collision estimates, calculated using the sCRM, are presented by biological season. A summary of the annual outputs and the corresponding increase in the annual baseline mortality rate is also presented. Parameters used in the sCRM were agreed with Natural England during the ETG process and are described in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES.

774. Based on the mean collision rates, the annual total of breeding adult Arctic terns from the Anglesey Terns/Morwenoliaid Ynys Môn SPA at risk of collision as a result of the Project is 0.00. This would result in no detectable increase (0.00%) in the existing mortality of the SPA breeding population.

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*Table 8.39 Predicted seasonal and annual collision mortality (Stochastic model Option 2, avoidance rate 0.991 (±0.0004)) for breeding adult Arctic terns at the windfarm site, apportioned to Anglesey Terns/Morwenoliaid Ynys Môn SPA, with corresponding increases to baseline mortality of the population*

| | **Breeding Season** | **Autumn Migration** | **Non-breeding/winter** | **Spring Migration** | **Annual** |
|---|---|---|---|---|---|
| **Period** | **Jun-Jul** | **Aug-Sep** | **Oct-Mar** | **Apr-May** | **Jan-Dec** |
| Total collisions (mean and 95% Cis) | 0.35<br>(0.01-1.57) | 0.02<br>(0.00-0.09) | 0.00<br>(0.00-0.00) | 0.00<br>(0.00-0.00) | 0.37<br>(0.01-1.66) |
| % apportioned to the SPA | 0.0% | 1.54% | 0.0% | 1.54% | - |
| Total SPA collisions (mean and 95% Cis) | 0.00<br>(0.00-0.00) | 0.00<br>(0.00-0.00) | 0.00<br>(0.00-0.00) | 0.00<br>(0.00-0.00) | 0.00<br>(0.00-0.00) |
| **Mortality increase**[1] (**mean** and 95% Cis) | **0.00%**<br>(0.00-0.00%) | **0.00%**<br>(0.00-0.00%) | **0.00%**<br>(0.00-0.00%) | **0.00%**<br>(0.00-0.00%) | **0.00%**<br>(0.00-0.00%) |

[1] Assuming predicted annual SPA adult mortality of 1,045 birds (6,412 x 0.163)

775. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that no detectable changes in mortality rates would occur on this population for the mean monthly collision estimates for the Project, or for the upper 95% CI collision estimate.

776. **It is concluded that predicted Arctic tern mortality due to collision at the windfarm site would not adversely affect the integrity of the Anglesey Terns/Morwenoliaid Ynys Môn SPA.**

777. The confidence in the assessment is high. The evidence used to define the CRM input parameters presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES is of high applicability and quality. Whilst there is uncertainty around some of the input parameters (e.g. avoidance rate), the rates selected are considered to be sufficiently precautionary based on expert opinion to provide confidence that collision rates are not underestimated.

*In-combination*

778. As no measurable effects on Arctic tern are predicted as a result of the Project-alone, there would be no material contribution to the effects of other plans or projects in-combination. **It is therefore concluded that is no potential for the Project-alone or in-combination to have an adverse effect on the integrity of the Anglesey Terns/Morwenoliaid Ynys Môn SPA**.

## 8.11 Bowland Fells SPA

779. Bowland Fells SPA is located approximately 53km from the windfarm site.

### 8.11.1 Description of designation

780. Bowland Fells SPA encompasses the main upland block within the area of Lancashire known as the Forest of Bowland. These extensive upland fells support the largest expanse of blanket bog and heather moorland in Lancashire and provide suitable habitat for a diverse upland breeding bird community which includes the Annex I species hen harrier and merlin for which the SPA is classified. The site also qualifies as it supports more than 1% of the biogeographic population of breeding lesser black-backed gull.

### 8.11.2 Conservation objectives

781. Bowland Fells SPA's conservation objectives are to ensure that the integrity of the site is maintained or restored as appropriate, and ensure that the site contributes to achieving the aims of the Wild Birds Directive, by maintaining or restoring:

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

### 8.11.3 Assessment

#### 8.11.3.1 Hen harrier

##### Status

782. Breeding hen harrier is listed as a qualifying feature of Bowland Fells SPA. The SPA was designated in 1993 for holding an average of at least 12 breeding pairs. Three breeding pairs were counted in 2018, with a five-year mean of 1.4 breeding pairs in 2014-2018 (information from RSPB, in Natural England, 2019b). Recent years have seen an upturn in numbers, with 16 nesting attempts by 15 females in the SPA in 2022, the first time in over 10 years that the SPA has reached the minimum number of pairs it was designated for (RSPB, 2022). Four males were polygamous, giving an SPA breeding population of 26 adults in 2022.

### Functional linkage and seasonal apportionment of potential effects

783. Hen harrier has been screened into the Appropriate Assessment due to the risk of potential impacts occurring during the spring and autumn migration seasons. This species was not recorded in the aerial survey study area during the baseline surveys undertaken at the windfarm site. However, it is recognised that there is the potential that it may pass through the habitat in the windfarm site during migration periods and may have been missed by the surveys.

784. The apportioning of impacts to this designated site was calculated by dividing the number of collisions calculated at the national level by the proportion of the national population that were members of the designated site population at citation. The numbers used to define the national populations were the Great Britain populations presented in Wright *et al.,* (2012), comprising 570 pairs or 1,140 individuals during the breeding season. Designated site populations were obtained from the SPA citation, comprising 12 pairs or 24 individuals. Accordingly, 2.1% of impacts to this species were apportioned to Bowland Fells SPA.

### Potential effects on the qualifying feature from the Project-alone

785. Hen harrier has been screened into the Appropriate Assessment due to the potential risk of collision. The magnitude of potential collision impacts has been investigated using the SOSSMAT tool (Wright *et al*., 2012).

786. The unapportioned annual collision risk calculated using the SOSSMAT tool (refer to **Appendix 12.1** of the ES), is estimated at less than 0.01 birds, assuming an avoidance rate of 0.980. Zero mortality is therefore apportioned to Bowland Fells SPA. Such impacts would consequently not result in any measurable effect.

787. **It is concluded that the predicted hen harrier mortality would not adversely affect the integrity of the Bowland Fells SPA.**

788. Whilst extensive information exists on the responses of raptors to onshore OWFs, there is substantial uncertainty regarding the effects on migratory bird movements at sea. The confidence level assigned to this section of the assessment is therefore medium. However, since such low levels of collision are predicted, an adverse effect on the integrity of the site is considered highly unlikely, even in the unlikely event that impacts have been underestimated.

### Potential effects on the qualifying feature in-combination with other projects

789. As no measurable effects are predicted as a result of the project alone, there would be no contribution to in-combination effects. **It is concluded that predicted hen harrier mortality due to collision at the windfarm site,**

**alone and in-combination with other projects, would not adversely affect the integrity of the Bowland Fells SPA**.

### 8.11.3.2 Merlin

#### Status

790. Breeding merlin is listed as a qualifying feature of Bowland Fells SPA. The SPA was designated in 1993 for holding an average of 21 pairs of breeding merlin. Stroud *et al* (2016) gave a breeding population of 16 pairs in 2008. The most recent evidence points to the SPA holding approximately 8-12 pairs (2018 RSPB survey, Waterman Infrastructure & Environment Limited 2017, Bowland Ecology 2016, in Natural England 2019) which gives a maximum of 24 breeding adults.

#### Functional linkage and seasonal apportionment of potential effects

791. Merlin has been screened into the Appropriate Assessment due to the risk of potential impacts occurring during the spring and autumn migration seasons. This species was not recorded in the aerial survey study area during the baseline surveys undertaken at the windfarm site. However, it is recognised that there is the potential that it may pass through the habitat in the windfarm site during migration periods, and may have been missed by the surveys.

792. The apportioning of impacts to this designated site was calculated by dividing the number of collisions calculated at the national level by the proportion of the national population that were members of the designated site population at citation. The numbers used to define the national populations were the Great Britain populations presented in Wright *et al.,* (2012), comprising 1,330 pairs or 2,660 individuals during the breeding season. Designated site populations were obtained from the SPA citation, comprising 21 pairs or 42 individuals. Accordingly, 1.6% of impacts to this species were apportioned to Bowland Fells SPA.

#### Potential effects on the qualifying feature from the Project-alone

793. Merlin has been screened into the Appropriate Assessment due to the potential risk of collision. The magnitude of potential collision impacts has been investigated using the SOSSMAT tool (Wright *et al.,* 2012).

794. The unapportioned annual collision risk calculated using the SOSSMAT tool (refer to **Appendix 12.1** of the ES), is estimated at less than 0.01 birds, assuming an avoidance rate of 0.980. Zero mortality is therefore apportioned to Bowland Fells SPA. Such impacts would consequently not result in any measurable effect.

795. **It is concluded that the predicted merlin mortality would not adversely affect the integrity of the Bowland Fells SPA.**

796. Whilst extensive information exists on the responses of raptors to onshore OWFs, there is substantial uncertainty regarding the effects on migratory bird movements at sea. The confidence level assigned to this section of the assessment is therefore medium. However, since such low levels of collision are predicted, an adverse effect on the integrity of the site is considered highly unlikely, even in the unlikely event that impacts have been underestimated.

### Potential effects on the qualifying feature in-combination with other projects

797. As no measurable effects are predicted as a result of the project alone, there would be no contribution to in-combination effects. **It is concluded that predicted merlin mortality due to collision at the windfarm site, alone and in-combination with other projects, would not adversely affect the integrity of the Bowland Fells SPA**.

### 8.11.3.3 Lesser black-backed gull

### Status

798. Breeding lesser black-backed gull is listed as a qualifying feature of Bowland Fells SPA. The SPA population was cited as 13,900 pairs in 1998 (Furness, 2015, Stroud *et al*., 2016). Furness (2015) proposed a breeding population of 4,575 pairs in 2013. The most recent count (2018) was 14,341 AON, or 28,682 breeding adults (JNCC, 2023a).

799. Based on the most recent SPA population of breeding adults, and an annual breeding adult baseline mortality rate of 0.115 (1-0.885 (annual survival rate); Horswill and Robinson (2015)), 3,298 breeding adults from the SPA population would be expected to die each year.

### Functional linkage and seasonal apportionment of potential effects

800. The mean maximum foraging range of lesser black-backed gull is 127km (±109km) and the maximum foraging range is 533km (Woodward *et al*., 2019). The Project is located approximately 53km from Bowland Fells SPA, which means the Project is within the mean maximum foraging range of breeding lesser black-backed gulls from this SPA.

801. As described in **Section 8.5.2.2**, tracking studies on birds from Bowland Fells SPA (Clewley *et al*., 2017) have found that birds from these colonies utilise terrestrial habitats almost exclusively, with a small proportion of trips to near-shore areas. It is therefore considered unlikely that birds from this SPA will occur at the windfarm site during the breeding season, although it is recognised that some birds may forage more widely.

802. The windfarm site is located within the mean maximum foraging range +1SD of a total of 168 lesser black-backed gull colonies (both SPA and non-SPA), with a total count of 72,320 individuals. For a review of these other SPAs and colonies see **Section 8.5.2.2**. It is therefore likely that birds present at the windfarm site during the breeding season may originate from a number of different colonies within the mean maximum +1SD foraging range for this species.

803. In addition, some of the lesser black-backed gulls recorded at the windfarm site during the breeding season would have been sub-adult birds. Based on review of raw survey data, 286 lesser black-backed gulls were recorded during the two-year baseline digital aerial surveys. Of these, 177 birds were able to be assigned to an age class, and of these, 126 birds (71.2% of those assigned to an age class) were classified as adults. It is therefore assumed, on a precautionary basis, that 71.2% of lesser black-backed gulls apportioned to the SPA during the full breeding season are breeding adult birds from Bowland Fells SPA.

804. The NatureScot apportioning tool (NatureScot, 2018) has been used to estimate the proportion of lesser black-backed gulls from each of the relevant SPAs present at the windfarm site during the breeding season (see **Appendix 12.1** of the ES for apportioning outputs). As there was uncertainty regarding the extent to which inland colonies may contribute to the population present in coastal/offshore areas (i.e. at the windfarm site), two estimates for breeding season apportioning have been calculated. The first has used all breeding colonies (both coastal and inland, including Bowland Fells SPA) within mean maximum +1SD of the windfarm site. The second approach has included only coastal sites (defined as being located within 5km of the coast), which excluded Bowland Fells SPA from the calculation (i.e. assumes that no birds from Bowland Fells SPA occur at the windfarm site during the breeding season).

805. When all lesser black-backed gull colonies are included, it is estimated that 39.69% of birds present at the windfarm site are apportioned to Bowland Fells SPA during the breeding season. If only coastal sites are included, no birds (i.e. 0.00%) are apportioned to the SPA.

806. During the pre- and post-breeding periods, breeding lesser black-backed gulls from the Bowland Fells SPA migrate through UK waters, whilst some birds remain in the UK during the winter. The relevant reference population is considered to be the UK Western Waters BDMPS. This consists of 163,304 individuals during autumn migration (September to October), 41,159 individuals during winter (November to February) and 163,304 individuals during spring migration (March) (Furness, 2015).

807. Furness (2015) estimated that 50% of the Bowland Fells SPA breeding adults were present within the UK Western Waters BDMPS during the autumn and spring migration periods, representing 4,575 birds. During the winter period 20% of the population is estimated to be present, which is 1,830 birds. This represents 2.80% of the BDMPS population for the autumn and spring periods, and 4.45% during winter. During autumn migration, winter, and spring migration, 2.80%, 4.45%, and 2.80% of impacts are considered to affect birds from the SPA respectively (Furness, 2015).

## Potential effects on the qualifying feature from the Project-alone

808. The lesser black-backed gull qualifying feature of the Bowland Fells SPA has been screened into the assessment due to the potential risk of collision.

### Operation and maintenance phase collision risk

809. Information on collision risk for breeding adult lesser black-backed gulls belonging to the Bowland Fells SPA population is presented in **Table 8.40**. Collision estimates, calculated using Option 2 of the sCRM (McGregor *et al.*, 2018), are presented by biological season. A summary of the annual outputs and the corresponding increase in the annual baseline mortality rate is also presented. Parameters used in the sCRM, together with the avoidance rate applied were agreed with Natural England during the ETG process and are described in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES.

810. If all lesser black-backed gull colonies are included in the breeding season apportioning, and based on the mean collision rates, the annual total of breeding adult lesser black backed gulls from the Bowland Fells SPA at risk of collision due to the Project is 0.62. This would increase the existing mortality of the SPA breeding population by 0.02%. If only coastal sites are used for apportioning, annual mortality would be 0.05, and increase of 0.00%.

*Table 8.40 Predicted seasonal and annual collision mortality (Stochastic model Option 2, avoidance rate 0.994 (±0.0004)) for breeding adult lesser black-backed gulls at the windfarm site, apportioned to Bowland Fells SPA, with corresponding increases to baseline mortality of the population*

| Period | Breeding Season (all sites apportioned) Apr-Aug | Breeding Season (only coastal sites apportioned) Apr-Aug | Autumn Migration Sep-Oct | Non-breeding/winter Nov-Feb | Spring Migration Mar | Annual (all sites apportioned) Jan-Dec | Annual (only coastal sites apportioned) Jan-Dec |
|---|---|---|---|---|---|---|---|
| Total collisions[1] (mean and 95% Cis) | 1.44 (0.00-4.53) | 1.44 (0.00-4.53) | 1.25 (0.00-5.63) | 0.15 (0.00-0.80) | 0.15 (0.00-0.94) | **2.98 (0.00-11.90)** | **2.98 (0.00-11.90)** |
| % apportioned to the SPA | 39.69% | 0.00% | 2.80% | 4.45% | 2.80% | **-** | **-** |
| Total SPA collisions (mean and 95% Cis) | 0.57 (0.00-1.80) | 0.00 (0.00-0.00) | 0.04 (0.00-0.16) | 0.01 (0.00-0.04) | 0.00 (0.00-0.03) | **0.62 (0.00-2.02)** | **0.05 (0.00-0.22)** |
| Mortality increase[2] (mean and 95% Cis) | 0.02% (0.00-0.05%) | 0.00% (0.00-0.00%) | 0.00% (0.00-0.00%)_ | 0.00% (0.00-0.00%) | 0.00% (0.00-0.00%) | **0.02% (0.00-0.06%)** | **0.00% (0.00-0.01%)** |

[1] Breeding season collision values reduced to 71.9% of modelled value to reflect proportion of adult birds recorded during site surveys.
[2] Assuming predicted annual SPA mortality of 3,298 birds (28,682 x 0.115)

811. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that no detectable changes in mortality rates would occur in this population from the mean monthly collision estimates for the Project. This applies for both apportioning approaches considered, and for the upper 95% CI estimate.

812. **For the Project-alone, it is concluded that predicted lesser black-backed gull mortality due to collision at the windfarm site would not adversely affect the integrity of the Bowland Fells SPA.**

813. The confidence in the assessment is high. The evidence used to define the CRM input parameters presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES is of high applicability and quality. Whilst there is uncertainty around some of the input parameters (e.g. avoidance rate), the rates selected are considered to be sufficiently precautionary based on expert opinion to provide confidence that collision rates are not underestimated.

### Potential effects on the qualifying feature in-combination with other projects

814. As the Project would have no measurable effect on lesser black-backed gull populations from the Bowland Fells SPA (irrespective of apportioning approach), there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of Bowland Fells SPA, when assessed in-combination with other plans or projects.**

## 8.12 Mersey Estuary SPA and Ramsar

815. Mersey Estuary SPA and Ramsar site is located approximately 53km from the windfarm site.

### 8.12.1 Description of designation

816. Mersey Estuary SPA encompasses all or parts of Mersey Estuary SSSI and New Ferry SSSI. It is a large, sheltered estuary which comprises large areas of saltmarsh and extensive intertidal sand and mudflats, with limited areas of brackish marsh, rocky shoreline and boulder clay cliffs, within a rural and industrial environment. The intertidal flats and saltmarshes provide feeding and roosting sites for large and internationally important populations of waterfowl. During the winter, the site is of major importance for duck and waders. The site is also important during spring and autumn migration periods, particularly for wader populations moving along the west coast of Britain.

### 8.12.2 Conservation objectives

817. The SPA's conservation objectives are to ensure that the integrity of the site is maintained or restored as appropriate, subject to natural change, and ensure that the site contributes to achieving the aims of the Wild Birds Directive, by maintaining or restoring:

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

### 8.12.3 Assessment

818. The qualifying features of Mersey Estuary SPA and Ramsar site screened into the Appropriate Assessment are listed in **Table 5.2**.

#### 8.12.3.1 Migratory waterbird qualifying features

##### Status

819. The status of each migratory waterbird qualifying feature screened into the Appropriate Assessment for this site is presented in **Table 8.41**. This consists of the site population at designation, national population in 2012 (Wright *et al*., 2012) and latest five-year peak mean WeBS count (Frost *et al.,* 2021).

## Functional linkage and seasonal apportionment of potential effects

820. All qualifying features of this designated site have been screened into the Appropriate Assessment due to the risk of potential impacts occurring during the spring and autumn migration seasons. Other than grey plover and dunlin (which were both recorded on one occasion in the first year of surveys only), the qualifying features were not recorded in the aerial survey study area during the baseline surveys undertaken at the windfarm site. However, it is recognised that the qualifying features may pass through the habitat in the windfarm site during migration periods and may have been missed by the surveys.

821. The apportioning of impacts to this designated site was calculated for each qualifying feature by dividing the number of collisions calculated at the national level by the proportion of the national population that were members of the designated site population at citation. The numbers used to define the national populations were the Great Britain populations presented in Wright *et al.,* (2012). Designated site populations were obtained from the SPA citation, or the Ramsar site population if the SPA citation did not include a population estimate.

## Potential effects on the qualifying features

822. The qualifying features of this designated site have been screened into the Appropriate Assessment due to the potential risk of collision.

823. The magnitudes of potential collision impacts have been investigated using the SOSSMAT tool (Wright *et al*., 2012).

## Potential effects on the qualifying feature from the Project-alone

### Collision risk

824. The estimated annual collision risk for of each qualifying feature from this designated site, along with the conclusion of the assessment based on this annual collision rate, is presented in **Table 8.41**. An avoidance rate of 0.980 has been assumed for all species.

825. The number of annual collisions predicted for all qualifying features is very low. It is expected that the increases to existing mortality rates for each qualifying feature due to this impact would be undetectable within the site population. Such impacts would consequently not result in any measurable effect.

826. **It is concluded that the predicted mortality of all qualifying features due to collision at the Project windfarm site would not adversely affect the integrity of the Mersey Estuary SPA and Ramsar site.**

827. Whilst extensive information exists on the responses of waterbirds to onshore OWFs, there is substantial uncertainty regarding waterbird movements at sea. The confidence level assigned to this section of the assessment is therefore medium. However, since such low levels of collision are predicted, an adverse effect on the integrity of the site is considered highly unlikely, even in the unlikely event that impacts have been underestimated.

## Potential effects on the qualifying feature in-combination with other projects

### Collision risk

828. The migration corridors identified by Wright *et al.,* (2012) indicated that migration activity of all qualifying features from this designated site would be widespread across UK waters. Similarly low numbers of birds, and hence collisions, are therefore expected at other OWFs in UK waters. The total collision mortality of non-breeding waterbirds at all UK OWFs is still likely to be small in the context of their respective national populations, and the number of collisions associated with this designated site will be smaller still. It is expected that the increases to existing mortality rates for each qualifying feature due to this impact would be undetectable within the site population. Such impacts would consequently not result in any measurable effect.

829. **It is concluded that predicted mortality of all qualifying features due to collision at the Project windfarm site, in-combination with other projects, would not adversely affect the integrity of the Mersey Estuary SPA and Ramsar site**.

*Table 8.41 Information to support the Appropriate Assessment for Mersey Estuary SPA and Ramsar site (migratory waterbird qualifying features)*

| Qualifying feature | GB population (Wright *et al.*, 2012) | SPA population (citation/ standard data form) | Ramsar site population (citation) | Five-year peak mean 2015/16 – 2019/20 | Apportioning rate | Unapportioned predicted mean annual collisions (avoidance rate 0.980) | Annual collisions apportioned to SPA | Conclusion of adverse effect on site integrity |
|---|---|---|---|---|---|---|---|---|
| Great crested grebe | 19,000 | 136 | - | 51 | 0.7% | 0.01 | 0.00 | No. Numbers of collisions so small that effects on population would be negligible. It would not be possible for impacts of this magnitude to have an effect at the site level given the background populations |
| Shelduck | 61,000 | 6,476 | 12,676 | 10,697 | 10.6% | 0.03 | 0.00 | |
| Wigeon | 440,000 | 11,886 | 8,268 | 1,701 | 2.7% | 0.23 | 0.01 | |
| Teal | 210,000 | 11,723 | 10,613 | 2,792 | 5.6% | 0.08 | 0.00 | |
| Pintail | 29,000 | 1,169 | 565 | 147 | 4.0% | 0.01 | 0.00 | |
| Ringed plover | 34,000 | 505 | 429 | 1,626 | 1.5% | 0.02 | 0.00 | |
| Golden plover | 400,000 | 3,040 | - | 1,712 | 0.8% | 0.26 | 0.00 | |
| Grey plover | 43,000 | 1,010 | - | 416 | 2.3% | 0.03 | 0.00 | |
| Lapwing | 620,000 | 10,544 | - | 8,507 | 1.7% | 0.43 | 0.01 | |
| Curlew | 140,000 | 1,300 | 2,010 | 1,502 | 0.9% | 0.10 | 0.00 | |
| Redshank (p) | 120,000 | 4,513 | 6,651 | 4,189 | 3.8% | 0.08 | 0.00 | |
| Redshank (w) | 120,000 | 4,993 | - | 4,189 | 4.2% | 0.08 | 0.00 | |
| Black-tailed godwit | 43,000 | 976 | 2,011 | 2,912 | 2.3% | 0.03 | 0.00 | |
| Dunlin | 350,000 | 48,789 | 48,364 | 51,456 | 13.9% | 0.38 | 0.05 | |
| Waterbird assemblage | - | 104,599 | - | - | - | - | - | No. Based on the small number of collisions predicted for named qualifying features, no adverse effect on integrity is anticipated |

## 8.13 Ynys Seiriol/Puffin Island SPA

830. Ynys Seiriol/Puffin Island SPA is located approximately 55km from the windfarm site.

### 8.13.1 Description of designation

831. Ynys Seiriol/Puffin Island SPA covers an area of 31.6ha and is just off the eastern tip of the Isle of Anglesey in North Wales. It is a Carboniferous limestone block rising to 55m with steep cliffs on all sides. The site is of European importance for its breeding population of cormorant, which feed in the surrounding waters outside the SPA. The island is also of interest for other nesting seabirds breeding both on its sea-cliffs and open grassland areas.

### 8.13.2 Conservation objectives

832. As the SPA has only one qualifying feature (cormorant), the overarching conservation objective for this feature is 'to achieve and maintain favourable conservation status, in which all the following conditions are satisfied:

- The number of breeding cormorants within the SPA are stable or increasing
- The abundance and distribution of prey species are sufficient to support this number of breeding pairs and for successful breeding
- The management and control of activities or operations likely to adversely affect the cormorants, is appropriate for maintaining the feature in favourable condition and is secure in the long term'

### 8.13.3 Assessment

833. One qualifying feature of Ynys Seiriol/Puffin Island SPA has been screened into the Appropriate Assessment (**Table 5.2**): breeding cormorant.

#### 8.13.3.1 Cormorant

#### Status

834. The Ynys Seiriol/Puffin Island SPA breeding cormorant population was cited as 556 pairs, or 1,112 breeding adults for the period 1996-2000 (Furness, 2015). Furness (2015) gave a breeding population of 448 pairs, or 896 breeding adults in 2013. The most recent count was 402 pairs (AON), or 804 breeding adults in 2021 (JNCC, 2023a); this is used as the reference population for the assessment.

835. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.132 (1 – 0.868; Horswill

and Robinson 2015), the expected annual mortality from the SPA population would be 106 breeding adults.

### Functional linkage and seasonal apportionment of potential effects

836. The mean maximum foraging range of cormorant is 25.6km (±8.3km) and the maximum foraging range is 35km (Woodward *et al*., 2019). The Project is located approximately 55km from Ynys Seiriol/Puffin Island SPA, which means the Project is beyond the maximum foraging range of cormorants from the SPA. No impacts during the season from the Project are therefore apportioned to cormorants breeding at this SPA.

837. Outside the breeding season, breeding cormorants from the SPA are assumed to range widely and to mix with birds of all ages from breeding colonies in the UK and beyond. However, as no cormorants were recorded within the windfarm site or 2km buffer, it can be concluded that no birds from Ynys Seiriol/Puffin Island SPA are likely to occur at the windfarm site.

### Potential effects on the qualifying feature from the Project-alone

838. No effects on cormorants from Ynys Seiriol/Puffin Island SPA are predicted. **Therefore, it is concluded that there would be no adverse effect on the integrity of the Ynys Seiriol/Puffin Island SPA for the Project-alone**.

### Potential effects on the qualifying feature in-combination with other projects

839. As the Project would have no measurable effect on cormorant populations from the Ynys Seiriol/Puffin Island SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of Ynys Seiriol/Puffin Island SPA, when assessed in-combination with other plans or projects**.

# 8.14 Leighton Moss Ramsar

840. Leighton Moss Ramsar site is located approximately 59km from the windfarm site.

## 8.14.1 Description of designation

841. Originally wet peatland, the area was drained and cultivated in the 19th century before being re-flooded in 1917. The base-rich water has produced a rich vegetation consisting of large areas of sedge and reedbeds, fen communities, wet Salix scrub, and woodland. The site supports nationally important populations of breeding and wintering waterbirds.

## 8.14.2 Assessment

### 8.14.2.1 Migratory waterbird qualifying features

#### Status

842. Leighton Moss Ramsar site supports nationally important populations of breeding birds including bittern (5-6 pairs), marsh harrier (1-2 pairs) and bearded tit (20-30 pairs; 1990 counts). Average peak counts for the five winters 1987/88 to 1991/92 included nationally important numbers of teal (960) and shoveler (179).

#### Functional linkage and seasonal apportionment of potential effects

843. All qualifying features of this designated site identified above, except for bearded tit, have been screened into the Appropriate Assessment due to the risk of potential impacts occurring during the spring and autumn migration seasons. Bearded tit is not assessed as this species is predominantly sedentary, and therefore extremely unlikely to occur at the windfarm site. None of the remaining qualifying features were recorded in the aerial survey study area during the baseline surveys undertaken at the windfarm site. However, it is recognised that these species may pass through the habitat in the windfarm site during migration periods and may have been missed by the surveys.

844. The apportioning of impacts to this designated site was calculated for each assessed qualifying feature by dividing the number of collisions calculated at the national level by the proportion of the national population that were members of the designated site population at citation. The numbers used to define the national populations were the Great Britain populations presented in Wright *et al.,* (2012). Designated site populations were obtained from the Ramsar site population.

### Potential effects on the qualifying feature from the Project-alone

845. The qualifying features of this designated site have been screened into the Appropriate Assessment due to the potential risk of collision.

846. The magnitudes of potential collision impacts have been investigated using the SOSSMAT tool (Wright *et al*., 2012).

#### Collision risk

847. The estimated annual collision risk for each qualifying feature from this designated site, along with the conclusion of the assessment based on this annual collision rate, is presented in **Table 8.42**. An avoidance rate of 0.980 has been assumed for all species.

848. The number of annual collisions predicted for all qualifying features is very low. It is expected that the increases to existing mortality rates for each qualifying feature due to this impact would be undetectable within the site population. Such impacts would consequently not result in any measurable effect.

849. **It is concluded that the predicted mortality of all qualifying features due to collision at the Project windfarm site would not adversely affect the integrity of the Leighton Moss Ramsar site**.

850. Whilst extensive information exists on the responses of waterbirds to onshore OWFs, there is substantial uncertainty regarding waterbird movements at sea. The confidence level assigned to this section of the assessment is therefore medium. However, since such low levels of collision are predicted, an adverse effect on the integrity of the site is considered highly unlikely, even in the unlikely event that impacts have been underestimated.

### Potential effects on the qualifying feature in-combination with other projects

#### Collision risk

851. The migration corridors identified by Wright *et al.,* (2012) indicate that migration activity of all qualifying features from this designated site is widespread across UK waters. Similarly low numbers of birds, and hence collisions, are therefore expected at other OWFs in UK waters. The total collision mortality of non-breeding waterbirds at all UK OWFs is still likely to be small in the context of their respective national populations, and the number of collisions associated with this designated site will be smaller still. It is expected that the increases to existing mortality rates for each qualifying feature due to this impact would be undetectable within the site population. Such impacts would consequently not result in any measurable effect.

852. **It is concluded that predicted mortality of all qualifying features due to collision at the Project windfarm site, in-combination with other projects,**

**would not adversely affect the integrity of the Leighton Moss Ramsar site.**

*Table 8.42 Information to support the Appropriate Assessment for Leighton Moss Ramsar site (migratory waterbird qualifying features)*

| Qualifying feature | GB population (Wright *et al.*, 2012) | Ramsar site population (citation) | Apportioning rate | Unapportioned predicted mean annual collisions (avoidance rate 0.980) | Annual collisions apportioned to SPA | Conclusion of adverse effect on site integrity |
|---|---|---|---|---|---|---|
| Bittern | 600 | 12 | 2.0% | 0.26 | 0.01 | No adverse effect on site integrity. Numbers of collisions so small that effects on population would be negligible. It would not be possible for impacts of this magnitude to have an effect at the site level given the background populations. |
| Marsh harrier | 402 | 4 | 1.0% | 0.00 | 0.00 | |
| Teal | 210,000 | 960 | 0.5% | 4.20 | 0.02 | |
| Shoveler | 18,000 | 179 | 1.0% | 0.41 | 0.00 | |

## 8.15 Traeth Lafan/Lavan Sands, Conway Bay SPA

853. Traeth Lafan/Lavan Sands, Conway Bay SPA is located approximately 59km from the windfarm site.

### 8.15.1 Description of designation

854. Traeth Lafan/Lavan Sands SPA is situated in Conwy Bay between Bangor and Llanfairfechan in north-west Wales. This large area of intertidal sand- and mud-flats lies at the eastern edge of the Menai Strait. The area has a range of exposures and a diversity of conditions, enhanced by freshwater streams that flow across the flats. The site is of importance for wintering waterbirds, especially oystercatcher and curlew. In conditions of severe winter weather, Traeth Lafan acts as a refuge area for oystercatchers displaced from the Dee Estuary. The site is also an important moulting roost for great crested grebe in late summer/early autumn.

### 8.15.2 Conservation objectives

855. The conservation objectives for this SPA specifically relate to oystercatcher:

- The 5 year mean peak of the number of wintering oystercatchers is at least 4,000.
- The abundance and distribution of cockles of 15mm or larger and other suitable food are maintained at levels sufficient to support the population with a 5 year mean peak of 4,000 individuals
- Oystercatchers are not disturbed in ways that prevent them spending enough time feeding for survival
- Roost sites, including high tide roost sites, remain suitable for oystercatchers to roost undisturbed
- The management and control of activities or operations likely to adversely affect the oystercatchers, is appropriate for maintaining the feature in favourable condition and is secure in the long term

### 8.15.3 Assessment

#### 8.15.3.1 Migratory waterbird qualifying features

##### Status

856. The status of each migratory waterbird qualifying feature screened into the Appropriate Assessment for this site is presented in **Table 8.43**. This consists of the site population at designation, national population in 2012 (Wright *et al*., 2012) and latest five-year peak mean WeBS count (Frost *et al*., 2021).

### Functional linkage and seasonal apportionment of potential effects

857. All qualifying features of this designated site have been screened into the Appropriate Assessment due to the risk of potential impacts occurring during the spring and autumn migration seasons. Other than grey plover and dunlin (which were both recorded on one occasion in the first year of surveys only), the qualifying features were not recorded in the aerial survey study area during the baseline surveys undertaken at the windfarm site. However, it is recognised that the qualifying features may pass through the habitat in the windfarm site during migration periods and may have been missed by the surveys.

858. The apportioning of impacts to this designated site was calculated for each qualifying feature by dividing the number of collisions calculated at the national level by the proportion of the national population that were members of the designated site population at citation. The numbers used to define the national populations were the Great Britain populations presented in Wright *et al.,* (2012). Designated site populations were obtained from the SPA citation, or the Ramsar site population if the SPA citation did not include a population estimate.

### Potential effects on the qualifying features

859. The qualifying features of this designated site have been screened into the Appropriate Assessment due to the potential risk of collision.

860. The magnitudes of potential collision impacts have been investigated using the SOSSMAT tool (Wright *et al*., 2012).

### Potential effects on the qualifying feature from the Project-alone

#### Collision risk

861. The estimated annual collision risk for of each qualifying feature from this designated site, along with the conclusion of the assessment based on this annual collision rate, is presented in **Table 8.43**. An avoidance rate of 0.980 has been assumed for all species.

862. The number of annual collisions predicted for all qualifying features is very low. It is expected that the increases to existing mortality rates for each qualifying feature due to this impact would be undetectable within the site population. Such impacts would consequently not result in any measurable effect.

863. **It is concluded that the predicted mortality of all qualifying features due to collision at the Project windfarm site would not adversely affect the integrity of the Traeth Lafan/Lavan Sands, Conway Bay SPA.**

864. Whilst extensive information exists on the responses of waterbirds to onshore OWFs, there is substantial uncertainty regarding waterbird movements at sea. The confidence level assigned to this section of the assessment is therefore medium. However, since such low levels of collision are predicted, an adverse effect on the integrity of the site is considered highly unlikely, even in the unlikely event that impacts have been underestimated.

## Potential effects on the qualifying feature in-combination with other projects

### Collision risk

865. The migration corridors identified by Wright *et al.,* (2012) indicate that migration activity of all qualifying features from this designated site is widespread across UK waters. Similarly low numbers of birds, and hence collisions, are therefore expected at other OWFs in UK waters. The total collision mortality of non-breeding waterbirds at all UK OWFs is still likely to be small in the context of their respective national populations, and the number of collisions associated with this designated site will be smaller still. It is expected that the increases to existing mortality rates for each qualifying feature due to this impact would be undetectable within the site population. Such impacts would consequently not result in any measurable effect.

866. **It is concluded that predicted mortality of all qualifying features due to collision at the Project windfarm site, in-combination with other projects, would not adversely affect the integrity of the Traeth Lafan/Lavan Sands, Conway Bay SPA**.

*Table 8.43 Information to support the Appropriate Assessment for Traeth Lafan/Lavan Sands, Conway Bay SPA (migratory waterbird qualifying features)*

| Qualifying feature | GB population (Wright *et al*., 2012) | SPA population (citation/standard data form) | Five-year peak mean 2015/16 – 2019/20 | Apportioning rate | Unapportioned predicted mean annual collisions (avoidance rate 0.980) | Annual collisions apportioned to SPA | Conclusion of adverse effect on site integrity |
|---|---|---|---|---|---|---|---|
| Great crested grebe | 19,000 | 500 | 158 | 2.6% | 0.01 | 0.00 | No adverse effect on site integrity. Numbers of collisions so small that effects on population would be negligible. It would not be possible for impacts of this magnitude to have an effect at the site level given the background populations |
| Oystercatcher | 320,000 | 5,500 | 5,789 | 1.7% | 0.23 | 0.00 | |
| Red-breasted merganser | 8,400 | 120 | 58 | 1.4% | 0.00 | 0.00 | |
| Curlew | 140,000 | 1,500 | 1,916 | 1.1% | 0.10 | 0.00 | |
| Redshank | 120,000 | 1,200 | 1,361 | 1.0% | 0.08 | 0.00 | |

# 8.16 Solway Firth SPA

867. Solway Firth SPA is located approximately 76km from the windfarm site.

## 8.16.1 Description of designation

868. The Solway Firth SPA is a large estuarine/marine site on west coast of Great Britain. The SPA includes the classified Upper Solway Flats and Marshes SPA with extensive areas of intertidal mudflats, fringing saltmarshes and grazing marshes. The offshore sediments of the marine extension are substantially sand, associated with mud and gravel towards the edges of the firth, especially in the smaller tributary estuaries. The series of sandbanks north east of the Isle of Man is the result of strong currents and an abundant supply of sand. The inner firth is shallow, as is Wigtown Bay, but further west towards the north-eastern Irish Sea the water deepens steadily to over 40m.

## 8.16.2 Conservation objectives

869. The conservation of the SPA are:

- To ensure that the qualifying features of Solway Firth SPA are in favourable condition and make an appropriate contribution to achieving Favourable Conservation Status
- To ensure that the integrity of Solway Firth SPA is maintained or restored as appropriate, in the context of environmental changes by meeting objectives 2a, 2b and 2c for each qualifying feature:
  - 2a. The populations of the qualifying features are viable components of the site
  - 2b. The distributions of the qualifying features throughout the site are maintained, or where appropriate, restored by avoiding significant disturbance of the species
  - 2c. The supporting habitats and processes relevant to the qualifying features and their prey/food resources are maintained or where appropriate, restored

## 8.16.3 Assessment

870. The qualifying features of Solway Firth SPA screened into the Appropriate Assessment are listed in **Table 5.2**.

#### 8.16.3.1 Migratory waterbird qualifying features

##### Status

871. The status of each migratory waterbird qualifying feature screened into the Appropriate Assessment for this site is presented in **Table 8.44**. This consists of the site population at designation, national population in 2012 (Wright *et al*., 2012) and latest five-year peak mean WeBS count (Frost *et al*., 2021).

##### Functional linkage and seasonal apportionment of potential effects

872. All qualifying features of this designated site have been screened into the Appropriate Assessment due to the risk of potential impacts occurring during the spring and autumn migration seasons. Other than grey plover and dunlin (which were both recorded on one occasion in the first year of surveys only), the qualifying features were not recorded in the aerial survey study area during the baseline surveys undertaken at the windfarm site. However, it is recognised that the qualifying features may pass through the habitat in the windfarm site during migration periods and may have been missed by the surveys.

873. The apportioning of impacts to this designated site was calculated for each qualifying feature by dividing the number of collisions calculated at the national level by the proportion of the national population that were members of the designated site population at citation. The numbers used to define the national populations were the Great Britain populations presented in Wright *et al.,* (2012). Designated site populations were obtained from the SPA citation.

##### Potential effects on the qualifying features

874. The qualifying features of this designated site have been screened into the Appropriate Assessment due to the potential risk of collision.

875. The magnitudes of potential collision impacts have been investigated using the SOSSMAT tool (Wright *et al*., 2012).

##### Potential effects on the qualifying feature from the Project-alone

**Collision risk**

876. The estimated annual collision risk for of each qualifying feature from this designated site, along with the conclusion of the assessment based on this annual collision rate, is presented in **Table 8.44**. An avoidance rate of 0.980 has been assumed for all species.

877. The number of annual collisions predicted for all qualifying features is very low. It is expected that the increases to existing mortality rates for each qualifying feature due to this impact would be undetectable within the site

population. Such impacts would consequently not result in any measurable effect.

878. **It is concluded that the predicted mortality of all qualifying features due to collision at the Project windfarm site would not adversely affect the integrity of the Solway Firth SPA.**

879. Whilst extensive information exists on the responses of waterbirds to onshore OWFs, there is substantial uncertainty regarding waterbird movements at sea. The confidence level assigned to this section of the assessment is therefore medium. However, since such low levels of collision are predicted, an adverse effect on the integrity of the site is considered highly unlikely, even in the unlikely event that impacts have been underestimated.

## Potential effects on the qualifying feature in-combination with other projects

### Collision risk

880. The migration corridors identified by Wright *et al.,* (2012) indicate that migration activity of all qualifying features from this designated site is widespread across UK waters. Similarly low numbers of birds, and hence collisions, are therefore expected at other OWFs in UK waters. The total collision mortality of non-breeding waterbirds at all UK OWFs is still likely to be small in the context of their respective national populations, and the number of collisions associated with this designated site will be smaller still. It is expected that the increases to existing mortality rates for each qualifying feature due to this impact would be undetectable within the site population. Such impacts would consequently not result in any measurable effect.

881. **It is concluded that predicted mortality of all qualifying features due to collision at the Project windfarm site, in-combination with other projects, would not adversely affect the integrity of the Solway Firth SPA**.

*Table 8.44 Information to support the Appropriate Assessment for Solway Firth SPA (migratory waterbird qualifying features)*

| Qualifying feature | GB population (Wright *et al.*, 2012) | SPA population (citation/standard data form) | Five-year peak mean 2015/16 – 2019/20* | Apportioning rate | Unapportioned predicted mean annual collisions (avoidance rate 0.980) | Annual collisions apportioned to SPA | Conclusion of adverse effect on site integrity |
|---|---|---|---|---|---|---|---|
| Red-throated diver | 17,000 | 521 | 5 | 3.1% | 0.00 | 0.00 | No adverse effect on site integrity. Numbers of collisions so small that effects on population would be negligible. It would not be possible for impacts of this magnitude to have an effect at the site level given the background populations |
| Cormorant* | 35,000 | 581 | 237 | 1.7% | 0.00 | 0.00 | |
| Whooper swan | 11,000 | 250 | 303 | 2.3% | 0.03 | 0.00 | |
| Pink-footed goose | 360,000 | 14,900 | 11,346 | 4.1% | 0.01 | 0.00 | |
| Barnacle goose | 33,000 | 12,300 | 40,958 | 37.3% | 0.00 | 0.00 | |
| Shelduck* | 61,000 | 1,600 | 2,772 | 2.6% | 0.03 | 0.00 | |
| Teal* | 210,000 | 1,400 | 3,357 | 0.7% | 0.08 | 0.00 | |
| Pintail* | 29,000 | 1,400 | 3,042 | 4.8% | 0.01 | 0.00 | |
| Shoveler* | 18,000 | 120 | 197 | 0.7% | 0.01 | 0.00 | |
| Scaup* | 5,200 | 2,300 | 596 | 44.2% | 0.00 | 0.00 | |
| Common scoter* | 100,000 | 1,588 | 271 | 1.6% | 0.00 | 0.00 | |
| Goldeneye* | 20,000 | 300 | 54 | 1.5% | 0.01 | 0.00 | |
| Goosander* | 12,000 | 146 | 89 | 1.2% | 0.00 | 0.00 | |
| Oystercatcher | 320,000 | 33,850 | 26,672 | 10.6% | 0.23 | 0.02 | |
| Ringed plover | 34,000 | 981 | 964 | 2.9% | 0.02 | 0.00 | |
| Golden plover | 400,000 | 3,380 | 5,395 | 0.8% | 0.26 | 0.00 | |
| Grey plover* | 43,000 | 720 | 292 | 1.7% | 0.03 | 0.00 | |
| Lapwing* | 620,000 | 5,037 | 4,224 | 0.8% | 0.11 | 0.00 | |
| Knot | 320,000 | 15,300 | 8,227 | 4.8% | 0.43 | 0.02 | |
| Sanderling* | 16,000 | 260 | 455 | 1.6% | 0.01 | 0.00 | |
| Bar-tailed godwit | 38,000 | 4,800 | 552 | 12.6% | 0.04 | 0.01 | |
| Curlew | 140,000 | 6,700 | 2,183 | 4.8% | 0.10 | 0.00 | |
| Redshank* | 120,000 | 2,100 | 2,836 | 1.8% | 0.08 | 0.00 | |
| Turnstone* | 48,000 | 600 | 222 | 1.3% | 0.03 | 0.00 | |
| Dunlin* | 350,000 | 11,900 | 17,418 | 3.4% | 0.38 | 0.01 | |
| Black-headed gull* | 2,200,000 | 13,732 | 3,436 | 0.6% | 0.96 | 0.01 | |
| Common gull* | 700,000 | 12,486 | 2,158 | 1.8% | 0.64 | 0.01 | |
| Herring gull* | 730,000 | 3,034 | 1,898 | 0.4% | 0.79 | 0.00 | |
| Waterbird assemblage | - | 122,200 | - | - | - | - | No adverse effect on site integrity. Based on the small number of collisions predicted for named qualifying features, no adverse effect on integrity is anticipated |

[1]Solway Estuary WeBS peak mean; * named qualifying feature of the waterbird assemblage

## 8.17 Migneint-Arenig-Dduallt SPA

882. Migneint-Arenig-Dduallt SPA is located approximately 79km from the windfarm site.

### 8.17.1 Description of designation

883. Migneint-Arenig-Dduallt is a large upland site that stretches between Ysbyty Ifan and Penmachno in the north down to Rhydymain in the south, and from Trawsfynnydd in the west to just east of Llyn Celyn, ranging in altitude from 300 m to 712 m. Habitats include blanket bog, dry heath, wet heath, lakes and woodland. The SPA is designated for its breeding populations of hen harrier, merlin and peregrine.

### 8.17.2 Conservation objectives

884. The conservation objectives for hen harrier are as follows:

- The size of the population is at least 8 breeding pairs and preferably increasing
- Hen Harrier nesting distribution within the site is maintained or expanded, so that breeding occurs in all appropriate habitats. There are appropriate and sufficient food sources for terns within access of the SPA
- Hen Harrier breeding success is at least one young fledged per nest
- There is sufficient nesting and roosting tall heather habitat to support the population in the long-term
- There is sufficient hunting habitat, often in mosaic and including areas of grassland, bogs, flushes, short heath and bracken with low trees/scrub present. There is an adequate supply of prey species in the form of small birds and small mammals to maintain successful breeding. Prey supply cannot be easily monitored or assessed but may be an important attribute, for research and study, if productivity is low
- All factors affecting the achievement of these conditions are under control

885. The conservation objectives for merlin are as follows:

- The size of the population is at least 9 breeding pairs and preferably increasing
- Merlin nesting distribution within the site is maintained or expanded, so that breeding occurs in all appropriate habitats. There are appropriate and sufficient food sources for terns within access of the SPA

- Merlin breeding success is at least one young fledged per nest when sample monitoring is carried out
- There is sufficient nesting and roosting tall heather, individual trees often with crows' nests and forestry edge habitat to support the population in the long-term
- There is sufficient hunting habitat, often in mosaic and including areas of grassland, bogs, flushes, short heath and bracken with low trees/scrub present. There is an adequate supply of prey species in the form of small birds (commonly meadow pipit and skylark) and large insects to maintain successful breeding. Prey supply cannot be easily monitored or assessed but may be an important attribute, for research and study, if productivity is low. All factors affecting the achievement of these conditions are under control
- All factors affecting the achievement of these conditions are under control

886. The conservation objectives for peregrine are as follows:

- The size of the population is at least 9 breeding pairs and preferably increasing
- Peregrine nesting distribution within the site is maintained or expanded, so that breeding occurs in all appropriate nest site
- Peregrine breeding success is at least one young fledged per nest when sample population monitoring is carried out
- There are sufficient cliff and crag with ledges suitable for nesting usually known traditional nest sites to support the population in the long-term
- There is a sufficient hunting habitat and prey. Prey supply cannot be easily monitored or assessed but may be an important attribute, for research and study, if peregrine productivity is low
- All factors affecting the achievement of these conditions are under control

## 8.17.3 Assessment

### 8.17.3.1 Hen harrier

#### Status

887. The Migneint-Arenig-Dduallt SPA breeding hen harrier population stood at 10 pairs in 2003, and 14 pairs in 2010 (Stroud *et al*., 2016).

### Functional linkage and seasonal apportionment of potential effects

888. Hen harrier has been screened into the Appropriate Assessment due to the risk of potential impacts occurring during the spring and autumn migration seasons. This species was not recorded in the aerial survey study area during the baseline surveys undertaken at the windfarm site. However, it is recognised that there is the potential that it may pass through the habitat in the windfarm site during migration periods, and may have been missed by the surveys.

889. The apportioning of impacts to this designated site was calculated by dividing the number of collisions calculated at the national level by the proportion of the national population that were members of the designated site population at citation. The numbers used to define the national populations were the Great Britain populations presented in Wright *et al.,* (2012), comprising 570 pairs or 1,140 individuals during the breeding season. Designated site populations were obtained from the SPA citation, comprising 10 pairs or 20 individuals. Accordingly, 1.8% of impacts to this species were apportioned to Migneint-Arenig-Dduallt SPA.

### Potential effects on the qualifying feature from the Project-alone

890. Hen harrier has been screened into the Appropriate Assessment due to the potential risk of collision. The magnitude of potential collision impacts has been investigated using the SOSSMAT tool (Wright *et al*., 2012).

891. The unapportioned annual collision risk calculated using the SOSSMAT tool (refer to **Appendix 12.1** of the ES), is estimated at less than 0.01 birds, assuming an avoidance rate of 0.980. Zero mortality is therefore apportioned to Migneint-Arenig-Dduallt SPA. Such impacts would consequently not result in any measurable effect.

892. **It is concluded that the predicted hen harrier mortality would not adversely affect the integrity of the Migneint-Arenig-Dduallt SPA.**

893. Whilst extensive information exists on the responses of raptors to onshore OWFs, there is substantial uncertainty regarding the effects on migratory bird movements at sea. The confidence level assigned to this section of the assessment is therefore medium. However, since such low levels of collision are predicted, an adverse effect on the integrity of the site is considered highly unlikely, even in the unlikely event that impacts have been underestimated.

### Potential effects on the qualifying feature in-combination with other projects

894. As no measurable effects are predicted as a result of the project alone, there would be no contribution to in-combination effects. **It is concluded that predicted hen harrier mortality due to collision at the windfarm site,**

**alone and in-combination with other projects, would not adversely affect the integrity of the Migneint-Arenig-Dduallt SPA**.

### 8.17.3.2 Merlin

#### Status

895. The Migneint-Arenig-Dduallt SPA breeding merlin stood at seven pairs at classification (2003) and seven pairs in 2008 (Stroud *et al.,* 2016).

#### Functional linkage and seasonal apportionment of potential effects

896. Merlin has been screened into the Appropriate Assessment due to the risk of potential impacts occurring during the spring and autumn migration seasons. This species was not recorded in the aerial survey study area during the baseline surveys undertaken at the windfarm site. However, it is recognised that there is the potential that it may pass through the habitat in the windfarm site during migration periods, and may have been missed by the surveys.

897. The apportioning of impacts to this designated site was calculated by dividing the number of collisions calculated at the national level by the proportion of the national population that were members of the designated site population at citation. The numbers used to define the national populations were the Great Britain populations presented in Wright *et al.,* (2012), comprising 1,330 pairs or 2,660 individuals during the breeding season. Designated site populations were obtained from the SPA citation, comprising seven pairs or 14 individuals. Accordingly, 0.5% of impacts to this species were apportioned to Migneint-Arenig-Dduallt SPA.

#### Potential effects on the qualifying feature from the Project-alone

898. Merlin has been screened into the Appropriate Assessment due to the potential risk of collision. The magnitude of potential collision impacts has been investigated using the SOSSMAT tool (Wright *et al*., 2012).

899. The unapportioned annual collision risk calculated using the SOSSMAT tool (refer to **Appendix 12.1** of the ES), is estimated at less than 0.01 birds, assuming an avoidance rate of 0.980. Zero mortality is therefore apportioned to Migneint-Arenig-Dduallt SPA. Such impacts would consequently not result in any measurable effect.

900. **It is concluded that the predicted merlin mortality would not adversely affect the integrity of the Migneint-Arenig-Dduallt SPA.**

901. Whilst extensive information exists on the responses of raptors to onshore OWFs, there is substantial uncertainty regarding the effects on migratory bird movements at sea. The confidence level assigned to this section of the assessment is therefore medium. However, since such low levels of collision

are predicted, an adverse effect on the integrity of the site is considered highly unlikely, even in the unlikely event that impacts have been underestimated.

### Potential effects on the qualifying feature in-combination with other projects

902. As no measurable effects are predicted as a result of the project alone, there would be no contribution to in-combination effects. **It is concluded that predicted merlin mortality due to collision at the windfarm site, alone and in-combination with other projects, would not adversely affect the integrity of the Migneint-Arenig-Dduallt SPA**.

#### 8.17.3.3 Peregrine

### Status

903. The Migneint-Arenig-Dduallt SPA breeding peregrine population stood at eight pairs in 2002 (Stroud *et al*., 2016).

### Functional linkage and seasonal apportionment of potential effects

904. Peregrine was screened into the Appropriate Assessment due to the risk of potential impacts occurring during the spring and autumn migration seasons. However, peregrine populations in the UK are largely sedentary (Cramp and Simmons, 1980), and therefore it is considered very unlikely that birds from the Migneint-Arenig-Dduallt SPA population would pass through the windfarm site. Accordingly, no impacts are apportioned to Migneint-Arenig-Dduallt SPA.

### Potential effects on the qualifying feature alone and in-combination with other projects

905. As no peregrines from the Migneint-Arenig-Dduallt SPA population are likely to occur at the windfarm site, there would be no measurable effect on this species, and there would be no contribution to any potential in-combination effects. **It is therefore concluded that there would be no adverse effect on the integrity of the Migneint-Arenig-Dduallt SPA, either alone or in-combination with other projects**.

## 8.18 Berwyn SPA

906. Berwyn SPA is located approximately 81km from the windfarm site.

### 8.18.1 Description of designation

907. Berwyn is a large area of upland moorland containing blanket bog, dry heath, transition mires and calcareous vegetation. It is considered the most important upland in Wales for breeding birds, supporting a wide range of species including internationally significant numbers of hen harrier, merlin, peregrine and red kite, as well as significant proportions of the Welsh populations of other species including short-eared owl, golden plover, red grouse and black grouse.

### 8.18.2 Conservation objectives

908. The conservation objectives for hen harrier are as follows:

- The size of the population must be being maintained at eleven breeding pairs or increased beyond this
- There will be sufficient appropriate habitat to support the population in the long-term including patches of tall heather available for nesting and roosting, areas grasslands, bracken of low trees/scrub for feeding with an adequate supply of prey species in the form of small birds and small mammals to maintain successful breeding
- Distribution of species within site is maintained
- Distribution and extent of habitats supporting the species is maintained
- Developments should not be permitted where they can be shown to have likely adverse impacts upon hen harrier
- Populations of legally controllable predator species, such as foxes and carrion crows, will not pose a threat to ground nesting birds
- Hunting territories will be managed by controlled grazing to improve structural diversity within the grasslands. This will increase seed production and maximise prey availability e.g. small passerines
- There will be no disturbance of any nest location
- Illegal human persecution of protected bird species should not occur
- All factors affecting the achievement of these conditions are under control

909. The conservation objectives for merlin are as follows:

- The size of the population must be being maintained at 13 breeding pairs or increased beyond this
- There will be sufficient appropriate habitat to support the population in the long-term including patches of tall heather available for nesting and roosting, areas grasslands, bracken of low trees/scrub for feeding with an adequate supply of prey species in the form of small birds and small mammals to maintain successful breeding
- Distribution of species within site is maintained
- Distribution and extent of habitats supporting the species is maintained
- Developments should not be permitted where they can be shown to have likely adverse impacts upon merlin
- Populations of legally controllable predator species, such as foxes and carrion crows, should not pose a threat to ground nesting birds
- Adjoining hunting territories will be managed by controlled grazing to improve structural diversity within the grasslands. This will increase seed production and maximise prey availability e.g. small passerines
- There will be no disturbance of any nest location
- Illegal human persecution of protected bird species should not occur
- All factors affecting the achievement of these conditions are under control

910. The conservation objectives for peregrine are as follows:

- The size of the population must be being maintained at 13 breeding pairs or increased beyond this
- Mountainous and moorland terrain with cliffs, crags and quarries for nesting and roosting plus grasslands, bracken of low trees/scrub for feeding with an adequate supply of prey species in the form of small birds and small mammals to maintain successful breeding
- The range of the population must not be contracting
- Distribution and extent of habitats supporting the species is maintained
- Developments should not be permitted where they can be shown to have likely adverse impacts upon peregrine
- Populations of legally controllable predator species, such as foxes and carrion crows, should not pose a threat to ground nesting birds

- Adjoining hunting territories will be managed by controlled grazing to improve structural diversity within the grasslands. This will increase seed production and maximise prey availability e.g. small passerines
- There will be no disturbance of any nest location
- Illegal human persecution of protected bird species should not occur
- All factors affecting the achievement of these conditions are under control

911. The conservation objectives for red kite are as follows:

- The size of the population must be being maintained at 2 breeding pairs or increased beyond this
- Sufficient broadleaf woodland required for nesting and roosting plus heath and rough grassland for feeding with an adequate supply of prey species in the form of carrion, small birds and small mammals to maintain successful breeding
- Developments should not be permitted where they can be shown to have likely adverse impacts upon red kite
- Adjoining hunting territories will be managed by controlled grazing to improve structural diversity within the grasslands. This will increase seed production and maximise prey availability e.g. small passerines
- There will be no disturbance of any nest location
- Illegal human persecution of protected bird species should not occur
- All factors affecting the achievement of these conditions are under control

### 8.18.3 Assessment

#### 8.18.3.1 Hen harrier

##### Status

912. The Berwyn SPA breeding hen harrier population stood at 14 pairs at classification (1998), and 20 pairs in 2010 (Stroud *et al*., 2016).

##### Functional linkage and seasonal apportionment of potential effects

913. Hen harrier has been screened into the Appropriate Assessment due to the risk of potential impacts occurring during the spring and autumn migration seasons. This species was not recorded in the aerial survey study area during the baseline surveys undertaken at the windfarm site. However, it is recognised that there is the potential that it may pass through the habitat in

the windfarm site during migration periods, and may have been missed by the surveys.

914. The apportioning of impacts to this designated site was calculated by dividing the number of collisions calculated at the national level by the proportion of the national population that were members of the designated site population at citation. The numbers used to define the national populations were the Great Britain populations presented in Wright *et al.,* (2012), comprising 570 pairs or 1,140 individuals during the breeding season. Designated site populations were obtained from the SPA citation, comprising 14 pairs or 28 individuals. Accordingly, 2.5% of impacts to this species were apportioned to Berwyn SPA.

### Potential effects on the qualifying feature from the Project-alone

915. Hen harrier has been screened into the Appropriate Assessment due to the potential risk of collision. The magnitude of potential collision impacts has been investigated using the SOSSMAT tool (Wright *et al*., 2012).

916. The unapportioned annual collision risk calculated using the SOSSMAT tool (refer to **Appendix 12.1** of the ES), is estimated at less than 0.01 birds, assuming an avoidance rate of 0.980. Zero mortality is therefore apportioned to Berwyn SPA. Such impacts would consequently not result in any measurable effect.

917. **It is concluded that the predicted hen harrier mortality would not adversely affect the integrity of the Berwyn SPA.**

918. Whilst extensive information exists on the responses of raptors to onshore OWFs, there is substantial uncertainty regarding the effects on migratory bird movements at sea. The confidence level assigned to this section of the assessment is therefore medium. However, since such low levels of collision are predicted, an adverse effect on the integrity of the site is considered highly unlikely, even in the unlikely event that impacts have been underestimated.

### Potential effects on the qualifying feature in-combination with other projects

919. As no measurable effects are predicted as a result of the project alone, there would be no contribution to in-combination effects. **It is concluded that predicted hen harrier mortality due to collision at the windfarm site, alone and in-combination with other projects, would not adversely affect the integrity of the Berwyn SPA**.

#### 8.18.3.2 Merlin

##### Status

920. The Berwyn SPA breeding merlin population stood at 14 pairs at classification (1998) and two pairs in 2011 (Stroud *et al*., 2016).

##### Functional linkage and seasonal apportionment of potential effects

921. Merlin has been screened into the Appropriate Assessment due to the risk of potential impacts occurring during the spring and autumn migration seasons. This species was not recorded in the aerial survey study area during the baseline surveys undertaken at the windfarm site. However, it is recognised that there is the potential that merlin may pass through the habitat in the windfarm site during migration periods, and may have been missed by the surveys.

922. The apportioning of impacts to this designated site was calculated by dividing the number of collisions calculated at the national level by the proportion of the national population that were members of the designated site population at citation. The numbers used to define the national populations were the Great Britain populations presented in Wright *et al.,* (2012), comprising 1,330 pairs or 2,660 individuals during the breeding season. Designated site populations were obtained from the SPA citation, comprising 14 pairs or 28 individuals. Accordingly, 1.1% of impacts to this species were apportioned to Berwyn SPA.

##### Potential effects on the qualifying feature from the Project-alone

923. Merlin has been screened into the Appropriate Assessment due to the potential risk of collision. The magnitude of potential collision impacts has been investigated using the SOSSMAT tool (Wright *et al*., 2012).

924. The unapportioned annual collision risk calculated using the SOSSMAT tool (refer to **Appendix 12.1** of the ES), is estimated at less than 0.01 birds, assuming an avoidance rate of 0.980. Zero mortality is therefore apportioned to Berwyn SPA. Such impacts would consequently not result in any measurable effect.

925. **It is concluded that the predicted merlin mortality would not adversely affect the integrity of the Berwyn SPA.**

926. Whilst extensive information exists on the responses of raptors to onshore OWFs, there is substantial uncertainty regarding the effects on migratory bird movements at sea. The confidence level assigned to this section of the assessment is therefore medium. However, since such low levels of collision

are predicted, an adverse effect on the integrity of the site is considered highly unlikely, even in the unlikely event that impacts have been underestimated.

#### Potential effects on the qualifying feature in-combination with other projects

927. As no measurable effects are predicted as a result of the project alone, there would be no contribution to in-combination effects. **It is concluded that predicted merlin mortality due to collision at the windfarm site, alone and in-combination with other projects, would not adversely affect the integrity of the Berwyn SPA**.

### 8.18.3.3 Peregrine

#### Status

928. The Berwyn SPA breeding peregrine population stood at 18 pairs at classification (1998), and seven pairs in 2002 (Stroud *et al*., 2016).

#### Functional linkage and seasonal apportionment of potential effects

929. Peregrine was screened into the Appropriate Assessment due to the risk of potential impacts occurring during the spring and autumn migration seasons. However, peregrine populations in the UK are largely sedentary (Cramp and Simmons, 1980), and therefore it is considered very unlikely that birds from the Berwyn SPA population would pass through the windfarm site. Accordingly, no impacts are apportioned to Berwyn SPA.

#### Potential effects on the qualifying feature alone and in-combination with other projects

930. As no peregrines from the Berwyn SPA population are likely to occur at the windfarm site, there would be no measurable effect on this species, and there would be no contribution to any potential in-combination effects**. It is therefore concluded that there would be no adverse effect on the integrity of the Berwyn SPA, either alone or in-combination with other projects**.

### 8.18.3.4 Red kite

#### Status

931. The Berwyn SPA breeding red kite population stood at two to three pairs at classification (1998).

#### Functional linkage and seasonal apportionment of potential effects

932. Red kite was screened into the Appropriate Assessment due to the risk of potential impacts occurring during the spring and autumn migration seasons. However, the native Welsh red kite populations are largely sedentary (Cramp

and Simmons, 1980), and therefore it is considered very unlikely that birds from the Berwyn SPA population would pass through the windfarm site. Accordingly, no impacts are apportioned to Berwyn SPA.

### Potential effects on the qualifying feature alone and in-combination with other projects

933. As no red kites from the Berwyn SPA population are likely to occur at the windfarm site, there would be no measurable effect on this species, and there would be no contribution to any potential in-combination effects. **It is therefore concluded that there would be no adverse effect on the integrity of the Berwyn SPA, either alone or in-combination with other projects**.

## 8.19 South Pennine Moors Phase 2 SPA

934. South Pennine Moors Phase 2 SPA is located approximately 87km from the windfarm site.

### 8.19.1 Description of designation

935. South Pennine Moors Phase 2 SPA includes two discrete blocks of moorland, one south of Ilkley and another on the watershed between Bradford and Burnley and stretching south to Marsden at the northern edge of the Peak District. It covers extensive tracts of semi-natural moorland habitats including upland heath and blanket mire. The site was designated for its breeding merlin and golden plover populations, together with its breeding assemblage of migratory species, including short-eared owl.

### 8.19.2 Conservation objectives

936. South Pennine Moors Phase 2 SPA's conservation objectives are to ensure that the integrity of the site is maintained or restored as appropriate, and ensure that the site contributes to achieving the aims of the Wild Birds Directive, by maintaining or restoring:

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

### 8.19.3 Assessment

#### 8.19.3.1 Merlin

##### Status

937. Breeding merlin is listed as a qualifying feature South Pennine Moors Phase 2 SPA. At the time of its classification (1997) the SPA supported 28 breeding pairs (Natural England, 2018b).

##### Functional linkage and seasonal apportionment of potential effects

938. Merlin has been screened into the Appropriate Assessment due to the risk of potential impacts occurring during the spring and autumn migration seasons. This species was not recorded in the aerial survey study area during the

baseline surveys undertaken at the windfarm site. However, it is recognised that there is the potential that it may pass through the habitat in the windfarm site during migration periods, and may have been missed by the surveys.

939. The apportioning of impacts to this designated site was calculated by dividing the number of collisions calculated at the national level by the proportion of the national population that were members of the designated site population at citation. The numbers used to define the national populations were the Great Britain populations presented in Wright *et al.,* (2012), comprising 1,330 pairs or 2,660 individuals during the breeding season. Designated site populations were obtained from the SPA citation, comprising 28 pairs or 56 individuals. Accordingly, 2.1% of impacts to this species were apportioned to South Pennine Moors Phase 2 SPA.

### Potential effects on the qualifying feature from the Project-alone

940. Merlin has been screened into the Appropriate Assessment due to the potential risk of collision. The magnitude of potential collision impacts has been investigated using the SOSSMAT tool (Wright *et al*., 2012).

941. The unapportioned annual collision risk calculated using the SOSSMAT tool (refer to **Appendix 12.1** of the ES), is estimated at less than 0.01 birds, assuming an avoidance rate of 0.980. Zero mortality is therefore apportioned to South Pennine Moors Phase 2 SPA. Such impacts would consequently not result in any measurable effect.

942. **It is concluded that the predicted merlin mortality would not adversely affect the integrity of the South Pennine Moors Phase 2 SPA.**

943. Whilst extensive information exists on the responses of raptors to onshore OWFs, there is substantial uncertainty regarding the effects on migratory bird movements at sea. The confidence level assigned to this section of the assessment is therefore medium. However, since such low levels of collision are predicted, an adverse effect on the integrity of the site is considered highly unlikely, even in the unlikely event that impacts have been underestimated.

### Potential effects on the qualifying feature in-combination with other projects

944. As no measurable effects are predicted as a result of the project alone, there would be no contribution to in-combination effects. **It is concluded that predicted merlin mortality due to collision at the windfarm site, alone and in-combination with other projects, would not adversely affect the integrity of the South Pennine Moors Phase 2 SPA**.

#### 8.19.3.2 Golden plover

##### Status

945. Breeding golden plover is listed as a qualifying feature of South Pennine Moors Phase 2 SPA. At the time of its classification (1997) the SPA supported 292 breeding pairs (Natural England, 2018b).

##### Functional linkage and seasonal apportionment of potential effects

946. Golden plover has been screened into the Appropriate Assessment due to the risk of potential impacts occurring during the spring and autumn migration seasons. This species was not recorded in the aerial survey study area during the baseline surveys undertaken at the windfarm site. However, it is recognised that there is the potential that it may pass through the habitat in the windfarm site during migration periods, and may have been missed by the surveys.

947. The apportioning of impacts to this designated site was calculated by dividing the number of collisions calculated at the national level by the proportion of the national population that were members of the designated site population at citation. The numbers used to define the national populations were the Great Britain populations presented in Wright *et al.,* (2012), comprising 22,600 pairs or 45,200 individuals during the breeding season. Designated site populations were obtained from the SPA citation, comprising 292 pairs or 584 individuals. Accordingly, 1.3% of impacts to this species were apportioned to South Pennine Moors Phase 2 SPA.

##### Potential effects on the qualifying feature from the Project-alone

948. Golden plover has been screened into the Appropriate Assessment due to the potential risk of collision. The magnitude of potential collision impacts has been investigated using the SOSSMAT tool (Wright *et al*., 2012).

949. The unapportioned annual collision risk calculated using the SOSSMAT tool (refer to **Appendix 12.1** of the ES), is estimated at 0.03 birds, assuming an avoidance rate of 0.980. Zero mortality is therefore apportioned to South Pennine Moors Phase 2 SPA. Such impacts would consequently not result in any measurable effect.

950. **It is concluded that the predicted golden plover mortality would not adversely affect the integrity of the South Pennine Moors Phase 2 SPA.**

951. Whilst extensive information exists on the responses of birds to onshore OWFs, there is substantial uncertainty regarding the effects on migratory bird movements at sea. The confidence level assigned to this section of the assessment is therefore medium. However, since such low levels of collision

are predicted, an adverse effect on the integrity of the site is considered highly unlikely, even in the unlikely event that impacts have been underestimated.

### Potential effects on the qualifying feature in-combination with other projects

952. As no measurable effects are predicted as a result of the project alone, there would be no contribution to in-combination effects. **It is concluded that predicted golden plover mortality due to collision at the windfarm site, alone and in-combination with other projects, would not adversely affect the integrity of the South Pennine Moors Phase 2 SPA**.

### 8.19.3.3 Short-eared owl

### Status

953. Breeding short-eared owl is listed as a component of the qualifying assemblage of species for South Pennine Moors Phase 2 SPA. The population of the South Pennine Moors was 25 pairs in 1990 (Stroud *et al*., 2016).

### Functional linkage and seasonal apportionment of potential effects

954. Short-eared owl has been screened into the Appropriate Assessment due to the risk of potential impacts occurring during the spring and autumn migration seasons. This species was not recorded in the aerial survey study area during the baseline surveys undertaken at the windfarm site. However, it is recognised that there is the potential that it may pass through the habitat in the windfarm site during migration periods, and may have been missed by the surveys.

955. The apportioning of impacts to this designated site was calculated by dividing the number of collisions calculated at the national level by the proportion of the national population that were members of the designated site population at citation. The numbers used to define the national populations were the Great Britain populations presented in Wright *et al.,* (2012), comprising 1,000 pairs or 2,000 individuals during the breeding season. Designated site populations were obtained from the SPA citation, comprising 25 pairs or 50 individuals. Accordingly, 2.5% of impacts to this species were apportioned to South Pennine Moors Phase 2 SPA.

### Potential effects on the qualifying feature from the Project-alone

956. Short-eared owl has been screened into the Appropriate Assessment due to the potential risk of collision. The magnitude of potential collision impacts has been investigated using the SOSSMAT tool (Wright *et al*., 2012).

957. The unapportioned annual collision risk calculated using the SOSSMAT tool (refer to **Appendix 12.1** of the ES), is estimated at 0.01 birds, assuming an avoidance rate of 0.980. Zero mortality is therefore apportioned to South Pennine Moors Phase 2 SPA. Such impacts would consequently not result in any measurable effect.

958. **It is concluded that the predicted short-eared owl mortality would not adversely affect the integrity of the South Pennine Moors Phase 2 SPA.**

959. Whilst extensive information exists on the responses of birds to onshore OWFs, there is substantial uncertainty regarding the effects on migratory bird movements at sea. The confidence level assigned to this section of the assessment is therefore medium. However, since such low levels of collision are predicted, an adverse effect on the integrity of the site is considered highly unlikely, even in the unlikely event that impacts have been underestimated.

### Potential effects on the qualifying feature in-combination with other projects

960. As no measurable effects are predicted as a result of the Project alone, there would be no contribution to in-combination effects. **It is concluded that predicted short-eared owl mortality due to collision at the windfarm site, alone and in-combination with other projects, would not adversely affect the integrity of the South Pennine Moors Phase 2 SPA**.

## 8.20 North Pennine Moors SPA

961. North Pennine Moors SPA is located approximately 98km from the windfarm site.

### 8.20.1 Description of designation

962. The North Pennine Moors SPA includes parts of the Pennine moorland massif between the Tyne Gap (Hexham) and the Ribble-Aire corridor (Skipton). It encompasses extensive tracts of semi-natural moorland habitats including upland heath and blanket bog. The southern end of the North Pennine Moors SPA is within 10 km of the South Pennine Moors SPA, which supports a similar assemblage of upland breeding species.

### 8.20.2 Conservation objectives

963. North Pennine Moors SPA's conservation objectives are to ensure that the integrity of the site is maintained or restored as appropriate, and ensure that the site contributes to achieving the aims of the Wild Birds Directive, by maintaining or restoring:

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

### 8.20.3 Assessment

#### 8.20.3.1 Hen harrier

##### Status

964. Breeding hen harrier is listed as a component of North Pennine Moors SPA. At the time of classification (1998), it was estimated that the site supported 11 pairs, but this had declined to two pairs in 2006 (Natural England, 2019c).

##### Functional linkage and seasonal apportionment of potential effects

965. Hen harrier has been screened into the Appropriate Assessment due to the risk of potential impacts occurring during the spring and autumn migration seasons. This species was not recorded in the aerial survey study area during the baseline surveys undertaken at the windfarm site. However, it is

recognised that there is the potential that it may pass through the habitat in the windfarm site during migration periods and may have been missed by the surveys.

966. The apportioning of impacts to this designated site was calculated by dividing the number of collisions calculated at the national level by the proportion of the national population that were members of the designated site population at citation. The numbers used to define the national populations were the Great Britain populations presented in Wright *et al.,* (2012), comprising 570 pairs or 1,140 individuals during the breeding season. Designated site populations were obtained from the SPA citation, comprising 11 pairs or 22 individuals. Accordingly, 1.9% of impacts to this species were apportioned to North Pennine Moors SPA.

### Potential effects on the qualifying feature from the Project-alone

967. Hen harrier has been screened into the Appropriate Assessment due to the potential risk of collision. The magnitude of potential collision impacts has been investigated using the SOSSMAT tool (Wright *et al*., 2012).

968. The unapportioned annual collision risk calculated using the SOSSMAT tool (refer to **Appendix 12.1** of the ES), is estimated at less than 0.01 birds, assuming an avoidance rate of 0.980. Zero mortality is therefore apportioned to North Pennine Moors SPA. Such impacts would consequently not result in any measurable effect.

969. **It is concluded that the predicted hen harrier mortality would not adversely affect the integrity of the North Pennine Moors SPA.**

970. Whilst extensive information exists on the responses of raptors to onshore OWFs, there is substantial uncertainty regarding the effects on migratory bird movements at sea. The confidence level assigned to this section of the assessment is therefore medium. However, since such low levels of collision are predicted, an adverse effect on the integrity of the site is considered highly unlikely, even in the unlikely event that impacts have been underestimated.

### Potential effects on the qualifying feature in-combination with other projects

971. As no measurable effects are predicted as a result of the project alone, there would be no contribution to in-combination effects. **It is concluded that predicted hen harrier mortality due to collision at the windfarm site, alone and in-combination with other projects, would not adversely affect the integrity of the North Pennine Moors SPA**.

#### 8.20.3.2 Merlin

##### Status

972. Breeding merlin is listed as a qualifying feature of North Pennine Moors SPA. At the time of the SPA classification, surveys in 1993 and 1994 estimated that the site supported 136 pairs, although this had declined to 65 territories in 2006 (Natural England, 2019c).

##### Functional linkage and seasonal apportionment of potential effects

973. Merlin has been screened into the Appropriate Assessment due to the risk of potential impacts occurring during the spring and autumn migration seasons. This species was not recorded in the aerial survey study area during the baseline surveys undertaken at the windfarm site. However, it is recognised that there is the potential that it may pass through the habitat in the windfarm site during migration periods and may have been missed by the surveys.

974. The apportioning of impacts to this designated site was calculated by dividing the number of collisions calculated at the national level by the proportion of the national population that were members of the designated site population at citation. The numbers used to define the national populations were the Great Britain populations presented in Wright *et al.,* (2012), comprising 1,330 pairs or 2,660 individuals during the breeding season. Designated site populations were obtained from the SPA citation, comprising 136 pairs or 272 individuals. Accordingly, 10.2% of impacts to this species were apportioned to North Pennine Moors SPA.

##### Potential effects on the qualifying feature from the Project-alone

975. Merlin has been screened into the Appropriate Assessment due to the potential risk of collision. The magnitude of potential collision impacts has been investigated using the SOSSMAT tool (Wright *et al*., 2012).

976. The unapportioned annual collision risk calculated using the SOSSMAT tool (refer to **Appendix 12.1** of the ES), is estimated at less than 0.01 birds, assuming an avoidance rate of 0.980. Zero mortality is therefore apportioned to North Pennine Moors SPA. Such impacts would consequently not result in any measurable effect.

977. **It is concluded that the predicted merlin mortality would not adversely affect the integrity of the North Pennine Moors SPA.**

978. Whilst extensive information exists on the responses of raptors to onshore OWFs, there is substantial uncertainty regarding the effects on migratory bird movements at sea. The confidence level assigned to this section of the assessment is therefore medium. However, since such low levels of collision

are predicted, an adverse effect on the integrity of the site is considered highly unlikely, even in the unlikely event that impacts have been underestimated.

#### Potential effects on the qualifying feature in-combination with other projects

979. As no measurable effects are predicted as a result of the project alone, there would be no contribution to in-combination effects. **It is concluded that predicted merlin mortality due to collision at the windfarm site, alone and in-combination with other projects, would not adversely affect the integrity of the North Pennine Moors SPA**.

### 8.20.3.3 Peregrine

#### Status

980. Breeding peregrine is listed as a qualifying feature of North Pennine Moors SPA. At the time of classification, the SPA supported 15 breeding pairs, although this had declined to four territories in 2006 (Natural England, 2019c).

#### Functional linkage and seasonal apportionment of potential effects

981. Peregrine was screened into the Appropriate Assessment due to the risk of potential impacts occurring during the spring and autumn migration seasons. However, peregrine populations in the UK are largely sedentary (Cramp and Simmons, 1980), and therefore it is considered very unlikely that birds from the North Pennine Moors SPA population would pass through the windfarm site. Accordingly, no impacts are apportioned to North Pennine Moors SPA.

#### Potential effects on the qualifying feature alone and in-combination with other projects

982. As no peregrines from the North Pennine Moors SPA population are likely to occur at the windfarm site, there would be no measurable effect on this species, and there would be no contribution to any potential in-combination effects. **It is therefore concluded that there would be no adverse effect on the integrity of the North Pennine Moors SPA, either alone or in-combination with other projects**.

### 8.20.3.4 Golden plover

#### Status

983. Breeding golden plover is listed as a qualifying feature of North Pennine Moors SPA. At the time of classification, the SPA was estimated to support 1,400 pairs (Natural England, 2019c). The 2005-2007 North Pennine Moors SPA survey (Shepherd, 2007) recorded 4,171 pairs across the site.

### Functional linkage and seasonal apportionment of potential effects

984. Golden plover has been screened into the Appropriate Assessment due to the risk of potential impacts occurring during the spring and autumn migration seasons. This species was not recorded in the aerial survey study area during the baseline surveys undertaken at the windfarm site. However, it is recognised that there is the potential that it may pass through the habitat in the windfarm site during migration periods and may have been missed by the surveys.

985. The apportioning of impacts to this designated site was calculated by dividing the number of collisions calculated at the national level by the proportion of the national population that were members of the designated site population at citation. The numbers used to define the national populations were the Great Britain populations presented in Wright *et al.,* (2012), comprising 22,600 pairs or 45,200 individuals during the breeding season. Designated site populations were obtained from the SPA citation, comprising 1,400 pairs or 2,800 individuals. Accordingly, 6.2% of impacts to this species were apportioned to North Pennine Moors SPA.

### Potential effects on the qualifying feature from the Project-alone

986. Golden plover has been screened into the Appropriate Assessment due to the potential risk of collision. The magnitude of potential collision impacts has been investigated using the SOSSMAT tool (Wright *et al*., 2012).

987. The unapportioned annual collision risk calculated using the SOSSMAT tool (refer to **Appendix 12.1** of the ES), is estimated at 0.03 birds, assuming an avoidance rate of 0.980. Zero mortality is therefore apportioned to North Pennine Moors SPA. Such impacts would consequently not result in any measurable effect.

988. **It is concluded that the predicted golden plover mortality would not adversely affect the integrity of the North Pennine Moors SPA.**

989. Whilst extensive information exists on the responses of birds to onshore OWFs, there is substantial uncertainty regarding the effects on migratory bird movements at sea. The confidence level assigned to this section of the assessment is therefore medium. However, since such low levels of collision are predicted, an adverse effect on the integrity of the site is considered highly unlikely, even in the unlikely event that impacts have been underestimated.

### Potential effects on the qualifying feature in-combination with other projects

990. As no measurable effects are predicted as a result of the project alone, there would be no contribution to in-combination effects. **It is concluded that predicted golden plover mortality due to collision at the windfarm site,**

**alone and in-combination with other projects, would not adversely affect the integrity of the North Pennine Moors SPA**.

# 8.21 Glannau Aberdaron ac Ynys Enlli/Aberdaron Coast and Bardsey Island SPA

991. Glannau Aberdaron ac Ynys Enlli/Aberdaron Coast and Bardsey Island SPA is located approximately 125km from the windfarm site.

## 8.21.1 Description of designation

992. Aberdaron Coast and Bardsey Island is located at the very tip of the Llŷn Peninsula in north-west Wales. The site consists of Ynys Enlli/Bardsey Island and a length of adjacent coastline together with two small islands Ynysoedd y Gwylanod/Gwylan Islands, in addition to an area of sea extending approximately 9km out from Bardsey. The coastline is rocky, with many crags and low cliffs, heather-covered hills and grassy valleys in a distinctive landscape of small fields and “cloddiau” (stone-faced banks).

993. The site supports a population of chough which depend on the low intensity pastoral management of this mix of habitats. Bardsey Island holds a large breeding colony of Manx shearwaters which forage widely across the ocean and loaf on adjacent areas of the sea for a number of essential activities, such as preening, bathing and displaying, before attempting their hazardous approach to the nest site after nightfall.

## 8.21.2 Conservation objectives

994. The overarching conservation objectives for each of the qualifying features of the SPA are:

- The size of the population should be stable or increasing, allowing for natural variability, and sustainable in the long term
- The distribution of the population should be being maintained, or where appropriate increasing
- There should be sufficient habitat, of sufficient quality, to support the population in the long term
- Factors affecting the population or its habitat should be under appropriate control

## 8.21.3 Assessment

### 8.21.3.1 Manx shearwater

#### Status

995. Manx shearwater is listed as a qualifying species of this SPA.

996. The SPA population has been cited as 6,930, in 1996 (Furness, 2015, Stroud *et al*., 2016). Furness (2015) proposed a breeding population of 16,183 pairs in 2001. The most recent count in the SMP database (2001) was 16,183 apparently occupied sites (AOS; burrows or crevices), or 32,366 breeding adults (JNCC, 2022).

997. Based on the most recent SPA population of breeding adults, and an annual breeding adult baseline mortality rate of 0.13 (Horswill and Robinson, 2015), 4,208 breeding adults from the SPA population would be expected to die each year.

**Functional linkage and seasonal apportionment of potential effects**

998. The windfarm site is situated approximately 135km from Bardsey Island, at its nearest point; the across-sea distance is approximately 146km. The mean maximum foraging range of Manx shearwater is 1,347km (±1,019km) (Woodward *et al*., 2019). The windfarm site is therefore within the mean maximum foraging range of Manx shearwaters from the Glannau Aberdaron ac Ynys Enlli/Aberdaron Coast and Bardsey Island SPA.

999. There was limited published tracking information for Manx shearwaters from Bardsey Island, but some recent tracking studies are documented in grey literature. Porter (2017) recorded a number of Manx shearwater tracks during the breeding period in 2017, which showed that birds travelled northwards from the breeding colony into the Irish Sea. The recorded tracks showed that some birds travelled to the Irish Sea Front SPA area, located to the south west of the Isle of Man, while other birds were shown to continue northwards, past the east coast of Northern Ireland, to feed off the west coast of Scotland. The Welsh Ornithological Society (WOS, 2022) has reported similar results from tracking studies undertaken in 2022, with tracks suggesting concentrations of activity around the Irish Sea Front SPA, the west coast of the Isle of Man and west coast of Scotland. Some activity was also recorded to the east of the Isle of Man, but no tracks passed close to the windfarm site itself.

1000. Four further UK SPAs designated for Manx shearwater are located within the UK Western Waters BDMPS area, all of which are within the mean maximum foraging range of this species (straight-line distance from windfarm site and most recent population count (AOS) in brackets):

- Copeland Islands SPA (149km; 4,850 AOS (2007))
- Skomer, Skokholm and the Seas off Pembrokeshire/Sgomer, Sgogwm a Moroedd Penfro SPA (246km; 455,156 AOS (2018))
- Rum SPA (374km; 120,000 AOS (2001))

- St Kilda SPA (526km; 4,802 AOS (1999[20]))

1001. In addition to UK SPA colonies, the SMP database identifies five Irish SPAs and a number of non-SPA colonies. A total of 38 Manx shearwater colonies in and around the UK Western Waters BDMPS area have been identified, with a total count of 1,299,546 adult birds (based on the most recently available post-1999 counts (together with the 1999 St Kilda count) and including SPA colonies). It is therefore likely that any birds present at the windfarm site during the breeding season may originate from a number of different colonies within this region.

1002. The NatureScot apportioning tool (NatureScot, 2018) has been used to estimate the proportion of Manx shearwaters from each of the relevant SPAs present at the windfarm site during the breeding season. The apportioning to SPA and non-SPA colonies is set out in **Table 8.45**; refer also to **Appendix 12.1** of the ES for further information on the apportioning approach and results. Accordingly, 8.63% of impacts at the windfarm site during the breeding season are apportioned to Glannau Aberdaron ac Ynys Enlli/Aberdaron Coast and Bardsey Island SPA.

*Table 8.45 Manx shearwater breeding season apportioning*

| Site | Apportioning rate |
|---|---|
| **Glannau Aberdaron ac Ynys Enlli/Aberdaron Coast and Bardsey Island SPA** | **8.63%** |
| Copeland Islands SPA | 2.22% |
| Skomer, Skokholm and the Seas off Pembrokeshire / Sgomer, Sgogwm a Moroedd Penfro SPA | 76.54% |
| Rum SPA | 8.44% |
| St Kilda SPA | 0.20% |
| Cruagh Island SPA (transboundary site) | 0.10% |
| Blasket Islands SPA (transboundary site) | 0.61% |
| Deenish Island and Scariff Island SPA (transboundary site) | 0.08% |
| Puffin Island SPA (transboundary site) | 0.22% |
| Skelligs SPA (transboundary site) | 0.03% |
| Non-SPA colonies | 2.91% |

[20] From Furness (2015) – no post-1999 counts are available on the SMP database.

1003. During the pre- and post-breeding periods, breeding Manx shearwaters from the Glannau Aberdaron ac Ynys Enlli/Aberdaron Coast and Bardsey Island SPA migrate through UK waters. The relevant reference population is considered to be the UK Western Waters BDMPS. This consists of 1,580,895 individuals during the post-breeding (August-early October) and return migration (late march-May) periods (Furness, 2015).

1004. Estimates of the proportion of Manx shearwaters present at the windfarm site which originate from the Glannau Aberdaron ac Ynys Enlli/Aberdaron Coast and Bardsey Island SPA during the post-breeding and return migration periods (and therefore the proportion of predicted mortalities from the SPA population) are based on the SPA population (i.e. 32,366 adults) as a proportion of the UK Western Waters BDMPS during the relevant season. During the post-breeding and return migration periods, 2.05% of impacted birds are considered to originate from the SPA (Furness, 2015).

### Potential effects on the qualifying feature from the Project-alone

1005. The Manx shearwater qualifying feature of the Glannau Aberdaron ac Ynys Enlli/ Aberdaron Coast and Bardsey Island SPA has been screened into the assessment due to the potential risk of disturbance, displacement and barrier effects during the construction and decommissioning, and operation and maintenance phases of the Project.

#### *Construction and decommissioning phase disturbance/displacement/barrier effects*

1006. Effects during the construction and decommissioning phases of the Project are considered unlikely, given the transient presence of the species and low susceptibility to disturbance related impacts; refer to **Paragraph 1009**. However, in accordance with feedback received from Natural England and NRW, a precautionary estimation of construction and decommissioning phase disturbance, displacement and barrier effects has been undertaken assuming 50% of the operational phase effect.

1007. A displacement rate of 30-70% and mortality rate of 1-10% has been presented. Given that 10% would represent a rate close to the expected 'natural' annual mortality (0.13), this rate is considered very unlikely. Accordingly, a 1% mortality rate is considered to be most appropriate, with the upper end of this range likely to be precautionary. Given the very extensive foraging range of this species (Woodward *et al.,* 2019), there may be no mortality costs to displacement from the relatively very small footprints of OWFs.

1008. Applying 50% reduction to the operational values presented in **Table 8.46**, and based on mean density, predicted mortality would be between one and 17 birds (30-70% displacement and 1-10% mortality of displaced birds). Using realistic values (i.e. mean density, 50% displacement and 1% mortality), there would be an annual increase in mortality of 1.2 birds, which is equivalent to a 0.03% increase in background mortality for the SPA population. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. **Accordingly, no significant effects on Manx shearwater are predicted during the construction and decommissioning phases, and it is concluded that there would be no adverse effect on the integrity of Glannau Aberdaron ac Ynys Enlli/ Aberdaron Coast and Bardsey Island SPA**.

*Operation and maintenance phase disturbance/displacement/barrier effects*

1009. Manx shearwater is generally considered to have a low susceptibility to disturbance and displacement (Furness *et al*., 2013). Dierschke *et al*., (2016) described Manx shearwater as “weakly avoiding wind farms”, although also noted that evidence was lacking for the species. Bradbury *et al*., (2014) classified Manx shearwater as having “very low” population vulnerability to displacement.

1010. Dierschke *et al*., (2016) suggested that Manx shearwater were avoiding North Hoyle OWF, stating that an obvious distribution gap was observed at the OWF, although evidence for this appeared limited. Dierschke *et al*., (2016) also noted that Manx shearwater had been recorded within Robin Rigg OWF.

1011. Displacement effects for Manx shearwater for the Project were assessed during the breeding, autumn migration and spring migration periods, based on an unapportioned peak mean population of 4,705, 2,650 and 1,617 birds respectively, calculated for the windfarm site and a 2km buffer, in line with recommendations within the SNCB guidance (SNCBs, 2017). The displacement matrices used to calculate potential impacts are presented in **Appendix 12.1** of the ES and summarised in **Table 8.46**. The application of the same displacement rate to the OWF and the 2km buffer, to determine the total number of birds subject to displacement, is precautionary, as in reality the displacement rate is likely to fall with distance from the windfarm site.

*Table 8.46 Manx shearwater – predicted operation and maintenance phase displacement and mortality from Glannau Aberdaron ac Ynys Enlli/ Aberdaron Coast and Bardsey Island SPA*

| Mean peak abundance estimate type | Mean peak abundance estimate | Number of SPA breeding adults present by season[1] | Annual mortality range[2] | Annual baseline mortality increase range[3] |
|---|---|---|---|---|
| Upper 95% CI | 10,010 (breeding)<br>4,447 (autumn)<br>4,711 (spring)<br>19,168 (year round) | 864 (breeding)<br>91 (autumn)<br>96 (spring)<br>1,051 (year round) | 3-74 | 0.07-1.75% |
| Mean | 4,705 (breeding)<br>2,650 (autumn)<br>1,617 (spring)<br>8,972 (year round) | 406 (breeding)<br>54 (autumn)<br>33 (spring)<br>493 (year round) | 1-35 | 0.04-0.82% |
| Lower 95% CI | 783 (breeding)<br>1,308 (autumn)<br>0 (spring)<br>2,092 (year round) | 68 (breeding)<br>27 (autumn)<br>0 (spring)<br>94 (year round) | 0-7 | 0.01-0.16% |

[1] During the breeding season, assumes 8.6% of recorded birds are adults from the SPA population (32,366), and 2.05% during the autumn and spring migration periods

[2] Assumes displacement rates of 30-70% and mortality rates of 1-10%

[3] Background population is Glannau Aberdaron ac Ynys Enlli/ Aberdaron Coast and Bardsey Island SPA breeding adults (32,366 individuals), adult age class annual mortality rate of 13% (Horswill and Robinson, 2015)

1012. Using realistic values (i.e. mean density, 50% displacement and 1% mortality), there would be an annual increase in mortality of 2.47 birds/0.06%. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. **Accordingly, no significant effects on Manx shearwater are predicted during the operation and maintenance phase, and it is concluded that there would be no adverse effect on the integrity of Glannau Aberdaron ac Ynys Enlli/Aberdaron Coast and Bardsey Island SPA.**

1013. The confidence in the assessment is high. The evidence used to set the displacement rates presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES is of high applicability and quality. Whilst there is limited available evidence to inform mortality rates, 1% is considered to be sufficiently precautionary, based on expert opinion.

1014. A review of the potential effects of artificial light on Manx shearwaters is presented in Section 12.6.3.1 of **Chapter 12 Offshore Ornithology**. This concludes that lighting associated with the Project is very unlikely to significantly affect disturbance and displacement effects on Manx shearwater, and therefore the conclusions of the assessment are unchanged.

**Potential effects in-combination with other projects**

1015. No in-combination effects are predicted during the construction and decommissioning phases. This is because it is unlikely that there would be significant temporal and/or spatial overlap with other plans or projects, and due to the negligible effects predicted from the project alone.

1016. During the operation and maintenance phase, the in-combination assessment for Manx shearwaters from Glannau Aberdaron ac Ynys Enlli/ Aberdaron Coast and Bardsey Island SPA has been undertaken in accordance with the approach presented in **Section 8.1**. The total population apportioned to the SPA at risk of displacement is estimated to be 740 breeding adults (**Appendix 12.1** of the ES). Annual in-combination displacement and mortality rates for birds from Glannau Aberdaron ac Ynys Enlli/ Aberdaron Coast and Bardsey Island SPA are presented in **Table 8.47**.

1017. Assuming a displacement rate of 70% and a mortality rate of 10% of displaced birds, 52 breeding adult SPA birds would be lost to displacement annually. This would increase the existing mortality within the SPA population (4,208 breeding adult birds per year) by 1.23%. Using a realistic displacement rate of 50%, and a mortality rate for displaced birds of 1%, the annual in-combination mortality would be 4 birds. This would increase the existing mortality within this population by 0.09%. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that detectable changes in mortality rates will not occur due to the level or mortality predicted if the more realistic rates for mortality are used.

1018. It is concluded that predicted Manx shearwater mortality due to of operational phase displacement impacts of the Project in-combination with other projects, would not adversely affect the integrity of Glannau Aberdaron ac Ynys Enlli/ Aberdaron Coast and Bardsey Island SPA. This accords with the conclusions of the Round 4 offshore wind leasing HRA (NIRAS, 2021), which concluded no adverse effect on site integrity (for all SPAs) on the basis of the low vulnerability to OWFs and low density of this species within Round 4 areas.

1019. It is noted that limited or no data are available from five historic projects that may have the potential to contribute to the in-combination effect on this feature (Burbo Bank, Walney 1&2, Gwynt y Môr, Rhyl Flats and Robin Rigg). As set out in the cumulative assessment for Manx shearwater presented in **Chapter 12 Offshore Ornithology** of the ES, in each case the assessments for these projects concluded no impact, or 'low' or 'very low' significance effects on this species. In order to reach a threshold where a significant effect might be possible (i.e. an increase in background mortality >1% affecting the SPA Manx shearwater population, assuming realistic displacement rates of 50%/1%), these historic projects would need contribute approximately 7,675 additional birds annually to the total potentially impacted population. This would equate to approximately 1,535 birds apportioned to the SPA at each project site annually. Given that the largest contribution from single project where data are available is 493 birds (for the Project), it is considered extremely unlikely that such high contributions could arise from these historic projects. Accordingly, it is concluded that these additional projects would not affect the conclusion of the assessment.

*Table 8.47 In-combination year-round displacement matrix for Manx shearwater from Glannau Aberdaron ac Ynys Enlli/Aberdaron Coast and Bardsey Island SPA*

| **Annual** | **Mortality** | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| **Displacement** | **1%** | **2%** | **3%** | **4%** | **5%** | **10%** | **20%** | **30%** | **50%** | **80%** | **100%** |
| **10%** | 0 | 1 | 1 | 2 | 2 | 5 | 10 | 15 | 25 | 40 | 49 |
| **20%** | 1 | 2 | 3 | 4 | 5 | 10 | 20 | 30 | 49 | 79 | 99 |
| **30%** | **1** | **3** | **4** | **6** | **7** | **15** | 30 | 44 | 74 | 119 | 148 |
| **40%** | **2** | **4** | **6** | **8** | **10** | **20** | 40 | 59 | 99 | 158 | 198 |
| **50%** | **2** | **5** | **7** | **10** | **12** | **25** | 49 | 74 | 123 | 198 | 247 |
| **60%** | **3** | **6** | **9** | **12** | **15** | **30** | 59 | 89 | 148 | 237 | 296 |
| **70%** | **3** | **7** | **10** | **14** | **17** | **35** | 69 | 104 | 173 | 277 | 346 |
| **80%** | 4 | 8 | 12 | 16 | 20 | 40 | 79 | 119 | 198 | 316 | 395 |
| **90%** | 4 | 9 | 13 | 18 | 22 | 44 | 89 | 133 | 222 | 356 | 444 |
| **100%** | 5 | 10 | 15 | 20 | 25 | 49 | 99 | 148 | 247 | 395 | 494 |

Note: The cells show the number of birds predicted to die (rounded to the nearest integer) at a given rate of displacement and mortality. Highlighted cells are considered to be the most realistic scenario, in accordance with SNCB advice (SNCBs, 2022).

## 8.22 Strangford Lough SPA and Ramsar

1020. Strangford Lough SPA and Ramsar site is located approximately 129km from the windfarm site.

### 8.22.1 Description of designation

1021. Strangford Lough is a large (150km$^2$) marine inlet on the east coast of County Down, of which about 50km$^2$ lies between high water mark mean tide and low water mark mean tide. It is connected to the open sea by the Strangford Narrows, an 8 km long channel with a minimum width of 0.5km. The Lough is 30km long from head to mouth and up to 8km wide. The tidal flats of Strangford Lough form extensive areas around the northern and north-eastern shorelines. The Lough supports an impressive range of marine habitats and communities with over 2,000 recorded species. It is important for marine invertebrates, algae and saltmarsh plants, for a range of wintering and breeding waterbirds, and for marine mammals.

### 8.22.2 Conservation objectives

1022. The overarching conservation objective for the SPA is 'to maintain each feature in favourable condition.' For the qualifying features, the objectives are:

- To maintain or enhance the population of the qualifying species
- Fledging success sufficient to maintain or enhance population
- To maintain or enhance the range of habitats utilised by the qualifying species
- To ensure that the integrity of the site is maintained
- To ensure there is no significant disturbance of the species
- To ensure that the following are maintained in the long term:
    - Population of the species as a viable component of the site
    - Distribution of the species within the site
    - Distribution and extend of habitats supporting the species
    - Structure, function and supporting processes of habitats supporting the species

### 8.22.3 Assessment

1023. Two qualifying features of Strangford Lough SPA and Ramsar site have been screened into the Appropriate Assessment (**Table 5.2**). These are breeding Sandwich tern and breeding common tern.

### 8.22.3.1 Sandwich tern

#### Status

1024. The mean Strangford Lough SPA breeding Sandwich tern population at classification was 593 pairs, or 1,186 breeding adults, for the period 1993 to 1997 (Department of the Environment (DoE), 1998). Furness (2015) gave the breeding population as 771 pairs or 1,542 breeding adults. The most recent count was 310 pairs (AON), or 620 breeding adults, in 2022 (JNCC, 2023a); this is used as the reference population for the assessment.

1025. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult mortality rate of 0.102 (1 – 0.898; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 63 breeding adults.

#### Functional linkage and seasonal apportionment of potential effects

1026. The mean maximum foraging range of Sandwich tern is 34.3km (±23.2km) and the maximum foraging range is 80km (Woodward *et al*., 2019). The Project is located approximately 129km from Strangford Lough SPA, which means that the Project is beyond the maximum foraging range of Sandwich terns from the SPA. No impacts during the breeding season from the Project are therefore apportioned to Sandwich terns breeding at this SPA.

1027. Outside the breeding season, breeding Sandwich terns are assumed to range widely and to mix with birds of all ages from breeding colonies in the UK and further afield. The relevant background population is considered to be the UK Western waters BDMPS, consisting of 10,761 individuals during autumn migration (July to September), and spring migration (March to May) (Furness, 2015).

1028. Estimates of the proportion of Sandwich terns present at the Project site during the autumn and spring migration seasons which originate from the Strangford Lough SPA site are based on the SPA population as a proportion of the UK Western waters BDMPS (Furness, 2015). During both autumn and spring migration seasons, breeding adult Sandwich terns from Strangford Lough SPA make up 14.33% of the total BDMPS population. The same percentage of impacts are therefore attributable to birds from this SPA during these times of year.

#### Potential effects on the qualifying feature from the Project-alone

1029. The Sandwich tern qualifying feature of the Strangford Lough SPA has been screened into the assessment due to the potential risk of collision.

### Operation and maintenance phase collision risk

*Project-alone*

1030. Information for collision risk on breeding adult Sandwich terns belonging to the Strangford Lough SPA population is presented in **Table 8.13**. Collision estimates, calculated using the sCRM, are presented by biological season. A summary of the annual outputs and the corresponding increase in the annual baseline mortality rate is also presented. Parameters used in the sCRM were agreed with Natural England during the ETG process and are described in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES.

1031. Based on the mean collision rates, the annual total of breeding adult Sandwich terns from the Strangford Lough SPA at risk of collision as a result of the Project is 0.05. This would result in no detectable increase (0.07%) in the existing mortality of the SPA breeding population.

*Table 8.48 Predicted seasonal and annual collision mortality (Stochastic model Option 2, avoidance rate 0.991 (±0.0004)) for breeding adult Sandwich terns at the windfarm site, apportioned to Strangford Lough SPA, with corresponding increases to baseline mortality of the population*

| | Breeding Season | Autumn Migration | Non-breeding/winter | Spring Migration | Annual |
|---|---|---|---|---|---|
| **Period** | **Apr-Aug** | **Sep** | **Oct-Feb** | **Mar** | **Jan-Dec** |
| **Total collisions (mean and 95% Cis)** | 0.00<br>(0.00-0.00) | 0.33<br>(0.02-1.07) | 0.00<br>(0.00-0.00) | 0.00<br>(0.00-0.00) | 0.33<br>(0.02-1.07) |
| **% apportioned to the SPA** | 0.0% | 14.33% | 0.0% | 14.33% | - |
| **Total SPA collisions (mean and 95% Cis)** | 0.00<br>(0.00-0.00) | 0.05<br>(0.00-0.15) | 0.00<br>(0.00-0.00) | 0.00<br>(0.00-0.00) | 0.05<br>(0.00-0.15) |
| **Mortality increase[1] (mean and 95% Cis)** | **0.00%**<br>(0.00-0.00%) | **0.07%**<br>(0.00-0.24%) | **0.00%**<br>(0.00-0.00%) | **0.00%**<br>(0.00-0.00%) | **0.07%**<br>(0.00-0.24%) |

[1] Assuming predicted annual SPA adult mortality of 63 birds (620 x 0.102)

1032. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that no detectable changes in mortality rates would occur on this population for the mean monthly collision estimates for the Project, or for the upper 95% CI collision estimate.

1033. **It is concluded that predicted Sandwich tern mortality due to collision at the windfarm site would not adversely affect the integrity of the Strangford Lough SPA and Ramsar site**.

1034. The confidence in the assessment is high. The evidence used to define the CRM input parameters presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES is of high applicability and quality. Whilst there is uncertainty around some of the input parameters (e.g. avoidance rate), the rates selected are considered to be sufficiently precautionary based on expert opinion to provide confidence that collision rates are not underestimated.

*In-combination*

1035. As no measurable effects on Sandwich tern are predicted as a result of the Project-alone, there would be no material contribution to the effects of other plans or projects in-combination. **It is therefore concluded that is no potential for the Project-alone or in-combination to have an adverse effect on the integrity of the Strangford Lough SPA and Ramsar site**.

#### 8.22.3.2 Common tern

##### Status

1036. The mean Strangford Lough SPA breeding common tern population at classification was 603 pairs, or 1,206 breeding adults, for the period 1993 to 1997 (DoE, 1998). Furness (2015) gave a breeding population of 352 pairs or 704 individuals in 2013. The most recent count was 449 pairs (AON), or 898 breeding adults, in 2022 (JNCC, 2023a); this is used as the reference population for the assessment.

1037. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.117 (1 – 0.883; Horswill and Robinson, 2015), the expected annual mortality of the SPA population would be 105 breeding adults.

##### Functional linkage and seasonal apportionment of potential effects

1038. The mean maximum breeding season foraging range of common tern is 18.0km (±8.9km) and the maximum foraging range is 30km (Woodward *et al.,* 2019). The Project is located approximately 129km from Strangford Lough SPA, which means that the Project is beyond the maximum foraging range of common terns from the SPA. No impacts during the breeding season from the Project are therefore apportioned to common terns breeding at this SPA.

1039. Outside the breeding season, breeding common terns are assumed to range widely and to mix with birds of all ages from breeding colonies in the UK and further afield. The relevant background population is considered to be the UK Western waters BDMPS, consisting of 64,659 individuals during autumn migration (late July to early September), and spring migration (April to May) (Furness, 2015).

1040. Estimates of the proportion of common terns present at the Project site during the autumn and spring migration seasons which originate from the Strangford Lough SPA site are based on the SPA population as a proportion of the UK Western waters BDMPS (Furness, 2015). During both autumn and spring migration seasons, breeding adult common terns from the Strangford Lough SPA make up 1.09% of the total BDMPS population. The same percentage of impacts are therefore attributable to birds from this SPA during these times of year.

## Potential effects on the qualifying feature from the Project-alone

1041. The common tern qualifying feature of the Strangford Lough SPA has been screened into the assessment due to the potential risk of collision.

### Operation and maintenance phase collision risk

*Project-alone*

1042. Information for collision risk on breeding adult common terns belonging to the Strangford Lough SPA population is presented in **Table 8.13**. Collision estimates, calculated using the sCRM, are presented by biological season. A summary of the annual outputs and the corresponding increase in the annual baseline mortality rate is also presented. Parameters used in the sCRM were agreed with Natural England during the ETG process and are described in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES.

1043. Based on the mean collision rates, the annual total of breeding adult common terns from the Strangford Lough SPA at risk of collision as a result of the Project is 0.00. This would result in no detectable increase (0.14%) in the existing mortality of the SPA breeding population.

*Table 8.49 Predicted seasonal and annual collision mortality (Stochastic model Option 2, avoidance rate 0.991 (±0.0004)) for breeding adult common terns at the windfarm site, apportioned to Strangford Lough SPA, with corresponding increases to baseline mortality of the population*

| | Breeding Season | Autumn Migration | Non-breeding/winter | Spring Migration | Annual |
|---|---|---|---|---|---|
| **Period** | **Jun-Jul** | **Aug-Sep** | **Oct-Mar** | **Apr-May** | **Jan-Dec** |
| **Total collisions (mean and 95% Cis)** | 0.00 (0.00-0.00) | 0.14 (0.01-0.37) | 0.00 (0.00-0.00) | 0.08 (0.00-0.22) | 0.22 (0.01-0.60) |
| **% apportioned to the SPA** | 0.0% | 1.09% | 0.0% | 1.09% | - |
| **Total SPA collisions (mean and 95% Cis)** | 0.00 (0.00-0.00) | 0.00 (0.00-0.00) | 0.00 (0.00-0.00) | 0.00 (0.00-0.00) | 0.00 (0.00-0.01) |
| **Mortality increase[2] (mean and 95% Cis)** | **0.00%** (0.00-0.00%) | **0.09%** (0.00-0.25%) | **0.00%** (0.00-0.00%) | **0.05%** (0.00-0.15%) | **0.14%** (0.01-0.40%) |

[1] May overlaps breeding and spring migration period, has been included in migration period as birds present at the windfarm site are considered most likely to be migrants.

[2] Assuming predicted annual SPA adult mortality of 105 birds (898 x 0.117)

1044. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that no detectable changes in mortality rates would occur on this population for the mean monthly collision estimates for the Project, or for the upper 95% CI collision estimate.

1045. **It is concluded that predicted common tern mortality due to collision at the windfarm site would not adversely affect the integrity of the Strangford Lough SPA and Ramsar**.

1046. The confidence in the assessment is high. The evidence used to define the CRM input parameters presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES is of high applicability and quality. Whilst there is uncertainty around some of the input parameters (e.g. avoidance rate), the rates selected are considered to be sufficiently precautionary based on expert opinion to provide confidence that collision rates are not underestimated.

*In-combination*

1047. As no measurable effects on common tern are predicted as a result of the Project-alone, there would be no material contribution to the effects of other plans or projects in-combination. **It is therefore concluded that is no potential for the Project-alone or in-combination to have an adverse effect on the integrity of the Strangford Lough SPA and Ramsar**.

## 8.23 Copeland Islands SPA

1048. Copeland Islands SPA is located approximately 149km from the windfarm site.

### 8.23.1 Description of designation

1049. Copeland Islands SPA is composed of three islands, Big Copeland, Light House Island and Mew Island, lying off the north-east coast of the Outer Ards SPA. The islands are sites for breeding seabirds, with Big Copeland and Lighthouse Island being home to the main colonies. Important breeding and wintering populations of Eider Duck occur. Notable breeding populations of wader species also occur on Big Copeland.

#### 8.23.1.1 Conservation objectives

1050. The overarching conservation objective for the SPA is 'to maintain each feature in favourable condition.' For the qualifying features, the objectives are:

- To maintain or enhance the population of the qualifying species
- Fledging success sufficient to maintain or enhance population
- To maintain or enhance the range of habitats utilised by the qualifying species
- To ensure that the integrity of the site is maintained
- To ensure there is no significant disturbance of the species
- To ensure that the following are maintained in the long term:
    - Population of the species as a viable component of the site
    - Distribution of the species within site
    - Distribution and extent of habitats supporting the species
    - Structure, function and supporting processes of habitats supporting the species

### 8.23.2 Assessment

#### 8.23.2.1 Manx shearwater

**Status**

1051. Manx shearwater is listed as a qualifying species of this SPA.

1052. The SPA population has been cited as 4,800 pairs (2000-02) (Northern Ireland Environment Agency (NIEA), 2010), and the conservation objectives document identified a five-year mean (to 2010) of 5,903 pairs (11,806 adults) (NIEA, 2015). No more recent counts were identified in the SMP database.

1053. Based on the most recent SPA population of breeding adults, and an annual breeding adult baseline mortality rate of 0.13 (Horswill and Robinson, 2015), 1,535 breeding adults from the SPA population would be expected to die each year.

**Functional linkage and seasonal apportionment of potential effects**

1054. The windfarm site is situated approximately 149km from Copeland Islands SPA, at its nearest point; the across-sea distance is approximately 162km. The mean maximum foraging range of Manx shearwater is 1,347km (±1,019km) (Woodward *et al*., 2019). The windfarm site is therefore within the mean maximum foraging range of Manx shearwaters from the Copeland Islands SPA.

1055. Two studies have been identified that document results of tracking of Manx shearwaters from the Copeland Islands SPA colony; Freeman *et al.,* (2013) and Padget *et al.,* (2019). Both studies indicated that birds from the Copeland Islands colony predominantly foraged in the areas to the north of the colony, off the west coast of Scotland, and to the south, between the Isle of Man and the Northern Ireland coast. The Padget *et al.,* (2019) study also showed some activity to the east of the Isle of Man, but no tracks passed close to the windfarm site itself.

1056. A number of SPA and non-SPA Manx shearwater colonies are located in and around the UK Western Waters BDMPS area, all of which are within the mean maximum foraging range of this species. For a review of these sites see **Section 8.21.3.1**.

1057. The NatureScot apportioning tool (NatureScot, 2018) has been used to estimate the proportion of Manx shearwaters from each of the relevant SPAs present at the windfarm site during the breeding season. The apportioning to SPA and non-SPA colonies is set out in **Table 8.50**; refer also to **Appendix 12.1** of the ES for further information on the apportioning approach and results. Accordingly, 2.22% of impacts at the windfarm site during the breeding season are apportioned to Copeland Islands SPA.

*Table 8.50 Manx shearwater breeding season apportioning*

| Site | Apportioning rate |
|---|---|
| Glannau Aberdaron ac Ynys Enlli/Aberdaron Coast and Bardsey Island SPA | 8.63% |
| **Copeland Islands SPA** | **2.22%** |

| Site | Apportioning rate |
|---|---|
| Skomer, Skokholm and the Seas off Pembrokeshire / Sgomer, Sgogwm a Moroedd Penfro SPA | 76.54% |
| Rum SPA | 8.44% |
| St Kilda SPA | 0.20% |
| Cruagh Island SPA (transboundary site) | 0.10% |
| Blasket Islands SPA (transboundary site) | 0.61% |
| Deenish Island and Scariff Island SPA (transboundary site) | 0.08% |
| Puffin Island SPA (transboundary site) | 0.22% |
| Skelligs SPA (transboundary site) | 0.03% |
| Non-SPA colonies | 2.91% |

1058. During the pre- and post-breeding periods, breeding Manx shearwaters from the Copeland Islands SPA migrate through UK waters. The relevant reference population is considered to be the UK Western Waters BDMPS. This consists of 1,580,895 individuals during the post-breeding (August-early October) and return migration (late march-May) periods.

1059. Estimates of the proportion of Manx shearwaters present at the windfarm site which originate from the Copeland Islands SPA during the post-breeding and return migration periods (and therefore the proportion of predicted mortalities from the SPA population) are based on the SPA population (i.e. 11,806 adults) as a proportion of the UK Western Waters BDMPS during the relevant season. During the post-breeding and return migration periods, 0.75% of impacts are considered to affect birds from the SPA (Furness, 2015).

### Potential effects on the qualifying feature from the Project-alone

1060. The Manx shearwater qualifying feature of the Copeland Islands SPA has been screened into the assessment due to the potential risk of disturbance, displacement and barrier effects during the construction and decommissioning, and operation and maintenance phases of the Project.

#### *Construction and decommissioning phase disturbance/displacement/barrier effects*

1061. Effects during the construction and decommissioning phases of the Project are considered unlikely, given the transient presence of the species and low susceptibility to disturbance related impacts; refer to **Paragraph 1063**. However, in accordance with feedback received from Natural England and NRW, a precautionary estimation of construction and decommissioning phase disturbance, displacement and barrier effects has been undertaken assuming 50% of the operational phase effect.

1062. Applying 50% reduction to the operational values presented in **Table 8.51**, and based on mean density, predicted mortality would be between zero and five birds (30-70% displacement and 1-10% mortality of displaced birds). Using realistic values (i.e. mean density, 50% displacement and 1% mortality), there would be an annual increase in mortality of less than one (0.3) birds, which is equivalent to a 0.02% increase in background mortality for the SPA population. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. **Accordingly, no significant effects on Manx shearwater are predicted during the construction and decommissioning phases, and it is concluded that there would be no adverse effect on the integrity of Copeland Islands SPA**.

**Operation and maintenance phase disturbance/displacement/barrier effects**

1063. Manx shearwater are generally considered to have a low susceptibility to disturbance and displacement (Furness *et al.*, 2013). See **Section 8.21.3.1** for summary of effects from Dierschke *et al*., (2016) and Bradbury *et al.,* (2014).

1064. Displacement effects for Manx shearwater for the Project were assessed during the breeding, autumn migration and spring migration periods, based on an unportioned peak mean population of 4,705, 2,650 and 1,617 birds respectively, calculated for the windfarm site and a 2km buffer, in line with recommendations within the SNCB guidance (SNCBs, 2017). The displacement matrices used to calculate potential impacts are presented in **Appendix 12.1** of the ES and summarised in **Table 8.51**. The application of the same displacement rate to the OWF and the 2km buffer, to determine the total number of birds subject to displacement, is precautionary, as in reality the displacement rate is likely to fall with distance from the windfarm site.

1065. A displacement rate of 30-70% and mortality rate of 1-10% has been presented. Given that 10% would represent a rate close to the expected ‘natural’ annual mortality (0.13), this rate is considered very unlikely. Accordingly, a 1% mortality rate is considered to be most appropriate, with the upper end of this range likely to be precautionary. Given the very extensive foraging range of this species (Woodward *et al.*, 2019), there may be no mortality costs to displacement from the relatively very small footprints of OWFs.

*Table 8.51 Manx shearwater – predicted operation and maintenance phase displacement and mortality from Copeland Islands SPA*

| Mean peak abundance estimate type | Mean peak abundance estimate | Number of SPA breeding adults present by season[1] | Annual mortality range[2] | Annual baseline mortality increase range[3] |
|---|---|---|---|---|
| Upper 95% CI | 10,010 (breeding)<br>4,447 (autumn)<br>4,711 (spring)<br>19,168 (year round) | 222 (breeding)<br>33 (autumn)<br>35 (spring)<br>291 (year round) | 1-20 | 0.06-1.33% |
| Mean | 4,705 (breeding)<br>2,650 (autumn)<br>1,617 (spring)<br>8,972 (year round) | 104 (breeding)<br>20 (autumn)<br>12 (spring)<br>136 (year round) | 0-10 | 0.03-0.62% |
| Lower 95% CI | 783 (breeding)<br>1,308 (autumn)<br>0 (spring)<br>2,092 (year round) | 17 (breeding)<br>(autumn)<br>0 (spring)<br>27 (year round) | 0-2 | 0.01-0.12% |

[1] During the breeding season, assumes 2.2% of recorded birds are adults from the SPA population (11,806), and 0.75% during the autumn and spring migration periods

[2] Assumes displacement rates of 30-70% and mortality rates of 1-10%

[3] Background population is Copeland Islands SPA breeding adults (11,806 individuals), adult age class annual mortality rate of 13% (Horswill and Robinson, 2015)

1066. Using realistic values (i.e. mean density, 50% displacement and 1% mortality), there would be an annual increase in mortality of <1 bird, representing a 0.04% increase in mortality rate. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. **Accordingly, no significant effects on Manx shearwater are predicted during the operation and maintenance phase, and it is concluded that there is no potential for the Project to have an adverse effect on the integrity of Copeland Islands SPA**.

1067. A review of the potential effects of artificial light on Manx shearwaters is presented in Section 12.6.3.1 of **Chapter 12 Offshore Ornithology** of the ES. This concludes that lighting associated with the Project is very unlikely to significantly affect disturbance and displacement effects on Manx shearwater, and therefore the conclusions of the assessment are unchanged.

1068. The confidence in the assessment is high. The evidence used to set the displacement rates presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES is of high applicability and quality. Whilst there was limited available evidence to inform mortality rates, 1% is considered to be sufficiently precautionary based on expert opinion.

### Potential effects in-combination with other projects

1069. No in-combination effects are predicted during the construction and decommissioning phases. This is because it is unlikely that there would be significant temporal and/or spatial overlap with other plans or projects, and due to the negligible effects predicted from the project alone.

1070. During the operation and maintenance phase, the in-combination assessment for Manx shearwaters from Copeland Islands SPA has been undertaken in accordance with the approach presented in **Section 8.1**. The total population apportioned to the SPA at risk of displacement is estimated to be 194 breeding adults (**Appendix 12.1** of the ES). Annual in-combination displacement and mortality rates for birds from Copeland Islands SPA are presented in **Table 8.52**.

1071. Assuming a displacement rate of 70% and a mortality rate of 10% of displaced birds, 14 breeding adult SPA birds would be lost to displacement annually. This would increase the existing mortality within the SPA population (1,535 breeding adult birds per year) by 0.89%. Using a realistic displacement rate of 50%, and a mortality rate for displaced birds of 1%, the annual in-combination mortality would be <1 bird. This would increase the existing mortality within this population by 0.06%. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that detectable changes in mortality rates will not occur due to the level or mortality predicted if the more realistic rates for mortality are used.

1072. It is concluded that predicted Manx shearwater mortality due to of operational phase displacement impacts of the Project in-combination with other projects, would not adversely affect the integrity of Copeland Islands SPA. This accords with the conclusions of the Round 4 offshore wind leasing HRA (NIRAS, 2021), which concluded no adverse effect on site integrity (for all SPAs) on the basis of the low vulnerability to OWFs and low density of this species within Round 4 areas.

1073. It is noted that limited or no data are available from five historic projects that may have the potential to contribute to the in-combination effect on this feature (Burbo Bank, Walney 1&2, Gwynt y Môr, Rhyl Flats and Robin Rigg). As set out in the cumulative assessment for Manx shearwater presented in **Chapter 12 Offshore Ornithology** of the ES, in each case the assessments for these projects concluded no impact, or 'low' or 'very low' significance effects on this species. In order to reach a threshold where a significant effect might be possible (i.e. an increase in background mortality >1% affecting the SPA Manx shearwater population, assuming realistic displacement rates of 50%/1%), these historic projects would need contribute approximately 2,875 additional birds annually to the total potentially impacted population. This would equate to approximately 575 birds apportioned to the SPA at each project site annually. Given that the largest contribution from single project where data are available is 136 birds (for the Project), it is considered extremely unlikely that such high contributions could arise from these historic projects. Accordingly, it is concluded that these additional projects would not affect the conclusion of the assessment.

*Table 8.52 In-combination year-round displacement matrix for Manx shearwater from Copeland Islands SPA*

| Annual | Mortality | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| **Displacement** | **1%** | **2%** | **3%** | **4%** | **5%** | **10%** | **20%** | **30%** | **50%** | **80%** | **100%** |
| **10%** | 0 | 0 | 1 | 1 | 1 | 2 | 4 | 6 | 10 | 16 | 19 |
| **20%** | 0 | 1 | 1 | 2 | 2 | 4 | 8 | 12 | 19 | 31 | 39 |
| **30%** | **1** | **1** | **2** | **2** | **3** | **6** | 12 | 18 | 29 | 47 | 58 |
| **40%** | **1** | **2** | **2** | **3** | **4** | **8** | 16 | 23 | 39 | 62 | 78 |
| **50%** | **1** | **2** | **3** | **4** | **5** | **10** | 19 | 29 | 49 | 78 | 97 |
| **60%** | **1** | **2** | **4** | **5** | **6** | **12** | 23 | 35 | 58 | 93 | 117 |
| **70%** | **1** | **3** | **4** | **5** | **7** | **14** | 27 | 41 | 68 | 109 | 136 |
| **80%** | 2 | 3 | 5 | 6 | 8 | 16 | 31 | 47 | 78 | 124 | 156 |
| **90%** | 2 | 4 | 5 | 7 | 9 | 18 | 35 | 53 | 88 | 140 | 175 |
| **100%** | 2 | 4 | 6 | 8 | 10 | 19 | 39 | 58 | 97 | 156 | 194 |

Note: The cells show the number of birds predicted to die (rounded to the nearest integer) at a given rate of displacement and mortality. Highlighted cells are considered to be the most realistic scenario, in accordance with SNCB advice (SNCBs, 2022).

## 8.24 Larne Lough SPA and Ramsar

1074. Larne Lough SPA and Ramsar site is located approximately 166km from the windfarm site.

### 8.24.1 Description of designation

1075. Larne Lough is situated on the County Antrim coast in the east of Northern Ireland. The SPA covers the inter-tidal area and all islands within the Lame Lough estuary south of the harbour area. Breeding seabirds occur on both the natural island known as Swan Island and the artificial island known as Blue Circle Island. The site boundary is entirely coincident with that of the Lame Lough Area of Special Scientific Interest. The SPA boundary is also entirely coincident with that of the Lame Lough Ramsar Site.

### 8.24.2 Conservation objectives

1076. The overarching conservation objective for the SPA is 'to maintain each feature in favourable condition.' For the qualifying features, the objectives are:

- To maintain or enhance the population of the qualifying species
- Fledging success sufficient to maintain or enhance population
- To maintain or enhance the range of habitats utilised by the qualifying species
- To ensure that the integrity of the site is maintained
- To ensure there is no significant disturbance of the species
- To ensure that the following are maintained in the long term:
    - Population of the species as a viable component of the site
    - Distribution of the species within the site
    - Distribution and extend of habitats supporting the species
    - Structure, function and supporting processes of habitats supporting the species

### 8.24.3 Assessment

1077. One qualifying feature of Larne Lough SPA and Ramsar site has been screened into the Appropriate Assessment (**Table 5.2**): Sandwich tern.

#### 8.24.3.1 Sandwich tern

##### Status

1078. The Larne Lough SPA breeding Sandwich tern population at classification was 192 pairs, or 384 breeding adults, for the period 1993 – 1997. Following renotification in 2015 the mean population was cited as 413 pairs, or 826 breeding adults, for the period 2010 – 2014 (DoE, 2015). Furness (2015) gave the SPA population as 257 birds or 514 adults in 2013. The most recent count was 1,113 pairs (AON), or 2,226 breeding adults, in 2019 (JNCC, 2023a); this is used as the reference population for the assessment.

1079. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult mortality rate of 0.102 (1 – 0.898; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 227 breeding adults.

##### Functional linkage and seasonal apportionment of potential effects

1080. The mean maximum foraging range of Sandwich tern is 34.3km (±23.2km) and the maximum foraging range is 80km (Woodward *et al*., 2019). The Project is located approximately 166km from Larne Lough SPA, which means that the Project is beyond the maximum foraging range of Sandwich terns from the SPA. No impacts during the breeding season from the Project are therefore apportioned to Sandwich terns breeding at this SPA.

1081. Outside the breeding season breeding Sandwich terns are assumed to range widely and to mix with birds of all ages from breeding colonies in the UK and further afield. The relevant background population is considered to be the UK Western waters BDMPS, consisting of 10,761 individuals during autumn migration (July to September), and spring migration (March to May) (Furness, 2015).

1082. Estimates of the proportion of Sandwich terns present at the Project site during the autumn and spring migration seasons which originate from the Larne Lough SPA site are based on the SPA population as a proportion of the UK Western waters BDMPS (Furness, 2015). During both autumn and spring migration seasons, breeding adult Sandwich terns from Strangford Lough SPA make up 4.78% of the total BDMPS population. The same percentage of impacts are therefore attributable to birds from this SPA during these times of year.

##### Potential effects on the qualifying feature from the Project-alone

1083. The Sandwich tern qualifying feature of the Larne Lough SPA has been screened into the assessment due to the potential risk of collision.

## Operation and maintenance phase collision risk

*Project-alone*

1084. Information for collision risk on breeding adult Sandwich terns belonging to the Larne Lough SPA population is presented in **Table 8.53**. Collision estimates, calculated using the sCRM, are presented by biological season. A summary of the annual outputs and the corresponding increase in the annual baseline mortality rate is also presented. Parameters used in the sCRM were agreed with Natural England during the ETG process and are described in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES.

1085. Based on the mean collision rates, the annual total of breeding adult Sandwich terns from the Larne Lough SPA at risk of collision as a result of the Project is 0.02. This would result in no detectable increase (0.01%) in the existing mortality of the SPA breeding population.

*Table 8.53 Predicted seasonal and annual collision mortality (Stochastic model Option 2, avoidance rate 0.991 (±0.0004)) for breeding adult Sandwich terns at the windfarm site, apportioned to Larne Lough SPA, with corresponding increases to baseline mortality of the population*

| | Breeding Season | Autumn Migration | Non-breeding/winter | Spring Migration | Annual |
|---|---|---|---|---|---|
| Period | Apr-Aug | Sep | Oct-Feb | Mar | Jan-Dec |
| Total collisions (mean and 95% Cis) | 0.00 (0.00-0.00) | 0.33 (0.02-1.07) | 0.00 (0.00-0.00) | 0.00 (0.00-0.00) | 0.33 (0.02-1.07) |
| % apportioned to the SPA | 0.0% | 4.78% | 0.0% | 4.78% | - |
| Total SPA collisions (mean and 95% Cis) | 0.00 (0.00-0.00) | 0.02 (0.00-0.05) | 0.00 (0.00-0.00) | 0.00 (0.00-0.00) | 0.02 (0.00-0.05) |
| **Mortality increase**[1] (**mean** and 95% Cis) | **0.00%** (0.00-0.00%) | **0.01%** (0.00-0.02%) | **0.00%** (0.00-0.00%) | **0.00%** (0.00-0.00%) | **0.01%** (0.00-0.02%) |
| [1] Assuming predicted annual SPA adult mortality of 227 birds (2,226 x 0.102) | | | | | |

1086. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that no detectable changes in mortality rates would occur on this population for the mean monthly collision estimates for the Project, or for the upper 95% CI collision estimate.

1087. **It is concluded that predicted Sandwich tern mortality due to collision at the windfarm site would not adversely affect the integrity of the Larne Lough SPA and Ramsar site.**

1088. The confidence in the assessment is high. The evidence used to define the CRM input parameters presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES is of high applicability and quality. Whilst there is uncertainty around some of the input parameters (e.g. avoidance rate), the rates selected are considered to be sufficiently precautionary based on expert opinion to provide confidence that collision rates are not underestimated.

*In-combination*

1089. As no measurable effects on Sandwich tern are predicted as a result of the Project-alone, there would be no material contribution to the effects of other plans or projects in-combination. **It is therefore concluded that is no potential for the Project-alone or in-combination to have an adverse effect on the integrity of the Larne Lough SPA and Ramsar site**.

## 8.25 Ailsa Craig SPA

1090. Ailsa Craig SPA is located approximately 177km from the windfarm site.

### 8.25.1 Description of designation

1091. Ailsa Craig SPA is an island situated in the outer part of the Firth of Clyde. Cliffs up to 100 metres encircle the island and provide nesting sites for a variety of seabirds, notably one of the largest Northern gannet colonies in the world. The boundary of Ailsa Craig SPA is coincident with Ailsa Craig SSSI. The seaward elements extend approximately 2km into the marine environment to include the seabed, water column and surface.

#### 8.25.1.1 Conservation objectives

1092. The overarching conservation objectives of the site are:

- To avoid deterioration of the habitats of the qualifying species or significant disturbance to the qualifying species, thus ensuring that the integrity of the site is maintained; and
- To ensure for the qualifying species that the following are maintained in the long term:
  - Population of the species as a viable component of the site
  - Distribution of the species within site
  - Distribution and extent of habitats supporting the species
  - Structure, function and supporting processes of habitats supporting the species; and
  - No significant disturbance of the species

### 8.25.2 Assessment

#### 8.25.2.1 Gannet

##### Status

1093. Gannet is listed as a qualifying species of this SPA.

1094. The SPA population at classification was cited as 32,460 pairs in 1995 (Furness, 2015, Stroud *et al*., 2016). Furness (2015) gave the breeding population of 27,130 pairs in 2004, or 54,260 individuals. The most recent count (2015) was 33,226 AOS, or 66,452 breeding adults (JNCC, 2022).

1095. Based on the most recent SPA population of breeding adults, and an annual breeding adult baseline mortality rate of 0.081 (Horswill and Robinson, 2015), 5,383 breeding adults from the SPA population would be expected to die each year.

**Functional linkage and seasonal apportionment of potential effects**

1096. The windfarm site is 177km from Ailsa Craig SPA. The mean maximum foraging range of gannet is 315.2km (±194.2km). The windfarm site is therefore within the mean maximum foraging range of gannets from the Ailsa Craig SPA.

1097. Modelled at-sea utilisation distributions of breeding adult birds during the breeding season have been published, based on GPS tracking data (Wakefield *et al*., 2013). These suggest that the windfarm site is on the edge of the core foraging range for breeding adult birds from Ailsa Craig SPA.

1098. One further UK SPA designated for gannet is located within the UK Western Waters BDMPS area and within the mean maximum foraging range +1SD; Grassholm SPA. This site is located approximately 239km from the windfarm site which is within the mean maximum foraging range of this species. The most recent population count for this site from the SMP database is 36,011 AOS (2015). Data presented by Wakefield *et al.,* (2013) indicate that the foraging ranges of gannets from different breeding colonies tend not to overlap, and that the windfarm site is located outside of the core foraging area for adult birds from Grassholm SPA. One transboundary site is also located within the mean maximum foraging range +1SD; Saltee Islands SPA, which is located approximately 265km from the windfarm site. As with Grassholm SPA, data presented in Wakefield *et al.,* (2013) indicated that birds from Saltee Islands SPA are unlikely to occur at the windfarm site during the breeding season.

1099. Two further UK sites are within the straight-line foraging distance of the windfarm site; Flamborough and Filey Coast SPA (212km) and Forth Islands SPA (239km). However, both sites are on the eastern UK coast, with an across-sea distance of >1000km, and, as gannets will not typically fly across land, are therefore considered geographically isolated from the windfarm site during the breeding season.

1100. All breeding adult gannets present at the windfarm site during the full breeding season (March to September (Furness, 2015)) are therefore assumed to originate from the Ailsa Craig SPA, even though non-breeding adults from a range of breeding colonies are also likely to be present.

1101. In addition, some of the gannets recorded at the windfarm site during the breeding season will be sub-adult birds. During the full breeding season, 1,255 gannets were recorded during the baseline surveys. Of these, 572 birds were able to be assigned to an age class, and of these, 422 birds (73.8% of those assigned to an age class) were classified as adults. It is therefore assumed that this proportion of gannets recorded at the windfarm site during the full breeding season were breeding adult birds from Ailsa Craig SPA.

1102. Outside the breeding season breeding gannets, including those from the Ailsa Craig SPA, are not constrained by requirements to visit nests to incubate eggs or provision chicks. At this time, they are assumed to range more widely and to mix with gannets of all age classes from breeding colonies in the UK and further afield. The background population during these seasons is the UK Western Waters BDMPS. This consists of 545,954 individuals during autumn migration (September to November), and 661,888 individuals during spring migration (December to March) (Furness, 2015).

1103. Estimates of the proportion of gannets present at the windfarm site which originate from the Ailsa Craig SPA during the non-breeding season (and therefore the proportion of predicted mortalities from the SPA population) are based on the SPA population (i.e. 54,260 breeding adults) as a proportion of the UK Western Waters BDMPS during the relevant season. During autumn migration and spring migration, 9.9%, and 8.2% of impacted birds are considered to originate from the SPA respectively (Furness, 2015).

## Potential effects on the qualifying feature

1104. The gannet qualifying feature of the Ailsa Craig SPA has been screened into the assessment due to the potential risk of collision and operational phase displacement/barrier effects during the operation and maintenance phase of the Project.

### Operation and maintenance phase displacement/barrier effects

1105. Gannets have shown a low level of sensitivity to ship and helicopter traffic (Garthe and Hüppop, 2004, Furness and Wade, 2012, Furness *et al.*, 2013), but appeared to be more sensitive to displacement from structures such as offshore wind turbines (Wade *et al.*, 2016). Cook *et al.,* (2018) reviewed a number of studies of displacement of gannets from offshore windfarms. Where quantified, macro-avoidance rates (the percentage of birds taking action to avoid entering the wind turbine array) of 64-100% were reported. 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 Egmond aan Zee offshore windfarm (Krijgsveld *et al.*, 2011) was appropriate for this species. A study of seabird flight behaviour at Thanet offshore windfarm, not included in the above review, found a macro-avoidance rate of 79.7% for gannets approaching within 3km of the windfarm (Skov *et al*., 2018).

1106. Displacement effects for gannet for the Project were assessed during the breeding, autumn migration and spring migration periods, based on a peak mean population of 809, 189 and 16 birds respectively, calculated for the windfarm site and a 2km buffer, in line with recommendations within the SNCB guidance (SNCB 2017). The displacement matrices used to calculate potential impacts are presented in **Appendix 12.1** of the ES, and summarised in **Table 8.54**. The inclusion of all birds within the 2km buffer, to determine the total number of birds subject to displacement, is precautionary, as in reality the avoidance rate is likely to fall with distance from the windfarm site.

1107. A displacement rate of 60-80% and mortality rate of 1% has been presented. A maximum 1% mortality value has been selected firstly because gannet are known to possess high habitat flexibility (Furness and Wade, 2012). This suggests that displaced birds will readily find alternative habitats including foraging areas. Secondly, no evidence of displacement-induced mortality has been identified, which means there is limited justification for setting predicted mortality rates at a higher level. Given the extensive foraging range of this species (Woodward *et al.*, 2019), there may be no mortality costs to displacement from the relatively very small footprints of OWFs.

Table 8.54 Gannet – predicted operation and maintenance phase displacement and mortality from Ailsa Craig SPA

| Mean peak abundance estimate type | Mean peak abundance estimate | Number of SPA breeding adults present by season[1] | Annual mortality range[2] | Annual baseline mortality increase range[3] |
|---|---|---|---|---|
| Upper 95% CI | 809 (breeding)<br>189 (autumn)<br>16 (spring)<br>1,014 (year round) | 597 (breeding)<br>19 (autumn)<br>1 (spring)<br>605 (year round) | 4-5 | 0.07-0.09% |
| Mean | 541 (breeding)<br>124 (autumn)<br>(spring)<br>673 (year round) | 399 (breeding)<br>12 (autumn)<br>1 (spring)<br>404 (year round) | 2-3 | 0.05-0.06% |
| Lower 95% CI | 160 (breeding)<br>0 (autumn)<br>0 (spring)<br>160 (year round) | 118 (breeding)<br>0 (autumn)<br>0 (spring)<br>116 (year round) | 1-1 | 0.01-0.02% |

[1] During the breeding season, assumes 73.8% of recorded birds are adults, and 100% of these are from the SPA population (66,452). During autumn and spring migration periods, 9.9% and 8.2% of birds are assumed to be breeding adults from the SPA population.

[2] Assumes displacement rates of 60-80% and mortality rate of 1%

[3] Background population is Ailsa Craig SPA breeding adults (66,452 individuals), adult age class annual mortality rate of 8.1% (Horswill and Robinson, 2015)

1108. Using the maximum potential mortality value, there would be an annual increase in mortality of 0.09%. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. Accordingly, no significant effects on gannet are predicted during the operation and maintenance phase, and **it is concluded that there would be no potential for the Project-alone to have an adverse effect on the integrity of Ailsa Craig SPA.**

1109. The confidence in the assessment is high for several reasons. Firstly, the evidence used to set the displacement rates presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES submitted alongside this RIAA is of high applicability and quality. Whilst there is limited available evidence to inform mortality rates, 1% is considered to be sufficiently precautionary based on expert opinion. Finally, the conclusion of the assessment is the same irrespective of whether the mean or 95% upper CI mean peak abundances are used to calculate potential mortality and increases in the baseline mortality rate of the background population.

**Operation and maintenance phase collision risk**

1110. Information for collision risk on breeding adult gannets belonging to the Ailsa Craig SPA population is presented in **Table 8.55**. Collision estimates, calculated using the sCRM, are presented by biological season. A summary of the annual outputs and the corresponding increase in the annual baseline mortality rate is also presented. Parameters used in the sCRM were agreed with Natural England during the ETG process and are described in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES. In accordance with Natural England advice, a 70% macro-avoidance correction was applied to gannet abundance data used in the sCRM.

1111. Based on the mean collision rates, the annual total of breeding adult gannets from Ailsa Craig SPA at risk of collision as a result of the Project is 0.97. This would increase the existing mortality of the SPA breeding population by 0.02%.

*Table 8.55 Predicted seasonal and annual collision mortality (Stochastic model Option 2, avoidance rate 0.993 (±0.0003), plus 70% macro-avoidance) for breeding adult gannets at the windfarm site, apportioned to Ailsa Craig SPA, with corresponding increases to baseline mortality of the population*

| Period | Breeding Season<br>Apr-Aug | Autumn Migration<br>Sep-Nov | Non-breeding/winter<br>- | Spring Migration<br>Dec-Mar | Annual<br>Jan-Dec |
|---|---|---|---|---|---|
| Total collisions[1] (mean and 95% Cis) | 0.83<br>(0.00-3.35) | 0.14<br>(0.00-0.74) | - | 0.00 | 0.97<br>(0.00-4.10) |
| % apportioned to the SPA | 100.0% | 8.2% | - | 9.9% | - |
| Total SPA collisions (mean and 95% CIs) | 0.83<br>(0.00-3.35) | 0.01<br>(0.00-0.06) | - | 0.00<br>(0.00-0.00) | 0.84<br>(0.00-3.41) |
| Mortality increase[2] (mean and 95% CIs) | 0.02%<br>(0.00-0.06%) | 0.00%<br>(0.00-0.00%) | - | 0.00%<br>(0.00-0.00%) | 0.02%<br>(0.00-0.06%) |

[1] Breeding season collision values reduced to 73.8% of modelled value to reflect proportion of adult birds recorded during site surveys

[2] Assuming predicted annual SPA mortality of 5,382.61 birds (66,452 x 0.081)

1112. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that no detectable changes in mortality rates would occur on this population for the mean monthly collision estimates for the Project, or for the upper 95% CI collision estimate. Comments received from RSPB during the ETG process, indicating that they do not accept the 70% macro-avoidance rate for collision risk recommended by Natural England, are noted. However, even in the absence of this correction factor, the net increase in mortality would be well below the 1% threshold (0.05% (0.00-0.21%)).

1113. **It is concluded that based on predicted gannet mortality due to collision at the windfarm site there is no potential for the Project to have an adverse effect on the integrity of the Ailsa Craig SPA.**

1114. The confidence in the assessment is high. The evidence used to define the CRM input parameters presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** as part of the ES submitted alongside this RIAA is of high applicability and quality. Whilst there is uncertainty around some of the input parameters (e.g. avoidance rate), the rates selected are considered to be sufficiently precautionary based on expert opinion to provide confidence that collision rates are not underestimated.

### Combined displacement/barrier effects and collision risk

1115. The mean combined displacement and collision rates for breeding adult gannet from the Ailsa Craig SPA are presented in **Table 8.56**.

*Table 8.56 Predicted annual mean and 95% CI displacement and collision mortality of Ailsa Craig SPA breeding adult gannets, along with increases to existing annual mortality of the population*

| Annual displacement mortality[1] | Annual collision mortality | Annual displacement and collision mortality | Annual mortality increase[2] |
|---|---|---|---|
| 3<br>(1-4) | 0.84<br>(0.00-3.41) | 3.84<br>(1.00-7.41) | 0.07%<br>(0.02-0.14%) |
| [1] Assumes displacement rate of 0.70 and 1% mortality of displaced birds<br>[2] Background population is Ailsa Craig SPA breeding adults (66,452 individuals), adult age class annual mortality rate of 8.1% (Horswill and Robinson, 2015) | | | |

1116. The annual combined mortality of breeding adult gannets from the Ailsa Craig SPA is 3.84 (95% CIs 1.00-7.41). This would increase the existing mortality of the SPA breeding population by a maximum of 0.14%. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that no detectable changes in mortality rates are likely in a typical year of impacts due to the Project.

1117. **It is concluded that based on predicted gannet mortality due to the combined effects of operational phase displacement and collision there is no potential for the Project-alone or in-combination to have an adverse effect on the integrity of the Ailsa Craig SPA.**

1118. The confidence in the assessment is high, for the reasons provided in the individual displacement and collision assessments.

## Potential effects in-combination with other projects

### In-combination operation and maintenance phase displacement/barrier effects

1119. The in-combination assessment for gannets from Ailsa Craig SPA due to displacement and barrier effects has been undertaken in accordance with the approach presented in **Section 8.1**. The total population apportioned to Ailsa Craig SPA at risk of displacement is estimated to be 1,849 breeding adults (**Appendix 12.1** of the ES). Annual in-combination displacement and mortality rates for birds from Ailsa Craig SPA are presented in **Table 8.57**.

1120. Assuming a displacement rate of 70% and a mortality rate of 1% of displaced birds, 13 breeding adult SPA birds would be lost to displacement annually. This would increase the existing mortality within the SPA population (5,383 breeding adult birds per year) by 0.24%. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation.

*Table 8.57 In-combination year-round displacement matrix for gannet from Ailsa Craig SPA*

| Annual | Mortality | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| **Displacement** | **1%** | **2%** | **3%** | **4%** | **5%** | **10%** | **20%** | **30%** | **50%** | **80%** | **100%** |
| **10%** | 2 | 4 | 6 | 7 | 9 | 18 | 37 | 55 | 92 | 148 | 185 |
| **20%** | 4 | 7 | 11 | 15 | 18 | 37 | 74 | 111 | 185 | 296 | 370 |
| **30%** | 6 | 11 | 17 | 22 | 28 | 55 | 111 | 166 | 277 | 444 | 555 |
| **40%** | 7 | 15 | 22 | 30 | 37 | 74 | 148 | 222 | 370 | 592 | 740 |
| **50%** | 9 | 18 | 28 | 37 | 46 | 92 | 185 | 277 | 462 | 740 | 925 |
| **60%** | **11** | 22 | 33 | 44 | 55 | 111 | 222 | 333 | 555 | 888 | 1110 |
| **70%** | **13** | 26 | 39 | 52 | 65 | 129 | 259 | 388 | 647 | 1036 | 1295 |
| **80%** | **15** | 30 | 44 | 59 | 74 | 148 | 296 | 444 | 740 | 1184 | 1479 |
| **90%** | 17 | 33 | 50 | 67 | 83 | 166 | 333 | 499 | 832 | 1332 | 1664 |
| **100%** | 18 | 37 | 55 | 74 | 92 | 185 | 370 | 555 | 925 | 1479 | 1849 |

Note: The cells show the number of birds predicted to die (rounded to the nearest integer) at a given rate of displacement and mortality. Highlighted cells are considered to be the most realistic scenario, in accordance with SNCB advice (SNCBs, 2022).

### In-combination operation and maintenance phase collision risk

1121. The in-combination assessment for gannets from Ailsa Craig SPA due to collision risk has been undertaken in accordance with the approach presented in **Section 8.1**. **Table 8.58** sets out the predicted annual mortality for relevant projects, where data are available.

*Table 8.58 Gannet – predicted in-combination collision mortality from Ailsa Craig SPA*

| Project name | Predicted annual mortality (assuming 70% macro-avoidance for all projects) |
|---|---|
| Burbo Bank Extension | 1.09 |
| Ormonde | 0.94 |
| Walney 3&4 | 4.61 |
| Awel y Môr | 1.83 |
| Erebus | 0.41 |
| Twin Hub | 0.33 |
| Morgan Generation Assets | 0.82 |
| Mona | 0.78 |
| West of Orkney | 0.53 |
| White Cross | 0.09 |
| Morlais (underwater collision) | 0.00 |
| The Project | 0.84 |
| **Total** | **12.27** |

1122. The loss of 12.27 breeding adult SPA birds would increase the existing mortality within the SPA population (5,383 breeding adult birds per year) by 0.23%. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation.

### In-combination combined displacement/barrier effects and collision risk

1123. For the operation and maintenance phase, in-combination mortality values (for disturbance and displacement, assuming 1% mortality, and collision risk combined) for gannet at Ailsa Craig would be 25 birds.

1124. Based on the Ailsa Craig SPA population of 66,452 birds and a background mortality of 0.081 (5,383 birds per annum), an increase in mortality of 25 birds would increase background mortality by 0.47%.

1125. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. Accordingly, no significant effects on gannet are predicted during the operation and maintenance phase, and **it is concluded that there would no potential for the Project to have an adverse effect on the integrity of Ailsa Craig SPA, when considering the Project in-combination with other plans or projects.** This accords with the conclusions of the Round 4 offshore wind leasing HRA (NIRAS, 2021), which concluded no effect on site integrity on Ailsa Craig SPA, both alone and in-combination with other plans and projects.

1126. It is noted that limited or no data are available from some historic projects that may have the potential to contribute to the in-combination effect on this feature (Walney 1&2, West of Duddon Sands, Gwynt y Môr, Rhyl Flats, Robin Rigg and Burbo Bank). As set out in the cumulative assessment for gannet presented in **Chapter 12 Offshore Ornithology** of the ES, in all cases the assessments for these projects concluded 'low' or 'negligible' significance effects on this species. In order to reach a threshold where a significant effect might be possible (i.e. an increase in background mortality >1% affecting the Ailsa Craig gannet population), each historic project would need to cause an average annual mortality of approximately 4.8 gannets from the SPA. Given that the average mortality for projects where data are available is approximately 2.1 birds per annum, it is considered very unlikely that such high levels of mortality would result from the historic projects. Accordingly, it is concluded that these additional projects would not affect the conclusion of the assessment.

### 8.25.2.2 Lesser black-backed gull

#### Status

1127. The Ailsa Craig SPA breeding lesser black-backed gull population at classification was 1,800 pairs, or 3,600 breeding adults, in 1990 (Stroud *et al*., 2016). Having undergone a significant decline, Furness (2015) gave a population of 183 pairs, or 366 breeding adults, in 2010. The most recent count was 189 pairs (AON), or 378 breeding adults, in 2019 (JNCC, 2023a); this is used as the reference population for the assessment.

1128. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.115 (1 – 0.885; Horswill and Robinson; 2015), the expected annual mortality from the SPA population would be 43 breeding adults.

#### Functional linkage and seasonal apportionment of potential effects

1129. The mean maximum foraging range of lesser black-backed gull is 127km (±109km) and the maximum foraging range is 533km (Woodward *et al*., 2019).

The Project is located approximately 177km from Ailsa Craig SPA, which means the Project is beyond the mean maximum foraging range of breeding lesser black-backed gulls from the SPA, but within the mean maximum foraging range +1SD, and the maximum foraging range.

1130. The NatureScot apportioning tool (NatureScot, 2018) has been used to estimate the proportion of lesser black-backed gulls from each of the relevant SPAs present at the windfarm site during the breeding season. The apportioning to SPA and non-SPA colonies is set out in **Appendix 12.1** of the ES. 0.10% of impacts at the windfarm site during the breeding season are apportioned to Ailsa Craig SPA, assuming that only lesser black-backed gulls from coastal colonies are present at the windfarm site (the worst-case scenario, when compared to apportioning both coastal and inland colonies).

1131. Outside the breeding season, breeding lesser black-backed gulls from the SPA are assumed to range widely and to mix with lesser black-backed gulls of all ages from breeding colonies in the UK and beyond. The relevant non-breeding season reference population is the UK Western Waters BDMPS, consisting of 163,304 individuals during spring and autumn migration (March and September to October) and 41,159 during winter (November to February) (Furness, 2015).

1132. Furness (2015) estimated that 50% of the Ailsa Craig SPA breeding adults were present within the UK Western Waters and Channel BDMPS during the autumn and spring migration periods, representing 183 birds. During the winter period 20% of the population is estimated to be present, amounting to 73 birds. This represents 0.11% of the BDMPS population for the autumn and spring periods, and 0.18% during the winter period. Impacts to birds from the SPA during these periods have therefore been apportioned accordingly.

## Potential effects on the qualifying feature from the Project-alone

1133. The lesser black-backed gull qualifying feature of the Ailsa Craig SPA has been screened into the Appropriate Assessment due to the potential risk of collision.

### Operation and maintenance phase collision risk

1134. Information for collision risk on breeding adult lesser black-backed gulls belonging to the Ailsa Craig SPA population is presented in **Table 8.59**. Collision estimates, calculated using the sCRM, are presented by biological season. A summary of the annual outputs and the corresponding increase in the annual baseline mortality rate is also presented. Parameters used in the sCRM were agreed with Natural England during the ETG process and are described in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES.

1135. Based on the mean collision rates, the annual total of breeding adult lesser black-backed gulls from Ailsa Craig SPA at risk of collision as a result of the Project is less than one bird (0.00). This would increase the existing mortality of the SPA breeding population by 0.01%.

*Table 8.59 Predicted seasonal and annual collision mortality (Stochastic model Option 2, avoidance rate 0.994 (±0.0004)), for breeding adult lesser black-backed gulls at the windfarm site, apportioned to Ailsa Craig SPA, with corresponding increases to baseline mortality of the population*

| Period | Breeding Season Apr-Aug | Autumn Migration Sep-Oct | Non-breeding/winter Nov-Feb | Spring Migration Mar | Annual Jan-Dec |
|---|---|---|---|---|---|
| Total collisions[1] (mean and 95% Cis) | 1.44 (0.00-4.53) | 1.25 (0.00-5.63) | 0.15 (0.00-0.80) | 0.15 (0.00-0.94) | 2.98 (0.00-11.90) |
| % apportioned to the SPA | 0.10% | 0.11% | 0.18% | 0.11% | - |
| Total SPA collisions (mean and 95% Cis) | 0.00 (0.00-0.00) | 0.00 (0.00-0.00) | 0.00 (0.00-0.00) | 0.00 (0.00-0.00) | 0.00 (0.00-0.01) |
| Mortality increase[2] (mean and 95% Cis) | 0.00% (0.00-0.01%) | 0.00% (0.00-0.00%) | 0.00 (0.00-0.00) | 0.00% (0.00-0.00%) | 0.01% (0.00-0.03%) |

[1] Breeding season collision values reduced to 71.9% of modelled value to reflect proportion of adult birds recorded during site surveys

[2] Assuming predicted annual SPA mortality of 43 birds (378 x 0.115)

1136. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that no detectable changes in mortality rates would occur on this population for the mean monthly collision estimates for the Project, or for the upper 95% CI collision estimate.

1137. **It is concluded that based on predicted lesser black-backed gull mortality due to collision at the windfarm site there is no potential for the Project to have an adverse effect on the integrity of the Ailsa Craig SPA.**

1138. The confidence in the assessment is high. The evidence used to define the CRM input parameters presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** as part of the ES submitted alongside this RIAA is of high applicability and quality. Whilst there is uncertainty around some of the input parameters (e.g. avoidance rate), the rates selected are considered to be sufficiently precautionary, based on expert opinion, to provide confidence that collision rates are not underestimated.

### Potential effects on the qualifying feature in-combination with other projects

1139. As the Project would have no measurable effect on lesser black-backed gull populations from the Ailsa Craig SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of Ailsa Craig SPA, when assessed in-combination with other plans or projects**.

### 8.25.2.3 Kittiwake

### Status

1140. The Ailsa Craig SPA breeding kittiwake population at classification was 3,100 pairs, or 6,200 breeding adults, in 1990 (Stroud *et al*., 2016). Furness (2015) gave a population of 489 pairs, or 978 breeding adults, in 2013. The most recent count was 490 pairs, or 980 breeding adults, in 2021 (JNCC, 2023a); this is used as the reference population for the assessment.

1141. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.146 (1 – 0.854; Horswill and Robinson, 2015), the expected annual mortality from the SPA population would be 143 breeding adults.

### Functional linkage and seasonal apportionment of potential effects

1142. The mean maximum foraging range of kittiwake is 156.1km (±144.5km) and the maximum foraging range is 770km (Woodward *et al*., 2019). The Project is located approximately 177km from Ailsa Craig SPA, which means the Project is beyond the mean maximum foraging range of breeding kittiwakes

from the SPA, but within the mean maximum foraging range +1SD, and the maximum foraging range.

1143. The NatureScot apportioning tool (NatureScot, 2018) has been used to estimate the proportion of kittiwakes from each of the relevant SPAs present at the windfarm site during the breeding season. The apportioning to SPA and non-SPA colonies is set out in **Appendix 12.1** of the ES. 0.27% of impacts at the windfarm site during the breeding season are apportioned to Ailsa Craig SPA.

1144. Outside the breeding season, breeding kittiwakes from the SPA are assumed to range widely and to mix with kittiwakes of all ages from breeding colonies in the UK and beyond. The relevant non-breeding season reference population is the UK Western Waters and Channel BDMPS, consisting of 911,586 individuals during autumn migration (September to December) and 627,816 during spring migration (January to February) (Furness, 2015).

1145. Furness (2015) estimated that 60% of the Ailsa Craig SPA breeding adults are present within the UK Western Waters and Channel BDMPS during the autumn migration period, which is 587 birds. During the spring migration period 80% of the population is estimated to be present, which is 782 birds. This represents 0.06% and 0.12% of the BDMPS population for the autumn and spring periods respectively. During autumn migration and spring migration, 0.06%, and 0.12% of impacts are therefore considered to affect birds from the SPA (Furness, 2015).

## Potential effects on the qualifying feature from the Project-alone

1146. The kittiwake qualifying feature of the Ailsa Craig SPA has been screened into the Appropriate Assessment due to the potential risk of collision.

### Operation and maintenance phase collision risk

1147. Information for collision risk on breeding adult kittiwakes belonging to the Ailsa Craig SPA population is presented in **Table 8.60**. Collision estimates, calculated using the sCRM, are presented by biological season. A summary of the annual outputs and the corresponding increase in the annual baseline mortality rate is also presented. Parameters used in the sCRM were agreed with Natural England during the ETG process and are described in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES.

1148. Based on the mean collision rates, the annual total of breeding adult kittiwakes from Ailsa Craig SPA at risk of collision as a result of the Project is 0.05. This would increase the existing mortality of the SPA breeding population by 0.03%.

*Table 8.60 Predicted seasonal and annual collision mortality (Stochastic model Option 2, avoidance rate 0.993 (±0.0003)), for breeding adult kittiwakes at the windfarm site, apportioned to Ailsa Craig SPA, with corresponding increases to baseline mortality of the population*

| Period | Breeding Season<br>Mar-Aug | Autumn Migration<br>Sep-Dec | Non-breeding/winter<br>- | Spring Migration<br>Jan-Feb | Annual<br>Jan-Dec |
|---|---|---|---|---|---|
| Total collisions[1] (mean and 95% Cis) | 15.36 (4.17-33.87) | 8.49 (2.20-18.85) | - | 0.63 (0.00-1.46) | 24.48 (6.37-54.17) |
| % apportioned to the SPA | 0.27% | 0.06% | - | 0.12% | - |
| Total SPA collisions (mean and 95% Cis) | 0.04 (0.01-0.09) | 0.01 (0.00-0.06) | - | 0.00 (0.00-0.00) | 0.05 (0.01-0.11) |
| Mortality increase[2] (mean and 95% Cis) | 0.03% (0.01-0.06%) | 0.00% (0.00-0.00%) | - | 0.00% (0.00-0.00%) | 0.03% (0.01-0.07%) |

[1] Breeding season collision values reduced to 94.1% of modelled value to reflect proportion of adult birds recorded during site surveys

[2] Assuming predicted annual SPA mortality of 143 birds (980 x 0.146)

1149. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that no detectable changes in mortality rates would occur on this population for the mean monthly collision estimates for the Project, or for the upper 95% CI collision estimate.

1150. **It is concluded that based on predicted kittiwake mortality due to collision at the windfarm site there is no potential for the Project to have an adverse effect on the integrity of the Ailsa Craig SPA.**

1151. The confidence in the assessment is high. The evidence used to define the CRM input parameters presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** as part of the ES submitted alongside this RIAA is of high applicability and quality. Whilst there is uncertainty around some of the input parameters (e.g. avoidance rate), the rates selected are considered to be sufficiently precautionary, based on expert opinion, to provide confidence that collision rates are not underestimated.

### Potential effects on the qualifying feature in-combination with other projects

1152. As the Project would have no measurable effect on kittiwake populations from Ailsa Craig, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of Ailsa Craig SPA, when assessed in-combination with other plans or projects**.

### 8.25.2.4 Herring gull

### Status

1153. The Ailsa Craig SPA breeding herring gull population at classification was 2,250 pairs, or 5,500 breeding adults, in 1990 (Stroud *et al*., 2016). Having undergone a significant decline, Furness (2015) gave a population of 129 pairs, or 258 breeding adults, in 2013. The most recent count was 213 pairs (AON), or 426 breeding adults, in 2019 (JNCC, 2023a); this is used as the reference population for the assessment.

1154. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.166 (1 – 0.834; Horswill and Robinson, 2015), the expected annual mortality from the SPA population would be 71 breeding adults.

### Functional linkage and seasonal apportionment of potential effects

1155. The mean maximum foraging range of herring gull is 58.8km (±26.8km) and the maximum foraging range is 92km (Woodward *et al*., 2019). The Project is located approximately 177km from Ailsa Craig SPA, which means the Project is beyond the maximum foraging range of herring gulls from the SPA. No impacts during the breeding season from the Project are therefore apportioned to herring gulls breeding at this SPA.

1156. Outside the breeding season, herring gulls from the SPA are assumed to range widely and to mix with herring gulls of all ages from breeding colonies in the UK and beyond. The relevant non-breeding season reference population is the UK Western Waters BDMPS, consisting of 173,299 individuals during the non-breeding period (September to February) (Furness, 2015).

1157. Furness (2015) estimated that 80% of the Ailsa Craig SPA breeding adults are present within the UK Western Waters BDMPS during the non-breeding period, which is 206 birds. This represents 0.12% of the BDMPS population 0.12% of impacts to birds from the SPA are therefore apportioned during the non-breeding season.

## Potential effects on the qualifying feature from the Project-alone

1158. The herring gull qualifying feature of the Ailsa Craig SPA has been screened into the Appropriate Assessment due to the potential risk of collision.

### Operation and maintenance phase collision risk

1159. Information for collision risk on breeding adult herring gulls belonging to the Ailsa Craig SPA population is presented in **Table 8.61**. Collision estimates, calculated using the sCRM, are presented by biological season. A summary of the annual outputs and the corresponding increase in the annual baseline mortality rate is also presented. Parameters used in the sCRM were agreed with Natural England during the ETG process and are described in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES.

1160. Based on the mean collision rates, the annual total of breeding adult herring gulls from Ailsa Craig SPA at risk of collision as a result of the Project is less than one bird (0.00). This would increase the existing mortality of the SPA breeding population by 0.00%.

*Table 8.61 Predicted seasonal and annual collision mortality (Stochastic model Option 2, avoidance rate 0.994 (±0.0004)), for breeding adult herring gulls at the windfarm site, apportioned to Ailsa Craig SPA, with corresponding increases to baseline mortality of the population*

| Period | Breeding Season Apr-Aug | Autumn Migration Sep-Oct | Non-breeding/winter Nov-Feb | Spring Migration Mar | Annual Jan-Dec |
|---|---|---|---|---|---|
| Total collisions[1] (mean and 95% Cis) | 0.85 (0.00-7.72) | - | 2.38 (0.00-9.70) | - | 3.23 (0.00-13.41) |
| % apportioned to the SPA | 0.00% | - | 0.12% | - | - |
| Total SPA collisions (mean and 95% Cis) | 0.00 (0.00-0.00) | - | 0.00 (0.00-0.01) | - | 0.00 (0.00-0.01) |
| Mortality increase[2] (mean and 95% Cis) | 0.00% (0.00-0.00%) | - | 0.00% (0.00-0.02%) | - | 0.00% (0.00-0.02%) |

[1] Breeding season collision values reduced to 48.0% of modelled value to reflect proportion of adult birds recorded during site surveys

[2] Assuming predicted annual SPA mortality of 71 birds (426 x 0.166)

1161. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that no detectable changes in mortality rates would occur on this population for the mean monthly collision estimates for the Project, or for the upper 95% CI collision estimate.

1162. **It is concluded that based on predicted herring gull mortality due to collision at the windfarm site there is no potential for the Project to have an adverse effect on the integrity of the Ailsa Craig SPA**.

1163. The confidence in the assessment is high. The evidence used to define the CRM input parameters presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** as part of the ES submitted alongside this RIAA is of high applicability and quality. Whilst there is uncertainty around some of the input parameters (e.g. avoidance rate), the rates selected are considered to be sufficiently precautionary based on expert opinion to provide confidence that collision rates are not underestimated.

### Potential effects on the qualifying feature in-combination with other projects

1164. As the Project would have no measurable effect on herring gull populations from the Ailsa Craig SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of Ailsa Craig SPA, when assessed in-combination with other plans or projects**.

### 8.25.2.5 Guillemot

### Status

1165. The Ailsa Craig SPA breeding guillemot population at classification was 3,350 pairs, or 6,700 breeding adults, in 1990 (Stroud *et al*., 2016). Furness (2015) gave a population of 5,247 pairs, or 10,494 breeding adults, in 2013. The most recent count was 7,140 individuals in 2021 (JNCC, 2023a); this is used as the reference population for the assessment.

1166. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.061 (1 – 0.939; Horswill and Robinson, 2015), the expected annual mortality rate from the SPA population would be 436 breeding adults.

### Functional linkage and seasonal apportionment of potential effects

1167. The mean maximum foraging range of guillemot is 73.2km (±80.5km) and the maximum foraging range is 338km (Woodward *et al*., 2019). The Project is located approximately 177km from Ailsa Craig SPA, which means the Project is beyond the mean maximum foraging range +1SD of guillemots breeding at this SPA, but within the maximum foraging range. The maximum foraging range is a poor indicator of typical foraging behaviour. It would be expected that few birds or foraging trips will occur at this distance from the colony, and even fewer with any regularity. No impacts during the breeding season are therefore apportioned to birds breeding at this colony.

1168. Outside the breeding season, guillemots from the SPA are assumed to range widely and to mix with guillemots of all ages from breeding colonies in the UK and beyond. The relevant non-breeding season reference population is the UK Western Waters BDMPS, consisting of 1,139,220 individuals (August to February) (Furness, 2015). During the non-breeding season, it is estimated that 0.9% of birds present are considered to be breeding adults from the Ailsa Craig SPA, and impacts are apportioned accordingly. This is based on the SPA adult population from Furness (2015) as a proportion of the total UK Western Waters BDMPS.

## Potential effects on the qualifying feature

### Operation and maintenance phase disturbance / displacement / barrier effects

*Project-alone*

1169. The mean peak abundance of guillemots present within the windfarm site and 2km buffer during the non-breeding season was 8,315 (6,085-12,047) individuals (refer to **Appendix 12.1** of the ES). Of these, 75 birds (55-108) were likely to be breeding adults from the Ailsa Craig SPA.

1170. **Table 8.62** sets out the predicted impacts on guillemots from Ailsa Craig SPA during the non-breeding season. Displacement rates of 30% to 70% are considered for this species, along with a range of mortality rates of 1% to 10% of displaced birds (UK SNCBs, 2017).

1171. The available evidence suggested that the upper ranges of these displacement and mortality rates may be excessively precautionary. Whilst it is true that guillemots and razorbills tend to be displaced by OWFs they do not avoid them completely, and displacement rates vary between sites (MacArthur Green, 2019b). On average it was concluded that densities within OWFs are around half of the density found in the habitats around the OWF. At some OWFs there was also displacement of birds from a buffer zone surrounding it. The size of the buffer zone varied between OWFs and was generally less than 2km, with auk density increasing across the buffer zone as distance from turbines increased, up to the density in the wider area. Another recent review (APEM, 2022) found that the current displacement rates suggested by the SNCBs for guillemot and razorbill (30% to 70%) did not account for the quality of or confidence in the studies which were used to inform this position, and that studies where no significant effects were recorded were not accounted for during the provision of the advice. APEM (2022) suggested that in the case of Hornsea Project Four, an evidence-based displacement rate of 50% was appropriate. However, the study also recognised that larger displacement effects are possible.

1172. Mortality due to displacement could arise due to increased energy costs and/or decreased energy intake, if displacement results in increased flying time to avoid OWFs, and/or increased bird densities and competition for prey in areas of unimpacted habitat outside the OWF. Given that UK OWFs in the Irish and Celtic Seas represent a very small proportion of the available foraging habitat for guillemots and razorbills within UK Western Waters, the increase in density of auks outside OWFs due to displacement is likely to be negligible (MacArthur Green, 2019b). It is considered unlikely that the mortality rate due to displacement would be as high as the empirically estimated 6.1% annual mortality for adult guillemots that occurs due to the combination of 'natural' factors and anthropogenic activities (Horswill and Robinson, 2015). Indeed, it may be much lower; MacArthur Green (2019b) recommended precautionary rates of 50% displacement for guillemot and 1% mortality of displaced birds based on a review of available evidence. Modelling undertaken by APEM (2022) found that in the case of Hornsea Project Four, a mortality rate of 1% for displaced auks was considered to be precautionary, and that this may be an overestimate given that Hornsea Project Four is located towards the upper end of the mean maximum foraging range for guillemot from the closest designated sites. It is also noted that in the recent decision on Hornsea Project Four, the SoS determined that a rate of 70% displacement/2% mortality was appropriate to inform the assessment of effects on guillemots from Flamborough and Filey Coast SPA (DESNZ, 2023b).

1173. The full ranges of recommended displacement and mortality effects are considered by the assessment, along with evidence-based displacement and mortality rates of 50% and 1%, respectively (APEM, 2022; MacArthur Green, 2019b).

*Table 8.62 Guillemot – predicted operation and maintenance phase displacement and mortality from Ailsa Craig SPA*

| Mean peak abundance estimate type | Mean peak abundance estimate | Number of Ailsa Craig SPA breeding adults present (non-breeding season)[1] | Annual mortality range[2] | Annual baseline mortality increase range[3] |
|---|---|---|---|---|
| Upper 95% CI | 12,047 | 108 | 0-8 | 0.07-1.74% |
| Mean | 8,315 | 75 | 0-5 | 0.05-1.20% |
| Lower 95% CI | 6,085 | 55 | 0-4 | 0.04-0.88% |

[1] Assumes 0.9% of birds present during the non-breeding season are Ailsa Craig SPA breeding adults

[2] Assumes displacement rates of 30-70% and mortality rates of 1-10%

[3] Background mortality rate of 6.1% (Horswill and Robinson, 2015)

1174. Based on the mean peak abundances, the annual total of guillemots from the Ailsa Craig SPA at risk of displacement is 75 birds (**Table 8.62**). At displacement rates of 30% to 70%, and mortality rates of 1% to 10% for displaced birds, 0 to 5 SPA breeding adults would be predicted to die each year due to displacement from the Project.

1175. Assuming a displacement rate of 70% and a mortality rate of 10% of displaced birds, annual mortality within the SPA breeding adult population would increase by 1.20%. Using an evidence-based displacement rate of 50%, and a mortality rate for displaced birds of 1%, annual mortality in the population would instead increase by 0.09% (<1 bird).

1176. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that no detectable changes in mortality rates would occur under almost any combination of displacement and mortality rates when the mean peak abundance estimate assessments are considered. Mortality rate increases of over 1% are predicted for mean peak abundance estimate assessments only when a displacement rate of 70% and a mortality rate of 10% is considered. These displacement and mortality rates are much higher than evidence suggested would actually be the case. Use of the evidence-based displacement (50%) and mortality rate (1%) would result in a mortality increase of significantly less than 1%, as would a rate of 70%/2% agreed by the SoS in respect of Hornsea Project Four (DESNZ, 2023b).

1177. Increases of over 1% are also predicted if the upper 95% CIs for mean peak abundances are used as inputs to the assessment alongside a 10% mortality rate for displaced birds. The probability of this occurring is extremely small for two reasons. Firstly, the upper 95% CI for the mean peak abundances are highly unlikely to occur regularly at the windfarm site. Secondly, mortality rates for displaced birds of 10% are much higher than evidence suggested would actually be the case, and use of the evidence-based displacement (50%) and mortality rate (1%) (and also 70%/2%) would again result in a mortality increase of significantly less than 1%.

1178. **It is concluded that predicted guillemot mortality due to operational phase displacement for the project alone would not adversely affect the integrity of the Ailsa Craig SPA.**

1179. The confidence in the assessment is high for several reasons. Firstly, the evidence used to inform the evidence-based displacement rates is of high applicability and quality (based on the criteria discussed in **Chapter 12 Offshore Ornithology** of the ES). Whilst there is limited available evidence to inform mortality rates, 1% is considered to be sufficiently precautionary, based on expert opinion. This species is not regarded as being highly specialised in its habitat requirements (Bradbury *et al*., 2014; Furness and Wade, 2012; Garthe and Hüppop, 2004), and it is therefore anticipated that displaced birds will find alternative habitat in the vast majority of cases. Finally, the conclusion of the assessment is the same irrespective of whether the mean or 95% upper CI mean peak abundances are used to calculate potential mortality and increases in the baseline mortality rate of the background population, provided the evidence-based displacement and mortality rates are used.

*In-combination*

1180. The in-combination assessment for guillemots from Ailsa Craig SPA has been undertaken in accordance with the approach presented in **Section 8.1**. The total population apportioned to Ailsa Craig SPA at risk of displacement is estimated to be 447 breeding adults (**Appendix 12.1** of the ES). Annual in-combination displacement and mortality rates for birds from Ailsa Craig SPA are presented in **Table 8.63**.

1181. Assuming a displacement rate of 70% and a mortality rate of 10% of displaced birds, 31 breeding adult SPA birds would be lost to displacement annually. Including additional apportioned mortality from the Morlais and Holyhead tidal projects (total 0.28 birds), this would increase the existing mortality within the SPA population (436 breeding adult birds per year) by 7.26%. Using an evidence-based displacement rate of 50%, and a mortality rate for displaced birds of 1%, the annual in-combination displacement mortality would be 2 birds. This would increase the existing mortality within this population by 0.58%. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that detectable changes in mortality rates will not occur due to the level or mortality predicted if the more realistic rates for mortality are used.

1182. **It is concluded that predicted guillemot mortality due to of operational phase displacement impacts of the Project in-combination with other projects, would not adversely affect the integrity of Ailsa Craig SPA.**

*Table 8.63 In-combination year-round displacement matrix for guillemot from Ailsa Craig SPA*

| **Annual** | **Mortality** | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| **Displacement** | **1%** | **2%** | **3%** | **4%** | **5%** | **10%** | **20%** | **30%** | **50%** | **80%** | **100%** |
| **10%** | 0 | 1 | 1 | 2 | 2 | 4 | 9 | 13 | 22 | 36 | 45 |
| **20%** | 1 | 2 | 3 | 4 | 4 | 9 | 18 | 27 | 45 | 72 | 89 |
| **30%** | **1** | **3** | **4** | **5** | **7** | **13** | 27 | 40 | 67 | 107 | 134 |
| **40%** | **2** | **4** | **5** | **7** | **9** | **18** | 36 | 54 | 89 | 143 | 179 |
| **50%** | **2** | **4** | **7** | **9** | **11** | **22** | 45 | 67 | 112 | 179 | 224 |
| **60%** | **3** | **5** | **8** | **11** | **13** | **27** | 54 | 81 | 134 | 215 | 268 |
| **70%** | **3** | **6** | **9** | **13** | **16** | **31** | 63 | 94 | 157 | 251 | 313 |
| **80%** | 4 | 7 | 11 | 14 | 18 | 36 | 72 | 107 | 179 | 286 | 358 |
| **90%** | 4 | 8 | 12 | 16 | 20 | 40 | 81 | 121 | 201 | 322 | 403 |
| **100%** | 4 | 9 | 13 | 18 | 22 | 45 | 89 | 134 | 224 | 358 | 447 |

Note: The cells show the number of birds predicted to die (rounded to the nearest integer) at a given rate of displacement and mortality. Highlighted cells are considered to be the most realistic scenario, in accordance with SNCB advice (SNCBs, 2022).

# 8.26 Coquet Island SPA

1183. Coquet Island SPA is located approximately 210km from the windfarm site.

## 8.26.1 Description of designation

1184. Coquet Island is situated 1km off the coast of Northumberland. It is a small, flat-topped island with a plateau extent of approximately 7ha. The island consists of sandy soil and peat over a soft sandstone base. Low cliffs of up to 3.7m high result from earlier quarrying. Surrounding the island is a rocky upper shore and intertidal covering 15ha when fully exposed. There is a sandy beach on the south-west of the island and the southeast corner is shingle and rock. The qualifying bird species for Coquet Island SPA are Sandwich tern, roseate tern, common tern and Arctic tern.

## 8.26.2 Conservation objectives

1185. The SPA's conservation objectives are to ensure that the integrity of the site is maintained or restored as appropriate, and that the site contributes to achieving the aims of the Wild Birds Directive, by maintaining or restoring:

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

## 8.26.3 Assessment

1186. One qualifying features of Coquet Island SPA has been screened into the Appropriate Assessment (**Table 5.2**): breeding common tern. This species has also been screened in as a component of the seabird assemblage.

### 8.26.3.1 Common tern

#### Status

1187. The Coquet Island SPA breeding common tern population at classification was 1,189 pairs, or 2,378 breeding adults, for the period 2010 to 2014 (Natural England, 2017a). Furness (2015) gave the SPA population as 1,041 pairs or 2,082 individuals in 2013. The most recent count was 1,776 pairs (AON), or 3,552 breeding adults, in 2021 (JNCC, 2023a); this is used as the reference population for the assessment.

1188. Based on the published adult common tern mortality rate of 0.117 (1 – 0.883; Horswill and Robinson, 2015), the expected annual mortality of the SPA population is 416 breeding adults.

### Functional linkage and seasonal apportionment of potential effects

1189. The mean maximum breeding season foraging range of common tern is 18.0km (±8.9km) and the maximum foraging range is 30km (Woodward *et al.,* 2019). The Project is located approximately 210km from Coquet Island SPA, which means the Project is beyond the maximum foraging range of common terns from the SPA. No impacts from the Project during the breeding season are therefore apportioned to the Coquet Island SPA common tern colony.

1190. Outside the breeding season, breeding common terns are assumed to range widely and to mix with birds of all ages from breeding colonies in the UK and further afield. The relevant background population is considered to be the UK Western waters BDMPS, consisting of 64,659 individuals during autumn migration (late July to early September), and spring migration (April to May) (Furness, 2015).

1191. Estimates of the proportion of common terns present at the Project site during the autumn and spring migration seasons originating from the Coquet Island SPA site are based on the SPA population as a proportion of the UK Western waters BDMPS (Furness, 2015). Furness (2015) estimated that 30% of the Ailsa Craig SPA breeding adults are present within the UK Western waters BDMPS during the autumn and spring migration periods, which is 625 birds. Therefore, during both autumn and spring migration seasons, breeding adult common terns from the Coquet Island SPA make up 0.97% of the total BDMPS population. The same percentage of impacts are therefore attributable to birds from this SPA during these times of year.

### Potential effects on the qualifying feature from the Project-alone

1192. The common tern qualifying feature of the Coquet Island SPA has been screened into the assessment due to the potential risk of collision.

#### Operation and maintenance phase collision risk

*Project-alone*

1193. Information for collision risk on breeding adult common terns belonging to the Coquet Island SPA population is presented in **Table 8.64**. Collision estimates, calculated using the sCRM, are presented by biological season. A summary of the annual outputs and the corresponding increase in the annual baseline mortality rate is also presented. Parameters used in the sCRM were agreed with Natural England during the ETG process and are described in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES.

1194. Based on the mean collision rates, the annual total of breeding adult common terns from the Coquet Island SPA at risk of collision as a result of the Project is 0.00. This would result in no detectable increase (0.00%) in the existing mortality of the SPA breeding population.

*Table 8.64 Predicted seasonal and annual collision mortality (Stochastic model Option 2, avoidance rate 0.991 (±0.0004)) for breeding adult common terns at the windfarm site, apportioned to Coquet Island SPA, with corresponding increases to baseline mortality of the population*

| | Breeding Season | Autumn Migration | Non-breeding/winter | Spring Migration | Annual |
|---|---|---|---|---|---|
| Period | Jun-Jul | Aug-Sep | Oct-Mar | Apr-May | Jan-Dec |
| Total collisions (mean and 95% Cis) | 0.00 (0.00-0.00) | 0.14 (0.01-0.37) | 0.00 (0.00-0.00) | 0.08 (0.00-0.22) | 0.22 (0.01-0.60) |
| % apportioned to the SPA | 0.0% | 0.34% | 0.0% | 0.34% | - |
| Total SPA collisions (mean and 95% Cis) | 0.00 (0.00-0.00) | 0.00 (0.00-0.00) | 0.00 (0.00-0.00) | 0.00 (0.00-0.00) | 0.00 (0.00-0.01) |
| **Mortality increase**[2] (**mean** and 95% Cis) | **0.00%** (0.00-0.00%) | **0.00%** (0.00-0.01%) | **0.00%** (0.00-0.00%) | **0.00%** (0.00-0.00%) | **0.00%** (0.00-0.01%) |

[1] May overlaps breeding and spring migration period, has been included in migration period as birds present at the windfarm site are considered most likely to be migrants.

[2] Assuming predicted annual SPA adult mortality of 416 birds (3,552 x 0.117)

1195. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that no detectable changes in mortality rates would occur on this population for the mean monthly collision estimates for the Project, or for the upper 95% CI collision estimate.

1196. **It is concluded that predicted common tern mortality due to collision at the windfarm site would not adversely affect the integrity of the Coquet Island SPA.**

1197. The confidence in the assessment is high. The evidence used to define the CRM input parameters presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES is of high applicability and quality. Whilst there is uncertainty around some of the input parameters (e.g. avoidance rate), the rates selected are considered to be sufficiently precautionary, based on expert opinion, to provide confidence that collision rates are not underestimated.

*In-combination*

1198. As no measurable effects on common tern are predicted as a result of the Project-alone, there would be no material contribution to the effects of other plans or projects in-combination. **It is therefore concluded that is no potential for the Project-alone or in-combination to have an adverse effect on the integrity of the Coquet Island SPA**.

## 8.27 Flamborough and Filey Coast SPA

1199. Flamborough and Filey Coast (FFC) SPA is located approximately 212km from the windfarm site (straight line distance), but is over 1,000km from the site across sea.

### 8.27.1 Description of designation

1200. FFC SPA was designated in 2018. It is located on the Yorkshire coast between Bridlington and Scarborough and is composed of two sections. The northern section runs from Cunstone Nab to Filey Brigg, and the southern section from Speeton, around Flamborough Head, to South Landing. The seaward boundary extends 2km offshore and applies to both sections of the SPA. It is a geographical extension to the former Flamborough Head and Bempton Cliffs SPA, which was designated in 1993 (Natural England, 2018c).

1201. The predominantly chalk cliffs of Flamborough Head and Bempton Cliffs rise to 135m and support internationally important breeding populations of seabirds. The marine extension includes areas close to the colonies used by seabirds for maintenance behaviours (loafing, preening etc).

### 8.27.2 Conservation objectives

1202. The SPA's conservation objectives are to ensure that the integrity of the site is maintained or restored as appropriate, and that the site contributes to achieving the aims of the Wild Birds Directive, by maintaining or restoring:

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

### 8.27.3 Assessment

1203. Two qualifying features of FFC SPA have been screened into the Appropriate Assessment (**Table 5.2**): breeding gannet and breeding kittiwake. These species have also been screened in as components of the seabird assemblage.

#### 8.27.3.1 Kittiwake

##### Status

1204. At the time of the classification of the former Flamborough Head and Bempton Cliffs SPA in 1993, the kittiwake breeding population was cited as 83,370 breeding pairs. This was based on a count carried out in 1987. The breeding adult kittiwake population of the FFC SPA at classification in 2018 was cited as 44,420 pairs or 89,040 breeding adults. This was based on counts carried out between 2008 and 2011 (Natural England, 2018c).

1205. Recent counts indicated increases in the kittiwake breeding population since 2008, with estimates of 51,001 pairs, or 102,002 breeding adults, in 2016 (Babcock *et al*., 2016) and 51,535 pairs, or 103,070 breeding adults, in 2017 (Aitken *et al*., 2017). The latter was a complete census of the colony and is considered to represent the best available evidence of the current population size.

1206. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult mortality rate of 0.146 (1 – 0.854; Horswill and Robinson, 2015), the expected annual mortality from the SPA population would be 15,048 breeding adults.

##### Functional linkage and seasonal apportionment of potential effects

1207. The mean maximum foraging range of kittiwake is 156.1km (±144.5km) and the maximum foraging range is 770km (Woodward *et al*., 2019). The Project is located over 1,000km from FFC SPA by sea, which means the Project is beyond the mean maximum foraging range of kittiwakes breeding at this SPA. No impacts during the breeding season are therefore apportioned to birds breeding at this colony.

1208. Outside the breeding season, breeding kittiwakes from the SPA are assumed to range widely and to mix with kittiwakes of all ages from breeding colonies in the UK and beyond. The relevant non-breeding season reference population is the UK Western Waters and Channel BDMPS, consisting of 911,586 individuals during autumn migration (September to December) and 627,816 during spring migration (January to February) (Furness, 2015).

1209. Furness (2015) estimated that 20% of the FFC SPA breeding adults are present within the UK Western Waters and Channel BDMPS during the autumn migration period, which is 15,047 birds. During the spring migration period 30% of the population is estimated to be present, which is 22,570 birds. This represents 1.65% and 3.60% of the BDMPS population for the autumn and spring periods respectively. During autumn migration and spring migration, 1.65%, and 3.60% of impacts are therefore considered to affect birds from the SPA.

## Potential effects on the qualifying feature from the Project-alone

1210. The kittiwake qualifying feature of the FFC SPA has been screened into the Appropriate Assessment due to the potential risk of collision.

### Operation and maintenance phase collision risk

1211. Information for collision risk on breeding adult kittiwakes belonging to the FFC SPA population is presented in **Table 8.65**. Collision estimates, calculated using the sCRM, are presented by biological season. A summary of the annual outputs and the corresponding increase in the annual baseline mortality rate is also presented. Parameters used in the sCRM were agreed with Natural England during the ETG process and are described in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES.

1212. Based on the mean collision rates, the annual total of breeding adult kittiwakes from FFC SPA at risk of collision as a result of the Project is less than one bird (0.16). This would increase the existing mortality of the SPA breeding population by 0.00%.

*Table 8.65 Predicted seasonal and annual collision mortality (Stochastic model Option 2, avoidance rate 0.993 (±0.0003)), for breeding adult kittiwakes at the windfarm site, apportioned to Flamborough and Filey Coast SPA, with corresponding increases to baseline mortality of the population*

| | **Breeding Season** | **Autumn Migration** | **Non-breeding/winter** | **Spring Migration** | **Annual** |
|---|---|---|---|---|---|
| **Period** | **Mar-Aug** | **Sep-Dec** | **-** | **Jan-Feb** | **Jan-Dec** |
| Total collisions[1] (mean and 95% Cis) | 15.36 (4.17-33.87) | 8.49 (2.20-18.85) | - | 0.63 (0.00-1.46) | 24.48 (6.37-54.17) |
| % apportioned to the SPA | 0.00% | 1.65% | - | 3.60% | - |
| Total SPA collisions (mean and 95% Cis) | 0.00 (0.00-0.00) | 0.14 (0.04-0.31) | - | 0.02 (0.00-0.05) | 0.16 (0.04-0.36) |
| Mortality increase[2] (mean and 95% Cis) | 0.00% (0.00-0.00%) | 0.00% (0.00-0.00%) | - | 0.00% (0.00-0.00%) | 0.00% (0.00-0.00%) |

[1] Breeding season collision values reduced to 94.1% of modelled value to reflect proportion of adult birds recorded during site surveys

[2] Assuming predicted annual SPA mortality of 15,048 birds (103,070 x 0.146)

1213. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that no detectable changes in mortality rates would occur on this population for the mean monthly collision estimates for the Project, or for the upper 95% CI collision estimate.

1214. **It is concluded that based on predicted kittiwake mortality due to collision at the windfarm site there is no potential for the Project to have an adverse effect on the integrity of the Flamborough and Filey Coast SPA.**

1215. The confidence in the assessment is high. The evidence used to define the CRM input parameters presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** as part of the ES submitted alongside this RIAA is of high applicability and quality. Whilst there is uncertainty around some of the input parameters (e.g. avoidance rate), the rates selected are considered to be sufficiently precautionary based on expert opinion to provide confidence that collision rates are not underestimated.

### Potential effects on the qualifying feature in-combination with other projects

1216. As the Project would have no measurable effect on kittiwake populations from the FFC SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of Flamborough and Filey Coast SPA, when assessed in-combination with other plans or projects**.

### 8.27.3.2 Gannet

### Status

1217. Within the FFC SPA, gannets nest along a 5km stretch of Bempton Cliffs where numbers have increased in recent years. Natural England (2020d) gave counts of 3,940 pairs, or 7,880 breeding adults, in 2004, and 7,859 pairs, or 15,718 breeding adults, in 2009. Furness (2015) used a count of 11,061 pairs (22,122 breeding adults) in 2012. The most recent count was 13,392 pairs (AOS), or 26,784 breeding adults, in 2017 (JNCC, 2023a); this is used as the reference population for the assessment.

1218. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.081 (1 – 0.919; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 2,170 breeding adults.

### Functional linkage and seasonal apportionment of potential effects

1219. The mean maximum foraging range of gannet is 315.2km (±194.2km), and the maximum foraging range is 709km (Woodward *et al*., 2019). The straight-line distance between the Project and from FFC SPA is approximately 212km, and therefore theoretically within the mean maximum foraging range and the maximum foraging range for this species. However, the across-sea distance exceeds 1,000km, and, as gannets will not typically fly across land, no breeding season connectivity between the Site and SPA gannet population is predicted.

1220. Outside of the breeding season, breeding gannets, including those from the FFC SPA, are not constrained by requirements to visit nests to incubate eggs or provision chicks. At this time, they are assumed to range more widely and to mix with gannets of all age classes from breeding colonies in the UK and further afield. The background population during these seasons is the UK Western Waters BDMPS. This consists of 545,954 individuals during the autumn migration season (September to November) and 661,888 individuals during the spring migration season (December to March).

1221. Furness (2015) estimated that 30% of the FFC SPA breeding adults (22,122) are present within the UK Western Waters BDMPS during the spring migration season, which is 6,637 birds. This represents 1.0% of the BDMPS population for this period (661,888). During the autumn migration season, Furness (2015) estimated that no birds from FFC SPA are present within the UK Western Waters BDMPS. It is therefore assumed that 1.0% of gannets present at the Project site during the spring migration period are breeding adults from FFC SPA, but none are present during autumn migration.

### Potential effects on the qualifying feature – Project-alone

1222. The gannet qualifying feature of the FFC SPA has been screened into the assessment due to the potential risk of collision and displacement/barrier effects during the operation and maintenance phase of the Project.

#### Operation and maintenance phase displacement/barrier effects

1223. Displacement effects for gannet for the Project were assessed during the spring migration period, based on a peak mean population of eight birds, calculated for the windfarm site and a 2km buffer, in line with recommendations within the SNCB guidance (SNCB 2017). As set out above, no gannets present at the windfarm site have been apportioned to the FFC SPA during the breeding or autumn migration seasons. The displacement matrices used to calculate potential impacts are presented in **Appendix 12.1** of the ES and summarised in **Table 8.66**. The inclusion of all birds within the 2km buffer, to determine the total number of birds subject to displacement, is precautionary, as in reality the avoidance rate is likely to fall with distance from the windfarm site.

1224. A displacement rate of 60-80% and mortality rate of 1% has been presented. A maximum 1% mortality value has been selected firstly because gannet is known to possess high habitat flexibility (Furness and Wade, 2012). This suggests that displaced birds will readily find alternative habitats including foraging areas. Secondly, no evidence of displacement-induced mortality has been identified, which means there is limited justification for setting predicted mortality rates at a higher level. Given the extensive foraging range of this species (Woodward *et al.*, 2019), there may be no mortality costs to displacement from the relatively very small footprints of OWFs.

*Table 8.66 Gannet – predicted operation and maintenance phase displacement and mortality from Flamborough and Filey Coast SPA*

| Mean peak abundance estimate type | Mean peak abundance estimate (spring migration) | Number of SPA breeding adults present by season[1] | Annual mortality range[2] | Annual baseline mortality increase range[3] |
|---|---|---|---|---|
| Upper 95% CI | 16.0 | 0.2 | 0-0 | 0.00-0.00% |
| Mean | 7.9 | 0.1 | 0-0 | 0.00-0.00% |
| Lower 95% CI | 0.0 | 0.0 | 0-0 | 0.00-0.00% |

[1] During spring migration period, 1.0% of birds are assumed to be breeding adults from the SPA population.

[2]Assumes displacement rates of 60-80% and mortality rate of 1%

[3] Background population is FFC SPA breeding adults (26,784 individuals), adult age class annual mortality rate of 8.1% (Horswill and Robinson, 2015)

1225. Using the maximum potential mortality value, there would be no measurable increase in gannet mortality. Accordingly, no significant effects on gannet are predicted during the operation and maintenance phase, and **it is concluded that there would be no potential for the Project to have an adverse effect on the integrity of the Flamborough and Filey Coast SPA.**

1226. The confidence in the assessment is high for several reasons. Firstly, the evidence used to set the displacement rates presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES is of high applicability and quality. Whilst there is limited available evidence to inform mortality rates, 1% is considered to be sufficiently precautionary, based on expert opinion. Finally, the conclusion of the assessment is the same irrespective of whether the mean or 95% upper CI mean peak abundances are used to calculate potential mortality and increases in the baseline mortality rate of the background population.

**Operation and maintenance phase collision risk**

1227. Information to support the Appropriate Assessment for collision risk on breeding adult gannets belonging to the FFC SPA population is presented in **Table 8.67**. Collision estimates, calculated using the sCRM, are presented by biological season. A summary of the annual outputs and the corresponding increase in the annual baseline mortality rate is also presented. Parameters used in the sCRM were agreed with Natural England during the ETG process and are described in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES. In accordance with Natural England advice, a 70% macro-avoidance correction was applied to gannet abundance data used in the sCRM.

1228. Based on the mean collision rates, no breeding adult gannets from the FFC SPA are considered at risk of collision as a result of the Project. Therefore, there would be no measurable increase in the existing mortality of the SPA breeding population.

*Table 8.67 Predicted seasonal and annual collision mortality (Stochastic model Option 2, avoidance rate 0.993 (±0.0003), plus 70% macro-avoidance) for breeding adult gannets at the windfarm site, apportioned to the Flamborough and Filey Coast SPA, with corresponding increases to baseline mortality of the population*

| Period | Breeding Season<br>Apr-Aug | Autumn Migration<br>Sep-Nov | Non-breeding/winter<br>- | Spring Migration<br>Dec-Mar | Annual<br>Jan-Dec |
|---|---|---|---|---|---|
| Total collisions[1] (mean and 95% Cis) | 0.83<br>(0.00-3.35) | 0.14<br>(0.00-0.74) | - | 0.00 | 0.97<br>(0.00-4.10) |
| % apportioned to the SPA | 0.0% | 0.0% | - | 1.0% | - |
| Total SPA collisions (mean and 95% Cis) | 0.00<br>(0.00-0.00) | 0.00<br>(0.00-0.00) | - | 0.00<br>(0.00-0.00) | 0.00<br>(0.00-0.00) |
| Mortality increase[2] (mean and 95% Cis) | 0.00%<br>(0.00-0.00%) | 0.00%<br>(0.00-0.00%) | - | 0.00%<br>(0.00-0.00%) | 0.00%<br>(0.00-0.00%) |

[1] Breeding season collision values reduced to 73.8% of modelled value to reflect proportion of adult birds recorded during site surveys

[2] Assuming predicted annual SPA mortality of 2,170 birds (26,784 x 0.081)

1229. Accordingly, no significant effects on gannet are predicted during the operation and maintenance phase, and it is concluded that there would be no potential for the Project to have an adverse effect on the integrity of the FFC SPA. Comments received from RSPB during the ETG process, indicating that they do not accept the 70% macro-avoidance rate for collision risk recommended by Natural England, are noted. However, even in the absence of this correction factor, the net increase in mortality would be unchanged (i.e. zero).

1230. The confidence in the assessment is high. The evidence used to define the CRM input parameters presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES submitted alongside this RIAA is of high applicability and quality. Whilst there is uncertainty around some of the input parameters (e.g. avoidance rate), the rates selected are considered to be sufficiently precautionary based on expert opinion to provide confidence that collision rates are not underestimated.

**Combined displacement/barrier effects and collision risk**

1231. As no measurable increase in mortality is predicted for both displacement and collision risk, the mean combined displacement and collision rates for breeding adult gannet from the FFC SPA would be zero. Therefore, there would be no net increase in existing mortality rates.

1232. **It is concluded that based on predicted gannet mortality due to the combined effects of operational phase displacement and collision there is no potential for the Project-alone or in-combination to have an adverse effect on the integrity of the Flamborough & Filey Coast SPA.**

1233. The confidence in the assessment is high, for the reasons provided in the individual displacement and collision assessments.

### Potential effects in-combination with other projects

1234. As no measurable effects of displacement/barrier and collision on gannet are predicted as a result of the Project-alone, there would be no contribution to other plans or projects in-combination. **It is therefore concluded that there would be no potential for the Project-alone or in-combination to have an adverse effect on the integrity of the Flamborough and Filey Coast SPA.**

## 8.28 Rathlin Island SPA

1235. Rathlin Island SPA is located approximately 223km from the windfarm site.

### 8.28.1 Description of designation

1236. Rathlin Island is a large inhabited marine island situated some 4km from the north Antrim coast. There are basalt and chalk cliffs, some as high as 100m, as well as several sea stacks on the north and west shores of the island. The south and east shores are more gently sloping with areas of maritime grassland and rocky shore. The length of the coastline is approximately 30km. The cliffs are principally important for the seabird colonies, most notably around the area of West Light, but also along sections of the north coast. This extensive habitat also supports a notable breeding population of peregrine.

### 8.28.2 Conservation objectives

1237. The overarching conservation objective for the SPA is 'to maintain each feature in favourable condition.' For the qualifying features, the objectives are:

- To maintain or enhance the population of the qualifying species
- Fledging success sufficient to maintain or enhance population
- To maintain or enhance the range of habitats utilised by the qualifying species
- To ensure that the integrity of the site is maintained
- To ensure there is no significant disturbance of the species
- To ensure that the following are maintained in the long term:
  - Population of the species as a viable component of the site
  - Distribution of the species within the site
  - Distribution and extend of habitats supporting the species
  - Structure, function and supporting processes of habitats supporting the species

### 8.28.3 Assessment

1238. Breeding kittiwake, breeding guillemot and breeding razorbill associated with Rathlin Island SPA have been screened into the Appropriate Assessment (**Table 5.2**). These species, together with fulmar, lesser black-backed gull and puffin, have also been screened in as components of the seabird assemblage.

#### 8.28.3.1 Kittiwake

##### Status

1239. The Rathlin Island SPA breeding kittiwake population was cited at 6,822 pairs, or 13,644 breeding adults, in 1985 (Furness, 2015; Stroud *et al*., 2016). Furness (2015) gave the breeding population of 7,922 pairs, or 15,844 breeding adults, in 2011. The most recent count was 13,706 pairs (AON), or 27,412 breeding adults, in 2021 (JNCC, 2023a); this is used as the reference population for the assessment.

1240. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.146 (1 – 0.854; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 27,412 breeding adults.

##### Functional linkage and seasonal apportionment of potential effects

1241. The mean maximum foraging range of kittiwake is 156.1km (±144.5km) and the maximum foraging range is 770km (Woodward *et al*., 2019). The Project is located approximately 223km from Rathlin Island SPA, which means the Project is beyond the mean maximum foraging range of kittiwakes breeding at this SPA, but within the mean maximum foraging range +1SD, and the maximum foraging range.

1242. The NatureScot apportioning tool (NatureScot, 2018) has been used to estimate the proportion of kittiwakes from each of the relevant SPAs present at the windfarm site during the breeding season. The apportioning to SPA and non-SPA colonies is set out in **Appendix 12.1** of the ES. 6.27% of impacts at the windfarm site during the breeding season are apportioned to Rathlin Island SPA.

1243. Outside the breeding season, kittiwakes from the SPA are assumed to range widely and to mix with kittiwakes of all ages from breeding colonies in the UK and beyond. The relevant non-breeding season reference population is the UK Western Waters and Channel BDMPS, consisting of 911,586 individuals during autumn migration (September to December) and 627,816 during spring migration (January to February) (Furness, 2015).

1244. Furness (2015) estimated that 60% of the Rathlin Island SPA breeding adults are present within the UK Western Waters and Channel BDMPS during the autumn migration period, which is 9,506 birds. During the spring migration period 80% of the population is estimated to be present, which is 12,675 birds. This represents 1.04% and 2.02% of the BDMPS population for the autumn and spring periods respectively. During autumn migration and spring migration, 1.04%, and 2.02% of impacts are therefore considered to affect birds from the SPA (Furness, 2015).

## Potential effects on the qualifying feature from the Project-alone

1245. The kittiwake qualifying feature of the Rathlin Island SPA has been screened into the Appropriate Assessment due to the potential risk of collision.

### Operation and maintenance phase collision risk

1246. Information for collision risk on breeding adult kittiwakes belonging to the Rathlin Island SPA population is presented in **Table 8.68**. Collision estimates, calculated using the sCRM, are presented by biological season. A summary of the annual outputs and the corresponding increase in the annual baseline mortality rate is also presented. Parameters used in the sCRM were agreed with Natural England during the ETG process and are described in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES.

1247. Based on the mean collision rates, the annual total of breeding adult kittiwakes from Rathlin Island SPA at risk of collision as a result of the Project is one bird (1.06). This would increase the existing mortality of the SPA breeding population by 0.03%.

*Table 8.68 Predicted seasonal and annual collision mortality (Stochastic model Option 2, avoidance rate 0.993 (±0.0003)), for breeding adult kittiwakes at the windfarm site, apportioned to Rathlin Island SPA, with corresponding increases to baseline mortality of the population*

| | Breeding Season | Autumn Migration | Non-breeding/winter | Spring Migration | Annual |
|---|---|---|---|---|---|
| Period | Mar-Aug | Sep-Dec | - | Jan-Feb | Jan-Dec |
| Total collisions[1] (mean and 95% CIs) | 15.36 (4.17-33.87) | 8.49 (2.20-18.85) | - | 0.63 (0.00-1.46) | 24.48 (6.37-54.17) |
| % apportioned to the SPA | 6.27% | 1.04% | - | 2.02% | - |
| Total SPA collisions (mean and 95% CIs) | 0.96 (0.26-2.12) | 0.09 (0.02-0.20) | - | 0.01 (0.00-0.03) | 1.06 (0.28-2.35) |
| Mortality increase[2] (mean and 95% CIs) | 0.02% (0.01-0.05%) | 0.00% (0.00-0.00%) | - | 0.00% (0.00-0.00%) | 0.03% (0.01-0.06%) |

[1] Breeding season collision values reduced to 94.1% of modelled value to reflect proportion of adult birds recorded during site surveys

[2] Assuming predicted annual SPA mortality of 4,002 birds (27,412 x 0.146)

1248. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that no detectable changes in mortality rates would occur on this population for the mean monthly collision estimates for the Project, or for the upper 95% CI collision estimate.

1249. **It is concluded that based on predicted kittiwake mortality due to collision at the windfarm site there is no potential for the Project to have an adverse effect on the integrity of the Rathlin Island SPA.**

1250. The confidence in the assessment is high. The evidence used to define the CRM input parameters presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** as part of the ES submitted alongside this RIAA is of high applicability and quality. Whilst there is uncertainty around some of the input parameters (e.g. avoidance rate), the rates selected are considered to be sufficiently precautionary, based on expert opinion, to provide confidence that collision rates are not underestimated.

### Potential effects on the qualifying feature in-combination with other projects

1251. As the Project would have no measurable effect on kittiwake populations from the Rathlin Island SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of Rathlin Island SPA, when assessed in-combination with other plans or projects**.

### 8.28.3.2 Guillemot

### Status

1252. The Rathlin Island SPA breeding guillemot population was cited as 28,064 pairs, or 56,128 breeding adults in 1985 (Furness, 2015; Stroud *et al*., 2016). Furness (2015) gave the breeding population of 87,398 pairs, or 174,796 breeding adults, in 2013. The most recent count (2021) was 149,510 individuals (JNCC, 2023a); this is used as the reference population for the assessment.

1253. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.061 (1 – 0.939; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 9,120 breeding adults.

### Functional linkage and seasonal apportionment of potential effects

1254. The mean maximum foraging range of guillemot is 73.2km (±80.5km) and the maximum foraging range is 338km (Woodward *et al*., 2019). The Project is located approximately 223km from Rathlin Island SPA, which means the Project is beyond the mean maximum foraging range +1SD of guillemots breeding at this SPA, but within the maximum foraging range. The maximum foraging range is a poor indicator of typical foraging behaviour. It would be expected that few birds or foraging trips will occur at this distance from the colony, and even fewer with any regularity. No impacts during the breeding season are therefore apportioned to birds breeding at this colony.

1255. Outside the breeding season, guillemots from the SPA are assumed to range widely and to mix with guillemots of all ages from breeding colonies in the UK and beyond. The relevant non-breeding season reference population is the UK Western Waters BDMPS, consisting of 1,139,220 individuals (August to February) (Furness, 2015). During the non-breeding season, it is estimated that 15.3% of birds present are considered to be breeding adults from the Rathlin Island SPA, and impacts are apportioned accordingly. This is based on the SPA adult population from Furness (2015) as a proportion of the total UK Western Waters BDMPS.

## Potential effects on the qualifying feature

### Operation and maintenance phase disturbance/displacement/barrier effects

*Project-alone*

1256. The mean peak abundance of guillemots present within the windfarm site and 2km buffer during the non-breeding season was 8,315 (6,085-12,047) individuals (refer to **Appendix 12.1** of the ES). Of these, 1,272 birds (931-1,843) were likely to be breeding adults from the Rathlin Island SPA.

1257. **Table 8.69** sets out the predicted impacts on guillemots from Rathlin Island SPA during the non-breeding season. Displacement rates of 30% to 70% are considered for this species, along with a range of mortality rates of 1% to 10% of displaced birds (UK SNCBs, 2017).

1258. The available evidence suggests that the upper ranges of these displacement and mortality rates may be excessively precautionary. Whilst it is true that guillemots and razorbills tend to be displaced by OWFs they do not avoid them completely, and displacement rates have been found to vary between sites (MacArthur Green, 2019b). It was concluded that mean densities within OWFs were around half of the density found in the habitats around the OWF. At some OWFs there was also displacement of birds from a buffer zone surrounding it. The size of the buffer zone varied between OWFs and was generally less than 2km, with auk density increasing across the buffer zone as distance from turbines increased, up to the density in the wider area. Another recent review (APEM, 2022) found that the displacement rates suggested by the SNCBs for guillemot and razorbill (30% to 70%) did not account for the quality of or confidence in the studies which were used to inform this position, and that studies where no significant effects were recorded were not accounted for during the provision of the advice. APEM (2022) suggested that in the case of

Hornsea Project Four, an evidence-based displacement rate of 50% was appropriate. However, the study also recognised that larger displacement effects are possible.

1259. Mortality due to displacement could arise due to increased energy costs and/or decreased energy intake, if displacement results in increased flying time to avoid OWFs, and/or increased bird densities and competition for prey in areas of unimpacted habitat outside the OWF. Given that UK OWFs in the Irish and Celtic Seas represent a very small proportion of the available foraging habitat for guillemots and razorbills within UK Western Waters, the increase in density of auks outside OWFs due to displacement is likely to be negligible (MacArthur Green, 2019b). It is considered unlikely that the mortality rate due to displacement would be as high as the empirically estimated 6.1% annual mortality for adult guillemots that occurs due to the combination of 'natural' factors and anthropogenic activities (Horswill and Robinson, 2015). Indeed, it may be much lower; MacArthur Green (2019b) recommended precautionary rates of 50% displacement for guillemot and 1% mortality of displaced birds based on a review of available evidence. Modelling undertaken by APEM (2022) found that in the case of Hornsea Project Four, a mortality rate of 1% for displaced auks was considered to be precautionary, and that this may be an overestimate given that Hornsea Project Four is located towards the upper end of the mean maximum foraging range for guillemot from the closest designated sites. It is also noted that in the recent decision on Hornsea Project Four, the SoS determined that a rate of 70% displacement/2% mortality was appropriate to inform the assessment of effects on guillemots from Flamborough and Filey Coast SPA (DESNZ, 2023b).

1260. The full ranges of recommended displacement and mortality effects are considered by the assessment, along with evidence-based displacement and mortality rates of 50% and 1%, respectively (APEM, 2022; MacArthur Green, 2019b).

*Table 8.69 Guillemot – predicted operation and maintenance phase displacement and mortality from Rathlin Island SPA*

| Mean peak abundance estimate type | Mean peak abundance estimate | Number of North Rathlin Island SPA breeding adults present (non-breeding season)[1] | Annual mortality range[2] | Annual baseline mortality increase range[3] |
|---|---|---|---|---|
| Upper 95% CI | 12,047 | 1,843 | 6-129 | 0.05-1.21% |
| Mean | 8,315 | 1,272 | 4-89 | 0.04-0.84% |
| Lower 95% CI | 6,085 | 931 | 3-65 | 0.03-0.61% |

[1] Assumes 15.3% of birds present during the non-breeding season are Rathlin Island SPA breeding adults

[2] Assumes displacement rates of 30-70% and mortality rates of 1-10%

[3] Background mortality rate of 6.1% (Horswill and Robinson, 2015)

1261. Based on the mean peak abundances, the annual total of guillemots from the Rathlin Island SPA at risk of displacement is 1,272 birds (**Table 8.69**). At displacement rates of 30% to 70%, and mortality rates of 1% to 10% for displaced birds, 4 to 89 SPA breeding adults would be predicted to die each year due to displacement from the Project.

1262. Assuming a displacement rate of 70% and a mortality rate of 10% of displaced birds, annual mortality within the SPA breeding adult population would increase by 0.81%. Using an evidence-based displacement rate of 50%, and a mortality rate for displaced birds of 1%, annual mortality in the population would instead increase by 0.06% (6 birds).

1263. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that no detectable changes in mortality rates would occur under almost any combination of displacement and mortality rates when the mean peak abundance estimate assessments are considered.

1264. Increases of over 1% are predicted if the upper 95% CIs for mean peak abundances are used as inputs to the assessment alongside a 10% mortality rate for displaced birds. The probability of this occurring is extremely small for two reasons. Firstly, the upper 95% CI for the mean peak abundances is highly unlikely to occur regularly at the windfarm site. Secondly, mortality rates for displaced birds of 10% are much higher than evidence suggests will actually be the case. Use of the evidence-based displacement (50%) and mortality rate (1%) would result in a mortality increase of significantly less than 1%, as would a rate of 70%/2% agreed by the SoS in respect of Hornsea Project Four (DESNZ, 2023b).

1265. **It is concluded that predicted guillemot mortality due to operational phase displacement for the project alone would not adversely affect the integrity of the Rathlin Island SPA.**

1266. The confidence in the assessment is high for several reasons. Firstly, the evidence used to inform the evidence-based displacement rates is of high applicability and quality (based on the criteria discussed in **Chapter 12 Offshore Ornithology** of the ES). Whilst there is limited available evidence to inform mortality rates, 1% is considered to be sufficiently precautionary based on expert opinion. This species is not regarded as being highly specialised in its habitat requirements (Bradbury *et al*., 2014; Furness and Wade, 2012; Garthe and Hüppop, 2004), and it is therefore anticipated that displaced birds will find alternative habitat in the vast majority of cases. Finally, the conclusion of the assessment is the same irrespective of whether the mean or 95% upper CI mean peak abundances are used to calculate potential mortality and increases in the baseline mortality rate of the background population, provided the evidence-based displacement and mortality rates are used.

*In-combination*

1267. The in-combination assessment for guillemots from Rathlin Island SPA has been undertaken in accordance with the approach presented in Section 8.1. The total population apportioned to Rathlin Island SPA at risk of displacement is estimated to be 7,579 breeding adults (**Appendix 12.1** of the ES). Annual in-combination displacement and mortality rates for birds from Rathlin Island SPA are presented in Table 8.70.

1268. Assuming a displacement rate of 70% and a mortality rate of 10% of displaced birds, 531 breeding adult SPA birds would be lost to displacement annually. Including additional apportioned mortality from the Morlais and Holyhead tidal projects (total 4.28 birds), this would increase the existing mortality within the SPA population (9,120 breeding adult birds per year) by 5.87%. Using an evidence-based (for the reasons set out in the Project-alone assessment above) displacement rate of 50%, and a mortality rate for displaced birds of 1%, the annual in-combination mortality would be 43 birds. This would increase the existing mortality within this population by 0.47%. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that detectable changes in mortality rates will not occur due to the level or mortality predicted if the more realistic rates for mortality are used.

1269. **It is concluded that predicted guillemot mortality due to of operational phase displacement impacts of the Project in-combination with other projects, would not adversely affect the integrity of Rathlin Island SPA.**

*Table 8.70 In-combination year-round displacement matrix for guillemot from Rathlin SPA*

| Annual | Mortality | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| **Displacement** | **1%** | **2%** | **3%** | **4%** | **5%** | **10%** | **20%** | **30%** | **50%** | **80%** | **100%** |
| **10%** | 8 | 15 | 23 | 30 | 38 | 76 | 152 | 227 | 379 | 606 | 758 |
| **20%** | 15 | 30 | 45 | 61 | 76 | 152 | 303 | 455 | 758 | 1213 | 1516 |
| **30%** | 23 | 45 | 68 | 91 | 114 | 227 | 455 | 682 | 1137 | 1819 | 2274 |
| **40%** | 30 | 61 | 91 | 121 | 152 | 303 | 606 | 909 | 1516 | 2425 | 3032 |
| **50%** | 38 | 76 | 114 | 152 | 189 | 379 | 758 | 1137 | 1895 | 3032 | 3789 |
| **60%** | 45 | 91 | 136 | 182 | 227 | 455 | 909 | 1364 | 2274 | 3638 | 4547 |
| **70%** | 53 | 106 | 159 | 212 | 265 | 531 | 1061 | 1592 | 2653 | 4244 | 5305 |
| **80%** | 61 | 121 | 182 | 243 | 303 | 606 | 1213 | 1819 | 3032 | 4850 | 6063 |
| **90%** | 68 | 136 | 205 | 273 | 341 | 682 | 1364 | 2046 | 3410 | 5457 | 6821 |
| **100%** | 76 | 152 | 227 | 303 | 379 | 758 | 1516 | 2274 | 3789 | 6063 | 7579 |

Note: The cells show the number of birds predicted to die (rounded to the nearest integer) at a given rate of displacement and mortality. Highlighted cells are considered to be the most realistic scenario, in accordance with SNCB advice (SNCBs, 2022).

### 8.28.3.3 Razorbill

#### Status

1270. The Rathlin Island SPA breeding razorbill population was cited as 5,978 pairs, or 11,956 breeding adults, in 1985 (Furness, 2015; Stroud *et al*., 2016). Furness (2015) gave the breeding population of 15,393 pairs, or 30,786 breeding adults in 2011. The most recent count (2021) was 22,421 individuals (JNCC, 2023a).

1271. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.105 (1 – 0.895; Horswill and Robinson, 2015), the expected annual mortality from the SPA population would be 2,354 breeding adults.

#### Functional linkage and seasonal apportionment of potential effects

1272. The mean maximum foraging range of razorbill is 88.7km (±75.9km) and the maximum foraging range is 313km (Woodward *et al*., 2019). The Project is located approximately 223km from Rathlin Island SPA, which means the Project is beyond the mean maximum foraging range +1SD of razorbills breeding at this SPA, but within the maximum foraging range. The maximum foraging range is a poor indicator of typical foraging behaviour. It would be expected that few birds or foraging trips will occur at this distance from the colony, and even fewer with any regularity. No impacts during the breeding season are therefore apportioned to birds breeding at this colony.

1273. Outside the breeding season, razorbills from the SPA are assumed to range widely and to mix with razorbills of all ages from breeding colonies in the UK and beyond. The relevant non-breeding season reference population is the UK Western Waters BDMPS, consisting of 606,914 individuals during autumn and spring passage periods (August to October and January to March), and 341,422 individuals during winter (November and December) (Furness, 2015). During autumn and spring migration, 98% of the SPA breeding adults (30,170 individuals based on the 2011 population estimate) are assumed to be present in the BDMPS, representing 5.0% of the BDMPS population (606,914 individuals of all ages). During the winter season, 40% of the SPA breeding adults (12,314 individuals based on the 2011 population estimate) are assumed to be present in the BDMPS, representing 3.6% of the BDMPS population (341,422 individuals of all ages). These percentages (i.e. 5.0% and 3.6%) are the proportions of birds present at the windfarm site that are presumed to originate from the Rathlin Island SPA during the relevant seasons.

## Potential effects on the qualifying feature

### Operation and maintenance phase disturbance/displacement/barrier effects

*Project-alone*

1274. The year-round mean peak abundance of razorbills present within the windfarm site and 2km buffer is 1,979 (703-3,552) individuals (refer to **Appendix 12.1** of the ES). Of these, 77 birds (32-129) are likely to be breeding adults from the Rathlin Island SPA.

1275. **Table 8.71** sets out the predicted impacts on razorbills from Rathlin Island SPA. Displacement rates of 30% to 70% are considered for this species, along with a range of mortality rates of 1% to 10% of displaced birds (UK SNCBs, 2017).

1276. The available evidence suggested that the upper ranges of these displacement and mortality rates may be excessively precautionary. Whilst it is true that guillemots and razorbills tend to be displaced by OWFs they do not avoid them completely, and displacement rates vary between sites (MacArthur Green, 2019b). It was concluded that mean densities within OWFs were around half of the density found in the habitats around the OWF. At some OWFs there was also displacement of birds from a buffer zone surrounding it. The size of the buffer zone varied between OWFs and was generally less than 2km, with auk density increasing across the buffer zone as distance from turbines increased, up to the density in the wider area. Another recent review (APEM, 2022) found that the current displacement rates suggested by the SNCBs for guillemot and razorbill (30% to 70%) did not account for the quality of or confidence in the studies which were used to inform this position, and that studies where no significant effects were recorded were not accounted for during the provision of the advice. APEM (2022) suggested that in the case of Hornsea Project Four, an evidence-based displacement rate of 50% was appropriate. However, the study also recognised that larger displacement effects were possible.

1277. Mortality due to displacement could arise due to increased energy costs and/or decreased energy intake, if displacement results in increased flying time to avoid OWFs, and/or increased bird densities and competition for prey in areas of unimpacted habitat outside the OWF. Given that UK OWFs in the Irish and Celtic Seas represent a very small proportion of the available foraging habitat for guillemots and razorbills within UK Western Waters, the increase in density of auks outside OWFs due to displacement is likely to be negligible (MacArthur Green, 2019b). It is considered unlikely that the mortality rate due to displacement would be as high as the empirically estimated 10.5% annual mortality for adult razorbills that occurs due to the combination of ‘natural’ factors and anthropogenic activities (Horswill and Robinson, 2015). Indeed, it may be much lower; MacArthur Green (2019b) recommended

precautionary rates of 50% displacement for auks and 1% mortality of displaced birds based on a review of available evidence. Modelling undertaken by APEM (2022) found that in the case of Hornsea Project Four, a mortality rate of 1% for displaced auks was considered to be precautionary, and that this may be an overestimate given that Hornsea Project Four is located towards the upper end of the mean maximum foraging range for guillemot from the closest designated sites.

1278. The full ranges of recommended displacement and mortality effects are considered by the assessment, along with evidence-based displacement and mortality rates of 50% and 1%, respectively (APEM, 2022; MacArthur Green, 2019b).

*Table 8.71 Razorbill – predicted operation and maintenance phase displacement and mortality from Rathlin Island SPA*

| Mean peak abundance estimate type | Mean peak abundance estimate by season[1] | Number of Rathlin Island SPA breeding adults present by season)[2] | Annual mortality range[3] | Annual baseline mortality increase range[4] |
|---|---|---|---|---|
| Upper 95% CI | 605 (b)<br>1,070 (aut)<br>1,297 (win)<br>580 (spr)<br>3,552 (year round) | 0 (b)<br>53 (aut)<br>47 (win)<br>29 (spr)<br>129 (year round) | 0-9 (0) | 0.02-0.38% (0.03%) |
| Mean | 252 (b)<br>694 (aut)<br>651 (win)<br>381 (spr)<br>1,979 (year round) | 0 (b)<br>35 (aut)<br>23 (win)<br>19 (spr)<br>77 (year round) | 0-5 (0) | 0.01-0.23% (0.02%) |
| Lower 95% CI | 21 (b)<br>309 (aut)<br>159 (win)<br>214 (spr)<br>703 (year round) | 0 (b)<br>15 (aut)<br>6 (win)<br>11 (spr)<br>32 (year round) | 0-4 (0) | 0.00-0.09% (0.01%) |

[1] Breeding season = b, autumn migration season = aut, winter season = win, spring migration season = spr

[2] Assumes breeding adult apportioning of 0.0% (breeding season), 5.0% (spring and autumn migration) and 3.6% (winter) to Rathlin Island SPA.

[3] Assumes displacement rates of 30-70% and mortality rates of 1-10%. Evidence-based estimates assuming a 50% displacement rate and 1% mortality of displaced birds are presented in parentheses.

[4] Background mortality rate of 10.5% (Horswill and Robinson, 2015)

1279. Based on the mean peak abundances, the annual total of razorbills from the Rathlin Island SPA at risk of displacement is 77 birds (**Table 8.71**). At displacement rates of 30% to 70%, and mortality rates of 1% to 10% for displaced birds, 0 to 5 SPA breeding adults would be predicted to die each year due to displacement from the Project.

1280. Assuming a displacement rate of 70% and a mortality rate of 10% of displaced birds, annual mortality within the SPA breeding adult population would increase by 0.23%. Using an evidence-based displacement rate of 50%, and a mortality rate for displaced birds of 1%, annual mortality in the population would instead increase by 0.02% (<1 bird).

1281. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that no detectable changes in mortality rates would occur under almost any combination of displacement and mortality rates when the mean peak abundance estimate assessments are considered. This would be the case even when upper 95% CIs are considered.

1282. **It is concluded that predicted razorbill mortality due to operational phase displacement for the project alone would not adversely affect the integrity of the Rathlin Island SPA.**

1283. The confidence in the assessment is high for several reasons. Firstly, the evidence used to inform the evidence-based displacement rates is of high applicability and quality (based on the criteria discussed in **Chapter 12 Offshore Ornithology** of the ES). Whilst there is limited available evidence to inform mortality rates, 1% is considered to be sufficiently precautionary based on expert opinion. This species is not regarded as being highly specialised in its habitat requirements (Bradbury *et al*., 2014; Furness and Wade, 2012; Garthe and Hüppop, 2004), and it is therefore anticipated that displaced birds will find alternative habitat in the vast majority of cases. Finally, the conclusion of the assessment is less than 1% mortality increase irrespective of whether the mean or 95% upper CI mean peak abundances are used to calculate potential mortality and increases in the baseline mortality rate of the background population, provided the evidence-based displacement and mortality rates are used.

*In-combination*

1284. The in-combination assessment for razorbills from Rathlin Island SPA has been undertaken in accordance with the approach presented in **Section 8.1**. The total population apportioned to Rathlin Island SPA at risk of displacement is estimated to be 494 breeding adults (**Appendix 12.1** of the ES). Annual in-combination displacement and mortality rates for birds from Rathlin Island SPA are presented in **Table 8.72**.

1285. Assuming a displacement rate of 70% and a mortality rate of 10% of displaced birds, 35 breeding adult SPA birds would be lost to displacement annually. Including additional apportioned mortality from the Morlais and Holyhead tidal projects (total 0.74 birds), this would increase the existing mortality within the SPA population (2,354 breeding adult birds per year) by 1.50%. Using an evidence-based displacement rate of 50%, and a mortality rate for displaced birds of 1%, the annual in-combination mortality would be 2 birds. This would increase the existing mortality within this population by 0.14%. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that detectable changes in mortality rates will not occur due to the level or mortality predicted if the more realistic rates for mortality are used.

1286. **It is concluded that predicted razorbill mortality due to operational phase displacement impacts of the Project in-combination with other projects, would not adversely affect the integrity of Rathlin Island SPA.**

*Table 8.72 In-combination year-round displacement matrix for razorbill from Rathlin Island SPA*

| Annual | Mortality | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Displacement | 1% | 2% | 3% | 4% | 5% | 10% | 20% | 30% | 50% | 80% | 100% |
| 10% | 0 | 1 | 1 | 2 | 2 | 5 | 10 | 15 | 25 | 40 | 49 |
| 20% | 1 | 2 | 3 | 4 | 5 | 10 | 20 | 30 | 49 | 79 | 99 |
| 30% | 1 | 3 | 4 | 6 | 7 | 15 | 30 | 44 | 74 | 119 | 148 |
| 40% | 2 | 4 | 6 | 8 | 10 | 20 | 40 | 59 | 99 | 158 | 198 |
| 50% | 2 | 5 | 7 | 10 | 12 | 25 | 49 | 74 | 123 | 198 | 247 |
| 60% | 3 | 6 | 9 | 12 | 15 | 30 | 59 | 89 | 148 | 237 | 296 |
| 70% | 3 | 7 | 10 | 14 | 17 | 35 | 69 | 104 | 173 | 277 | 346 |
| 80% | 4 | 8 | 12 | 16 | 20 | 40 | 79 | 119 | 198 | 316 | 395 |
| 90% | 4 | 9 | 13 | 18 | 22 | 44 | 89 | 133 | 222 | 356 | 444 |
| 100% | 5 | 10 | 15 | 20 | 25 | 49 | 99 | 148 | 247 | 395 | 494 |

Note: The cells show the number of birds predicted to die (rounded to the nearest integer) at a given rate of displacement and mortality. Highlighted cells are considered to be the most realistic scenario, in accordance with SNCB advice (SNCBs, 2022).

#### 8.28.3.4 Fulmar

##### Status

1287. The Rathlin Island SPA breeding fulmar population was cited as 1,482 pairs, or 2,964 breeding adults, in 1985 (Furness, 2015, Stroud *et al*., 2016). Furness (2015) gave the breeding population of 1,518 pairs, or 3,036 breeding adults, in 2011. The most recent count was 1,045 pairs (AOS), or 2,090 breeding adults, in 2021 (JNCC, 2023a); this is used as the reference population for the assessment.

1288. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.064 (1 – 0.936; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 134 breeding adults.

##### Functional linkage and seasonal apportionment of potential effects

1289. The mean maximum foraging range of fulmar is 542.3km (±657.9km) and the maximum foraging range is 2,736km (Woodward *et al*., 2019). The Project is located approximately 223km from Rathlin Island SPA, which means the Project is within the mean maximum foraging range of fulmars breeding at this SPA.

##### Potential effects on the qualifying feature from the Project-alone

1290. Fulmar are considered to have low vulnerability to both collision and displacement impacts from offshore windfarms (Bradbury *et al*., 2014). This species was recorded infrequently within the windfarm site (in only five of 24 surveys, all during the breeding season (January to August; Furness, 2015) and in low numbers, with densities in four of the five months less than 0.1 birds/km$^2$. A peak density of 0.23 birds/km$^2$ was recorded in May 2022.

1291. On the basis of the low vulnerability of fulmars to potential impacts, and the low frequency and abundance of birds present at the windfarm site, it is concluded that any effects on breeding populations at Rathlin Island SPA are very unlikely, both during and outside of the breeding season.

1292. **It is therefore concluded that there would be no measurable effects on fulmar due to the project alone, and no adverse effect on the integrity of the Rathlin Island SPA is predicted.**

##### Potential effects on the qualifying feature in-combination with other projects

1293. As the Project would have no measurable effect on fulmar populations from the Rathlin Island SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no**

**adverse effect on the integrity of Rathlin Island SPA, when assessed in-combination with other plans or projects.**

#### 8.28.3.5 Lesser black-backed gull

##### Status

1294. The Rathlin Island SPA breeding lesser black-backed gull population was cited as 155 pairs, or 310 breeding adults, in 1985 (Furness, 2015, Stroud *et al*., 2016). Furness (2015) gave the breeding population of 183 pairs, or 366 breeding adults, in 2010. The most recent count was 519 pairs (AON), or 1,038 breeding adults, in 2021 (JNCC, 2023a); this is used as the reference population for the assessment.

1295. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.115 (1 – 0.885; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 119 breeding adults.

##### Functional linkage and seasonal apportionment of potential effects

1296. The mean maximum foraging range of lesser black-backed gull is 127km (±109km) and the maximum foraging range is 533km (Woodward *et al*., 2019). The straight-line distance between the Project and Rathlin Island is approximately 223km, which means the Project is beyond the mean maximum foraging range of breeding lesser black-backed gulls from the SPA, but theoretically within the mean maximum foraging range +1SD, and the maximum foraging range. However, across-sea distance is approximately 236km, so on the limit of the mean maximum foraging range plus +1SD. The maximum foraging range is a poor indicator of typical foraging behaviour. It would be expected that few birds or foraging trips will occur at this distance from the colony, and even fewer with any regularity. No impacts during the breeding season are therefore apportioned to birds breeding at this colony.

1297. Outside the breeding season, breeding lesser black-backed gulls from the SPA are assumed to range widely and to mix with lesser black-backed gulls of all ages from breeding colonies in the UK and beyond. The relevant non-breeding season reference population is the UK Western Waters BDMPS, consisting of 163,304 individuals during spring and autumn migration (March and September to October) and 41,159 during winter (November to February) (Furness, 2015).

1298. Furness (2015) estimated that 50% of the Rathlin Island SPA breeding adults are present within the UK Western Waters and Channel BDMPS during the autumn and spring migration periods, which is 107 birds. During the winter period 20% of the population is estimated to be present, which is 43 birds. This represents 0.07% of the BDMPS population for the autumn and spring periods, and 0.10% during the winter period. Impacts to birds from the SPA during these periods are therefore apportioned accordingly.

## Potential effects on the qualifying feature from the Project-alone

1299. The lesser black-backed gull qualifying feature of the Rathlin Island SPA has been screened into the Appropriate Assessment due to the potential risk of collision.

### Operation and maintenance phase collision risk

1300. Information for collision risk on breeding adult lesser black-backed gulls belonging to the Rathlin Island SPA population is presented in **Table 8.73**. Collision estimates, calculated using the sCRM, are presented by biological season. A summary of the annual outputs and the corresponding increase in the annual baseline mortality rate is also presented. Parameters used in the sCRM were agreed with Natural England during the ETG process and are described in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES.

1301. Based on the mean collision rates, the annual total of breeding adult lesser black-backed gulls from Rathlin Island SPA at risk of collision as a result of the Project is less than one bird (0.00). This would increase the existing mortality of the SPA breeding population by 0.00%.

*Table 8.73 Predicted seasonal and annual collision mortality (Stochastic model Option 2, avoidance rate 0.994 (±0.0004)), for breeding adult lesser black-backed gulls at the windfarm site, apportioned to Rathlin Island SPA, with corresponding increases to baseline mortality of the population*

| Period | Breeding Season<br>Apr-Aug | Autumn Migration<br>Sep-Oct | Non-breeding/winter<br>Nov-Feb | Spring Migration<br>Mar | Annual<br>Jan-Dec |
|---|---|---|---|---|---|
| Total collisions[1] (mean and 95% CIs) | 1.44<br>(0.00-4.53) | 1.25<br>(0.00-5.63) | 0.15<br>(0.00-0.80) | 0.15<br>(0.00-0.94) | 2.98<br>(0.00-11.90) |
| % apportioned to the SPA | 0.00% | 0.07% | 0.11% | 0.07% | - |
| Total SPA collisions (mean and 95% CIs) | 0.00<br>(0.00-0.00) | 0.00<br>(0.00-0.00) | 0.00<br>(0.00-0.00) | 0.00<br>(0.00-0.00) | 0.00<br>(0.00-0.01) |
| Mortality increase[2] (mean and 95% CIs) | 0.00%<br>(0.00-0.00%) | 0.00%<br>(0.00-0.00%) | 0.00<br>(0.00-0.00) | 0.00%<br>(0.00-0.00%) | 0.00%<br>(0.00-0.00%) |

[1] Breeding season collision values reduced to 71.9% of modelled value to reflect proportion of adult birds recorded during site surveys

[2] Assuming predicted annual SPA mortality of 119 birds (1,038 x 0.115)

1302. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that no detectable changes in mortality rates would occur on this population for the mean monthly collision estimates for the Project, or for the upper 95% CI collision estimate.

1303. **It is concluded that based on predicted lesser black-backed gull mortality due to collision at the windfarm site there is no potential for the Project to have an adverse effect on the integrity of Rathlin Island SPA.**

1304. The confidence in the assessment is high. The evidence used to define the CRM input parameters presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** as part of the ES submitted alongside this RIAA is of high applicability and quality. Whilst there is uncertainty around some of the input parameters (e.g. avoidance rate), the rates selected are considered to be sufficiently precautionary based on expert opinion to provide confidence that collision rates are not underestimated.

### Potential effects on the qualifying feature in-combination with other projects

1305. As the Project would have no measurable effect on lesser black-backed gull populations from the Rathlin Island SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of Rathlin Island SPA, when assessed in-combination with other plans or projects**.

### 8.28.3.6 Puffin

### Status

1306. The Rathlin Island SPA breeding puffin population was cited at 2,398 pairs, or 4,796 breeding adults, in 1985. (Furness, 2015, Stroud *et al*., 2016). Furness (2015) gave a breeding population of 695 pairs, or 1,390 breeding adults, in 2011. The most recent count was 407 individuals in 2021 (JNCC, 2023a); this is used as the reference population for the assessment.

1307. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.094 (1 – 0.906; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 38 breeding adults.

### Functional linkage and seasonal apportionment of potential effects

1308. The mean maximum foraging range of puffin is 137.1km (±128.3km) and the maximum foraging range is 383km (Woodward *et al*., 2019). The Project is located approximately 223km from Rathlin Island SPA, which means the Project is beyond the mean maximum foraging range of breeding puffins from the SPA, but within the mean maximum foraging range +1SD, and the maximum foraging range.

1309. The NatureScot apportioning tool (NatureScot, 2018) has been used to estimate the proportion of puffins from each of the relevant SPAs present at the windfarm site during the breeding season. The apportioning to SPA and non-SPA colonies is set out in **Appendix 12.1** of the ES. The tool estimated that 0.63% of adult birds present are likely to originate from Rathlin Island SPA.

1310. Outside of the breeding season, breeding puffins, including those from the Rathlin Island SPA, are not constrained by requirements to visit nests to incubate eggs or provision chicks. At this time, they are assumed to range more widely and to mix with puffins of all age classes from breeding colonies in the UK and further afield. The background population during these seasons is the UK Western Waters BDMPS. This consists of 304,557 individuals during the non-breeding season (August to March).

1311. Furness (2015) estimated that 18% of the Rathlin Island SPA breeding adults are present within the UK Western Waters BDMPS during the non-breeding season, which is 250 birds. This represents 0.1% of the BDMPS population for this period (304,557). It is therefore assumed that 0.1% of puffins present at the Project site are breeding adults from Rathlin Island SPA.

**Operation and maintenance phase disturbance/displacement/barrier effects**

*Project-alone*

1312. During the breeding season, the mean peak abundance of puffins present within the windfarm site and 2km buffer was 38.7 (7.7-80.6) individuals (refer to **Appendix 12.1** of the ES). Of these, less than one bird (0.2 (0.1-0.5) was likely to be a breeding adult from the Rathlin Island SPA.

1313. The mean peak abundance of puffins present within the windfarm site and 2km buffer during the non-breeding season was 19.7 (1.9-50.8) individuals. Of these, less than one bird (0.0 (0.0-0.0)) was likely to be a breeding adult from the Rathlin Island SPA.

1314. **Table 8.74** sets out the predicted annual impacts on puffins from Rathlin Island SPA. This estimates that there would be no measurable increase in mortality from the SPA population, assuming a displacement rate of 30-70% and a mortality of 1-10% for displaced birds.

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*Table 8.74 Puffin – predicted operation and maintenance phase displacement and mortality from Rathlin Island SPA*

| Mean peak abundance estimate type | Mean peak abundance estimate | Number of Rathlin Island SPA breeding adults present [1] | Annual mortality range[2] | Annual baseline mortality increase range[3] |
|---|---|---|---|---|
| Upper 95% CI | 80.6 (breeding)<br>50.8 (non-breeding)<br>131.4 (year round) | 0.5 (breeding)<br>0.0 (non-breeding)<br>0.6 (year round) | 0-0 | 0.00-0.01% |
| Mean | 38.7 (breeding)<br>19.7 (non-breeding)<br>58.4 (year round) | 0.2 (breeding)<br>0.0 (non-breeding)<br>0.3 (year round) | 0-0 | 0.00-0.00% |
| Lower 95% CI | 7.7 (breeding)<br>1.9 (non-breeding)<br>9.5 (year round) | 0.1 (breeding)<br>0.0 (non-breeding)<br>0.1 (year round) | 0-0 | 0.00-0.00% |

[1] Assumes 0.6% of birds present during the breeding season and 0.1% during the non-breeding season are Rathlin Island SPA breeding adults

[2] Assumes displacement rates of 30-70% and mortality rates of 1-10%

[3] Background mortality rate of 9.4% (Horswill and Robinson, 2015)

1315. **It is concluded that predicted puffin mortality due to operational phase displacement would not adversely affect the integrity of the Rathlin Island SPA.**

1316. The confidence in the assessment is high for several reasons. Firstly, the evidence used to inform the evidence-based displacement rates is of high applicability and quality (based on the criteria discussed in **Chapter 12 Offshore Ornithology** of the ES). Whilst there is limited available evidence to inform mortality rates, 1% is considered to be sufficiently precautionary based on expert opinion. This species is not regarded as being highly specialised in its habitat requirements (Bradbury *et al*., 2014; Furness and Wade, 2012; Garthe and Hüppop, 2004), and it is therefore anticipated that displaced birds will find alternative habitat in the vast majority of cases. Finally, the conclusion of the assessment is the same irrespective of whether the mean or 95% upper CI mean peak abundances are used to calculate potential mortality and increases in the baseline mortality rate of the background population.

*In-combination*

1317. As the Project would have no measurable effect on puffin populations from the Rathlin Island SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that predicted puffin mortality due to operational phase displacement due to the Project in-combination with other plans or projects would not adversely affect the integrity of the Rathlin Island SPA**.

# 8.29 Sheep Island SPA

1318. Sheep Island SPA is located approximately 231km from the windfarm site.

## 8.29.1 Description of designation

1319. Sheep Island is a 3.5ha island located 500m off the North Antrim coast. The island is almost circular, with a lower promontory to the north-west, a near vertical cliff face rising between 20m and 30m above the above Mean High Water and a domed top overlaid with a thin layer of soil.

## 8.29.2 Conservation objectives

1320. The overarching conservation objective for the SPA is 'to maintain each feature in favourable condition.' For the qualifying features, the objectives are:

- To maintain or enhance the population of the qualifying species
- Fledging success sufficient to maintain or enhance population
- To maintain or enhance the range of habitats utilised by the qualifying species
- To ensure that the integrity of the site is maintained
- To ensure there is no significant disturbance of the species
- To ensure that the following are maintained in the long term:
  - Population of the species as a viable component of the site
  - Distribution of the species within the site
  - Distribution and extend of habitats supporting the species
  - Structure, function and supporting processes of habitats supporting the species

## 8.29.3 Assessment

1321. One qualifying feature of Sheep Island SPA has been screened into the Appropriate Assessment (**Table 5.2**): breeding cormorant.

### 8.29.3.1 Cormorant

#### Status

1322. The Sheep Island SPA breeding cormorant population was cited as 249 pairs, or 498 breeding adults for the period 1992 – 1996 (Furness, 2015, Stroud *et al*., 2016). Furness (2015) gave a breeding population of 112 pairs, or 224 breeding adults in 2013. The most recent count was 139 pairs (AON), or 278 breeding adults in 2021 (JNCC, 2023a); this is used as the reference population for the assessment.

1323. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.132 (1 – 0.868; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 37 breeding adults.

### Functional linkage and seasonal apportionment of potential effects

1324. The mean maximum foraging range of cormorant is 25.6km (±8.3km) and the maximum foraging range is 35km (Woodward *et al*., 2019). The Project is located approximately 231km from Sheep Island SPA, which means the Project is beyond the maximum foraging range of cormorants from the SPA. No impacts during the breeding season from the Project are therefore apportioned to cormorants breeding at this SPA.

1325. Outside the breeding season, breeding cormorants from the SPA are assumed to range widely and to mix with birds of all ages from breeding colonies in the UK and beyond. However, as no cormorants were recorded within the windfarm site or 2km buffer, it can be concluded that no birds from Sheep Island SPA are likely to occur at the windfarm site.

### Potential effects on the qualifying feature from the Project-alone

1326. No effects on cormorants from Sheep Island SPA are predicted. **Therefore, it is concluded that there would be no adverse effect on the integrity of the Sheep Island SPA for the Project-alone**.

### Potential effects on the qualifying feature in-combination with other projects

1327. As the Project would have no measurable effect on cormorant populations from the Sheep Island SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of Sheep Island SPA, when assessed in-combination with other plans or projects**.

## 8.30 Farne Islands SPA

1328. Farne Islands SPA is located approximately 232km from the windfarm site.

### 8.30.1 Description of designation

1329. The Farne Islands are a group of low-lying islands situated between 2km and 6km off the coast of Northumberland. The islands are important as nesting areas for a range of seabirds, especially terns, gulls and auks. Seabirds breeding at the SPA feed outside it in nearby waters, as well as more distantly in the North Sea.

### 8.30.2 Conservation objectives

1330. The SPA's conservation objectives are to ensure that the integrity of the site is maintained or restored as appropriate, subject to natural change, and ensure that the site contributes to achieving the aims of the Wild Birds Directive, by maintaining or restoring:

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

### 8.30.3 Assessment

1331. One named component of the Farne Islands SPA seabird assemblage has been screened into the Appropriate Assessment (**Table 5.2**): breeding puffin.

#### 8.30.3.1 Puffin

#### Status

1332. The Farne Islands SPA breeding puffin population at classification was cited at 76,798 individuals (Natural England, 2017b). Furness (2015) gave a breeding population of 79,924 breeding adults for 2013. The most recent published count was 43,752 pairs (AON), or 87,504 breeding adults, in 2019 (JNCC, 2023a).

1333. Based on the most recent SPA population of breeding adults, and an annual breeding adult baseline mortality rate of 0.094 (1 – 0.906; Horswill and Robinson 2015), 8,225 breeding adults from the SPA population would be expected to die each year.

### Functional linkage and seasonal apportionment of potential effects

1334. The mean maximum foraging range of puffin is 137.1km (±128.3km) and the maximum foraging range is 383km (Woodward *et al*., 2019). The straight-line distance between the Project and Farne Islands SPA is approximately 232km, and therefore theoretically within the mean maximum foraging range +1SD, and the maximum foraging range for this species. However, the across-sea distance is approximately 1,120km, and therefore no breeding season connectivity between the Site and SPA puffin population is predicted.

1335. Outside of the breeding season, breeding puffins, including those from the Farne Islands SPA, are not constrained by requirements to visit nests to incubate eggs or provision chicks. At this time, they are assumed to range more widely and to mix with puffins of all age classes from breeding colonies in the UK and further afield. The background population during these seasons is the UK Western Waters BDMPS. This consists of 304,557 individuals during the non-breeding season (August to March).

1336. Furness (2015) estimated that 7% of the Farne Islands breeding adults are present within the UK Western Waters BDMPS during the non-breeding season, which is 5,595 birds. This represents 1.8% of the BDMPS population for this period (304,557). It is therefore assumed that 1.8% of puffins present at the Project site are breeding adults from Farne Islands SPA.

### Potential effects on the qualifying feature

#### Operation and maintenance phase disturbance/displacement/barrier effects

*Project-alone*

1337. The mean peak abundance of puffins present within the windfarm site and 2km buffer during the non-breeding season was 19.7 (1.9-50.8) individuals (refer to **Appendix 12.1** of the ES). Of these, less than one bird (0.35 (0.03-0.91)) was likely to be a breeding adult from Farne Islands SPA.

1338. **Table 8.75** sets out the predicted impacts on puffins from Farne Islands SPA during the non-breeding season. This estimates that there would be no measurable increase in mortality from the SPA population, assuming a displacement rate of 30-70% and a mortality of 1-10% for displaced birds.

*Table 8.75 Puffin – predicted operation and maintenance phase displacement and mortality from Farne Islands SPA*

| Mean peak abundance estimate type | Mean peak abundance estimate | Number of Farne Island SPA breeding adults present (non-breeding season)[1] | Annual mortality range[2] | Annual baseline mortality increase range[3] |
|---|---|---|---|---|
| Upper 95% CI | 50.8 | 0.9 | 0-0 | 0.00-0.00% |
| Mean | 19.7 | 0.4 | 0-0 | 0.00-0.00% |
| Lower 95% CI | 1.9 | 0.0 | 0-0 | 0.00-0.00% |

[1] Assumes 1.8% of birds present during the non-breeding season are Farne Island SPA breeding adults

[2] Assumes displacement rates of 30-70% and mortality rates of 1-10%

[3] Background mortality rate of 9.4% (Horswill and Robinson, 2015)

1339. **It is concluded that predicted puffin mortality due to operational phase displacement would not adversely affect the integrity of the Farne Islands SPA.**

1340. The confidence in the assessment is high for several reasons. Firstly, the evidence used to inform the evidence-based displacement rates is of high applicability and quality (based on the criteria discussed in **Chapter 12 Offshore Ornithology** of the ES). Whilst there is limited available evidence to inform mortality rates, 1% is considered to be sufficiently precautionary based on expert opinion. This species is not regarded as being highly specialised in its habitat requirements (Bradbury *et al*., 2014; Furness and Wade, 2012; Garthe and Hüppop, 2004), and it is therefore anticipated that displaced birds will find alternative habitat in the vast majority of cases. Finally, the conclusion of the assessment is the same irrespective of whether the mean or 95% upper CI mean peak abundances are used to calculate potential mortality and increases in the baseline mortality rate of the background population.

*In-combination*

1341. As the Project would have no measurable effect on puffin populations from the Farne Islands SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that predicted puffin mortality due to operational phase displacement due to the Project in-combination with other plans or projects would not adversely affect the integrity of the Farne Islands SPA.**

# 8.31 Forth Islands SPA

1342. Forth Islands SPA is located approximately 239km from the windfarm site.

## 8.31.1 Description of designation

1343. The Forth Islands SPA consists of a series of islands supporting the main seabird colonies in the Firth of Forth. The seaward elements extend approximately 2km to include the seabed, water column and surface. Seabirds included within the designation feed both inside and outside the SPA in nearby waters, as well as more distantly in the wider North Sea.

## 8.31.2 Conservation objectives

1344. The overarching conservation objectives of the site are:

- To avoid deterioration of the habitats of the qualifying species or significant disturbance to the qualifying species, thus ensuring that the integrity of the site is maintained
- To ensure for the qualifying species that the following are maintained in the long term
- Population of the species as a viable component of the site
- Distribution of the species within site
- Distribution and extent of habitats supporting the species
- Structure, function and supporting processes of habitats supporting the species
- No significant disturbance of the species

## 8.31.3 Assessment

1345. The qualifying features of Forth Islands SPA screened into the Appropriate Assessment are listed in **Table 5.2**. These are breeding gannet and breeding puffin. The seabird assemblage has also been screened in for these species.

### 8.31.3.1 Gannet

#### Status

1346. The Forth Islands SPA breeding gannet population at classification was cited at 21,600 pairs, or 43,200 breeding adults, for the period 1986 – 1988 (SNH, 2018a). Furness (2015) gave a breeding population of 55,482 breeding pairs, or 110,964 breeding adults, for 2009. The most recent available count is 75,259 pairs, or 150,518 breeding adults, in 2014 (JNCC, 2023a).

1347. Based on the most recent SPA population of breeding adults and an annual baseline adult mortality rate of 0.081 (1 – 0.919; Horswill and Robinson 2015), 12,192 breeding adults from the SPA population would be expected to die each year.

### Functional linkage and seasonal apportionment of potential effects

1348. The mean maximum foraging range of gannet is 315.2km (±194.2km), and the maximum foraging range is 709km (Woodward *et al*., 2019). The straight-line distance between the Project and Forth Islands SPA is approximately 239km, and therefore theoretically within the mean maximum foraging range and the maximum foraging range for this species. However, the across-sea distance exceeds 1,000km, and therefore no breeding season connectivity between the Site and SPA gannet population is predicted.

1349. Outside of the breeding season, breeding gannets, including those from the Forth Islands SPA, are not constrained by requirements to visit nests to incubate eggs or provision chicks. At this time, they are assumed to range more widely and to mix with gannets of all age classes from breeding colonies in the UK and further afield. The background population during these seasons is the UK Western Waters BDMPS. This consists of 545,954 individuals during the autumn migration season (September to November) and 661,888 individuals during the spring migration season (December to March).

1350. Furness (2015) estimated that 30% of the Forth Islands breeding adults (110,964) are present within the UK Western Waters BDMPS during the spring migration season, which is 33,289 birds. This represents 5.0% of the BDMPS population for this period (661,888). During the autumn migration season, Furness (2015) estimated that no birds from Forth Islands SPA are present within the UK Western Waters BDMPS. It is therefore assumed that 5.0% of gannets present at the Project site during the spring migration period are breeding adults from Forth Islands SPA, but none are present during autumn migration.

### Potential effects on the qualifying feature – Project-alone

1351. The gannet qualifying feature of the Forth Islands SPA has been screened into the assessment due to the potential risk of collision and displacement/barrier effects during the operation and maintenance phase of the Project.

#### Operation and maintenance phase displacement/barrier effects

1352. Displacement effects for gannet for the Project were assessed during the spring migration period, based on a peak mean population of eight birds, calculated for the windfarm site and a 2km buffer, in line with recommendations within the SNCB guidance (SNCB 2017). As set out above, no gannets present at the windfarm site have been apportioned to the Forth Islands SPA during the breeding or autumn migration seasons. The displacement matrices used to calculate potential impacts are presented in **Appendix 12.1** of the ES and summarised in **Table 8.76**. The inclusion of all birds within the 2km buffer, to determine the total number of birds subject to displacement, is precautionary, as in reality the avoidance rate is likely to fall with distance from the windfarm site.

1353. A displacement rate of 60-80% and mortality rate of 1% has been presented. A maximum 1% mortality value has been selected firstly because gannet are known to possess high habitat flexibility (Furness and Wade, 2012). This suggests that displaced birds will readily find alternative habitats including foraging areas. Secondly, no evidence of displacement-induced mortality has been identified, which means there is limited justification for setting predicted mortality rates at a higher level. Given the extensive foraging range of this species (Woodward *et al.*, 2019), there may be no mortality costs to displacement from the relatively very small footprints of OWFs.

*Table 8.76 Gannet – predicted operation and maintenance phase displacement and mortality from Forth Islands SPA*

| Mean peak abundance estimate type | Mean peak abundance estimate (spring migration) | Number of SPA breeding adults present by season[1] | Annual mortality range[2] | Annual baseline mortality increase range[3] |
|---|---|---|---|---|
| Upper 95% CI | 16.0 | 0.8 | 0-0 | 0.00-0.00% |
| Mean | 7.9 | 0.4 | 0-0 | 0.00-0.00% |
| Lower 95% CI | 0.0 | 0.0 | 0-0 | 0.00-0.00% |

[1] During spring migration period, 5.0% of birds are assumed to be breeding adults from the SPA population.

[2] Assumes displacement rates of 60-80% and mortality rate of 1%

[3] Background population is Forth Islands SPA breeding adults (150,518 individuals), adult age class annual mortality rate of 8.1% (Horswill and Robinson, 2015)

1354. Using the maximum potential mortality value, there would be no measurable increase in gannet mortality. Accordingly, no significant effects on gannet are predicted during the operation and maintenance phase, and **it is concluded that there would be no potential for the Project to have an adverse effect on the integrity of the Forth Islands SPA.**

1355. The confidence in the assessment is high for several reasons. Firstly, the evidence used to set the displacement rates presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES is of high applicability and quality. Whilst there is limited available evidence to inform mortality rates, 1% is considered to be sufficiently precautionary based on expert opinion. Finally, the conclusion of the assessment is the same irrespective of whether the mean or 95% upper CI mean peak abundances are used to calculate potential mortality and increases in the baseline mortality rate of the background population.

**Operation and maintenance phase collision risk**

1356. Information to support the Appropriate Assessment for collision risk on breeding adult gannets belonging to the Forth Islands SPA population is presented in **Table 8.77**. Collision estimates, calculated using the sCRM, are presented by biological season. A summary of the annual outputs and the corresponding increase in the annual baseline mortality rate is also presented. Parameters used in the sCRM were agreed with Natural England during the ETG process and are described in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES. In accordance with Natural England advice, a 70% macro-avoidance correction was applied to gannet abundance data used in the sCRM.

1357. Based on the mean collision rates, no breeding adult gannets from the Forth Islands SPA are considered at risk of collision as a result of the Project. Therefore, there would be no measurable increase the existing mortality of the SPA breeding population.

*Table 8.77 Predicted seasonal and annual collision mortality (Stochastic model Option 2, avoidance rate 0.993 (±0.0003), plus 70% macro-avoidance) for breeding adult gannets at the windfarm site, apportioned to the Forth Islands SPA, with corresponding increases to baseline mortality of the population*

| Period | Breeding Season<br>Apr-Aug | Autumn Migration<br>Sep-Nov | Non-breeding/winter<br>- | Spring Migration<br>Dec-Mar | Annual<br>Jan-Dec |
|---|---|---|---|---|---|
| Total collisions[1] (mean and 95% CIs) | 0.83<br>(0.00-3.35) | 0.14<br>(0.00-0.74) | - | 0.00 | 0.97<br>(0.00-4.10) |
| % apportioned to the SPA | 0.0% | 0.0% | - | 5.0% | - |
| Total SPA collisions (mean and 95% CIs) | 0.00<br>(0.00-0.00) | 0.00<br>(0.00-0.00) | - | 0.00<br>(0.00-0.00) | 0.00<br>(0.00-0.00) |
| Mortality increase[2] (mean and 95% CIs) | 0.00%<br>(0.00-0.00%) | 0.00% | - | 0.00% | 0.00%<br>(0.00-0.00%) |

[1] Breeding season collision values reduced to 73.8% of modelled value to reflect proportion of adult birds recorded during site surveys
[2] Assuming predicted annual SPA mortality of 12,192 birds (150,518 x 0.081)

1358. Accordingly, no significant effects on gannet are predicted during the operation and maintenance phase, and **it is concluded that there would be no potential for the Project to have an adverse effect on the integrity of the Forth Islands SPA.** Comments received from RSPB during the ETG process, indicating that they do not accept the 70% macro-avoidance rate for collision risk recommended by Natural England, are noted. However, even in the absence of this correction factor, the net increase in mortality would be unchanged (i.e. zero).

1359. The confidence in the assessment is high. The evidence used to define the CRM input parameters presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES submitted alongside this RIAA is of high applicability and quality. Whilst there is uncertainty around some of the input parameters (e.g. avoidance rate), the rates selected are considered to be sufficiently precautionary based on expert opinion to provide confidence that collision rates are not underestimated.

**Combined displacement/barrier effects and collision risk**

1360. As no measurable increase in mortality is predicted for both displacement and collision risk, the mean combined displacement and collision rates for breeding adult gannet from the Forth Islands SPA would be zero. Therefore, there would be no net increase in existing mortality rates.

1361. **It is concluded that based on predicted gannet mortality due to the combined effects of operational phase displacement and collision there is no potential for the Project-alone or in-combination to have an adverse effect on the integrity of the Forth Islands SPA.**

1362. The confidence in the assessment is high, for the reasons provided in the individual displacement and collision assessments.

### Potential effects in-combination with other projects

1363. As no measurable effects of displacement /barrier and collision on gannet are predicted as a result of the Project-alone, there would be no contribution to other plans or projects in-combination. **It is therefore concluded that there would be no potential for the Project-alone or in-combination to have an adverse effect on the integrity of the Forth Islands SPA.**

### 8.31.3.2 Puffin

### Status

1364. The Forth Islands SPA breeding puffin population at classification was cited as 14,000 pairs, or 28,000 breeding adults, for the period 1986 – 1988 (SNH, 2018a). Furness (2015) gave a breeding population of 62,231 breeding pairs, or 124,462 adults, for the period 2008 – 2010; this is used as the reference population for this assessment.

1365. Based on the most recent SPA population of breeding adults, and an annual breeding adult baseline mortality rate of 0.094 (1 - 0.906; Horswill and Robinson, 2015), 11,699 breeding adults from the SPA population would be expected to die each year.

### Functional linkage and seasonal apportionment of potential effects

1366. The mean maximum foraging range of puffin is 137.1km (±128.3km) and the maximum foraging range is 383km (Woodward *et al*., 2019). The straight-line distance between the Project and the Forth Islands SPA is approximately 239km, and therefore theoretically within the mean maximum foraging range +1SD, and the maximum foraging range for this species. However, the across-sea distance is >1,000km, and therefore no breeding season connectivity between the Site and SPA puffin population is predicted.

1367. Outside of the breeding season, breeding puffins, including those from the Forth Islands SPA, are not constrained by requirements to visit nests to incubate eggs or provision chicks. At this time, they are assumed to range more widely and to mix with puffins of all age classes from breeding colonies in the UK and further afield. The background population during these seasons is the UK Western Waters BDMPS. This consists of 304,557 individuals during the non-breeding season (August to March).

1368. Furness (2015) estimated that 7% of the Forth Islands breeding adults are present within the UK Western Waters BDMPS during the non-breeding season, which is 8,712 birds. This represents 2.9% of the BDMPS population for this period (304,557). It is therefore assumed that 2.9% of puffins present at the Project site are breeding adults from the Forth Islands SPA.

### Potential effects on the qualifying feature

#### Operation and maintenance phase disturbance/displacement/barrier effects

*Project-alone*

1369. The mean peak abundance of puffins present within the windfarm site and 2km buffer during the non-breeding season was 19.7 (1.9-50.8) individuals (refer to **Appendix 12.1** of the ES). Of these, less than one bird (0.57 (0.05-1.47)) was likely to be a breeding adult from the Forth Islands SPA.

1370. **Table 8.78** sets out the predicted impacts on puffins from Forth Islands SPA during the non-breeding season. This estimates that there would be no measurable increase in mortality from the SPA population, assuming a displacement rate of 30-70% and a mortality of 1-10% for displaced birds. This predicts that there would be zero mortality on this feature, and consequently no measurable increase in background mortality.

*Table 8.78 Puffin – predicted operation and maintenance phase displacement and mortality from Forth Islands SPA*

| Mean peak abundance estimate type | Mean peak abundance estimate | Number of Farne Island SPA breeding adults present (non-breeding season)[1] | Annual mortality range[2] | Annual baseline mortality increase range[3] |
|---|---|---|---|---|
| Upper 95% CI | 50.8 | 1.5 | 0-0 | 0.00-0.00% |
| Mean | 19.7 | 0.6 | 0-0 | 0.00-0.00% |
| Lower 95% CI | 1.9 | 0.1 | 0-0 | 0.00-0.00% |

[1] Assumes 2.9% of birds present during the non-breeding season are Farne Island SPA breeding adults

[2] Assumes displacement rates of 30-70% and mortality rates of 1-10%

[3] Background mortality rate of 9.4% (Horswill and Robinson, 2015)

1371. **It is concluded that predicted puffin mortality due to operational phase displacement would not adversely affect the integrity of the Forth Islands SPA.**

1372. The confidence in the assessment is high for several reasons. Firstly, the evidence used to inform the evidence-based displacement rates is of high applicability and quality (based on the criteria discussed in **Chapter 12 Offshore Ornithology** of the ES). Whilst there was limited available evidence to inform mortality rates, 1% is considered to be sufficiently precautionary based on expert opinion. This species is not regarded as being highly specialised in its habitat requirements (Bradbury *et al*., 2014; Furness and Wade, 2012; Garthe and Hüppop, 2004), and it is therefore anticipated that displaced birds will find alternative habitat in the vast majority of cases. Finally, the conclusion of the assessment is the same irrespective of whether the mean or 95% upper CI mean peak abundances are used to calculate potential mortality and increases in the baseline mortality rate of the background population.

### *In-combination*

1373. As the Project would have no measurable effect on puffin populations from the Forth Islands SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that predicted puffin mortality due to operational phase displacement due to the Project in-combination with other plans or projects would not adversely affect the integrity of the Forth Islands SPA.**

## 8.32 Skomer, Skokholm and the Seas off Pembrokeshire/ Sgomer, Sgogwm a Moroedd Penfro SPA

1374. Skomer, Skokholm and the Seas off Pembrokeshire/Sgomer, Sgogwm a Moroedd Penfro SPA is located approximately 246km from the windfarm site.

### 8.32.1 Description of designation

1375. Skomer, Skokholm and the Seas off Pembrokeshire SPA is located off the south-west tip of Pembrokeshire in south-west Wales. This SPA extends beyond the 12 nautical mile boundary, lying partly in Welsh territorial waters and partly in UK offshore waters. The islands of Skomer and Skokholm support the largest concentration of breeding seabirds in England and Wales. They hold the largest breeding colony of Manx shearwater in the world, one of the largest colonies of lesser black-backed gull in Britain, as well as being important Welsh breeding sites for other seabird populations, such as razorbill, black-legged kittiwake, Atlantic puffin and common guillemot, supporting a breeding seabird assemblage of over 394,000 birds.

### 8.32.2 Conservation objectives

1376. The overarching conservation objectives for each of the qualifying features of the SPA are:

- The size of the population should be stable or increasing, allowing for natural variability, and sustainable in the long term
- The distribution of the population should be being maintained, or where appropriate increasing
- There should be sufficient habitat, of sufficient quality, to support the population in the long term
- Factors affecting the population or its habitat should be under appropriate control

### 8.32.3 Assessment

#### 8.32.3.1 Manx shearwater

##### Status

1377. Manx shearwater is listed as a qualifying species of this SPA.

1378. The SPA population at classification was cited as 150,968 pairs in 1998 (Furness, 2015, Stroud *et al*., 2016). Furness (2015) gave a breeding population of 350,000 pairs in 2011. The most recent count (2018) is 455,156

AOS (burrows or crevices; Skomer 349,663, Skokholm 88,945 and Middleholm 16,548), or 910,312 breeding adults (JNCC, 2022).

1379. Based on the most recent SPA population of breeding adults, and an annual breeding adult baseline mortality rate of 0.13 (1-0.870, Horswill and Robinson (2015)), 118,341 breeding adults from the SPA population would be expected to die each year.

**Functional linkage and seasonal apportionment of potential effects**

1380. The windfarm site is situated approximately 250km from Skomer (the closer of the two colonies), at its nearest point; the across-sea distance is approximately 270km. The mean maximum foraging range of Manx shearwater is 1,347km (±1,019km) (Woodward *et al*., 2019). The windfarm site is therefore within the mean maximum foraging range of Manx shearwaters from the Skomer, Skokholm and the Seas off Pembrokeshire / Sgomer, Sgogwm a Moroedd Penfro SPA.

1381. A tracking study of Manx shearwaters from the Skomer colony was documented in Guilford *et al.,* (2008). This indicated that the majority of birds foraged in areas to the north and west of the colony (typically within 100km). However, a proportion of trips continued further north into the Irish Sea, including the Irish Sea Front SPA area, the west coast of Northern Ireland, and as far north as the west coast of southern Scotland adjoining the Rhins of Galloway.

1382. A number of SPA and non-SPA Manx shearwater colonies are located in and around the UK Western Waters BDMPS area, all of which are within the mean maximum foraging range of this species. For a review of these sites see Section 8.21.3.1.

1383. The NatureScot apportioning tool (NatureScot, 2018) has been used to estimate the proportion of Manx shearwaters from each of the relevant SPAs present at the windfarm site during the breeding season. The apportioning to SPA and non-SPA colonies is set out in **Table 8.79**; refer also to **Appendix 12.1** of the ES for further information on the apportioning approach and results. Accordingly, 76.54% of impacts at the windfarm site during the breeding season are apportioned to Skomer, Skokholm and the Seas off Pembrokeshire / Sgomer, Sgogwm a Moroedd Penfro SPA.

*Table 8.79 Manx shearwater breeding season apportioning*

| Site | Apportioning rate |
|---|---|
| Glannau Aberdaron ac Ynys Enlli/Aberdaron Coast and Bardsey Island SPA | 8.63% |
| Copeland Islands SPA | 2.22% |

| Site | Apportioning rate |
|---|---|
| **Skomer, Skokholm and the Seas off Pembrokeshire / Sgomer, Sgogwm a Moroedd Penfro SPA** | **76.54%** |
| Rum SPA | 8.44% |
| St Kilda SPA | 0.20% |
| Cruagh Island SPA (transboundary site) | 0.10% |
| Blasket Islands SPA (transboundary site) | 0.61% |
| Deenish Island and Scariff Island SPA (transboundary site) | 0.08% |
| Puffin Island SPA (transboundary site) | 0.22% |
| Skelligs SPA (transboundary site) | 0.03% |
| Non-SPA colonies | 2.91% |

1384. During the pre- and post-breeding periods, breeding Manx shearwaters from the Skomer, Skokholm and the Seas off Pembrokeshire / Sgomer, Sgogwm a Moroedd Penfro SPA migrate through UK waters. The relevant reference population is considered to be the UK Western Waters BDMPS. This consists of 1,580,895 individuals during the post-breeding (August-early October) and return migration (late March-May) periods.

1385. Estimates of the proportion of Manx shearwaters present at the windfarm site which originate from the Skomer, Skokholm and the Seas off Pembrokeshire / Sgomer, Sgogwm a Moroedd Penfro SPA during the post-breeding and return migration periods (and therefore the proportion of predicted mortalities from the SPA population) are based on the SPA population (as published in Furness (2015); i.e. 700,000 adults) as a proportion of the UK Western Waters BDMPS during the relevant season (1,580,895 birds during the post-breeding and return migration periods). Therefore, during the post-breeding and return migration periods, 44.3% of impacts are considered to affect birds from the SPA (Furness, 2015).

### Potential effects on the qualifying feature from the Project-alone

1386. The Manx shearwater qualifying feature of the Skomer, Skokholm and the Seas off Pembrokeshire/Sgomer, Sgogwm a Moroedd Penfro SPA has been screened into the assessment due to the potential risk of disturbance, displacement and barrier effects during the construction and decommissioning, and operation and maintenance phases of the Project.

#### *Construction and decommissioning phase disturbance/displacement/barrier effects*

1387. Effects during the construction and decommissioning phases of the Project are considered unlikely, given the transient presence of the species and low susceptibility to disturbance related impacts; refer to **Paragraph 1389** below. However, in accordance with feedback received from Natural England and NRW, a precautionary estimation of construction and decommissioning phase disturbance, displacement and barrier effects has been undertaken assuming 50% of the operational phase effect.

1388. Applying 50% reduction to the operational values presented in **Table 8.80**, and based on mean density, predicted mortality would be between eight and 192 birds (30-70% displacement and 1-10% mortality of displaced birds). Using realistic values (i.e. mean density, 50% displacement and 1% mortality), there would be an annual increase in mortality of 14 birds, which is equivalent to a 0.01% increase in background mortality for the SPA population. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. Accordingly, no significant effects on Manx shearwater are predicted during the construction and decommissioning phases, and **it is concluded that there would be no adverse effect on the integrity of Skomer, Skokholm and the Seas off Pembrokeshire/Sgomer, Sgogwm a Moroedd Penfro SPA**.

**Operation and maintenance phase disturbance/displacement/barrier effects**

1389. Manx shearwater are generally considered to have a low susceptibility to disturbance and displacement (Furness *et al.,* 2013). Dierschke *et al*., (2016) described Manx shearwater as "weakly avoiding wind farms", although also noted that evidence was lacking for the species. Bradbury *et al.,* (2014) classified Manx shearwater as having "very low" population vulnerability to displacement.

1390. Dierschke *et al*., (2016) suggested that Manx shearwater were avoiding North Hoyle OWF, stating that an obvious distribution gap was observed at the OWF, although evidence for this appeared limited. Dierschke *et al.,* (2016) also noted that Manx shearwater had been recorded within Robin Rigg OWF.

1391. Displacement effects for Manx shearwater for the Project were assessed during the breeding, autumn migration and spring migration periods, based on an unapportioned peak mean population of 4,705, 2,650 and 1,617 birds respectively, calculated for the windfarm site and a 2km buffer, in line with recommendations within the SNCB guidance (SNCBs, 2017). The displacement matrices used to calculate potential impacts are presented in **Appendix 12.1** of the ES and summarised in **Table 8.80**. The application of the same displacement rate to the OWF and the 2km buffer, to determine the total number of birds subject to displacement, is precautionary, as in reality the displacement rate is likely to fall with distance from the windfarm site.

1392. A displacement rate of 30-70% and mortality rate of 1-10% has been presented. Given that 10% would represent a rate close to the expected 'natural' annual mortality (0.13), this rate is considered very unlikely.

Accordingly, a 1% mortality rate is considered to be most appropriate, with the upper end of this range likely to be precautionary. Given the very extensive foraging range of this species (Woodward *et al.*, 2019), there may be no mortality costs to displacement from the relatively very small footprints of OWFs.

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*Table 8.80 Manx shearwater – predicted operation and maintenance phase displacement and mortality from Skomer, Skokholm and the Seas off Pembrokeshire/Sgomer, Sgogwm a Moroedd Penfro SPA*

| Mean peak abundance estimate type | Mean peak abundance estimate | Number of SPA breeding adults present by season[1] | Annual mortality range[2] | Annual baseline mortality increase range[3] |
|---|---|---|---|---|
| Upper 95% CI | 10,010 (breeding)<br>4,447 (autumn)<br>4,711 (spring)<br>19,168 (year round) | 7,662 (breeding)<br>1,969 (autumn)<br>2,086 (spring)<br>11,717 (year round) | 35-820 | 0.03-0.69% |
| Mean | 4,705 (breeding)<br>2,650 (autumn)<br>1,617 (spring)<br>8,972 (year round) | 3,601 (breeding)<br>1,174 (autumn)<br>716 (spring)<br>5,491 (year round) | 16-384 | 0.01-0.32% |
| Lower 95% CI | 783 (breeding)<br>1,308 (autumn)<br>0 (spring)<br>2,092 (year round) | 600 (breeding)<br>580 (autumn)<br>0 (spring)<br>1,179 (year round) | 4-83 | 0.00-0.07% |

[1] During the breeding season, assumes 76.5% of recorded birds are adults from the SPA population (910,312), and 44.3% during the autumn and spring migration periods

[2] Assumes displacement rates of 30-70% and mortality rates of 1-10%

[3] Background population is Skomer, Skokholm and the Seas off Pembrokeshire / Sgomer, Sgogwm a Moroedd Penfro SPA breeding adults (910,312 individuals), adult age class annual mortality rate of 13% (Horswill and Robinson, 2015)

1393. Using realistic values (i.e. mean density, 50% displacement and 1% mortality), there would be an annual increase in mortality of 27.5 birds (0.02%). Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. Accordingly, no significant effects on Manx shearwater are predicted during the operation and maintenance phase, and **it is concluded that there would be no potential for the Project-alone to have an adverse effect on the integrity of Skomer, Skokholm and the Seas off Pembrokeshire / Sgomer, Sgogwm a Moroedd Penfro SPA**.

1394. A review of the potential effects of artificial light on Manx shearwaters is presented in Section 12.6.3.1 of **Chapter 12 Offshore Ornithology**. This concludes that lighting associated with the Project is very unlikely to significantly affect disturbance and displacement effects on Manx shearwater, and therefore the conclusions of the assessment are unchanged.

1395. The confidence in the assessment is high. The evidence used to set the displacement rates presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES is of high applicability and quality. Whilst there is limited available evidence to inform mortality rates, 1% is considered to be sufficiently precautionary based on expert opinion.

### Potential effects in-combination with other projects

1396. No in-combination effects are predicted during the construction and decommissioning phases. This is because it is unlikely that there would be significant temporal and/or spatial overlap with other plans or projects, and due to the negligible effects predicted from the project alone.

1397. During the operation and maintenance phase, the in-combination assessment for Manx shearwaters from Skomer, Skokholm and the Seas off Pembrokeshire/Sgomer, Sgogwm a Moroedd Penfro SPA has been undertaken in accordance with the approach presented in **Section 8.1**. The total population apportioned to the SPA at risk of displacement is estimated to be 17,152 breeding adults (**Appendix 12.1** of the ES). Annual in-combination displacement and mortality rates for birds from Skomer, Skokholm and the Seas off Pembrokeshire/Sgomer, Sgogwm a Moroedd Penfro SPA are presented in **Table 8.81**.

1398. Assuming a displacement rate of 70% and a mortality rate of 10% of displaced birds, 1,201 breeding adult SPA birds would be lost to displacement annually. This would increase the existing mortality within the SPA population (118,341 breeding adult birds per year) by 1.01%. Using a realistic displacement rate of 50%, and a mortality rate for displaced birds of 1%, the annual in-combination mortality would be 86 birds. This would increase the existing mortality within this population by 0.07%. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that detectable changes in mortality rates will not occur due to the level or mortality predicted if the more realistic rates for mortality are used.

1399. **It is concluded that predicted Manx shearwater mortality due to of operational phase displacement impacts of the Project in-combination with other projects, would not adversely affect the integrity of Skomer, Skokholm and the Seas off Pembrokeshire/Sgomer, Sgogwm a Moroedd Penfro SPA.** This accords with the conclusions of the Round 4 offshore wind leasing HRA (NIRAS, 2021), which concluded no effect on site integrity (for all SPAs) on the basis of the low vulnerability to OWFs and low density of this species within Round 4 areas.

1400. It is noted that limited or no data are available from five historic projects that may have the potential to contribute to the in-combination effect on this feature (Burbo Bank, Walney 1&2, Gwynt y Môr, Rhyl Flats and Robin Rigg). As set out in the cumulative assessment for Manx shearwater presented in **Chapter 12 Offshore Ornithology** of the ES, in each case the assessments for these projects concluded no impact, or ‘low’ or ‘very low’ significance effects on this species. In order to reach a threshold where a significant effect might be possible (i.e. an increase in background mortality >1% affecting the SPA Manx shearwater population, assuming realistic displacement rates of 50%/1%), these historic projects would need contribute approximately 219,529 additional birds annually to the total potentially impacted population. This would equate to approximately 43,906 birds apportioned to the SPA at each project site annually. Given that the largest contribution from single project where data are available is 5,491 birds (for the Project), it is considered extremely unlikely that such high contributions could arise from these historic projects. Accordingly, it is concluded that these additional projects would not affect the conclusion of the assessment.

*Table 8.81 In-combination year-round displacement matrix for Manx shearwater from Skomer, Skokholm and the Seas off Pembrokeshire / Sgomer, Sgogwm a Moroedd Penfro SPA*

| **Annual** | **Mortality** | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| **Displacement** | **1%** | **2%** | **3%** | **4%** | **5%** | **10%** | **20%** | **30%** | **50%** | **80%** | **100%** |
| **10%** | 17 | 34 | 51 | 69 | 86 | 172 | 343 | 515 | 858 | 1372 | 1715 |
| **20%** | 34 | 69 | 103 | 137 | 172 | 343 | 686 | 1029 | 1715 | 2744 | 3430 |
| **30%** | 51 | 103 | 154 | 206 | 257 | 515 | 1029 | 1544 | 2573 | 4116 | 5146 |
| **40%** | 69 | 137 | 206 | 274 | 343 | 686 | 1372 | 2058 | 3430 | 5489 | 6861 |
| **50%** | 86 | 172 | 257 | 343 | 429 | 858 | 1715 | 2573 | 4288 | 6861 | 8576 |
| **60%** | 103 | 206 | 309 | 412 | 515 | 1029 | 2058 | 3087 | 5146 | 8233 | 10291 |
| **70%** | 120 | 240 | 360 | 480 | 600 | 1201 | 2401 | 3602 | 6003 | 9605 | 12006 |
| **80%** | 137 | 274 | 412 | 549 | 686 | 1372 | 2744 | 4116 | 6861 | 10977 | 13721 |
| **90%** | 154 | 309 | 463 | 617 | 772 | 1544 | 3087 | 4631 | 7718 | 12349 | 15437 |
| **100%** | 172 | 343 | 515 | 686 | 858 | 1715 | 3430 | 5146 | 8576 | 13721 | 17152 |

Note: The cells show the number of birds predicted to die (rounded to the nearest integer) at a given rate of displacement and mortality. Highlighted cells are considered to be the most realistic scenario, in accordance with SNCB advice (SNCBs, 2022).

#### 8.32.3.2 European storm-petrel

##### Status

1401. The Skomer, Skokholm and the Seas off Pembrokeshire / Sgomer, Sgogwm a Moroedd Penfro SPA breeding European storm-petrel population was cited as 3,500 pairs, or 7,000 breeding adults, in 1982. Combined counts from Skomer in 2000 and Skokholm in 2001 give a total of 2,560 pairs (AOS), or 5120 breeding adults (JNCC, 2023a); this is the most recent complete survey data available and is used as the reference population for the assessment.

##### Functional linkage and seasonal apportionment of potential effects

1402. The mean maximum foraging range of European storm-petrel is 336km, as is the maximum foraging range (Woodward *et al*., 2019). The Project is located approximately 246km from Skomer, Skokholm and the Seas off Pembrokeshire/Sgomer, Sgogwm a Moroedd Penfro SPA, which means that the Project is within the mean maximum foraging range of European storm-petrels breeding at this SPA.

##### Potential effects on the qualifying feature from the Project-alone

1403. Storm petrel was not recorded during baseline surveys of the windfarm site (including buffer areas). It is therefore concluded that this species does not occur regularly in this area. It is noted that storm petrel is considered to have low vulnerability to collision risk and very low vulnerability to displacement impacts (Bradbury *et al*., 2014), and therefore the risk of significant effects would be low, even if this species occurred at the windfarm site.

1404. **It is therefore concluded that there would be no measurable effects on storm petrel due to the project alone, and no adverse effect on the integrity of the Skomer, Skokholm and the Seas off Pembrokeshire/Sgomer, Sgogwm a Moroedd Penfro SPA is predicted.**

##### Potential effects on the qualifying feature in-combination with other projects

1405. As the Project would have no measurable effect on storm petrel populations from the Skomer, Skokholm and the Seas off Pembrokeshire/Sgomer, Sgogwm a Moroedd Penfro SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of Skomer, Skokholm and the Seas off Pembrokeshire / Sgomer, Sgogwm a Moroedd Penfro SPA, when assessed in-combination with other plans or projects.**

### 8.32.3.3 Puffin

#### Status

1406. The Skomer, Skokholm and the Seas off Pembrokeshire / Sgomer, Sgogwm a Moroedd Penfro SPA breeding puffin population was cited as 9,500 pairs, or 19,000 breeding adults in the mid-1980s (Furness, 2015, Stroud *et al*., 2016). Furness (2015) gave a breeding population of 24,114 pairs, or 48,228 breeding adults, in 2013. The most recent count is 40,088 birds (individual birds, birds at sea and birds in the air combined) in 2022 (JNCC, 2023a).

1407. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.094 (1 – 0.906; Horswill and Robinson 2015); the expected annual mortality from the SPA population would be 3,768 breeding adults.

#### Functional linkage and seasonal apportionment of potential effects

1408. The mean maximum foraging range of puffin is 137.1km (±128.3km) and the maximum foraging range is 383km (Woodward *et al*., 2019). The Project is located approximately 246km from Skomer, Skokholm and the Seas off Pembrokeshire / Sgomer, Sgogwm a Moroedd Penfro SPA, which means that the Project is beyond the mean maximum foraging range of puffins breeding at this SPA, but within the mean maximum foraging range +1SD, and the maximum foraging range.

1409. The NatureScot apportioning tool (NatureScot, 2018) has been used to estimate the proportion of puffins from each of the relevant SPAs present at the windfarm site during the breeding season. The apportioning to SPA and non-SPA colonies is set out in **Appendix 12.1** of the ES. The tool estimates that 54.4% of adult birds present are likely to originate from Skomer, Skokholm and the Seas off Pembrokeshire/Sgomer, Sgogwm a Moroedd Penfro SPA. Tracking studies from Skomer (Fayet *et al.*, 2021) indicate that the core puffin forging area around the colony extends to only 52km (SE ±1.8) from the island for 'long trips', and therefore the apportioning estimate is likely to overestimate the proportion of SPA birds present at the windfarm site.

1410. Outside of the breeding season, breeding puffins, including those from the Skomer, Skokholm and the Seas off Pembrokeshire / Sgomer, Sgogwm a Moroedd Penfro SPA, are not constrained by requirements to visit nests to incubate eggs or provision chicks. At this time, they are assumed to range more widely and to mix with puffins of all age classes from breeding colonies in the UK and further afield. The background population during these seasons is the UK Western Waters BDMPS. This consists of 304,557 individuals during the non-breeding season (August to March).

1411. Furness (2015) estimated that 18% of the Skomer, Skokholm and the Seas off Pembrokeshire/Sgomer, Sgogwm a Moroedd Penfro SPA breeding adults are present within the UK Western Waters BDMPS during the non-breeding season, which is 8,681 birds. This represents 2.8% of the BDMPS population for this period (304,557). It is therefore assumed that 2.8% of puffins present at the Project site are breeding adults from Skomer, Skokholm and the Seas off Pembrokeshire / Sgomer, Sgogwm a Moroedd Penfro SPA.

**Operation and maintenance phase disturbance / displacement / barrier effects**

*Project-alone*

1412. During the breeding season, the mean peak abundance of puffins present within the windfarm site and 2km buffer was 38.7 (7.7-80.6) individuals (refer to **Appendix 12.1** of the ES). Of these, 21.0 birds (4.2-43.9) were likely to be breeding adults from the Skomer, Skokholm and the Seas off Pembrokeshire/Sgomer, Sgogwm a Moroedd Penfro SPA.

1413. The mean peak abundance of puffins present within the windfarm site and 2km buffer during the non-breeding season was 19.7 (1.9-50.8) individuals. Of these, less than one bird (0.55 (0.05-1.42)) was likely to be a breeding adult from the Skomer, Skokholm and the Seas off Pembrokeshire/Sgomer, Sgogwm a Moroedd Penfro SPA.

1414. **Table 8.82** sets out the predicted annual impacts on puffins from Skomer, Skokholm and the Seas off Pembrokeshire / Sgomer, Sgogwm a Moroedd Penfro SPA. This estimates that there would be no measurable increase in mortality from the SPA population, assuming a displacement rate of 30-70% and a mortality of 1-10% for displaced birds.

*Table 8.82 Puffin – predicted operation and maintenance phase displacement and mortality from Skomer, Skokholm and the Seas off Pembrokeshire / Sgomer, Sgogwm a Moroedd Penfro SPA*

| Mean peak abundance estimate type | Mean peak abundance estimate | Number of SPA breeding adults present [1] | Annual mortality range[2] | Annual baseline mortality increase range[3] |
|---|---|---|---|---|
| Upper 95% CI | 80.6 (breeding)<br>50.8 (non-breeding)<br>131.4 (year round) | 43.9 (breeding)<br>1.4 (non-breeding)<br>45.3 (year round) | 0-3 | 0.00-0.08% |
| Mean | 38.7 (breeding)<br>19.7 (non-breeding) | 21.0 (breeding)<br>0.6 (non-breeding) | 0-2 | 0.00-0.04% |

| Mean peak abundance estimate type | Mean peak abundance estimate | Number of SPA breeding adults present [1] | Annual mortality range[2] | Annual baseline mortality increase range[3] |
|---|---|---|---|---|
| | 58.4 (year round) | 21.6 (year round) | | |
| Lower 95% CI | 7.7 (breeding)<br>1.9 (non-breeding)<br>9.5 (year round) | 4.2 (breeding)<br>0.1 (non-breeding)<br>4.2 (year round) | 0-0 | 0.00-0.01% |

[1] Assumes 54.4% of birds present during the breeding season and 2.8% during the non-breeding season are Skomer, Skokholm and the Seas off Pembrokeshire / Sgomer, Sgogwm a Moroedd Penfro SPA breeding adults

[2] Assumes displacement rates of 30-70% and mortality rates of 1-10%

[3] Background mortality rate of 9.4% (Horswill and Robinson, 2015)

1415. **It is concluded that predicted puffin mortality due to operational phase displacement would not adversely affect the integrity of the Skomer, Skokholm and the Seas off Pembrokeshire/Sgomer, Sgogwm a Moroedd Penfro SPA.**

1416. The confidence in the assessment is high for several reasons. Firstly, the evidence used to inform the evidence-based displacement rates is of high applicability and quality (based on the criteria discussed in **Chapter 12 Offshore Ornithology** of the ES). Whilst there is limited available evidence to inform mortality rates, 1% is considered to be sufficiently precautionary, based on expert opinion. This species is not regarded as being highly specialised in its habitat requirements (Bradbury *et al*., 2014; Furness and Wade, 2012; Garthe and Hüppop, 2004), and it is therefore anticipated that displaced birds will find alternative habitat in the vast majority of cases. Finally, the conclusion of the assessment is the same irrespective of whether the mean or 95% upper CI mean peak abundances are used to calculate potential mortality and increases in the baseline mortality rate of the background population.

*In-combination*

1417. As the Project would have no measurable effect on puffin populations from the Skomer, Skokholm and the Seas off Pembrokeshire / Sgomer, Sgogwm a Moroedd Penfro SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that predicted puffin mortality due to operational phase displacement due to the Project in-combination with other plans or projects would not adversely affect the integrity of the Skomer, Skokholm and the Seas off Pembrokeshire / Sgomer, Sgogwm a Moroedd Penfro SPA**.

#### 8.32.3.4 Lesser black-backed gull

##### Status

1418. The Skomer, Skokholm and the Seas off Pembrokeshire / Sgomer, Sgogwm a Moroedd Penfro SPA breeding lesser black-backed gull population was cited as 20,300 pairs, or 40,600 breeding adults, for the period 1993 – 1997 (Furness, 2015, Stroud *et al*., 2016). Furness (2015) gave the breeding population of 9,640 pairs, or 19,280 in 2013. The most recent total count from Skomer and Skokholm is 8,437 breeding pairs, or 16,694 breeding adults, in 2021 (JNCC, 2023a); this is used as the reference population for the assessment.

1419. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.115 (1 – 0.885; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 1,920 breeding adults.

##### Functional linkage and seasonal apportionment of potential effects

1420. The mean maximum foraging range of lesser black-backed gull is 127km (±109km) and the maximum foraging range is 533km (Woodward *et al*., 2019). The Project is located approximately 246km from Skomer, Skokholm and the Seas off Pembrokeshire / Sgomer, Sgogwm a Moroedd Penfro SPA, which means that the Project is beyond the mean maximum foraging range +1SD of lesser black-backed gulls breeding at this SPA, but within the maximum foraging range. The maximum foraging range is a poor indicator of typical foraging behaviour. It would be expected that few birds or foraging trips will occur at this distance from the colony, and even fewer with any regularity. No impacts during the breeding season are therefore apportioned to birds breeding at this colony.

1421. Outside the breeding season, breeding lesser black-backed gulls from the SPA are assumed to range widely and to mix with lesser black-backed gulls of all ages from breeding colonies in the UK and beyond. The relevant non-breeding season reference population is the UK Western Waters BDMPS, consisting of 163,304 individuals during spring and autumn migration (March and September to October) and 41,159 during winter (November to February) (Furness, 2015).

1422. Furness (2015) estimated that 70% of the Skomer, Skokholm and the Seas off Pembrokeshire / Sgomer, Sgogwm a Moroedd Penfro SPA breeding adults are present within the UK Western Waters and Channel BDMPS during the autumn and spring migration periods, which is 13,496 birds. During the winter period 20% of the population is estimated to be present, which is 3,856 birds. This represents 8.26% of the BDMPS population for the autumn and spring periods, and 9.37% during the winter period. Impacts to birds from the SPA during these periods are therefore apportioned accordingly.

## Potential effects on the qualifying feature from the Project-alone

1423. The lesser black-backed gull qualifying feature of the Skomer, Skokholm and the Seas off Pembrokeshire/Sgomer, Sgogwm a Moroedd Penfro SPA has been screened into the Appropriate Assessment due to the potential risk of collision.

### Operation and maintenance phase collision risk

1424. Information for collision risk on breeding adult lesser black-backed gulls belonging to the Skomer, Skokholm and the Seas off Pembrokeshire/Sgomer, Sgogwm a Moroedd Penfro SPA population is presented in **Table 8.83**. Collision estimates, calculated using the sCRM, are presented by biological season. A summary of the annual outputs and the corresponding increase in the annual baseline mortality rate is also presented. Parameters used in the sCRM were agreed with Natural England during the ETG process and are described in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES.

1425. Based on the mean collision rates, the annual total of breeding adult lesser black-backed gulls from Skomer, Skokholm and the Seas off Pembrokeshire/Sgomer, Sgogwm a Moroedd Penfro SPA at risk of collision as a result of the Project is less than one bird (0.13). This would increase the existing mortality of the SPA breeding population by 0.11%.

*Table 8.83 Predicted seasonal and annual collision mortality (Stochastic model Option 2, avoidance rate 0.994 (±0.0004)), for breeding adult lesser black-backed gulls at the windfarm site, apportioned to Skomer, Skokholm and the Seas off Pembrokeshire / Sgomer, Sgogwm a Moroedd Penfro SPA, with corresponding increases to baseline mortality of the population*

| Period | Breeding Season<br>Apr-Aug | Autumn Migration<br>Sep-Oct | Non-breeding/winter<br>Nov-Feb | Spring Migration<br>Mar | Annual<br>Jan-Dec |
|---|---|---|---|---|---|
| Total collisions[1] (mean and 95% CIs) | 1.44<br>(0.00-4.53) | 1.25<br>(0.00-5.63) | 0.15<br>(0.00-0.80) | 0.15<br>(0.00-0.94) | 2.98<br>(0.00-11.90) |
| % apportioned to the SPA | 0.00% | 8.26% | 9.37% | 8.26% | - |
| Total SPA collisions (mean and 95% CIs) | 0.00<br>(0.00-0.00) | 0.10<br>(0.00-0.47) | 0.01<br>(0.00-0.07) | 0.01<br>(0.00-0.08) | 0.13<br>(0.00-0.62) |
| Mortality increase[2] (mean and 95% CIs) | 0.00%<br>(0.00-0.00%) | 0.09%<br>(0.00-0.40%) | 0.01%<br>(0.00-0.06%) | 0.01%<br>(0.00-0.07%) | 0.11%<br>(0.00-0.53%) |

[1] Breeding season collision values reduced to 71.9% of modelled value to reflect proportion of adult birds recorded during site surveys
[2] Assuming predicted annual SPA mortality of 1,920 birds (16,694 x 0.115)

1426. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that no detectable changes in mortality rates would occur on this population for the mean monthly collision estimates for the Project, or for the upper 95% CI collision estimate.

1427. **It is concluded that, based on predicted lesser black-backed gull mortality due to collision at the windfarm site, there is no potential for the Project to have an adverse effect on the integrity of the Skomer, Skokholm and the Seas off Pembrokeshire/Sgomer, Sgogwm a Moroedd Penfro SPA.**

1428. The confidence in the assessment is high. The evidence used to define the CRM input parameters presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** as part of the ES submitted alongside this RIAA is of high applicability and quality. Whilst there is uncertainty around some of the input parameters (e.g. avoidance rate), the rates selected are considered to be sufficiently precautionary based on expert opinion to provide confidence that collision rates are not underestimated.

### Potential effects on the qualifying feature in-combination with other projects

1429. The in-combination assessment for lesser black-backed gulls from Skomer, Skokholm and the Seas off Pembrokeshire/Sgomer, Sgogwm a Moroedd Penfro SPA due to collision risk has been undertaken in accordance with the approach presented in **Section 8.1** and **Appendix 12.1** of the ES. **Table 8.84** sets out the predicted annual mortality for relevant projects.

*Table 8.84 Lesser black-backed gull – predicted in-combination collision mortality from Skomer, Skokholm and the Seas off Pembrokeshire / Sgomer, Sgogwm a Moroedd Penfro SPA*

| Project name | Predicted annual mortality |
|---|---|
| Burbo Bank Extension | 2.28 |
| Ormonde | 1.15 |
| Walney 1 + 2 | 2.97 |
| Walney 3 + 4 | 1.52 |
| West of Duddon Sands | 2.72 |
| Gwynt y Môr | 0.26 |
| Rhyl Flats | 0.05 |
| Robin Rigg | 'low/negligible' |
| Awel y Môr | 0.00 |
| Erebus | 0.42 |

| Project name | Predicted annual mortality |
|---|---|
| Twin Hub | 0.42 |
| Morgan Generation Assets | 0.05 |
| Mona | 0.10 |
| Burbo Bank | 0.12 |
| West of Orkney | 0.00 |
| White Cross | 0.02 |
| **Sub-total excluding the Project** | **12.08** |
| **The Project (worst-case)** | **0.13** |
| **Total** | **12.21** |

1430. Based on the Skomer, Skokholm and the Seas off Pembrokeshire/Sgomer, Sgogwm a Moroedd Penfro SPA lesser black-backed gull breeding population of 16,694 adult birds and a background mortality of 0.115 (1,920 birds per annum), an increase in mortality of 12.21 birds would increase background mortality by 0.64%. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that no detectable changes in mortality rates would occur on this population, and **it is therefore concluded that there would be no adverse effect on the integrity of Skomer, Skokholm and the Seas off Pembrokeshire / Sgomer, Sgogwm a Moroedd Penfro SPA, when assessed in-combination with other plans or projects**.

#### 8.32.3.5 Kittiwake

#### Status

1431. The Skomer, Skokholm and the Seas off Pembrokeshire/Sgomer, Sgogwm a Moroedd Penfro SPA breeding kittiwake population was cited as 1,959 pairs, or 3,918 breeding adults, in 1982 (Stroud *et al*., 2001). Furness (2015) gave a population of 1,045 pairs, or 2,090 breeding adults in 2013. The most recent count is 1,007 pairs (AON), or 2,014 breeding adults, in 2022 (JNCC, 2023a); this is used as the reference population for the assessment.

1432. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.146 (1 – 0.854; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 294 breeding adults.

## Functional linkage and seasonal apportionment of potential effects

1433. The mean maximum foraging range of kittiwake is 156.1km (±144.5km) and the maximum foraging range is 770km (Woodward *et al*., 2019). The Project is approximately 246km from Skomer, Skokholm and the Seas off Pembrokeshire/Sgomer, Sgogwm a Moroedd Penfro SPA, which means which means that the Project is beyond the mean maximum foraging range of kittiwakes breeding at this SPA, but within the mean maximum foraging range +1SD, and the maximum foraging range.

1434. The NatureScot apportioning tool (NatureScot, 2018) has been used to estimate the proportion of kittiwakes from each of the relevant SPAs present at the windfarm site during the breeding season. The apportioning to SPA and non-SPA colonies is set out in **Appendix 12.1** of the ES. 0.34% of impacts at the windfarm site during the breeding season are apportioned to Skomer, Skokholm and the Seas off Pembrokeshire/Sgomer, Sgogwm a Moroedd Penfro SPA.

1435. Outside the breeding season, breeding kittiwakes from the SPA are assumed to range widely and to mix with kittiwakes of all ages from breeding colonies in the UK and beyond. The relevant non-breeding season reference population is the UK Western Waters and Channel BDMPS, consisting of 911,586 individuals during autumn migration (September to December) and 627,816 during spring migration (January to February) (Furness, 2015).

1436. Furness (2015) estimated that 60% of the Skomer, Skokholm and the Seas off Pembrokeshire/Sgomer, Sgogwm a Moroedd Penfro SPA breeding adults are present within the UK Western Waters and Channel BDMPS during the autumn migration period, which is 1,254 birds. During the spring migration period 80% of the population is estimated to be present, which is 1,672 birds. This represents 0.14% and 0.27% of the BDMPS population for the autumn and spring periods respectively. During autumn migration and spring migration, 0.15%, and 0.27% of impacts are therefore considered to affect birds from the SPA (Furness, 2015).

## Potential effects on the qualifying feature from the Project-alone

1437. The kittiwake qualifying feature of the Skomer, Skokholm and the Seas off Pembrokeshire/Sgomer, Sgogwm a Moroedd Penfro SPA has been screened into the Appropriate Assessment due to the potential risk of collision.

### Operation and maintenance phase collision risk

1438. Information for collision risk on breeding adult kittiwakes belonging to the Skomer, Skokholm and the Seas off Pembrokeshire/Sgomer, Sgogwm a Moroedd Penfro SPA population is presented in **Table 8.85**. Collision estimates, calculated using the sCRM, are presented by biological season. A summary of the annual outputs and the corresponding increase in the annual baseline mortality rate is also presented. Parameters used in the sCRM were agreed with Natural England during the ETG process and are described in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES.

1439. Based on the mean collision rates, the annual total of breeding adult kittiwakes from Skomer, Skokholm and the Seas off Pembrokeshire/Sgomer, Sgogwm a Moroedd Penfro SPA at risk of collision as a result of the Project is less than one bird (0.07). This would increase the existing mortality of the SPA breeding population by 0.02%.

*Table 8.85 Predicted seasonal and annual collision mortality (Stochastic model Option 2, avoidance rate 0.993 (±0.0003)), for breeding adult kittiwakes at the windfarm site, apportioned to Skomer, Skokholm and the Seas off Pembrokeshire/Sgomer, Sgogwm a Moroedd Penfro SPA, with corresponding increases to baseline mortality of the population*

| | Breeding Season | Autumn Migration | Non-breeding/winter | Spring Migration | Annual |
|---|---|---|---|---|---|
| Period | Mar-Aug | Sep-Dec | - | Jan-Feb | Jan-Dec |
| Total collisions[1] (mean and 95% CIs) | 15.36 (4.17-33.87) | 8.49 (2.20-18.85) | - | 0.63 (0.00-1.46) | 24.48 (6.37-54.17) |
| % apportioned to the SPA | 0.34% | 0.14% | - | 0.27% | - |
| Total SPA collisions (mean and 95% CIs) | 0.05 (0.01-0.12) | 0.01 (0.00-0.03) | - | 0.00 (0.00-0.00) | 0.07 (0.02-0.14) |
| Mortality increase[2] (mean and 95% CIs) | 0.02% (0.01-0.04%) | 0.00% (0.00-0.01%) | - | 0.00% (0.00-0.00%) | 0.02% (0.01-0.05%) |

[1] Breeding season collision values reduced to 94.1% of modelled value to reflect proportion of adult birds recorded during site surveys
[2] Assuming predicted annual SPA mortality of 294 birds (2014 x 0.146)

1440. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that no detectable changes in mortality rates would occur on this population for the mean monthly collision estimates for the Project, or for the upper 95% CI collision estimate.

1441. **It is concluded that based on predicted kittiwake mortality due to collision at the windfarm site there is no potential for the Project to have an adverse effect on the integrity of the Skomer, Skokholm and the Seas off Pembrokeshire/Sgomer, Sgogwm a Moroedd Penfro SPA.**

1442. The confidence in the assessment is high. The evidence used to define the CRM input parameters presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** as part of the ES submitted alongside this RIAA is of high applicability and quality. Whilst there is uncertainty around some of the input parameters (e.g. avoidance rate), the rates selected are considered to be sufficiently precautionary based on expert opinion to provide confidence that collision rates are not underestimated.

#### Potential effects on the qualifying feature in-combination with other projects

1443. As the Project would have no measurable effect on kittiwake populations from the SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of Skomer, Skokholm and the Seas off Pembrokeshire/Sgomer, Sgogwm a Moroedd Penfro SPA, when assessed in-combination with other plans or projects**.

### 8.32.3.6 Guillemot

#### Status

1444. The Skomer, Skokholm and the Seas off Pembrokeshire/Sgomer, Sgogwm a Moroedd Penfro SPA breeding guillemot population was cited as 7,067 pairs, or 14,134 breeding adults, in 1982 (Furness, 2015; Stroud *et al*., 2016). Furness (2015) gave the breeding population of 16,300 pairs, or 32,600 breeding adults, in 2013. The most recent combined count from Skokholm (2021), Skomer and Midholm (2022) is 27,578 individuals (JNCC, 2023a); this is used as the reference population for the assessment.

1445. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.061 (1 – 0.939; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 1,682 breeding adults.

## Functional linkage and seasonal apportionment of potential effects

1446. The mean maximum foraging range of guillemot is 73.2km (±80.5km) and the maximum foraging range is 338km (Woodward *et al*., 2019). The Project is located approximately 246km from Skomer, Skokholm and the Seas off Pembrokeshire / Sgomer, Sgogwm a Moroedd Penfro SPA, which means that the Project is beyond the mean maximum foraging range +1SD of guillemots breeding at this SPA, but within the maximum foraging range. The maximum foraging range is a poor indicator of typical foraging behaviour. It would be expected that few birds or foraging trips will occur at this distance from the colony, and even fewer with any regularity. No impacts during the breeding season are therefore apportioned to birds breeding at this colony.

1447. Outside the breeding season, breeding guillemots from the SPA are assumed to range widely and to mix with guillemots of all ages from breeding colonies in the UK and beyond. The relevant non-breeding season reference population is the UK Western Waters BDMPS, consisting of 1,139,220 individuals (August to February) (Furness, 2015).

1448. Furness (2015) estimated that 90% of the Skomer and Skokholm breeding adults are present within the UK Western Waters BDMPS during the non-breeding season, which is 29,340 birds. This represents 2.6% of the BDMPS population for this period (1,139,220). It is therefore assumed that 2.6% of guillemots present at the Project site are breeding adults from Skomer, Skokholm and the Seas off Pembrokeshire/Sgomer, Sgogwm a Moroedd Penfro SPA.

## Potential effects on the qualifying feature

### Operation and maintenance phase disturbance / displacement / barrier effects

*Project-alone*

1449. The mean peak abundance of guillemots present within the windfarm site and 2km buffer during the non-breeding season was 8,315 (6,085-12,047) individuals (refer to **Appendix 12.1** of the ES). Of these, 75 birds (55-108) were likely to be breeding adults from the Skomer, Skokholm and the Seas off Pembrokeshire/Sgomer, Sgogwm a Moroedd Penfro SPA.

1450. **Table 8.86** sets out the predicted impacts on guillemots from Skomer, Skokholm and the Seas off Pembrokeshire/Sgomer, Sgogwm a Moroedd Penfro SPA during the non-breeding season. Displacement rates of 30% to 70% are considered for this species, along with a range of mortality rates of 1% to 10% of displaced birds (UK SNCBs, 2017).

1451. The available evidence suggested that the upper ranges of these displacement and mortality rates may be excessively precautionary. Whilst it is true that guillemots and razorbills tend to be displaced by OWFs they do not avoid them completely, and displacement rates vary between sites (MacArthur Green, 2019b). On average it was concluded that densities within OWFs were around half of the density found in the habitats around the OWF. At some OWFs there was also displacement of birds from a buffer zone surrounding it. The size of the buffer zone varied between OWFs and was generally less than 2km, with auk density increasing across the buffer zone as distance from turbines increased, up to the density in the wider area. Another recent review (APEM, 2022) found that the current displacement rates suggested by the SNCBs for guillemot and razorbill (30% to 70%) did not account for the quality of or confidence in the studies which were used to inform this position, and that studies where no significant effects were recorded were not accounted for during the provision of the advice. APEM (2022) suggested that in the case of Hornsea Project Four, an evidence-based displacement rate of 50% was appropriate. However, the study also recognised that larger displacement effects were possible.

1452. Mortality due to displacement could arise due to increased energy costs and/or decreased energy intake, if displacement results in increased flying time to avoid OWFs, and/or increased bird densities and competition for prey in areas of unimpacted habitat outside the OWF. Given that UK OWFs in the Irish and Celtic Seas represent a very small proportion of the available foraging habitat for guillemots and razorbills within UK Western Waters, the increase in density of auks outside OWFs due to displacement is likely to be negligible (MacArthur Green, 2019b). It is considered unlikely that the mortality rate due to displacement would be as high as the empirically estimated 6.1% annual mortality for adult guillemots that occurs due to the combination of 'natural' factors and anthropogenic activities (Horswill and Robinson, 2015). Indeed, it may be much lower; MacArthur Green (2019b) recommended precautionary rates of 50% displacement for guillemot and 1% mortality of displaced birds based on a review of available evidence. Modelling undertaken by APEM (2022) found that in the case of Hornsea Project Four, a mortality rate of 1% for displaced auks was considered to be precautionary, and that this may be an overestimate given that Hornsea Project Four is located towards the upper end of the mean maximum foraging range for guillemot from the closest designated sites. It is also noted that in the recent decision on Hornsea Project Four, the SoS determined that a rate of 70% displacement/2% mortality was appropriate to inform the assessment of effects on guillemots from Flamborough and Filey Coast SPA (DESNZ, 2023b).

1453. The full ranges of recommended displacement and mortality effects are considered by the assessment, along with evidence-based displacement and mortality rates of 50% and 1%, respectively (APEM, 2022; MacArthur Green, 2019b).

*Table 8.86 Guillemot – predicted operation and maintenance phase displacement and mortality from Skomer, Skokholm and the Seas off Pembrokeshire / Sgomer, Sgogwm a Moroedd Penfro SPA*

| Mean peak abundance estimate type | Mean peak abundance estimate | Number of SPA breeding adults present (non-breeding season)[1] | Annual mortality range[2] | Annual baseline mortality increase range[3] |
|---|---|---|---|---|
| Upper 95% CI | 12,047 | 313 | 1-22 | 0.06-1.30% |
| Mean | 8,315 | 216 | 1-15 | 0.04-0.90% |
| Lower 95% CI | 6,085 | 158 | 0-11 | 0.03-0.66% |
| [1] Assumes 2.6% of birds present during the non-breeding season are Skomer, Skokholm and the Seas off Pembrokeshire / Sgomer, Sgogwm a Moroedd Penfro SPA breeding adults<br>[2] Assumes displacement rates of 30-70% and mortality rates of 1-10%<br>[3] Background mortality rate of 6.1% (Horswill and Robinson, 2015) | | | | |

1454. Based on the mean peak abundances, the annual total of guillemots from the Skomer, Skokholm and the Seas off Pembrokeshire / Sgomer, Sgogwm a Moroedd Penfro SPA at risk of displacement is 216 birds (**Table 8.86**). At displacement rates of 30% to 70%, and mortality rates of 1% to 10% for displaced birds, 1 to 15 SPA breeding adults would be predicted to die each year due to displacement from the Project.

1455. Assuming a displacement rate of 70% and a mortality rate of 10% of displaced birds, annual mortality within the SPA breeding adult population would increase by 0.90%. Using an evidence-based displacement rate of 50%, and a mortality rate for displaced birds of 1%, annual mortality in the population would instead increase by 0.06% (1 bird).

1456. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that no detectable changes in mortality rates would occur under any combination of displacement and mortality rates when the mean peak abundance estimate assessments are considered.

1457. Increases of over 1% are predicted if the upper 95% CIs for mean peak abundances are used as inputs to the assessment alongside a displacement rate ≥60% and 10% mortality rate for displaced birds. The probability of this occurring is extremely small for two reasons. Firstly, the upper 95% CI for the mean peak abundances are highly unlikely to occur regularly at the windfarm site. Secondly, mortality rates for displaced birds of 10% are much higher than evidence suggests will actually be the case, and use of the evidence-based displacement (50%) and mortality rate (1%) would result in a mortality increase of significantly less than 1%.

1458. **It is concluded that predicted guillemot mortality due to operational phase displacement for the project alone would not adversely affect the integrity of the Skomer, Skokholm and the Seas off Pembrokeshire/Sgomer, Sgogwm a Moroedd Penfro SPA.**

1459. The confidence in the assessment is high for several reasons. Firstly, the evidence used to inform the evidence-based displacement rates is of high applicability and quality (based on the criteria discussed in **Chapter 12 Offshore Ornithology** of the ES). Whilst there is limited available evidence to inform mortality rates, 1% is considered to be sufficiently precautionary based on expert opinion. This species is not regarded as being highly specialised in its habitat requirements (Bradbury *et al*., 2014; Furness and Wade, 2012; Garthe and Hüppop, 2004), and it is therefore anticipated that displaced birds will find alternative habitat in the vast majority of cases. Finally, the conclusion of the assessment is the same irrespective of whether the mean or 95% upper CI mean peak abundances are used to calculate potential mortality and increases in the baseline mortality rate of the background population, provided the evidence-based displacement and mortality rates are used.

*In-combination*

1460. The in-combination assessment for guillemots from Skomer, Skokholm and the Seas off Pembrokeshire/Sgomer, Sgogwm a Moroedd Penfro SPA has been undertaken in accordance with the approach presented in **Section 8.1**. The total population apportioned to Skomer, Skokholm and the Seas off Pembrokeshire/Sgomer, Sgogwm a Moroedd Penfro SPA at risk of displacement is estimated to be 4,280 breeding adults (**Appendix 12.1** of the ES). Annual in-combination displacement and mortality rates for birds from Skomer, Skokholm and the Seas off Pembrokeshire/Sgomer, Sgogwm a Moroedd Penfro SPA are presented in **Table 8.87**.

1461. Assuming a displacement rate of 70% and a mortality rate of 10% of displaced birds, 300 breeding adult SPA birds would be lost to displacement annually. Including additional apportioned mortality from the Morlais and Holyhead tidal projects (total 0.82 birds), this would increase the existing mortality within the SPA population (1,682 breeding adult birds per year) by 17.86%. Using an evidence-based displacement rate of 50%, and a mortality rate for displaced birds of 1%, the annual in-combination mortality would be 21 birds. This would increase the existing mortality within this population by 1.32%. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. Although marginally above this 1% threshold, it is considered very unlikely that this would actually have a measurable effect on the SPA population. This is because of the small number of potentially impacted birds due to displacement, and the recognition that, as guillemot is a dispersive rather than a fully migratory species, birds do not travel great distances from the breeding colony during the non-breeding season (MS-LOT, 2022), and therefore apportioning using the BDMPS is likely to significantly overestimate the numbers of birds from the SPA present at the Project site.

1462. **It is concluded that predicted guillemot mortality due to of operational phase displacement impacts of the Project in-combination with other projects, would not adversely affect the integrity of Skomer, Skokholm and the Seas off Pembrokeshire/Sgomer, Sgogwm a Moroedd Penfro SPA.**

*Table 8.87 In-combination year-round displacement matrix for guillemot from Skomer, Skokholm and the Seas off Pembrokeshire/Sgomer, Sgogwm a Moroedd Penfro SPA*

| Annual | Mortality | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| **Displacement** | **1%** | **2%** | **3%** | **4%** | **5%** | **10%** | **20%** | **30%** | **50%** | **80%** | **100%** |
| **10%** | 4 | 9 | 13 | 17 | 21 | 43 | 86 | 128 | 214 | 342 | 428 |
| **20%** | 9 | 17 | 26 | 34 | 43 | 86 | 171 | 257 | 428 | 685 | 856 |
| **30%** | 13 | 26 | 39 | 51 | 64 | 128 | 257 | 385 | 642 | 1027 | 1284 |
| **40%** | 17 | 34 | 51 | 68 | 86 | 171 | 342 | 514 | 856 | 1370 | 1712 |
| **50%** | 21 | 43 | 64 | 86 | 107 | 214 | 428 | 642 | 1070 | 1712 | 2140 |
| **60%** | 26 | 51 | 77 | 103 | 128 | 257 | 514 | 770 | 1284 | 2055 | 2568 |
| **70%** | 30 | 60 | 90 | 120 | 150 | 300 | 599 | 899 | 1498 | 2397 | 2996 |
| **80%** | 34 | 68 | 103 | 137 | 171 | 342 | 685 | 1027 | 1712 | 2739 | 3424 |
| **90%** | 39 | 77 | 116 | 154 | 193 | 385 | 770 | 1156 | 1926 | 3082 | 3852 |
| **100%** | 43 | 86 | 128 | 171 | 214 | 428 | 856 | 1284 | 2140 | 3424 | 4280 |

Note: The cells show the number of birds predicted to die (rounded to the nearest integer) at a given rate of displacement and mortality. Highlighted cells are considered to be the most realistic scenario, in accordance with SNCB advice (SNCBs, 2022).

### 8.32.3.7 Razorbill

#### Status

1463. The Skomer, Skokholm and the Seas off Pembrokeshire/Sgomer, Sgogwm a Moroedd Penfro SPA breeding razorbill population was cited as 2,854 pairs, or 5,708 breeding adults, in 1997 (Furness, 2015; Stroud *et al*., 2016). Furness (2015) gave the breeding population of 6,001 pairs, or 12,002 breeding adults, in 2013. The most recent combined count from Skokholm (2021), Skomer and Midholm (2022) is 9,545 individuals (JNCC, 2023a); this is used as the reference population for the assessment.

1464. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.105 (1 – 0.895; Horswill and Robinson, 2015), the expected annual mortality from the SPA population would be 1,002 breeding adults.

#### Functional linkage and seasonal apportionment of potential effects

1465. The mean maximum foraging range of razorbill is 88.7km (±75.9km) and the maximum foraging range is 313km (Woodward *et al*., 2019). The Project is located approximately 246km from Skomer, Skokholm and the Seas off Pembrokeshire/Sgomer, Sgogwm a Moroedd Penfro SPA, which means which means that the Project is beyond the mean maximum foraging range +1SD of razorbills breeding at this SPA, but within the maximum foraging range.

1466. The maximum foraging range is a poor indicator of typical foraging behaviour. It would be expected that few birds or foraging trips will occur at this distance from the colony, and even fewer with any regularity. No impacts during the breeding season are therefore apportioned to birds breeding at this colony.

1467. Outside the breeding season, breeding razorbills from the SPA are assumed to range widely and to mix with razorbills of all ages from breeding colonies in the UK and beyond. The relevant non-breeding season reference population is the UK Western Waters BDMPS, consisting of 606,914 individuals during autumn and spring passage periods (August to October and January to March), and 341,422 individuals during winter (November and December) (Furness, 2015). During autumn and spring migration, 98% of the SPA breeding adults (11,762 individuals based on the 2013 population estimate) are assumed to be present in the BDMPS, representing 1.9% of the BDMPS population (606,914 individuals of all ages). During the winter season, 30% of the SPA breeding adults (3,601 individuals based on the 2013 population estimate) are assumed to be present in the BDMPS, representing 1.1% of the BDMPS population (341,422 individuals of all ages). These percentages (i.e. 1.9% and 1.1%) are the proportions of birds present at the windfarm site that are presumed to originate from the Skomer, Skokholm and the Seas off Pembrokeshire/Sgomer, Sgogwm a Moroedd Penfro SPA during the relevant seasons.

## Potential effects on the qualifying feature

### Operation and maintenance phase disturbance / displacement / barrier effects

*Project-alone*

1468. The year-round mean peak abundance of razorbills present within the windfarm site and 2km buffer was 1,979 (703-3,552) individuals (refer to **Appendix 12.1** of the ES). Of these, 28 birds (12-46) were likely to be breeding adults from the Skomer, Skokholm and the Seas off Pembrokeshire/ Sgomer, Sgogwm a Moroedd Penfro SPA.

1469. **Table 8.88** sets out the predicted impacts on razorbills from Skomer, Skokholm and the Seas off Pembrokeshire/Sgomer, Sgogwm a Moroedd Penfro SPA. Displacement rates of 30% to 70% are considered for this species, along with a range of mortality rates of 1% to 10% of displaced birds (UK SNCBs, 2017).

1470. The available evidence suggested that the upper ranges of these displacement and mortality rates may be excessively precautionary. Whilst it is true that guillemots and razorbills tend to be displaced by OWFs they do not avoid them completely, and displacement rates have been noted to vary between sites (MacArthur Green, 2019b). On average it was concluded that densities within OWFs were around half of the density found in the habitats around the OWF. At some OWFs there was also displacement of birds from a buffer zone surrounding it. The size of the buffer zone varied between OWFs and was generally less than 2km, with auk density increasing across the buffer zone as distance from turbines increased, up to the density in the wider area. Another recent review (APEM, 2022) found that the displacement rates suggested by the SNCBs for guillemot and razorbill (30% to 70%) did not account for the quality of or confidence in the studies which were used to inform this position, and that studies where no significant effects were recorded were not accounted for during the provision of the advice. APEM (2022) suggested that in the case of Hornsea Project Four, an evidence-based displacement rate of 50% was appropriate. However, the study also recognised that larger displacement effects were possible.

1471. Mortality due to displacement could arise due to increased energy costs and/or decreased energy intake, if displacement results in increased flying time to avoid OWFs, and/or increased bird densities and competition for prey in areas of unimpacted habitat outside the OWF. Given that UK OWFs in the Irish and Celtic Seas represent a very small proportion of the available foraging habitat for guillemots and razorbills within UK Western Waters, the increase in density of auks outside OWFs due to displacement is likely to be negligible (MacArthur Green, 2019b). It is considered unlikely that the mortality rate due to displacement would be as high as the empirically estimated 10.5% annual mortality for adult razorbills that occurs due to the combination of 'natural' factors and anthropogenic activities (Horswill and Robinson, 2015). Indeed, it may be much lower; MacArthur Green (2019b) recommended precautionary rates of 50% displacement for auks and 1% mortality of displaced birds based on a review of available evidence. Modelling undertaken by APEM (2022) found that in the case of Hornsea Project Four, a mortality rate of 1% for displaced auks was considered to be precautionary, and that this may be an overestimate given that Hornsea Project Four is located towards the upper end of the mean maximum foraging range for guillemot from the closest designated sites.

1472. The full ranges of recommended displacement and mortality effects are considered by the assessment, along with evidence-based displacement and mortality rates of 50% and 1%, respectively (APEM, 2022; MacArthur Green, 2019b).

*Table 8.88 Razorbill – predicted operation and maintenance phase displacement and mortality from Skomer, Skokholm and the Seas off Pembrokeshire / Sgomer, Sgogwm a Moroedd Penfro SPA*

| **Mean peak abundance estimate type** | **Mean peak abundance estimate by season[1]** | **Number of SPA breeding adults present by season)[2]** | **Annual mortality range[3]** | **Annual baseline mortality increase range[4]** |
|---|---|---|---|---|
| Upper 95% CI | 605 (b)<br>1,070 (aut)<br>1,297 (win)<br>580 (spr)<br>3,552 (year round) | 0 (b)<br>20 (aut)<br>14 (win)<br>11 (spr)<br>46 (year round) | 0-3 (0) | 0.01-0.32% (0.02%) |
| Mean | 252 (b)<br>694 (aut)<br>651 (win)<br>381 (spr)<br>1,979 (year round) | 0 (b)<br>13 (aut)<br>7 (win)<br>7 (spr)<br>28 (year round) | 0-2 (0) | 0.01-0.19% (0.01%) |
| Lower 95% CI | 21 (b)<br>309 (aut)<br>159 (win)<br>214 (spr)<br>703 (year round) | 0 (b)<br>6 (aut)<br>2 (win)<br>4 (spr)<br>12 (year round) | 0-1 (0) | 0.00-0.08% (0.01%) |

| Mean peak abundance estimate type | Mean peak abundance estimate by season[1] | Number of SPA breeding adults present by season)[2] | Annual mortality range[3] | Annual baseline mortality increase range[4] |
|---|---|---|---|---|
| [1] Breeding season = b, autumn migration season = aut, winter season = win, spring migration season = spr<br>[2] Assumes breeding adult apportioning of 0.0% (breeding season), 1.9% (spring and autumn migration) and 1.1% (winter) to Skomer, Skokholm and the Seas off Pembrokeshire / Sgomer, Sgogwm a Moroedd Penfro SPA.<br>[3] Assumes displacement rates of 30-70% and mortality rates of 1-10%. Evidence-based estimates assuming a 50% displacement rate and 1% mortality of displaced birds are presented in parentheses.<br>[4] Background mortality rate of 10.5% (Horswill and Robinson, 2015) | | | | |

1473. Based on the mean peak abundances, the annual total of razorbills from the Skomer, Skokholm and the Seas off Pembrokeshire / Sgomer, Sgogwm a Moroedd Penfro SPA at risk of displacement is 28 birds (**Table 8.88**). At displacement rates of 30% to 70%, and mortality rates of 1% to 10% for displaced birds, 0 to 2 SPA breeding adults would be predicted to die each year due to displacement from the Project.

1474. Assuming a displacement rate of 70% and a mortality rate of 10% of displaced birds, annual mortality within the SPA breeding adult population would increase by 0.32%. Using an evidence-based displacement rate of 50%, and a mortality rate for displaced birds of 1%, annual mortality in the population would instead increase by 0.02% (<1 bird).

1475. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that no detectable changes in mortality rates would occur under almost any combination of displacement and mortality rates when the mean peak abundance estimate assessments are considered. This would be the case even when upper 95% CIs are considered.

1476. **It is concluded that predicted razorbill mortality due to operational phase displacement for the project alone would not adversely affect the integrity of the Skomer, Skokholm and the Seas off Pembrokeshire / Sgomer, Sgogwm a Moroedd Penfro SPA.**

1477. The confidence in the assessment is high for several reasons. Firstly, the evidence used to inform the evidence-based displacement rates is of high applicability and quality (based on the criteria discussed in **Chapter 12 Offshore Ornithology** of the ES). Whilst there is limited available evidence to inform mortality rates, 1% is considered to be sufficiently precautionary based on expert opinion. This species is not regarded as being highly specialised in its habitat requirements (Bradbury *et al*., 2014; Furness and Wade, 2012;

Garthe and Hüppop, 2004), and it is therefore anticipated that displaced birds will find alternative habitat in the vast majority of cases. Finally, the conclusion of the assessment is the same irrespective of whether the mean or 95% upper CI mean peak abundances are used to calculate potential mortality and increases in the baseline mortality rate of the background population, provided the evidence-based displacement and mortality rates are used.

*In-combination*

1478. The in-combination assessment for razorbills from Skomer, Skokholm and the Seas off Pembrokeshire / Sgomer, Sgogwm a Moroedd Penfro SPA has been undertaken in accordance with the approach presented in **Section 8.1**. The total population apportioned to the SPA at risk of displacement is estimated to be 602 breeding adults (**Appendix 12.1** of the ES). Annual in-combination displacement and mortality rates for birds from Skomer, Skokholm and the Seas off Pembrokeshire / Sgomer, Sgogwm a Moroedd Penfro SPA are presented in **Table 8.89**.

1479. Assuming a displacement rate of 70% and a mortality rate of 10% of displaced birds, 42 breeding adult SPA birds would be lost to displacement annually. Including additional apportioned mortality from the Morlais and Holyhead tidal projects (total 0.27 birds), this would increase the existing mortality within the SPA population (1,002 breeding adult birds per year) by 4.23%. Using an evidence-based displacement rate of 50%, and a mortality rate for displaced birds of 1%, the annual in-combination mortality would be 3 birds. This would increase the existing mortality within this population by 0.33%. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that detectable changes in mortality rates will not occur due to the level or mortality predicted if the more realistic rates for mortality are used.

1480. **It is concluded that predicted razorbill mortality due to of operational phase displacement impacts of the Project in-combination with other projects, would not adversely affect the integrity of Skomer, Skokholm and the Seas off Pembrokeshire / Sgomer, Sgogwm a Moroedd Penfro SPA.**

*Table 8.89 In-combination year-round displacement matrix for razorbill from Skomer, Skokholm and the Seas off Pembrokeshire / Sgomer, Sgogwm a Moroedd Penfro SPA*

| Annual | Mortality | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Displacement | 1% | 2% | 3% | 4% | 5% | 10% | 20% | 30% | 50% | 80% | 100% |
| 10% | 1 | 1 | 2 | 2 | 3 | 6 | 12 | 18 | 30 | 48 | 60 |
| 20% | 1 | 2 | 4 | 5 | 6 | 12 | 24 | 36 | 60 | 96 | 120 |
| 30% | 2 | 4 | 5 | 7 | 9 | 18 | 36 | 54 | 90 | 145 | 181 |
| 40% | 2 | 5 | 7 | 10 | 12 | 24 | 48 | 72 | 120 | 193 | 241 |
| 50% | 3 | 6 | 9 | 12 | 15 | 30 | 60 | 90 | 151 | 241 | 301 |
| 60% | 4 | 7 | 11 | 14 | 18 | 36 | 72 | 108 | 181 | 289 | 361 |
| 70% | 4 | 8 | 13 | 17 | 21 | 42 | 84 | 126 | 211 | 337 | 422 |
| 80% | 5 | 10 | 14 | 19 | 24 | 48 | 96 | 145 | 241 | 385 | 482 |
| 90% | 5 | 11 | 16 | 22 | 27 | 54 | 108 | 163 | 271 | 434 | 542 |
| 100% | 6 | 12 | 18 | 24 | 30 | 60 | 120 | 181 | 301 | 482 | 602 |

Note: The cells show the number of birds predicted to die (rounded to the nearest integer) at a given rate of displacement and mortality. Highlighted cells are considered to be the most realistic scenario, in accordance with SNCB advice (SNCBs, 2022).

# 8.33 Grassholm SPA

1481. Grassholm SPA is located approximately 256m from the windfarm site.

## 8.33.1 Description of designation

1482. Grassholm Island is situated 10 miles off the Pembrokeshire coast. Grassholm SPA is the only colony of gannets in Wales and is the third largest gannetry in Britain and Ireland. It holds 8.6% of the NE Atlantic population and supports approximately 7% of the world population (Murray, 2015). Grassholm SPA was first classified in 1986. In 2014 the site was extended to include adjacent sea areas that are used by birds from within the existing SPA for behaviours that are directly linked to their use of the breeding site.

## 8.33.2 Conservation objectives

1483. The overarching conservation objectives for the qualifying feature of the SPA (gannet) are:

- The population will not fall below 30,000 pairs in three consecutive years,
- It will not drop by more than 25% of the previous year's figures in any one year

1484. There will be no decline in this population significantly greater than any decline in the North Atlantic population as a whole.

## 8.33.3 Assessment

### 8.33.3.1 Gannet

#### Status

1485. Gannet is listed as a qualifying species of this SPA.

1486. The SPA population at classification was cited as 33,000 pairs in 1994/5 (Furness, 2015, Stroud *et al*., 2016). Furness (2015) gave the breeding population of 39,292 pairs, or 78,584 adults in 2009. The most recent count (2015) is 36,011 AOS, or 72,022 breeding adults (JNCC, 2022).

1487. Based on the most recent SPA population of breeding adults, and an annual breeding adult baseline mortality rate of 0.081 (1-0.919, Horswill and Robinson (2015)), 5,834 breeding adults from the SPA population would be expected to die each year.

**Functional linkage and seasonal apportionment of potential effects**

1488. The windfarm site is 239km from Grassholm SPA. The mean maximum foraging range of gannet is 315.2km (±194.2km). The windfarm site is therefore within the mean maximum foraging range of gannets from the Grassholm SPA.

1489. Modelled at-sea utilisation distributions of breeding adult birds during the breeding season have been published, based on GPS tracking data (Wakefield *et al.*, 2013). These suggest that the windfarm site is located outside of the core foraging range for breeding adult birds from Grassholm SPA.

1490. One further UK SPA designated for gannet is located within the UK Western Waters BDMPS area; Ailsa Craig SPA (**Section 8.25.2.1**). This site is located approximately 177km from the windfarm site which is within the mean maximum foraging range of this species. Data presented by Wakefield *et al.,* (2013) indicated that the foraging ranges of gannets from different breeding colonies tend not to overlap, and that the windfarm site is located on the edge of the core foraging area for adult birds from Ailsa Craig SPA, but outside of the foraging area for Grassholm SPA. One transboundary site is also located within the mean maximum foraging range +1SD; Saltee Islands SPA (**Section 8.69.3.3**), which is located approximately 265km from the windfarm site. As with Grassholm SPA, data presented in Wakefield *et al.,* (2013) indicated that birds from Saltee Islands SPA are unlikely to occur at the windfarm site during the breeding season.

1491. Two further UK sites are within the straight-line foraging distance of the windfarm site; Flamborough and Filey Coast SPA (212km; **Section 8.27.3.2**) and Forth Islands SPA (239km; **Section 8.31.3.1**). However, both sites are on the eastern UK coast, with an across-sea distance of >1,000km, and, as this species will not typically cross land, are therefore considered geographically isolated from the windfarm site during the breeding season.

1492. On the basis of the data presented by Wakefield *et al.,* (2013), it assumed that breeding adult gannets that were recorded at the windfarm site during the full breeding season (March to September (Furness, 2015)) originated from the Ailsa Craig SPA. Accordingly, no birds present during this period are considered to originate from Grassholm SPA.

1493. Outside the breeding season, adult gannets, including those from the Grassholm SPA, are not constrained by requirements to visit nests to incubate eggs or provision chicks. At this time, they are assumed to range more widely and to mix with gannets of all age classes from breeding colonies in the UK and further afield. The background population during these seasons is the UK Western Waters BDMPS. This consists of 545,954 individuals during autumn migration (September to November), and 661,888 individuals during spring migration (December to March) (Furness, 2015).

1494. Estimates of the proportion of gannets present at the windfarm site which originate from the Grassholm SPA during the non-breeding season (and therefore the proportion of predicted mortalities from the SPA population) are based on the SPA population (i.e. 78,584 breeding adults) as a proportion of the UK Western Waters BDMPS during the relevant season. During autumn migration and spring migration, 14.4%, and 11.9% of impacts are considered to affect birds from the SPA respectively (Furness, 2015).

### Potential effects on the qualifying feature

1495. The gannet qualifying feature of the Grassholm SPA has been screened into the assessment due to the potential risk of collision and operational phase displacement/barrier effects during the operation and maintenance phase of the Project.

#### Operation and maintenance phase displacement/barrier effects

1496. Displacement effects for gannet for the Project were assessed during the autumn and spring migration periods, based on an unapportioned peak mean population of 124 and eight birds respectively, calculated for the windfarm site and a 2km buffer, in line with recommendations within the SNCB guidance (SNCB 2017). As set out above, no gannets present at the windfarm site have been apportioned to Grassholm SPA during the breeding season. The displacement matrices used to calculate potential impacts are presented in **Appendix 12.1** of the ES and summarised in **Table 8.90**. The inclusion of all birds within the 2km buffer, to determine the total number of birds subject to displacement, is precautionary, as in reality the avoidance rate is likely to fall with distance from the windfarm site.

1497. A displacement rate of 60-80% and mortality rate of 1% has been presented. A maximum 1% mortality value has been selected firstly because gannet is known to possess high habitat flexibility (Furness and Wade, 2012). This suggests that displaced birds will readily find alternative habitats including foraging areas. Secondly, no evidence of displacement-induced mortality has been identified, which means there is limited justification for setting predicted mortality rates at a higher level. Given the extensive foraging range of this species (Woodward *et al.*, 2019), there may be no mortality costs to displacement from the relatively very small footprints of OWFs.

*Table 8.90 Gannet – predicted operation and maintenance phase displacement and mortality from Grassholm SPA*

| Mean peak abundance estimate type | Mean peak abundance estimate | Number of SPA breeding adults present by season[1] | Annual mortality range[2] | Annual baseline mortality increase range[3] |
|---|---|---|---|---|
| Upper 95% CI | 809 (breeding)<br>189 (autumn)<br>16 (spring)<br>1,014 (year round) | 0 (breeding)<br>27 (autumn)<br>2 (spring)<br>29 (year round) | 0-0 | 0.00-0.00% |
| Mean | 541 (breeding)<br>124 (autumn)<br>8 (spring)<br>673 (year round) | 0 (breeding)<br>18 (autumn)<br>1 (spring)<br>19 (year round) | 0-0 | 0.00-0.00% |
| Lower 95% CI | 160 (breeding)<br>0 (autumn)<br>0 (spring)<br>160 (year round) | 0 (breeding)<br>0 (autumn)<br>0 (spring)<br>0 (year round) | 0-0 | 0.00-0.00% |

[1] 14.4% and 11.9% of birds are assumed to be breeding adults from the SPA population during the autumn and spring migration periods respectively.

[2] Assumes displacement rates of 60-80% and mortality rate of 1%

[3] Background population is Grassholm SPA breeding adults (72,022 individuals), adult age class annual mortality rate of 8.1% (Horswill and Robinson, 2015)

1498. Using the maximum potential mortality value, there would be no measurable increase in gannet mortality. Accordingly, no significant effects on gannet are predicted during the operation and maintenance phase, and **it is concluded that there would be no potential for the Project to have an adverse effect on the integrity of Grassholm SPA.**

1499. The confidence in the assessment is high for several reasons. Firstly, the evidence used to set the displacement rates presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES is of high applicability and quality. Whilst there is limited available evidence to inform mortality rates, 1% is considered to be sufficiently precautionary based on expert opinion. Finally, the conclusion of the assessment is the same irrespective of whether the mean or 95% upper CI mean peak abundances are used to calculate potential mortality and increases in the baseline mortality rate of the background population.

**Operation and maintenance phase collision risk**

1500. Information to support the Appropriate Assessment for collision risk on breeding adult gannets belonging to the Grassholm SPA population is presented in **Table 8.91**. Collision estimates, calculated using the sCRM, are presented by biological season. A summary of the annual outputs and the corresponding increase in the annual baseline mortality rate is also presented. Parameters used in the sCRM were agreed with Natural England during the ETG process and are described in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES. In accordance with Natural England advice, a 70% macro-avoidance correction was applied to gannet abundance data used in the sCRM.

1501. Based on the mean collision rates, no breeding adult gannets from Grassholm SPA are considered at risk of collision as a result of the Project. Therefore, there would be no measurable increase in the existing mortality of the SPA breeding population.

*Table 8.91 Predicted seasonal and annual collision mortality (Stochastic model Option 2, avoidance rate 0.993 (±0.0003), plus 70% macro-avoidance) for breeding adult gannets at the windfarm site, apportioned to Grassholm SPA, with corresponding increases to baseline mortality of the population*

| Period | Breeding Season Apr-Aug | Autumn Migration Sep-Nov | Non-breeding/winter - | Spring Migration Dec-Mar | Annual Jan-Dec |
|---|---|---|---|---|---|
| Total collisions[1] (mean and 95% CIs) | 0.83 (0.00-3.35) | 0.14 (0.00-0.74) | - | 0.00 | 0.97 (0.00-4.10) |
| % apportioned to the SPA | 0.0% | 14.4% | - | 11.9% | - |
| Total SPA collisions (mean and 95% CIs) | 0.00 (0.00-0.00) | 0.02 (0.00-0.11) | - | 0.00 (0.00-0.00) | 0.02 (0.00-0.11) |
| Mortality increase[2] (mean and 95% CIs) | 0.00% (0.00-0.00%) | 0.00% | - | 0.00% | 0.00% (0.00-0.00%) |

[1] Breeding season collision values reduced to 73.8% of modelled value to reflect proportion of adult birds recorded during site surveys

[2] Assuming predicted annual SPA mortality of 5,383 birds (66,452 x 0.081)

1502. Accordingly, no significant effects on gannet are predicted during the operation and maintenance phase, and **it is concluded that there would be no potential for the Project to have an adverse effect on the integrity of Grassholm SPA.** Comments received from RSPB during the ETG process, indicating that they do not accept the 70% macro-avoidance rate for collision risk recommended by Natural England, are noted. However, even in the absence of this correction factor, the net increase in mortality would be unchanged (i.e. zero).

1503. The confidence in the assessment is high. The evidence used to define the CRM input parameters presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES submitted alongside this RIAA is of high applicability and quality. Whilst there is uncertainty around some of the input parameters (e.g. avoidance rate), the rates selected are considered to be sufficiently precautionary based on expert opinion to provide confidence that collision rates are not underestimated.

**Combined displacement/barrier effects and collision risk**

1504. As no measurable increase in mortality is predicted for both displacement and collision risk, the mean combined displacement and collision rates for breeding adult gannet from the Grasshom SPA would be zero. Therefore, there would be no net increase in existing mortality rates.

1505. It is concluded that based on predicted gannet mortality due to the combined effects of operational phase displacement and collision there is no potential for the Project-alone or in-combination to have an adverse effect on the integrity of the Grassholm SPA.

1506. The confidence in the assessment is high, for the reasons provided in the individual displacement and collision assessments.

### Potential effects in-combination with other projects

1507. As no measurable effects of displacement/barrier and collision on gannet are predicted as a result of the Project-alone, there would be no contribution to other plans or projects in-combination. **It is therefore concluded that there would be no potential for the Project-alone or in-combination to have an adverse effect on the integrity of Grassholm SPA.**

## 8.34 North Colonsay and Western Cliffs SPA

1508. North Colonsay and Western Cliffs SPA is located approximately 293km from the windfarm site (straight-line distance) and approximately 305km across sea.

### 8.34.1 Description of designation

1509. North Colonsay and Western Cliffs SPA covers an area of rocky coast, cliffs, and maritime heath on the island of Colonsay in Argyll, Scotland. It supports the northernmost stable population of chough in Europe, and is particularly significant to the maintenance of the breeding range of the chough in Britain and the EC. The SPA overlaps the boundaries of the North Colonsay SSSI and the West Colonsay Seabird Cliffs SSSI, and the seaward elements extend approximately 1km into the marine environment to include the seabed, water column and surface.

### 8.34.2 Conservation objectives

1510. The overarching conservation objectives for the site are:

- To avoid deterioration of the habitats of the qualifying species or significant disturbance to the qualifying species, thus ensuring that the integrity of the site is maintained
- To ensure for the qualifying species that the following are maintained in the long term:
  - Population of the species as a viable component of the site
  - Distribution of the species within site
  - Distribution and extent of habitats supporting the species
  - Structure, function and supporting processes of habitats supporting the species
  - No significant disturbance of the species

### 8.34.3 Assessment

1511. The qualifying features of North Colonsay and Western Cliffs SPA screened into the Appropriate Assessment are listed in **Table 5.2**. These are breeding guillemot and breeding kittiwake. These species are also screened in as named components of the seabird assemblage.

#### 8.34.3.1 Kittiwake

##### Status

1512. The North Colonsay and Western Cliffs SPA breeding kittiwake population was cited as 4,512 pairs, or 9,024 breeding adults, in 1997 (Furness, 2015; Stroud *et al*., 2016). Furness (2015) gave a breeding population of 5,563 breeding pairs, or 11,126 breeding adults, in 2000. The most recent complete count is 2,926 breeding pairs (AON), or 5,852 breeding adults, plus a further 454 individuals in 2016 (JNCC, 2023a); giving a total of 6,306 assumed breeding adults; this is used as the reference population for the assessment.

1513. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.146 (1 – 0.854; Horswill and Robinson, 2015), the expected annual mortality from the SPA population would be 921 breeding adults.

##### Functional linkage and seasonal apportionment of potential effects

1514. The mean maximum foraging range of kittiwake is 156.1km (±144.5km) and the maximum foraging range is 770km (Woodward *et al*., 2019). The Project is located approximately 305km from North Colonsay and Western Cliffs SPA, which means the Project is beyond the mean maximum +1SD foraging range of kittiwakes breeding at this SPA, but within the maximum foraging range.

1515. The maximum foraging range is a poor indicator of typical foraging behaviour. It would be expected that few birds or foraging trips will occur at this distance from the colony, and even fewer with any regularity. No impacts during the breeding season are therefore apportioned to birds breeding at this colony.

1516. Outside the breeding season, breeding kittiwakes from the SPA are assumed to range widely and to mix with kittiwakes of all ages from breeding colonies in the UK and beyond. The relevant non-breeding season reference population is the UK Western Waters and Channel BDMPS, consisting of 911,586 individuals during autumn migration (September to December) and 627,816 during spring migration (January to February) (Furness, 2015).

1517. Furness (2015) estimated that 60% of the North Colonsay SPA breeding adults are present within the UK Western Waters and Channel BDMPS during the autumn migration period, which is 6,676 birds. During the spring migration period 80% of the population is estimated to be present, which is 8,901 birds. This represents 0.73% and 1.42% of the BDMPS population for the autumn and spring periods respectively. During autumn migration and spring migration, 0.73%, and 1.42% of impacts are therefore considered to affect birds from the SPA (Furness, 2015).

## Potential effects on the qualifying feature from the Project-alone

1518. The kittiwake qualifying feature of the North Colonsay and Western Cliffs SPA has been screened into the Appropriate Assessment due to the potential risk of collision.

### Operation and maintenance phase collision risk

1519. Information for collision risk on breeding adult kittiwakes belonging to the North Colonsay and Western Cliffs SPA population is presented in **Table 8.92**. Collision estimates, calculated using the sCRM, are presented by biological season. A summary of the annual outputs and the corresponding increase in the annual baseline mortality rate is also presented. Parameters used in the sCRM were agreed with Natural England during the ETG process and are described in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES.

1520. Based on the mean collision rates, the annual total of breeding adult kittiwakes from North Colonsay and Western Cliffs SPA at risk of collision as a result of the Project is less than one bird (0.07). This would increase the existing mortality of the SPA breeding population by 0.01%.

*Table 8.92 Predicted seasonal and annual collision mortality (Stochastic model Option 2, avoidance rate 0.993 (±0.0003)), for breeding adult kittiwakes at the windfarm site, apportioned to North Colonsay and Western Cliffs SPA, with corresponding increases to baseline mortality of the population*

| | Breeding Season | Autumn Migration | Non-breeding/winter | Spring Migration | Annual |
|---|---|---|---|---|---|
| **Period** | **Mar-Aug** | **Sep-Dec** | **-** | **Jan-Feb** | **Jan-Dec** |
| Total collisions[1] (mean and 95% CIs) | 15.36<br>(4.17-33.87) | 8.49<br>(2.20-18.85) | - | 0.63<br>(0.00-1.46) | 24.48<br>(6.37-54.17) |
| % apportioned to the SPA | 0.00% | 0.73% | - | 1.42% | - |
| Total SPA collisions (mean and 95% CIs) | 0.00<br>(0.00-0.00) | 0.06<br>(0.02-0.14) | - | 0.01<br>(0.00-0.02) | 0.07<br>(0.02-0.16) |
| Mortality increase[2] (mean and 95% CIs) | 0.00%<br>(0.00-0.00%) | 0.01%<br>(0.00-0.01%) | - | 0.00%<br>(0.00-0.00%) | 0.01%<br>(0.00-0.02%) |

[1] Breeding season collision values reduced to 94.1% of modelled value to reflect proportion of adult birds recorded during site surveys

[2] Assuming predicted annual SPA mortality of 921 birds (980 x 0.146)

1521. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that no detectable changes in mortality rates would occur on this population for the mean monthly collision estimates for the Project, or for the upper 95% CI collision estimate.

1522. **It is concluded that based on predicted kittiwake mortality due to collision at the windfarm site there is no potential for the Project to have an adverse effect on the integrity of the North Colonsay and Western Cliffs SPA.**

1523. The confidence in the assessment is high. The evidence used to define the CRM input parameters presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** as part of the ES submitted alongside this RIAA is of high applicability and quality. Whilst there is uncertainty around some of the input parameters (e.g. avoidance rate), the rates selected are considered to be sufficiently precautionary based on expert opinion to provide confidence that collision rates are not underestimated.

### Potential effects on the qualifying feature in-combination with other projects

1524. As the Project would have no measurable effect on kittiwake populations from the SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of North Colonsay and Western Cliffs SPA, when assessed in-combination with other plans or projects**.

### 8.34.3.2 Guillemot

### Status

1525. The North Colonsay and Western Cliffs SPA breeding guillemot population is cited as 6,656 pairs, or 13,312 breeding adults, in 1997 (Furness, 2015; Stroud *et al*., 2016). Furness (2015) gave a breeding population of 13,500 pairs, or 27,000 breeding adults, in 2000. The most recent complete count is 18,739 individuals in 2018 (JNCC, 2023a); this is used as the reference population for the assessment.

1526. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.061 (1 – 0.939; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 1,143 breeding adults.

### Functional linkage and seasonal apportionment of potential effects

1527. The mean maximum foraging range of guillemot is 73.2km (±80.5km) and the maximum foraging range is 338km (Woodward *et al*., 2019). The Project is located approximately 293km from North Colonsay and Western Cliffs SPA.

While at the outer limit of the maximum foraging range recorded by Woodward *et al.,* (2019), this distance considerably exceeds the mean maximum foraging range +1SD for this species. The maximum foraging range is a poor indicator of typical foraging behaviour. It would be expected that few birds or foraging trips will occur at this distance from the colony, and even fewer with any regularity. No impacts during the breeding season are therefore apportioned to birds breeding at this colony.

1528. Outside the breeding season, breeding guillemots from the SPA are assumed to range widely and to mix with guillemots of all ages from breeding colonies in the UK and beyond. The relevant non-breeding season reference population is the UK Western Waters BDMPS, consisting of 1,139,220 individuals (August to February) (Furness, 2015). During the non-breeding season, it is estimated that 2.4% of birds present are considered to be breeding adults from the North Colonsay and Western Cliffs SPA, and impacts are apportioned accordingly. This is based on the SPA adult population from Furness (2015) as a proportion of the total UK Western Waters BDMPS.

## Potential effects on the qualifying feature

### Operation and maintenance phase disturbance/displacement/barrier effects

*Project-alone*

1529. The mean peak abundance of guillemots present within the windfarm site and 2km buffer during the non-breeding season was 8,315 (6,085-12,047) individuals (refer to **Appendix 12.1** of the ES). Of these, 200 birds (146-298) were likely to be breeding adults from the North Colonsay and Western Cliffs SPA.

1530. **Table 8.93** sets out the predicted impacts on guillemots from North Colonsay and Western Cliffs SPA during the non-breeding season. Displacement rates of 30% to 70% are considered for this species, along with a range of mortality rates of 1% to 10% of displaced birds (UK SNCBs, 2017).

1531. The available evidence suggests that the upper ranges of these displacement and mortality rates may be excessively precautionary. Whilst it is true that guillemots and razorbills tend to be displaced by OWFs they do not avoid them completely, and displacement rates vary between sites (MacArthur Green, 2019b). On average it was concluded that densities within OWFs were around half of the density found in the habitats around the OWF. At some OWFs there was also displacement of birds from a buffer zone surrounding it. The size of the buffer zone varied between OWFs and was generally less than 2km, with auk density increasing across the buffer zone as distance from turbines increased, up to the density in the wider area. Another recent review (APEM, 2022) found that the displacement rates suggested by the SNCBs for guillemot and razorbill (30% to 70%) did not account for the quality of or confidence in the studies which were used to inform this position, and that studies where no significant effects were recorded were not accounted for during the provision of the advice. APEM (2022) suggested that in the case of Hornsea Project Four, an evidence-based displacement rate of 50% was appropriate. However, the study also recognised that larger displacement effects are possible.

1532. Mortality due to displacement could arise due to increased energy costs and/or decreased energy intake, if displacement results in increased flying time to avoid OWFs, and/or increased bird densities and competition for prey in areas of unimpacted habitat outside the OWF. Given that UK OWFs in the Irish and Celtic Seas represent a very small proportion of the available foraging habitat for guillemots and razorbills within UK Western Waters, the increase in density of auks outside OWFs due to displacement is likely to be negligible (MacArthur Green, 2019b). It is considered unlikely that the mortality rate due to displacement would be as high as the empirically estimated 6.1% annual mortality for adult guillemots that occurs due to the combination of ‘natural’ factors and anthropogenic activities (Horswill and Robinson, 2015). Indeed, it may be much lower; MacArthur Green (2019b) recommended precautionary rates of 50% displacement for guillemot and 1% mortality of displaced birds based on a review of available evidence. Modelling undertaken by APEM (2022) found that in the case of Hornsea Project Four, a mortality rate of 1% for displaced auks was considered to be precautionary, and that this may be an overestimate given that Hornsea Project Four is located towards the upper end of the mean maximum foraging range for guillemot from the closest designated sites. It is also noted that in the recent decision on Hornsea Project Four, the SoS determined that a rate of 70% displacement/2% mortality was appropriate to inform the assessment of effects on guillemots from Flamborough and Filey Coast SPA (DESNZ, 2023b).

1533. The full ranges of recommended displacement and mortality effects are considered by the assessment, along with evidence-based displacement and mortality rates of 50% and 1%, respectively (APEM, 2022; MacArthur Green, 2019b).

*Table 8.93 Guillemot – predicted operation and maintenance phase displacement and mortality from North Colonsay and Western Cliffs SPA*

| Mean peak abundance estimate type | Mean peak abundance estimate | Number of North Colonsay and Western Cliffs SPA breeding adults present (non-breeding season)[1] | Annual mortality range[2] | Annual baseline mortality increase range[3] |
|---|---|---|---|---|
| Upper 95% CI | 12,047 | 289 | 1-20 | 0.08-1.77% |
| Mean | 8,315 | 200 | 1-14 | 0.05-1.22% |
| Lower 95% CI | 6,085 | 146 | 0-10 | 0.04-0.89% |

[1] Assumes 2.4% of birds present during the non-breeding season are North Colonsay and Western Cliffs SPA breeding adults

[2] Assumes displacement rates of 30-70% and mortality rates of 1-10%

[3] Background mortality rate of 6.1% (Horswill and Robinson, 2015)

1534. Based on the mean peak abundances, the annual total of guillemots from the North Colonsay and Western Cliffs SPA at risk of displacement is 200 birds (**Table 8.93**). At displacement rates of 30% to 70%, and mortality rates of 1% to 10% for displaced birds, 1 to 14 SPA breeding adults would be predicted to die each year due to displacement from the Project.

1535. Assuming a displacement rate of 70% and a mortality rate of 10% of displaced birds, annual mortality within the SPA breeding adult population would increase by 1.22%. Using an evidence-based displacement rate of 50%, and a mortality rate for displaced birds of 1%, annual mortality in the population would instead increase by 0.09% (1 bird).

1536. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that no detectable changes in mortality rates would occur under almost any combination of displacement and mortality rates when the mean peak abundance estimate assessments are considered. Mortality rate increases of over 1% are predicted for mean peak abundance estimate assessments only when a displacement rate of 70% and a mortality rate of 10% is considered. These displacement and mortality rates are much higher than evidence suggests will actually be the case. Use of the evidence-based displacement (50%) and mortality rate (1%) would result in a mortality increase of significantly less than 1%, as would a rate of 70%/2% agreed by the SoS in respect of Hornsea Project Four (DESNZ, 2023b).

1537. Increases of over 1% are also predicted if the upper 95% CIs for mean peak abundances are used as inputs to the assessment alongside a 10% mortality rate for displaced birds. The probability of this occurring is extremely small for two reasons. Firstly, the upper 95% CI for the mean peak abundances are highly unlikely to occur regularly at the windfarm site. Secondly, mortality rates for displaced birds of 10% are much higher than evidence suggests will actually be the case, and use of the evidence-based displacement (50%) and mortality rate (1%) (and also 70%/2%) would again result in a mortality increase of significantly less than 1%.

1538. **It is concluded that predicted guillemot mortality due to operational phase displacement for the project alone would not adversely affect the integrity of the North Colonsay and Western Cliffs SPA.**

1539. The confidence in the assessment is high for several reasons. Firstly, the evidence used to inform the evidence-based displacement rates is of high applicability and quality (based on the criteria discussed in **Chapter 12 Offshore Ornithology** of the ES). Whilst there is limited available evidence to inform mortality rates, 1% is considered to be sufficiently precautionary based on expert opinion. This species is not regarded as being highly specialised in its habitat requirements (Bradbury *et al*., 2014; Furness and Wade, 2012; Garthe and Hüppop, 2004), and it is therefore anticipated that displaced birds will find alternative habitat in the vast majority of cases. Finally, the conclusion of the assessment is the same irrespective of whether the mean or 95% upper CI mean peak abundances are used to calculate potential mortality and increases in the baseline mortality rate of the background population, provided the evidence-based displacement and mortality rates are used.

*In-combination*

1540. The in-combination assessment for guillemots from North Colonsay and Western Cliffs SPA has been undertaken in accordance with the approach presented in **Section 8.1**. The total population apportioned to Rathlin Island SPA at risk of displacement is estimated to be 1,190 breeding adults (**Appendix 12.1** of the ES). Annual in-combination displacement and mortality rates for birds from North Colonsay and Western Cliffs SPA are presented in **Table 8.94**.

1541. Assuming a displacement rate of 70% and a mortality rate of 10% of displaced birds, 83 breeding adult SPA birds would be lost to displacement annually. Including additional apportioned mortality from the Morlais and Holyhead tidal projects (total 0.76 birds), this would increase the existing mortality within the SPA population (1,143 breeding adult birds per year) by 7.35%. Using an evidence-based displacement rate of 50%, and a mortality rate for displaced birds of 1%, the annual in-combination mortality would be 7 birds. This would increase the existing mortality within this population by 0.59%. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that detectable changes in mortality rates will not occur due to the level or mortality predicted if the more realistic rates for mortality are used.

1542. **It is concluded that predicted guillemot mortality due to of operational phase displacement impacts of the Project in-combination with other projects, would not adversely affect the integrity of North Colonsay and Western Cliffs SPA.**

*Table 8.94 In-combination year-round displacement matrix for guillemot from North Colonsay and Western Cliffs SPA*

| Annual | Mortality | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Displacement | 1% | 2% | 3% | 4% | 5% | 10% | 20% | 30% | 50% | 80% | 100% |
| 10% | 1 | 2 | 4 | 5 | 6 | 12 | 24 | 36 | 59 | 95 | 119 |
| 20% | 2 | 5 | 7 | 10 | 12 | 24 | 48 | 71 | 119 | 190 | 238 |
| 30% | 4 | 7 | 11 | 14 | 18 | 36 | 71 | 107 | 178 | 286 | 357 |
| 40% | 5 | 10 | 14 | 19 | 24 | 48 | 95 | 143 | 238 | 381 | 476 |
| 50% | 6 | 12 | 18 | 24 | 30 | 59 | 119 | 178 | 297 | 476 | 595 |
| 60% | 7 | 14 | 21 | 29 | 36 | 71 | 143 | 214 | 357 | 571 | 714 |
| 70% | 8 | 17 | 25 | 33 | 42 | 83 | 167 | 250 | 416 | 666 | 833 |
| 80% | 10 | 19 | 29 | 38 | 48 | 95 | 190 | 286 | 476 | 762 | 952 |
| 90% | 11 | 21 | 32 | 43 | 54 | 107 | 214 | 321 | 535 | 857 | 1071 |
| 100% | 12 | 24 | 36 | 48 | 59 | 119 | 238 | 357 | 595 | 952 | 1190 |

Note: The cells show the number of birds predicted to die (rounded to the nearest integer) at a given rate of displacement and mortality. Highlighted cells are considered to be the most realistic scenario, in accordance with SNCB advice (SNCBs, 2022).

## 8.35 Treshnish Isles SPA

1543. Treshnish Isles SPA is located approximately 339km from the windfarm site.

### 8.35.1 Description of designation

1544. Treshnish Isles SPA comprises a string of islands and skerries about 5km off the west coast of the island of Mull. The site is rocky, with cliffs, screes and raised beaches, and supports strongly maritime grassland and heath. European storm-petrel is the only qualifying seabird species for this SPA.

### 8.35.2 Conservation objectives

1545. The overarching conservation objectives for the site are:

- To avoid deterioration of the habitats of the qualifying species or significant disturbance to the qualifying species, thus ensuring that the integrity of the site is maintained
- To ensure for the qualifying species that the following are maintained in the long term:
  - Population of the species as a viable component of the site
  - Distribution of the species within site
  - Distribution and extent of habitats supporting the species
  - Structure, function and supporting processes of habitats supporting the species
  - No significant disturbance of the species

### 8.35.3 Assessment

1546. One qualifying feature of Treshnish Isles SPA has been screened into the Appropriate Assessment (**Table 5.2**): European storm-petrel.

#### 8.35.3.1 European storm-petrel

##### Status

1547. The Treshnish Isles SPA European storm-petrel population was cited as 5,040 pairs, or 10,080 breeding adults, in 1996 (SNH, 2018b). The most recent count in 2018-19 identified 10,272 pairs (AOS), or 20,544 breeding adults (JNCC, 2023a); this is used as the reference population for the assessment.

### Functional linkage and seasonal apportionment of potential effects

1548. The mean maximum foraging range of European storm-petrel is 336km, as is the maximum foraging range (Woodward *et al*., 2019). The Project is located approximately 339km from Treshnish Isles SPA, which means that the Project is just outside the mean maximum foraging range of European storm-petrels breeding at this SPA.

### Potential effects on the qualifying feature from the Project-alone

1549. Storm petrel was not recorded during baseline surveys of the windfarm site (including buffer areas). It is therefore concluded that this species does not occur regularly in this area. It is noted that storm petrel is considered to have low vulnerability to collision risk and very low vulnerability to displacement impacts (Bradbury *et al*., 2014), and therefore the risk of significant effects would be low, even if this species occurred at the windfarm site.

1550. **It is therefore concluded that there would be no measurable effects on storm petrel due to the project alone, and no adverse effect on the integrity of the Treshnish Isles SPA is predicted**.

### Potential effects on the qualifying feature in-combination with other projects

1551. As the Project would have no measurable effect on storm petrel populations from the Treshnish Isles SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of Treshnish Isles SPA, when assessed in-combination with other plans or projects.**

## 8.36 Fowlsheugh SPA

1552. Fowlsheugh SPA is located approximately 351km from the windfarm site.

### 8.36.1 Description of designation

1553. Fowlsheugh SPA, located 4km south of Stonehaven on the east coast of Aberdeenshire, is a stretch of sheer cliffs between 30m and 60m high. Large numbers of seabirds nest on the cliffs. The seaward extension of the SPA extends 2km into the marine environment and includes the seabed, water column and surface. Seabirds included within the designation feed both inside and outside the SPA in nearby waters, as well as more distantly in the wider North Sea.

### 8.36.2 Conservation objectives

1554. The overarching conservation objectives for the site are:

- To avoid deterioration of the habitats of the qualifying species or significant disturbance to the qualifying species, thus ensuring that the integrity of the site is maintained
- To ensure for the qualifying species that the following are maintained in the long term:
  - Population of the species as a viable component of the site
  - Distribution of the species within site
  - Distribution and extent of habitats supporting the species
  - Structure, function and supporting processes of habitats supporting the species
  - No significant disturbance of the species

### 8.36.3 Assessment

1555. The qualifying features of Fowlsheugh SPA screened into the Appropriate Assessment are listed in **Table 5.2**. These are breeding fulmar and kittiwake. Both species are also screened in as named components of the seabird assemblage.

#### 8.36.3.1 Fulmar

#### Status

1556. The Fowlsheugh SPA breeding fulmar population at classification (1992) was cited as 1,170 pairs, or 2,340 breeding adults (SNH, 2009a). Furness (2015) gave a breeding population of 193 pairs, or 386 breeding adults, in 2009. The most recent count is 157 pairs, or 314 breeding adults, in 2018 (JNCC, 2023a); this is used as the reference population for the assessment.

1557. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.064 (1 – 0.936; Horswill and Robinson, 2015), the expected annual mortality of the SPA population would be 20 breeding adults.

### Functional linkage and seasonal apportionment of potential effects

1558. The mean maximum foraging range of fulmar is 542.3km (±657.9km) and the maximum foraging range is 2,736km (Woodward *et al*., 2019). The Project is located approximately 351km from Fowlsheugh SPA, which means that the Project is within the mean maximum foraging range of fulmars breeding at this SPA.

### Potential effects on the qualifying feature from the Project-alone

1559. Fulmar are considered to have low vulnerability to both collision and displacement impacts from offshore windfarms (Bradbury *et al*., 2014). This species was recorded infrequently within the windfarm site (in only five of 24 surveys, all during the breeding season (January to August; Furness, 2015) and in low numbers, with densities in four of the five months less than 0.1 birds/km$^2$. A peak density of 0.23 birds/km$^2$ was recorded in May 2022.

1560. On the basis of the low vulnerability of fulmars to potential impacts, and the low frequency and abundance of birds present at the windfarm site, it is concluded that any effects on breeding populations at Fowlsheugh SPA are very unlikely, both during and outside of the breeding season.

1561. **It is therefore concluded that there would be no measurable effects on fulmar due to the project alone, and no adverse effect on the integrity of the Fowlsheugh SPA is predicted.**

### Potential effects on the qualifying feature in-combination with other projects

1562. As the Project would have no measurable effect on fulmar populations from the Fowlsheugh SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of Fowlsheugh SPA, when assessed in-combination with other plans or projects.**

### 8.36.3.2 Kittiwake

#### Status

1563. The Fowlsheugh SPA kittiwake breeding population at classification (1992) was cited as 36,650 pairs, or 73,300 breeding adults (SNH, 2009a). Furness (2015) gave a breeding population of 9,337 breeding pairs, or 18,674 breeding adults, in 2012. The most recent available count is 14,039 breeding pairs (AON), or 28,078 breeding adults, in 2018 (JNCC, 2023a); this is used as the reference population for the assessment.

1564. Based on the published adult kittiwake mortality rate of 0.146 (1 – 0.854; Horswill and Robinson, 2015), the expected annual mortality of the SPA population would be 4,099 breeding adults.

#### Functional linkage and seasonal apportionment of potential effects

1565. The mean maximum foraging range of kittiwake is 156.1km (±144.5km) and the maximum foraging range is 770km (Woodward *et al*., 2019). The straight-line distance between the Project and Fowlsheugh SPA is approximately 351km and therefore theoretically within the maximum foraging range for this species. However, the across-sea distance is approximately 950km, and therefore no breeding season connectivity between the Site and SPA kittiwake population is predicted.

1566. Outside of the breeding season, breeding kittiwakes, including those from the Fowlsheugh SPA, are not constrained by requirements to visit nests to incubate eggs or provision chicks. At this time, they are assumed to range more widely and to mix with kittiwakes of all age classes from breeding colonies in the UK and further afield. The background population during these seasons is the UK Western Waters plus Channel BDMPS. This consists of 911,586 individuals during the autumn migration season (August to December) and 691,526 individuals during the spring migration season (January to April).

1567. Furness (2015) estimated that 20% of the Fowlsheugh breeding adults are present within the UK Western Waters plus Channel BDMPS during the autumn migration season, and 30% during spring migration, which is 3,735 and 5,602 birds respectively. This represents 0.41% of the BDMPS population during the autumn migration period (911,586), and 0.89% during spring migration (691,526). It is therefore assumed that 0.41% of kittiwakes present at the Project site during the autumn migration period are breeding adults from Fowlsheugh SPA, and 0.89% during spring migration.

## Potential effects on the qualifying feature from the Project-alone

1568. The kittiwake qualifying feature of the Fowlsheugh SPA has been screened into the Appropriate Assessment due to the potential risk of collision.

### Operation and maintenance phase collision risk

1569. Information for collision risk on breeding adult kittiwakes belonging to the Fowlsheugh SPA population is presented in **Table 8.95**. Collision estimates, calculated using the sCRM, are presented by biological season. A summary of the annual outputs and the corresponding increase in the annual baseline mortality rate is also presented. Parameters used in the sCRM were agreed with Natural England during the ETG process and are described in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES.

1570. Based on the mean collision rates, the annual total of breeding adult kittiwakes from Fowlsheugh SPA at risk of collision as a result of the Project is 0.04. This would increase the existing mortality of the SPA breeding population by 0.00%.

*Table 8.95 Predicted seasonal and annual collision mortality (Stochastic model Option 2, avoidance rate 0.993 (±0.0003)), for breeding adult kittiwakes at the windfarm site, apportioned to Fowlsheugh SPA, with corresponding increases to baseline mortality of the population*

| Period | Breeding Season<br>Mar-Aug | Autumn Migration<br>Sep-Dec | Non-breeding/winter<br>- | Spring Migration<br>Jan-Feb | Annual<br>Jan-Dec |
|---|---|---|---|---|---|
| Total collisions[1] (mean and 95% CIs) | 15.36<br>(4.17-33.87) | 8.49<br>(2.20-18.85) | - | 0.63<br>(0.00-1.46) | 24.48<br>(6.37-54.17) |
| % apportioned to the SPA | 0.00% | 0.41% | - | 0.89% | - |
| Total SPA collisions (mean and 95% CIs) | 0.00<br>(0.00-0.00) | 0.03<br>(0.01-0.08) | - | 0.01<br>(0.00-0.01) | 0.04<br>(0.01-0.09) |
| Mortality increase[2] (mean and 95% CIs) | 0.00%<br>(0.00-0.00%) | 0.00%<br>(0.00-0.00%) | - | 0.00%<br>(0.00-0.00%) | 0.00-0.00%<br>(0.00%) |

[1] Breeding season collision values reduced to 94.1% of modelled value to reflect proportion of adult birds recorded during site surveys

[2] Assuming predicted annual SPA mortality of 4,099 birds (28,078 x 0.146)

1571. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that no detectable changes in mortality rates would occur on this population for the mean monthly collision estimates for the Project, or for the upper 95% CI collision estimate.

1572. **It is concluded that based on predicted kittiwake mortality due to collision at the windfarm site there is no potential for the Project to have an adverse effect on the integrity of the Fowlsheugh SPA.**

1573. The confidence in the assessment is high. The evidence used to define the CRM input parameters presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** as part of the ES submitted alongside this RIAA is of high applicability and quality. Whilst there is uncertainty around some of the input parameters (e.g. avoidance rate), the rates selected are considered to be sufficiently precautionary based on expert opinion to provide confidence that collision rates are not underestimated.

### Potential effects on the qualifying feature in-combination with other projects

1574. As the Project would have no measurable effect on kittiwake populations from the Fowlsheugh SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of Fowlsheugh SPA, when assessed in-combination with other plans or projects.**

## 8.37 Rum SPA

1575. Rum SPA is located approximately 374km from the windfarm site.

### 8.37.1 Description of designation

1576. Rum SPA includes the Inner Hebridean Island of Rum, which has a largely rocky coast with cliffs rising to 210m, and adjacent coastal waters. The boundary of the SPA overlaps with Rum SSSI and the seaward elements extend approximately 4 km into the marine environment to include the seabed, water column and surface. The qualifying seabird species for the SPA comprise kittiwake, red-throated diver, guillemot and Manx shearwater.

### 8.37.2 Conservation objectives

1577. The overarching conservation objectives for the site are:

- To avoid deterioration of the habitats of the qualifying species or significant disturbance to the qualifying species, thus ensuring that the integrity of the site is maintained
- To ensure for the qualifying species that the following are maintained in the long term:
  - Population of the species as a viable component of the site
  - Distribution of the species within site
  - Distribution and extent of habitats supporting the species
  - Structure, function and supporting processes of habitats supporting the species
  - No significant disturbance of the species

### 8.37.3 Assessment

1578. One qualifying feature of Rum SPA has been screened into the Appropriate Assessment (**Table 5.2**): breeding Manx shearwater. This species has also been screened in as a component of the seabird assemblage.

#### 8.37.3.1 Manx shearwater

#### Status

1579. The Rum SPA breeding Manx shearwater population was cited as 61,000 pairs, or 122,000 breeding adults, in 1995 (SNH, 2020). Furness (2015) gave a breeding population of 120,000 pairs, or 240,000 breeding adults, in 2001. There were no recent (post-2001) counts on the SMP database (JNCC, 2033).

Therefore, the most recent accurate population estimate is taken to be 240,000 breeding adults; this is used as the reference population for the assessment.

1580. Based on the most recent SPA population of assumed breeding adults, and an annual adult baseline mortality rate of 0.130 (1 – 0.870; Horswill and Robinson 2015), the expected annual mortality of the SPA population would be 31,200 breeding adults.

### Functional linkage and seasonal apportionment of potential effects

1581. The mean maximum foraging range of Manx shearwater is 1,346.8km (±1,018.7km) and the maximum foraging range is 2890km. The Project is located approximately 374km from Rum SPA, which means that the Project is within the mean maximum foraging range of Manx shearwaters breeding at this SPA.

1582. A number of SPA and non-SPA Manx shearwater colonies are located in and around the UK Western Waters BDMPS area, all of which are within the mean maximum foraging range of this species. For a review of these sites see **Section 8.21.3.1**.

1583. The NatureScot apportioning tool (NatureScot, 2018) has been used to estimate the proportion of Manx shearwaters from each of the relevant SPAs present at the windfarm site during the breeding season. The apportioning to SPA and non-SPA colonies is set out in **Table 8.96**; refer also to **Appendix 12.1** of the ES for further information on the apportioning approach and results. Accordingly, 8.44% of impacts at the windfarm site during the breeding season are apportioned to Rum SPA.

*Table 8.96 Manx shearwater breeding season apportioning*

| Site | Apportioning rate |
|---|---|
| Glannau Aberdaron ac Ynys Enlli/Aberdaron Coast and Bardsey Island SPA | 8.63% |
| Copeland Islands SPA | 2.22% |
| Skomer, Skokholm and the Seas off Pembrokeshire / Sgomer, Sgogwm a Moroedd Penfro SPA | 76.54% |
| **Rum SPA** | **8.44%** |
| St Kilda SPA | 0.20% |
| Cruagh Island SPA (transboundary site) | 0.10% |
| Blasket Islands SPA (transboundary site) | 0.61% |
| Deenish Island and Scariff Island SPA (transboundary site) | 0.08% |

| Site | Apportioning rate |
|---|---|
| Puffin Island SPA (transboundary site) | 0.22% |
| Skelligs SPA (transboundary site) | 0.03% |
| Non-SPA colonies | 2.91% |

1584. During the pre- and post-breeding periods, breeding Manx shearwaters from the Rum SPA migrate through UK waters. The relevant reference population is considered to be the UK Western Waters BDMPS. This consists of 1,580,895 individuals during the post-breeding (August-early October) and return migration (late march-May) periods.

1585. Estimates of the proportion of Manx shearwaters present at the windfarm site which originate from the Rum SPA during the post-breeding and return migration periods (and therefore the proportion of predicted mortalities from the SPA population) are based on the SPA population (i.e. 240,000 adults) as a proportion of the UK Western Waters BDMPS during the relevant season. During the post-breeding and return migration periods, 15.2% of impacts are considered to affect birds from the SPA (Furness, 2015).

### Potential effects on the qualifying feature from the Project-alone

1586. The Manx shearwater qualifying feature of the Rum SPA has been screened into the assessment due to the potential risk of disturbance, displacement and barrier effects during the construction and decommissioning, and operation and maintenance phases of the Project.

#### Construction and decommissioning phase disturbance/displacement/barrier effects

*Project-alone*

1587. Effects during the construction and decommissioning phases of the Project are considered unlikely, given the transient presence of the species and low susceptibility to disturbance related impacts; refer to **Paragraph 1589** below. However, in accordance with feedback received from Natural England and NRW, a precautionary estimation of construction and decommissioning phase disturbance, displacement and barrier effects has been undertaken assuming 50% of the operational phase effect.

1588. Applying 50% reduction to the operational values presented in **Table 8.97**, and based on mean density, predicted mortality would be between two and 37 birds (30-70% displacement and 1-10% mortality of displaced birds). Using realistic values (i.e. mean density, 50% displacement and 1% mortality), there would be an annual increase in mortality of three birds, which is equivalent to a 0.01% increase in background mortality for the SPA population. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. Accordingly, no significant effects on Manx shearwater are predicted during the construction and decommissioning phases, and **it is concluded that there would be no adverse effect on the integrity of Rum SPA**.

### Operation and maintenance phase disturbance/displacement/barrier effects

1589. Manx shearwater are generally considered to have a low susceptibility to disturbance and displacement (Furness *et al.*, 2013). See **Section 8.21.3.1** for summary of effects from Dierschke *et al*., (2016) and Bradbury *et al.,* (2014).

1590. Displacement effects for Manx shearwater for the Project were assessed during the breeding, autumn migration and spring migration periods, based on an unapportioned peak mean population of 4,705, 2,650 and 1,617 birds respectively, calculated for the windfarm site and a 2km buffer, in line with recommendations within the SNCB guidance (SNCBs, 2017). The displacement matrices used to calculate potential impacts are presented in **Appendix 12.1** of the ES, and summarised in **Table 8.97.** The application of the same displacement rate to the OWF and the 2km buffer, to determine the total number of birds subject to displacement, is precautionary, as in reality the displacement rate is likely to fall with distance from the windfarm site.

1591. A displacement rate of 30-70% and mortality rate of 1-10% has been presented. Given that 10% would represent a rate close to the expected 'natural' annual mortality (0.13), this rate is considered very unlikely. Accordingly, a 1% mortality rate is considered to be most appropriate, with the upper end of this range likely to be precautionary. Given the very extensive foraging range of this species (Woodward *et al.*, 2019), there may be no mortality costs to displacement from the relatively very small footprints of OWFs.

*Table 8.97 Manx shearwater – predicted operation and maintenance phase displacement and mortality from Rum SPA*

| Mean peak abundance estimate type | Mean peak abundance estimate | Number of SPA breeding adults present by season[1] | Annual mortality range[2] | Annual baseline mortality increase range[3] |
|---|---|---|---|---|
| Upper 95% CI | 10,010 (breeding)<br>4,447 (autumn)<br>4,711 (spring)<br>19,168 (year round) | 845 (breeding)<br>675 (autumn)<br>715 (spring)<br>2,235 (year round) | 7-156 | 0.02-0.50% |
| Mean | 4,705 (breeding)<br>2,650 (autumn)<br>1,617 (spring)<br>8,972 (year round) | 397 (breeding)<br>402 (autumn)<br>246 (spring)<br>1,045 (year round) | 3-73 | 0.01-0.23% |
| Lower 95% CI | 783 (breeding)<br>1,308 (autumn)<br>0 (spring)<br>2,092 (year round) | 66 (breeding)<br>199 (autumn)<br>0 (spring)<br>265 (year round) | 1-19 | 0.00-0.06% |

[1] During the breeding season, assumes 8.4% of recorded birds are adults from the SPA population (11,806), and 15.2% during the autumn and spring migration periods

[2] Assumes displacement rates of 30-70% and mortality rates of 1-10%

[3] Background population is Rum SPA breeding adults (240,000 individuals), adult age class annual mortality rate of 13% (Horswill and Robinson, 2015)

1592. Using realistic values (i.e. mean density, 50% displacement and 1% mortality), there would be an annual increase in mortality of five birds, representing a 0.02% increase in mortality rate. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. Accordingly, no significant effects on Manx shearwater are predicted during the operation and maintenance phase, and **it is concluded that there is no potential for the Project to have an adverse effect on the integrity of Rum SPA**.

1593. A review of the potential effects of artificial light on Manx shearwaters is presented in Section 12.6.3.1 of **Chapter 12 Offshore Ornithology**. This concludes that lighting associated with the Project is very unlikely to significantly affect disturbance and displacement effects on Manx shearwater, and therefore the conclusions of the assessment are unchanged.

1594. The confidence in the assessment is high. The evidence used to set the displacement rates presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES is of high applicability and quality. Whilst there was limited available evidence to inform mortality rates, 1% is considered to be sufficiently precautionary based on expert opinion.

### Potential effects in-combination with other projects

1595. No in-combination effects are predicted during the construction and decommissioning phases. This is because of the negligible contribution of the Project-alone. It is also noted that there would be limited potential for significant temporal and/or spatial overlap with other plans or projects.

1596. During the operation and maintenance phase, the in-combination assessment for Manx shearwaters from Rum SPA has been undertaken in accordance with the approach presented in **Section 8.1**. The total population apportioned to the SPA at risk of displacement is estimated to be 1,561 breeding adults (**Appendix 12.1** of the ES). Annual in-combination displacement and mortality rates for birds from Rum SPA are presented in **Table 8.98**.

1597. Assuming a displacement rate of 70% and a mortality rate of 10% of displaced birds, 109 breeding adult SPA birds would be lost to displacement annually. This would increase the existing mortality within the SPA population (31,200 breeding adult birds per year) by 0.35%. Using a realistic displacement rate of 50%, and a mortality rate for displaced birds of 1%, the annual in-combination mortality would be 8 birds. This would increase the existing mortality within this population by 0.03%. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that detectable changes in mortality rates will not occur due to the level or mortality predicted if the more realistic rates for mortality are used.

1598. **It is concluded that predicted Manx shearwater mortality due to of operational phase displacement impacts of the Project in-combination with other projects, would not adversely affect the integrity of Rum SPA.**

*Table 8.98 In-combination year-round displacement matrix for Manx shearwater from Rum SPA*

| Annual | Mortality | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| **Displacement** | **1%** | **2%** | **3%** | **4%** | **5%** | **10%** | **20%** | **30%** | **50%** | **80%** | **100%** |
| **10%** | 2 | 3 | 5 | 6 | 8 | 16 | 31 | 47 | 78 | 125 | 156 |
| **20%** | 3 | 6 | 9 | 12 | 16 | 31 | 62 | 94 | 156 | 250 | 312 |
| **30%** | 5 | 9 | 14 | 19 | 23 | 47 | 94 | 140 | 234 | 375 | 468 |
| **40%** | 6 | 12 | 19 | 25 | 31 | 62 | 125 | 187 | 312 | 499 | 624 |
| **50%** | 8 | 16 | 23 | 31 | 39 | 78 | 156 | 234 | 390 | 624 | 780 |
| **60%** | 9 | 19 | 28 | 37 | 47 | 94 | 187 | 281 | 468 | 749 | 936 |
| **70%** | 11 | 22 | 33 | 44 | 55 | 109 | 219 | 328 | 546 | 874 | 1093 |
| **80%** | 12 | 25 | 37 | 50 | 62 | 125 | 250 | 375 | 624 | 999 | 1249 |
| **90%** | 14 | 28 | 42 | 56 | 70 | 140 | 281 | 421 | 702 | 1124 | 1405 |
| **100%** | 16 | 31 | 47 | 62 | 78 | 156 | 312 | 468 | 780 | 1249 | 1561 |

Note: The cells show the number of birds predicted to die (rounded to the nearest integer) at a given rate of displacement and mortality. Highlighted cells are considered to be the most realistic scenario, in accordance with SNCB advice (SNCBs, 2022).

# 8.38 Canna and Sanday SPA

1599. Canna and Sanday SPA is located approximately 394km from the windfarm site.

## 8.38.1 Description of designation

1600. The island of Canna is the most western of the Small Isles in the Inner Hebrides. The site also includes part of the smaller island of Sanday, which is connected to Canna at low tide. The coastline of Canna consists mainly of steep cliffs capped by a ridge of wet heath and blanket bog. Sanday and the more low-lying areas of Canna support a varied range of coastal grassland and heath communities. The boundary of the Special Protection Area overlaps with the boundary of Canna and Sanday SSSI, and the seaward elements extend approximately 1 km into the marine environment to include the seabed, water column and surface. The qualifying seabird species of the SPA are kittiwake, herring gull, guillemot and puffin.

## 8.38.2 Conservation objectives

1601. The overarching conservation objectives for the site are:

- To avoid deterioration of the habitats of the qualifying species or significant disturbance to the qualifying species, thus ensuring that the integrity of the site is maintained
- To ensure for the qualifying species that the following are maintained in the long term:
    - Population of the species as a viable component of the site
    - Distribution of the species within site
    - Distribution and extent of habitats supporting the species
    - Structure, function and supporting processes of habitats supporting the species
    - No significant disturbance of the species

## 8.38.3 Assessment

1602. One qualifying feature of Canna and Sanday SPA has been screened into the Appropriate Assessment (**Table 5.2**): breeding guillemot. This species has also been screened in as a named component of the seabird assemblage.

### 8.38.3.1 Guillemot

#### Status

1603. The Canna and Sanday SPA breeding guillemot population was cited as 3,858 pairs, or 7,716 breeding adults, in 1998 (Furness, 2015; Stroud *et al*., 2016). Furness (2015) gave the breeding population of 3,913 pairs, or 7,826 breeding adults, in 1999. The most recent count is 3,060 individuals in 2018 (JNCC, 2023a); this is used as the reference population for the assessment.

1604. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.061 (1 – 0.939; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 187 breeding adults.

#### Functional linkage and seasonal apportionment of potential effects

1605. The mean maximum foraging range of guillemot is 73.2km (±80.5km) and the maximum foraging range is 338km (Woodward *et al*., 2019). The Project is located approximately 394km from Canna and Sanday SPA, which means that the Project is beyond the maximum foraging range of guillemots from the SPA. No impacts during the breeding season from the Project are therefore apportioned to guillemots breeding at this SPA.

1606. Outside the breeding season, breeding guillemots from the SPA are assumed to range widely and to mix with guillemots of all ages from breeding colonies in the UK and beyond. The relevant non-breeding season reference population is the UK Western Waters BDMPS, consisting of 1,139,220 individuals (August to February) (Furness, 2015).

1607. Furness (2015) estimated that 95% of the Canna and Sanday SPA breeding adults are present within the UK Western Waters BDMPS during the non-breeding season, which is 7,435 birds. This represents 0.7% of the BDMPS population for this period (1,139,220). It is therefore assumed that 0.7% of guillemots present at the Project site are breeding adults from Canna and Sanday SPA.

#### Potential effects on the qualifying feature

#### Operation and maintenance phase disturbance / displacement / barrier effects

*Project-alone*

1608. The mean peak abundance of guillemots present within the windfarm site and 2km buffer during the non-breeding season was 8,315 (6,085-12,047) individuals (refer to **Appendix 12.1** of the ES). Of these, 58 birds (43-84) were likely to be breeding adults from the Canna and Sanday SPA.

1609. **Table 8.99** sets out the predicted impacts on guillemots from Canna and Sanday SPA during the non-breeding season. Displacement rates of 30% to 70% are considered for this species, along with a range of mortality rates of 1% to 10% of displaced birds (UK SNCBs, 2017).

1610. The available evidence suggests that the upper ranges of these displacement and mortality rates may be excessively precautionary. Whilst it is true that guillemots and razorbills tend to be displaced by OWFs they do not avoid them completely, and displacement rates vary between sites. MacArthur Green (2019b) concluded that, on average, densities within OWFs were around half of the density found in the habitats around the OWF. At some OWFs there was also displacement of birds from a buffer zone surrounding it. The size of the buffer zone varied between OWFs and was generally less than 2km, with auk density increasing across the buffer zone as distance from turbines increased, up to the density in the wider area. Another recent review (APEM, 2022) found that the current displacement rates suggested by the SNCBs for guillemot and razorbill (30% to 70%) did not account for the quality of or confidence in the studies which were used to inform this position, and that studies where no significant effects were recorded were not accounted for during the provision of the advice. APEM (2022) suggested that in the case of Hornsea Project Four, an evidence-based displacement rate of 50% was appropriate. However, the study also recognised that larger displacement effects were possible.

1611. Mortality due to displacement could arise due to increased energy costs and/or decreased energy intake, if displacement results in increased flying time to avoid OWFs, and/or increased bird densities and competition for prey in areas of unimpacted habitat outside the OWF. Given that UK OWFs in the Irish and Celtic Seas represent a very small proportion of the available foraging habitat for guillemots and razorbills within UK Western Waters, the increase in density of auks outside OWFs due to displacement is likely to be negligible (MacArthur Green, 2019b). It is considered unlikely that the mortality rate due to displacement would be as high as the empirically estimated 6.1% annual mortality for adult guillemots that occurs due to the combination of 'natural' factors and anthropogenic activities (Horswill and Robinson, 2015). Indeed, it may be much lower; MacArthur Green (2019b) recommended precautionary rates of 50% displacement for guillemot and 1% mortality of displaced birds based on a review of available evidence. Modelling undertaken by APEM (2022) found that in the case of Hornsea Project Four, a mortality rate of 1% for displaced auks was considered to be precautionary, and that this may be an overestimate given that Hornsea Project Four is located towards the upper end of the mean maximum foraging range for guillemot from the closest designated sites. It is also noted that in the recent decision on Hornsea Project Four, the SoS determined that a rate of 70% displacement and 2% mortality was appropriate to inform the assessment of effects on guillemots from Flamborough and Filey Coast SPA (DESNZ, 2023b).

1612. The full ranges of recommended displacement and mortality effects are considered by the assessment, along with evidence-based displacement and mortality rates of 50% and 1%, respectively (APEM, 2022; MacArthur Green, 2019b).

*Table 8.99 Guillemot – predicted operation and maintenance phase displacement and mortality from Canna and Sanday SPA*

| Mean peak abundance estimate type | Mean peak abundance estimate | Number of Canna and Sanday SPA breeding adults present (non-breeding season)[1] | Annual mortality range[2] | Annual baseline mortality increase range[3] |
|---|---|---|---|---|
| Upper 95% CI | 12,047 | 84 | 0-6 | 0.14-3.16% |
| Mean | 8,315 | 58 | 0-4 | 0.09-2.18% |
| Lower 95% CI | 6,085 | 43 | 0-3 | 0.07-1.60% |

[1] Assumes 0.9% of birds present during the non-breeding season are Canna and Sanday SPA breeding adults

[2] Assumes displacement rates of 30-70% and mortality rates of 1-10%

[3] Background mortality rate of 6.1% (Horswill and Robinson, 2015)

1613. Based on the mean peak abundances, the annual total of guillemots from the Canna and Sanday SPA at risk of displacement is 58 birds (**Table 8.99**). At displacement rates of 30% to 70%, and mortality rates of 1% to 10% for displaced birds, 0 to 4 SPA breeding adults would be predicted to die each year due to displacement from the Project.

1614. Assuming a displacement rate of 70% and a mortality rate of 10% of displaced birds, annual mortality within the SPA breeding adult population would increase by 2.18%. Using an evidence-based displacement rate of 50%, and a mortality rate for displaced birds of 1%, annual mortality in the population would instead increase by 0.16% (<1 bird).

1615. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that no detectable changes in mortality rates would occur under almost any combination of displacement and mortality rates when the mean peak abundance estimate assessments are considered. Mortality rate increases of over 1% are predicted for mean peak abundance estimate assessments only when a mortality rate of 10%, or a displacement of 5%/70% mortality, are considered. These displacement and mortality rates are much higher than evidence suggests will actually be the case. Use of the evidence-based displacement (50%) and mortality rate (1%) would result in a mortality increase of significantly less than 1%, as would a rate of 70%/2% agreed by the SoS in respect of Hornsea Project Four (DESNZ, 2023b).

1616. Increases of over 1% are predicted if the upper 95% CIs for mean peak abundances are used as inputs to the assessment alongside a 10% mortality rate for displaced birds. The probability of this occurring is extremely small for two reasons. Firstly, the upper 95% CI for the mean peak abundances are highly unlikely to occur regularly at the windfarm site. Secondly, mortality rates for displaced birds of 10% are much higher than evidence suggests will actually be the case, and use of the evidence-based displacement (50%) and mortality rate (1%) (and also 70%/2%) would again result in a mortality increase of significantly less than 1%.

1617. **It is concluded that predicted guillemot mortality due to operational phase displacement for the project alone would not adversely affect the integrity of the Canna and Sanday SPA.**

1618. The confidence in the assessment is high for several reasons. Firstly, the evidence used to inform the evidence-based displacement rates is of high applicability and quality (based on the criteria discussed in **Chapter 12 Offshore Ornithology** of the ES). Whilst there is limited available evidence to inform mortality rates, 1% is considered to be sufficiently precautionary based on expert opinion. This species is not regarded as being highly specialised in its habitat requirements (Bradbury *et al*., 2014; Furness and Wade, 2012; Garthe and Hüppop, 2004), and it is therefore anticipated that displaced birds will find alternative habitat in the vast majority of cases. Finally, the conclusion of the assessment is the same irrespective of whether the mean or 95% upper CI mean peak abundances are used to calculate potential mortality and increases in the baseline mortality rate of the background population, provided the evidence-based displacement and mortality rates are used.

### Potential effects in-combination with other projects

1619. The in-combination assessment for guillemots from Canna and Sanday SPA has been undertaken in accordance with the approach presented in **Section 8.1**. The total population apportioned to Canna and Sanday SPA at risk of displacement is estimated to be 346 breeding adults (**Appendix 12.1** of the ES). Annual in-combination displacement and mortality rates for birds from Canna and Sanday SPA are presented in **Table 8.100.**

1620. Assuming a displacement rate of 70% and a mortality rate of 10% of displaced birds, 24 breeding adult SPA birds would be lost to displacement annually. Including additional apportioned mortality from the Morlais and Holyhead tidal projects (total 0.22 birds), this would increase the existing mortality within the

SPA population (187 breeding adult birds per year) by 13.10%. Using an evidence-based displacement rate of 50%, and a mortality rate for displaced birds of 1%, the annual in-combination mortality would be two birds. This would increase the existing mortality within this population by 1.05%. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. Although marginally above this 1% threshold, it is considered very unlikely that this would actually have a measurable effect on the SPA population. This is because of the very small number of potentially impacted birds due to displacement (<2), and the recognition that, as guillemot is a dispersive rather than a fully migratory species, birds do not travel great distances from the breeding colony during the non-breeding season (MS-LOT, 2022), and therefore apportioning using the BDMPS is likely to significantly overestimate the numbers of birds from the SPA present at the Project site.

1621. **It is concluded that predicted guillemot mortality due to of operational phase displacement impacts of the Project in-combination with other projects, would not adversely affect the integrity of Canna and Sanday SPA.**

*Table 8.100 In-combination year-round displacement matrix for guillemot from Canna and Sanday SPA*

| Annual | Mortality | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| **Displacement** | 1% | 2% | 3% | 4% | 5% | 10% | 20% | 30% | 50% | 80% | 100% |
| **10%** | 0 | 1 | 1 | 1 | 2 | 3 | 7 | 10 | 17 | 28 | 35 |
| **20%** | 1 | 1 | 2 | 3 | 3 | 7 | 14 | 21 | 35 | 55 | 69 |
| **30%** | 1 | 2 | 3 | 4 | 5 | 10 | 21 | 31 | 52 | 83 | 104 |
| **40%** | 1 | 3 | 4 | 6 | 7 | 14 | 28 | 42 | 69 | 111 | 138 |
| **50%** | 2 | 3 | 5 | 7 | 9 | 17 | 35 | 52 | 87 | 138 | 173 |
| **60%** | 2 | 4 | 6 | 8 | 10 | 21 | 42 | 62 | 104 | 166 | 208 |
| **70%** | 2 | 5 | 7 | 10 | 12 | 24 | 48 | 73 | 121 | 194 | 242 |
| **80%** | 3 | 6 | 8 | 11 | 14 | 28 | 55 | 83 | 138 | 222 | 277 |
| **90%** | 3 | 6 | 9 | 12 | 16 | 31 | 62 | 93 | 156 | 249 | 312 |
| **100%** | 3 | 7 | 10 | 14 | 17 | 35 | 69 | 104 | 173 | 277 | 346 |

Note: The cells show the number of birds predicted to die (rounded to the nearest integer) at a given rate of displacement and mortality. Highlighted cells are considered to be the most realistic scenario, in accordance with SNCB advice (SNCBs, 2022).

## 8.39 Buchan Ness to Collieston Coast SPA

1622. Buchan Ness to Collieston Coast SPA is located approximately 401km from the windfarm site.

### 8.39.1 Description of designation

1623. Buchan Ness to Collieston Coast SPA is a stretch of south-east facing cliff in Aberdeenshire. The 15km stretch of cliffs, formed of granite, quartzite and other rocks, runs south of Peterhead, broken only by the sandy beach of Cruden Bay. The varied coastal vegetation on the ledges and the cliff tops includes maritime heath, grassland and brackish flushes. The boundary of the SPA follows the boundaries of Bullers of Buchan Coast SSSI and Collieston to Whinnyfold Coast SSSI, and the seaward elements extend approximately 2km into the marine environment to include the seabed, water column and surface. The qualifying seabird species for the SPA comprise fulmar, herring gull, kittiwake and guillemot.

### 8.39.2 Conservation objectives

1624. The overarching conservation objectives for the site are:

- To avoid deterioration of the habitats of the qualifying species or significant disturbance to the qualifying species, thus ensuring that the integrity of the site is maintained
- To ensure for the qualifying species that the following are maintained in the long term:
  - Population of the species as a viable component of the site
  - Distribution of the species within site
  - Distribution and extent of habitats supporting the species
  - Structure, function and supporting processes of habitats supporting the species
  - No significant disturbance of the species

### 8.39.3 Assessment

1625. Two qualifying features of Buchan Ness to Collieston Coast SPA have been screened into the Appropriate Assessment (**Table 5.2**): breeding fulmar and breeding kittiwake. These species have also been screened in as qualifying components of the seabird assemblage.

#### 8.39.3.1 Fulmar

##### Status

1626. The Buchan Ness to Collieston Coast SPA breeding fulmar population was cited as 1,765 pairs, or 3,530 breeding adults, in 1998 (Furness, 2015; Stroud *et al*., 2016). Furness (2015) gave a breeding population of 1,367 pairs, or 2,734 breeding adults, in 2007. The most recent count is 826 pairs, or 1,652 breeding adults, in 2019 (JNCC, 2023a) however this count did not include the full extent of the SPA. The 2007 count is therefore used as the reference population for the assessment.

1627. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.064 (1 – 0.936; Horswill and Robinson, 2015), the expected annual mortality of the SPA population would be 175 breeding adults.

##### Functional linkage and seasonal apportionment of potential effects

1628. The mean maximum foraging range of fulmar is 542.3km (±657.9km) and the maximum foraging range is 2,736km (Woodward *et al*., 2019). The Project is located approximately 401km from Buchan Ness to Collieston Coast SPA, which means that the Project is within the mean maximum foraging range of fulmars breeding at this SPA.

##### Potential effects on the qualifying feature from the Project-alone

1629. Fulmar are considered to have low vulnerability to both collision and displacement impacts from offshore windfarms (Bradbury *et al*., 2014). This species was recorded infrequently within the windfarm site (in only five of 24 surveys, all during the breeding season (January to August; Furness, 2015) and in low numbers, with densities in four of the five months less than 0.1 birds/km$^2$. A peak density of 0.23 birds/km$^2$ was recorded in May 2022.

1630. On the basis of the low vulnerability of fulmars to potential impacts, and the low frequency and abundance of birds present at the windfarm site, it is concluded that any effects on breeding populations at Buchan Ness to Collieston Coast SPA are very unlikely, both during and outside of the breeding season.

1631. **It is therefore concluded that there would be no measurable effects on fulmar due to the project alone, and no adverse effect on the integrity of the Buchan Ness to Collieston Coast SPA is predicted.**

##### Potential effects on the qualifying feature in-combination with other projects

1632. As the Project would have no measurable effect on fulmar populations from the Buchan Ness to Collieston Coast SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of Buchan Ness to Collieston Coast SPA, when assessed in-combination with other plans or projects**.

#### 8.39.3.2 Kittiwake

##### Status

1633. The Buchan Ness to Collieston Coast SPA breeding kittiwake population was cited as 30,452 pairs, or 60,904 breeding adults, in 1998 (Furness, 2015; Stroud *et al*., 2016). Furness (2015) gave a population of 12,542 pairs, or 25,084 breeding adults, in 2007. The most recent count is 11,295 pairs, or 22,590 breeding adults, in 2019 (JNCC, 2023a) however this count did not include the full extent of the SPA. The 2007 count is therefore used as the reference population for the assessment.

1634. Based on the most recent SPA population of assumed breeding adults, and an annual adult baseline mortality rate of 0.146 (1 – 0.854; Horswill and Robinson, 2015), the expected annual mortality of the SPA population would be 4,127 breeding adults.

##### Functional linkage and seasonal apportionment of potential effects

1635. The mean maximum foraging range of kittiwake is 156.1km (±144.5km) and the maximum foraging range is 770km (Woodward *et al*., 2019). The Project is located approximately 401km from Buchan Ness to Collieston Coast SPA, which means that the Project is beyond the mean maximum foraging range +1SD of kittiwakes breeding at this SPA, but within the maximum foraging range. The maximum foraging range is a poor indicator of typical foraging behaviour. It would be expected that few birds or foraging trips will occur at this distance from the colony, and even fewer with any regularity. No impacts during the breeding season are therefore apportioned to birds breeding at this colony.

1636. Outside the breeding season, breeding kittiwakes from the SPA are assumed to range widely and to mix with kittiwakes of all ages from breeding colonies in the UK and beyond. The relevant non-breeding season reference population is the UK Western Waters and Channel BDMPS, consisting of 911,586 individuals during autumn migration (September to December) and 627,816 during spring migration (January to February) (Furness, 2015).

1637. Furness (2015) estimated that 20% of the Buchan Ness to Collieston Coast SPA breeding adults are present within the UK Western Waters and Channel BDMPS during the autumn migration period, which is 5,017 birds. During the spring migration period 30% of the population is estimated to be present, which is 7,525 birds. This represents 0.55% and 1.20% of the BDMPS population for the autumn and spring periods respectively. During autumn migration and spring migration, 0.55%, and 1.20% of impacts are therefore considered to affect birds from the SPA.

### Potential effects on the qualifying feature from the Project-alone

1638. The kittiwake qualifying feature of the Buchan Ness to Collieston Coast SPA has been screened into the Appropriate Assessment due to the potential risk of collision.

#### Operation and maintenance phase collision risk

1639. Information for collision risk on breeding adult kittiwakes belonging to the Buchan Ness to Collieston Coast SPA population is presented in **Table 8.101**. Collision estimates, calculated using the sCRM, are presented by biological season. A summary of the annual outputs and the corresponding increase in the annual baseline mortality rate is also presented. Parameters used in the sCRM were agreed with Natural England during the ETG process and are described in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES.

1640. Based on the mean collision rates, the annual total of breeding adult kittiwakes from Buchan Ness to Collieston Coast SPA at risk of collision as a result of the Project is less than one bird (0.05). This would increase the existing mortality of the SPA breeding population by 0.00%.

*Table 8.101 Predicted seasonal and annual collision mortality (Stochastic model Option 2, avoidance rate 0.993 (±0.0003)), for breeding adult kittiwakes at the windfarm site, apportioned to Buchan Ness to Collieston Coast SPA, with corresponding increases to baseline mortality of the population*

| | Breeding Season | Autumn Migration | Non-breeding/winter | Spring Migration | Annual |
|---|---|---|---|---|---|
| **Period** | **Mar-Aug** | **Sep-Dec** | **-** | **Jan-Feb** | **Jan-Dec** |
| Total collisions[1] (mean and 95% CIs) | 15.36<br>(4.17-33.87) | 8.49<br>(2.20-18.85) | - | 0.63<br>(0.00-1.46) | 24.48<br>(6.37-54.17) |
| % apportioned to the SPA | 0.00% | 0.55% | - | 1.20% | - |
| Total SPA collisions (mean and 95% CIs) | 0.00<br>(0.00-0.00) | 0.05<br>(0.01-0.10) | - | 0.01<br>(0.00-0.02) | 0.05<br>(0.01-0.12) |
| Mortality increase[2] (mean and 95% CIs) | 0.00%<br>(0.00-0.00%) | 0.00%<br>(0.00-0.00%) | - | 0.00%<br>(0.00-0.00%) | 0.00%<br>(0.00-0.00%) |

[1] Breeding season collision values reduced to 94.1% of modelled value to reflect proportion of adult birds recorded during site surveys
[2] Assuming predicted annual SPA mortality of 3,662 birds (25,084 x 0.146)

1641. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that no detectable changes in mortality rates would occur on this population for the mean monthly collision estimates for the Project, or for the upper 95% CI collision estimate.

1642. **It is concluded that based on predicted kittiwake mortality due to collision at the windfarm site there is no potential for the Project to have an adverse effect on the integrity of the Buchan Ness to Collieston Coast SPA.**

1643. The confidence in the assessment is high. The evidence used to define the CRM input parameters presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** as part of the ES submitted alongside this RIAA is of high applicability and quality. Whilst there is uncertainty around some of the input parameters (e.g. avoidance rate), the rates selected are considered to be sufficiently precautionary based on expert opinion to provide confidence that collision rates are not underestimated.

### Potential effects on the qualifying feature in-combination with other projects

1644. As the Project would have no measurable effect on kittiwake populations from the Buchan Ness to Collieston Coast SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of Buchan Ness to Collieston Coast SPA, when assessed in-combination with other plans or projects**.

## 8.40 Mingulay and Berneray SPA

1645. Mingulay and Berneray SPA is located approximately 406km from the windfarm site.

### 8.40.1 Description of designation

1646. Mingulay and Berneray SPA consists of two adjacent islands at the southern end of the Outer Hebrides. They have a maritime flora and predominantly cliffed, rocky coastlines. The boundary of the SPA overlaps with the boundary of Mingulay and Berneray SSSI, and the seaward elements extend approximately 2km into the marine environment to include the seabed, water column and surface. The qualifying seabird species of the SPA comprise fulmar, kittiwake, guillemot and puffin.

### 8.40.2 Conservation objectives

1647. The overarching conservation objectives for the site are:

- To avoid deterioration of the habitats of the qualifying species or significant disturbance to the qualifying species, thus ensuring that the integrity of the site is maintained
- To ensure for the qualifying species that the following are maintained in the long term:
    - Population of the species as a viable component of the site
    - Distribution of the species within site
    - Distribution and extent of habitats supporting the species
    - Structure, function and supporting processes of habitats supporting the species
    - No significant disturbance of the species

### 8.40.3 Assessment

1648. The qualifying features of Mingulay and Berneray SPA screened into the Appropriate Assessment are listed in **Table 5.2**. These are breeding fulmar, breeding guillemot and breeding razorbill. The seabird assemblage has also been screened in for these species.

#### 8.40.3.1 Fulmar

##### Status

1649. The Mingulay and Berneray SPA breeding fulmar population is cited as 12,500 pairs, or 25,000 breeding adults, in 1994 (Furness, 2015; Stroud *et al*., 2016). Furness (2015) gave the breeding population of 9,046 pairs, or 18,092 breeding adults, in 2009. The most recent counts comprise 6,292 pairs (AOS), or 12,584 individuals, on Mingulay (2016) and 754 pairs (AOS), or 1,508 individuals, on Berneray (2021) (JNCC, 2023a), giving a combined total of 14,092 assumed breeding adults. This is used as the reference population for the assessment.

1650. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.064 (1 – 0.936; Horswill and Robinson, 2015), 902 breeding adults from the SPA population would be expected to die each year.

##### Functional linkage and seasonal apportionment of potential effects

1651. The mean maximum foraging range of fulmar is 542.3km (±657.9km) and the maximum foraging range is 2,736km (Woodward *et al*., 2019). The Project is located approximately 406km from Mingulay and Berneray SPA, which means that the Project is within the mean maximum foraging range of fulmars breeding at this SPA.

##### Potential effects on the qualifying feature from the Project-alone

1652. Fulmar are considered to have low vulnerability to both collision and displacement impacts from offshore windfarms (Bradbury *et al*., 2014). This species was recorded infrequently within the windfarm site (in only five of 24 surveys, all during the breeding season (January to August; Furness, 2015)) and in low numbers, with densities in four of the five months less than 0.1 birds/km$^2$. A peak density of 0.23 birds/km$^2$ was recorded in May 2022.

1653. On the basis of the low vulnerability of fulmars to potential impacts, and the low frequency and abundance of birds present at the windfarm site, it is concluded that any effects on breeding populations at Mingulay and Berneray SPA are very unlikely, both during and outside of the breeding season.

1654. **It is therefore concluded that there would be no measurable effects on fulmar due to the project alone, and no adverse effect on the integrity of the Mingulay and Berneray SPA is predicted.**

##### Potential effects on the qualifying feature in-combination with other projects

1655. As the Project would have no measurable effect on fulmar populations from the Mingulay and Berneray SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of Mingulay and Berneray SPA, when assessed in-combination with other plans or projects.**

### 8.40.3.2 Guillemot

#### Status

1656. The Mingulay and Berneray SPA breeding guillemot population is cited as 20,703 pairs, or 41,406 breeding adults, in 1994 (Furness, 2015, Stroud *et al*., 2016). Furness (2015) gave the breeding population of 13,527 pairs, or 27,054 breeding adults, in 2009. The most recent counts comprise 16,802 individuals on Mingulay (2016) and 18,393 individuals on Berneray (2021) (JNCC, 2023a), giving a combined total of 35,195 assumed breeding adults. This is used as the reference population for the assessment.

1657. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.061 (1 – 0.939; Horswill and Robinson 2015), 2,147 breeding adults from the SPA population would be expected to die each year.

#### Functional linkage and seasonal apportionment of potential effects

1658. The mean maximum foraging range of guillemot is 73.2km (±80.5km) and the maximum foraging range is 338km (Woodward *et al*., 2019). The Project is located approximately 406km from Mingulay and Berneray SPA, which means that the Project is beyond the maximum foraging range for guillemots from the SPA. No impacts during the breeding season from the Project are therefore apportioned to guillemots breeding at this SPA.

1659. Outside the breeding season, breeding guillemots from the SPA are assumed to range widely and to mix with guillemots of all ages from breeding colonies in the UK and beyond. The relevant non-breeding season reference population is the UK Western Waters BDMPS, consisting of 1,139,220 individuals (August to February) (Furness, 2015).

1660. Furness (2015) estimated that 95% of the Mingulay and Berneray SPA breeding adults are present within the UK Western Waters BDMPS during the non-breeding season, which is 25,701 birds. This represents 2.3% of the BDMPS population for this period (1,139,220). It is therefore assumed that 2.3% of guillemots present at the Project site are breeding adults from Mingulay and Berneray SPA.

## Potential effects on the qualifying feature

### Operation and maintenance phase disturbance/displacement/barrier effects

*Project-alone*

1661. The mean peak abundance of guillemots present within the windfarm site and 2km buffer during the non-breeding season was 8,315 (6,085-12,047) individuals (refer to **Appendix 12.1** of the ES). Of these, 191 birds (140-277) were likely to be breeding adults from the Mingulay and Berneray SPA.

1662. **Table 8.102** sets out the predicted impacts on guillemots from Mingulay and Berneray SPA during the non-breeding season. Displacement rates of 30% to 70% are considered for this species, along with a range of mortality rates of 1% to 10% of displaced birds (UK SNCBs, 2017).

1663. The available evidence suggests that the upper ranges of these displacement and mortality rates may be excessively precautionary. Whilst it is evident that guillemots and razorbills tend to be displaced by OWFs they do not avoid them completely, and displacement rates vary between sites. MacArthur Green, (2019b) found that, on average densities within OWFs were around half of the density found in the habitats around the OWF. At some OWFs there was also displacement of birds from a buffer zone surrounding it. The size of the buffer zone varied between OWFs and was generally less than 2km, with auk density increasing across the buffer zone as distance from turbines increased, up to the density in the wider area. Another recent review (APEM, 2022) found that the current displacement rates suggested by the SNCBs for guillemot and razorbill (30% to 70%) did not account for the quality of or confidence in the studies which were used to inform this position, and that studies where no significant effects were recorded were not accounted for during the provision of the advice. APEM (2022) suggested that in the case of Hornsea Project Four, an evidence-based displacement rate of 50% was appropriate. However, the study also recognised that larger displacement effects were possible.

1664. Mortality due to displacement could arise due to increased energy costs and/or decreased energy intake, if displacement results in increased flying time to avoid OWFs, and/or increased bird densities and competition for prey in areas of unimpacted habitat outside the OWF. Given that UK OWFs in the Irish and Celtic Seas represent a very small proportion of the available foraging habitat for guillemots and razorbills within UK Western Waters, the increase in density of auks outside OWFs due to displacement is likely to be negligible (MacArthur Green, 2019b). It is considered unlikely that the mortality rate due to displacement would be as high as the empirically estimated 6.1% annual mortality for adult guillemots that occurs due to the combination of 'natural' factors and anthropogenic activities (Horswill and Robinson, 2015). Indeed, it may be much lower; MacArthur Green (2019b) recommended precautionary rates of 50% displacement for guillemot and 1% mortality of displaced birds based on a review of available evidence. Modelling undertaken by APEM (2022) found that in the case of Hornsea Project Four, a mortality rate of 1% for displaced auks was considered to be precautionary, and that this may be an overestimate given that Hornsea Project Four is located towards the upper end of the mean maximum foraging range for guillemot from the closest designated sites. It is also noted that in the recent decision on Hornsea Project Four, the SoS determined that a rate of 70% displacement/2% mortality was appropriate to inform the assessment of effects on guillemots from Flamborough and Filey Coast SPA (DESNZ, 2023b).

1665. The full ranges of recommended displacement and mortality effects are considered by the assessment, along with evidence-based displacement and mortality rates of 50% and 1%, respectively (APEM, 2022; MacArthur Green, 2019b).

*Table 8.102 Guillemot – predicted operation and maintenance phase displacement and mortality from Mingulay and Berneray SPA*

| Mean peak abundance estimate type | Mean peak abundance estimate | Number of Mingulay and Bernaeray SPA breeding adults present (non-breeding season)[1] | Annual mortality range[2] | Annual baseline mortality increase range[3] |
|---|---|---|---|---|
| Upper 95% CI | 12,047 | 277 | 1-19 | 0.04-0.90% |
| Mean | 8,315 | 191 | 1-13 | 0.03-0.62% |
| Lower 95% CI | 6,085 | 140 | 0-10 | 0.02-0.46% |

[1] Assumes 2.3% of birds present during the non-breeding season are Mingluay and Berneray SPA breeding adults

[2] Assumes displacement rates of 30-70% and mortality rates of 1-10%

[3] Background mortality rate of 6.1% (Horswill and Robinson, 2015)

1666. Based on the mean peak abundances, the annual total of guillemots from the Mingulay and Berneray SPA at risk of displacement is 191 birds (**Table 8.102**). At displacement rates of 30% to 70%, and mortality rates of 1% to 10% for displaced birds, 1 to 13 SPA breeding adults would be predicted to die each year due to displacement from the Project.

1667. Assuming a displacement rate of 70% and a mortality rate of 10% of displaced birds, annual mortality within the SPA breeding adult population would increase by 0.62%. Using an evidence-based displacement rate of 50%, and a mortality rate for displaced birds of 1%, annual mortality in the population would instead increase by 0.04% (<1 bird).

1668. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that no detectable changes in mortality rates would occur under any combination of displacement and mortality rates when the mean peak abundance estimate assessments are considered. No increases of over 1% are also predicted if the upper 95% CIs for mean peak abundances are used as inputs to the assessment.

1669. **It is concluded that predicted guillemot mortality due to operational phase displacement for the project alone would not adversely affect the integrity of the Mingulay and Berneray SPA.**

*In-combination*

1670. The in-combination assessment for guillemots from Mingulay and Berneray SPA has been undertaken in accordance with the approach presented in **Section 8.1**. The total population apportioned to Mingulay and Berneray SPA at risk of displacement is estimated to be 1,139 breeding adults (**Appendix 12.1** of the ES). Annual in-combination displacement and mortality rates for birds from Mingulay and Berneray SPA are presented in **Table 8.103**.

1671. Assuming a displacement rate of 70% and a mortality rate of 10% of displaced birds, 80 breeding adult SPA birds would be lost to displacement annually. Including additional apportioned mortality from the Morlais and Holyhead tidal projects (total 0.73 birds), this would increase the existing mortality within the SPA population (2,147 breeding adult birds per year) by 3.75%. Using an evidence-based displacement rate of 50%, and a mortality rate for displaced birds of 1%, the annual in-combination displacement mortality would be six birds. This would increase the existing mortality within this population by 0.30%. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that detectable changes in mortality rates will not occur due to the level or mortality predicted if the more realistic rates for mortality are used.

1672. **It is concluded that predicted guillemot mortality due to operational phase displacement impacts of the Project in-combination with other projects, would not adversely affect the integrity of Mingulay and Berneray SPA.**

*Table 8.103 In-combination year-round displacement matrix for guillemot from Mingulay and Berneray SPA*

| Annual | Mortality | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Displacement | 1% | 2% | 3% | 4% | 5% | 10% | 20% | 30% | 50% | 80% | 100% |
| 10% | 1 | 2 | 3 | 5 | 6 | 11 | 23 | 34 | 57 | 91 | 114 |
| 20% | 2 | 5 | 7 | 9 | 11 | 23 | 46 | 68 | 114 | 182 | 228 |
| 30% | 3 | 7 | 10 | 14 | 17 | 34 | 68 | 102 | 171 | 273 | 342 |
| 40% | 5 | 9 | 14 | 18 | 23 | 46 | 91 | 137 | 228 | 364 | 455 |
| 50% | 6 | 11 | 17 | 23 | 28 | 57 | 114 | 171 | 285 | 455 | 569 |
| 60% | 7 | 14 | 20 | 27 | 34 | 68 | 137 | 205 | 342 | 547 | 683 |
| 70% | 8 | 16 | 24 | 32 | 40 | 80 | 159 | 239 | 399 | 638 | 797 |
| 80% | 9 | 18 | 27 | 36 | 46 | 91 | 182 | 273 | 455 | 729 | 911 |
| 90% | 10 | 20 | 31 | 41 | 51 | 102 | 205 | 307 | 512 | 820 | 1025 |
| 100% | 11 | 23 | 34 | 46 | 57 | 114 | 228 | 342 | 569 | 911 | 1139 |

Note: The cells show the number of birds predicted to die (rounded to the nearest integer) at a given rate of displacement and mortality. Highlighted cells are considered to be the most realistic scenario, in accordance with SNCB advice (SNCBs, 2022).

#### 8.40.3.3 Razorbill

##### Status

1673. The Mingulay and Berneray SPA breeding razorbill population is cited as 11,323 pairs, or 22,646 breeding adults, in 1985 (Furness, 2015, Stroud *et al*., 2016). Furness (2015) gave the breeding population of 10,111 pairs, or 20,222 breeding adults, in 2009. The most recent counts comprise of 11,453 birds on Mingulay (2017) and 8,524 birds in Berneray (2021) (JNCC, 2023a), giving a combined total of 19,977 breeding adults. This is used as the reference population for the assessment.

1674. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.105 (1 – 0.895; Horswill and Robinson 2015), 2,098 breeding adults from the SPA population would be expected to die each year.

##### Functional linkage and seasonal apportionment of potential effects

1675. The mean maximum foraging range of razorbill is 88.7km (±75.9km) and the maximum foraging range is 313km (Woodward *et al*., 2019). The Project is located approximately 406km from Mingulay and Berneray SPA, which means that the Project is beyond the maximum foraging range for razorbills from the SPA. No impacts during the breeding season from the Project are therefore apportioned to razorbills breeding at this SPA.

1676. Outside the breeding season, breeding razorbills from the SPA are assumed to range widely and to mix with razorbills of all ages from breeding colonies in the UK and beyond. The relevant non-breeding season reference population is the UK Western Waters BDMPS, consisting of 606,914 individuals during autumn and spring passage periods (August to October and January to March), and 341,422 individuals during winter (November and December) (Furness, 2015). During autumn and spring migration, 98% of the SPA breeding adults (19,818 individuals based on the 2009 population estimate) are assumed to be present in the BDMPS, representing 3.3% of the BDMPS population (606,914 individuals of all ages). During the winter season, 40% of the SPA breeding adults (8,089 individuals based on the 2009 population estimate) are assumed to be present in the BDMPS, representing 2.4% of the BDMPS population (341,422 individuals of all ages). These percentages (i.e. 3.3% and 2.4%) are the proportions of birds present at the windfarm site presumed to originate from the Mingulay and Berneray SPA during the relevant seasons.

## Potential effects on the qualifying feature

### Operation and maintenance phase disturbance/displacement/barrier effects

*Project-alone*

1677. The year-round mean peak abundance of razorbills present within the windfarm site and 2km buffer was 1,979 (703-3,552) individuals (refer to **Appendix 12.1** of the ES). Of these, 51 birds (21-86) were likely to be breeding adults from the Mingulay and Berneray SPA.

1678. **Table 8.104** sets out the predicted impacts on razorbills from Mingulay and Berneray SPA. Displacement rates of 30% to 70% are considered for this species, along with a range of mortality rates of 1% to 10% of displaced birds (UK SNCBs, 2017).

1679. The available evidence suggests that the upper ranges of these displacement and mortality rates may be excessively precautionary. Whilst it is true that guillemots and razorbills tend to be displaced by OWFs they do not avoid them completely, and displacement rates vary between sites. MacArthur Green (2019b) concluded that, on average, densities within OWFs were around half of the density found in the habitats around the OWF. At some OWFs there was also displacement of birds from a buffer zone surrounding it. The size of the buffer zone varied between OWFs and was generally less than 2km, with auk density increasing across the buffer zone as distance from turbines increased, up to the density in the wider area. Another recent review (APEM, 2022) found that the current displacement rates suggested by the SNCBs for guillemot and razorbill (30% to 70%) did not account for the quality of or confidence in the studies which were used to inform this position, and that studies where no significant effects were recorded were not accounted for during the provision of the advice. APEM (2022) suggested that in the case of Hornsea Project Four, an evidence-based displacement rate of 50% was appropriate. However, the study also recognised that larger displacement effects are possible.

1680. Mortality due to displacement could arise due to increased energy costs and/or decreased energy intake, if displacement results in increased flying time to avoid OWFs, and/or increased bird densities and competition for prey in areas of unimpacted habitat outside the OWF. Given that UK OWFs in the Irish and Celtic Seas represent a very small proportion of the available foraging habitat for guillemots and razorbills within UK Western Waters, the increase in density of auks outside OWFs due to displacement is likely to be negligible (MacArthur Green, 2019b). It is considered unlikely that the mortality rate due to displacement would be as high as the empirically estimated 10.5% annual mortality for adult razorbills that occurs due to the combination of ‘natural’ factors and anthropogenic activities (Horswill and Robinson, 2015). Indeed, it may be much lower; MacArthur Green (2019b) recommended

precautionary rates of 50% displacement for auks and 1% mortality of displaced birds based on a review of available evidence. Modelling undertaken by APEM (2022) found that in the case of Hornsea Project Four, a mortality rate of 1% for displaced auks was considered to be precautionary, and that this may be an overestimate given that Hornsea Project Four is located towards the upper end of the mean maximum foraging range for guillemot from the closest designated sites.

1681. The full ranges of recommended displacement and mortality effects are considered by the assessment, along with evidence-based displacement and mortality rates of 50% and 1%, respectively (APEM, 2022; MacArthur Green, 2019b).

*Table 8.104 Razorbill – predicted operation and maintenance phase displacement and mortality from Mingulay and Berneray SPA*

| Mean peak abundance estimate type | Mean peak abundance estimate by season[1] | Number of Mingulay and Berneray SPA breeding adults present by season)[2] | Annual mortality range[3] | Annual baseline mortality increase range[4] |
|---|---|---|---|---|
| Upper 95% CI | 605 (b)<br>1,070 (aut)<br>1,297 (win)<br>580 (spr)<br>3,552 (year round) | 0 (b)<br>35 (aut)<br>31 (win)<br>19 (spr)<br>86 (year round) | 0-6 (0) | 0.01-0.29% (0.02%) |
| Mean | 252 (b)<br>694 (aut)<br>651 (win)<br>381 (spr)<br>1,979 (year round) | 0 (b)<br>23 (aut)<br>16 (win)<br>13 (spr)<br>51 (year round) | 0-4 (0) | 0.01-0.17% (0.01%) |
| Lower 95% CI | 21 (b)<br>309 (aut)<br>159 (win)<br>214 (spr)<br>703 (year round) | 0 (b)<br>3 (aut)<br>4 (win)<br>7 (spr)<br>21 (year round) | 0-1 (0) | 0.00-0.07% (0.01%) |

[1] Breeding season = b, autumn migration season = aut, winter season = win, spring migration season = spr

[2] Assumes breeding adult apportioning of 0.0% (breeding season), 3.3% (spring and autumn migration) and 2.4% (winter) to Mingulay and Berneray SPA.

[3] Assumes displacement rates of 30-70% and mortality rates of 1-10%. Evidence-based estimates assuming a 50% displacement rate and 1% mortality of displaced birds are presented in parentheses.

[4] Background mortality rate of 10.5% (Horswill and Robinson, 2015)

1682. Based on the mean peak abundances, the annual total of razorbills from the Mingulay and Berneray SPA at risk of displacement is 51 birds (**Table 8.104**). At displacement rates of 30% to 70%, and mortality rates of 1% to 10% for displaced birds, 0 to 4 breeding adults from this SPA would be predicted to die each year due to displacement from the Project.

1683. Assuming a displacement rate of 70% and a mortality rate of 10% of displaced birds, annual mortality within the SPA breeding adult population would increase by 0.17%. Using an evidence-based displacement rate of 50%, and a mortality rate for displaced birds of 1%, annual mortality in the population would instead increase by 0.01% (<1 bird).

1684. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that no detectable changes in mortality rates would occur under almost any combination of displacement and mortality rates when the mean peak abundance estimate assessments are considered. This would be the case even when upper 95% CIs are considered.

1685. **It is concluded that predicted razorbill mortality due to operational phase displacement for the project alone would not adversely affect the integrity of the Mingulay and Berneray SPA.**

1686. The confidence in the assessment is high for several reasons. Firstly, the evidence used to inform the evidence-based displacement rates is of high applicability and quality (based on the criteria discussed in **Chapter 12 Offshore Ornithology** of the ES). Whilst there is limited available evidence to inform mortality rates, 1% is considered to be sufficiently precautionary based on expert opinion. This species is not regarded as being highly specialised in its habitat requirements (Bradbury *et al*., 2014; Furness and Wade, 2012; Garthe and Hüppop, 2004), and it is therefore anticipated that displaced birds will find alternative habitat in the vast majority of cases. Finally, the conclusion of the assessment is the same irrespective of whether the mean or 95% upper CI mean peak abundances are used to calculate potential mortality and increases in the baseline mortality rate of the background population, provided the evidence-based displacement and mortality rates are used.

*In-combination*

1687. The in-combination assessment for razorbills from Mingulay and Berneray SPA has been undertaken in accordance with the approach presented in **Section 8.1**. The total population apportioned to the SPA at risk of displacement is estimated to be 328 breeding adults (**Appendix 12.1** of the ES). Annual in-combination displacement and mortality rates for birds from Mingulay and Berneray SPA are presented in **Table 8.105**.

1688. Assuming a displacement rate of 70% and a mortality rate of 10% of displaced birds, 23 breeding adult SPA birds would be lost to displacement annually. Including additional apportioned mortality from the Morlais and Holyhead tidal projects (total 0.49 birds), this would increase the existing mortality within the SPA population (2,098 breeding adult birds per year) by 1.12%. Using an evidence-based displacement rate of 50%, and a mortality rate for displaced birds of 1%, the annual in-combination mortality would be 2 birds. This would increase the existing mortality within this population by 0.10%. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that detectable changes in mortality rates will not occur due to the level or mortality predicted if the more realistic rates for mortality are used.

1689. **It is concluded that predicted razorbill mortality due to of operational phase displacement impacts of the Project in-combination with other projects, would not adversely affect the integrity of Mingulay and Berneray SPA.**

*Table 8.105 In-combination year-round displacement matrix for razorbill from Mingulay and Berneray SPA*

| Annual | 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 |

Note: The cells show the number of birds predicted to die (rounded to the nearest integer) at a given rate of displacement and mortality. Highlighted cells are considered to be the most realistic scenario, in accordance with SNCB advice (SNCBs, 2022).

## 8.41 Troup, Pennan and Lion's Heads SPA

1690. Troup, Pennan and Lion's Heads SPA is located approximately 433km from the windfarm site.

### 8.41.1 Description of designation

1691. Troup, Pennan and Lion's Heads SPA is a 9km stretch of sea cliffs along the Aberdeenshire coast which support large colonies of breeding seabirds. The seaward extension of the SPA extends 2km into the marine environment and includes the seabed, water column and surface. Seabirds included within the designation (fulmar, kittiwake, herring gull and guillemot) feed both inside and outside the SPA in nearby waters, as well as more distantly in the wider North Sea.

### 8.41.2 Conservation objectives

1692. The overarching conservation objectives for the site are:

- To avoid deterioration of the habitats of the qualifying species or significant disturbance to the qualifying species, thus ensuring that the integrity of the site is maintained
- To ensure for the qualifying species that the following are maintained in the long term:
  - Population of the species as a viable component of the site
  - Distribution of the species within site
  - Distribution and extent of habitats supporting the species
  - Structure, function and supporting processes of habitats supporting the species
  - No significant disturbance of the species

### 8.41.3 Assessment

1693. The qualifying features of Troup, Pennan and Lion's Heads SPA screened into the Appropriate Assessment are listed in **Table 5.2**. These are breeding fulmar and breeding kittiwake. The seabird assemblage has also been screened in for these species.

#### 8.41.3.1 Fulmar

##### Status

1694. The Troup, Pennan and Lion's Heads SPA breeding fulmar population at the time of the site classification was cited as 4,400 pairs, or 8,800 breeding adults, in 1995 (SNH, 2009b). Furness (2015) gave a breeding population of 1,795 pairs, or 3,590 breeding adults, in 2007. The most recent available count is 1,894 pairs (AOS), or 3,788 breeding adults, in 2017 (JNCC, 2023a); this is used as the reference population for the assessment.

1695. Based on the most recent SPA population of assumed breeding adults and an annual baseline adult mortality rate of 0.064 (1 – 0.936; Horswill and Robinson 2015), 242 breeding adults from this SPA population would be expected to die each year.

##### Functional linkage and seasonal apportionment of potential effects

1696. The mean maximum foraging range of fulmar is 542.3km (±657.9km) and the maximum foraging range is 2,736km (Woodward *et al*., 2019). The Project located is approximately 433km from Troup, Pennan and Lion's Heads SPA, which means that the Project is within the mean maximum foraging range of fulmars breeding at this SPA.

##### Potential effects on the qualifying feature from the Project-alone

1697. Fulmar is considered to have low vulnerability to both collision and displacement impacts from offshore windfarms (Bradbury *et al*., 2014). This species was recorded infrequently within the windfarm site (in only five of 24 surveys, all during the breeding season (January to August; Furness, 2015) and in low numbers, with densities in four of the five months less than 0.1 birds/km$^2$. A peak density of 0.23 birds/km$^2$ was recorded in May 2022.

1698. On the basis of the low vulnerability of fulmars to potential impacts, and the low frequency and abundance of birds present at the windfarm site, it is concluded that any effects on breeding populations at Troup, Pennan and Lion's Heads SPA are very unlikely, both during and outside of the breeding season.

1699. **It is therefore concluded that there would be no measurable effects on fulmar due to the project alone, and no adverse effect on the integrity of the Troup, Pennan and Lion's Heads SPA is predicted.**

##### Potential effects on the qualifying feature in-combination with other projects

1700. As the Project would have no measurable effect on fulmar populations from the Troup, Pennan and Lion's Heads SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that**

**there would be no adverse effect on the integrity of Troup, Pennan and Lion's Heads SPA, when assessed in-combination with other plans or projects.**

### 8.41.3.2 Kittiwake

#### Status

1701. The Troup, Pennan and Lion's Heads SPA breeding kittiwake population at classification was cited as 31,600 pairs, or 63,200 breeding adults, in 1995 (SNH, 2009b). Furness (2015) gave a breeding population of 14,896 breeding pairs, or 29,792 adults, for 2007. The most recent available count is 10,616 pairs (AON), or 21,232 breeding adults, in 2017 (JNCC, 2023a); this is used as the reference population for the assessment.

1702. Based on the most recent SPA population of assumed breeding adults and an annual baseline adult mortality rate of 0.146 (1 – 0.854; Horswill and Robinson 2015), 3,100 breeding adults from this SPA population would be expected to die each year.

#### Functional linkage and seasonal apportionment of potential effects

1703. The mean maximum foraging range of kittiwake is 156.1km (±144.5km) and the maximum foraging range is 770km (Woodward *et al*., 2019). The Project is approximately 433km from Troup, Pennan and Lion's Heads SPA, which means that the Project is beyond the mean maximum foraging range +1SD of kittiwakes breeding at this SPA, but within the maximum foraging range.

1704. The maximum foraging range is a poor indicator of typical foraging behaviour. It would be expected that few birds or foraging trips will occur at this distance from the colony, and even fewer with any regularity. No impacts during the breeding season are therefore apportioned to birds breeding at this colony.

1705. Outside the breeding season, breeding kittiwakes from the SPA are assumed to range widely and to mix with kittiwakes of all ages from breeding colonies in the UK and beyond. The relevant non-breeding season reference population is the UK Western Waters and Channel BDMPS, consisting of 911,586 individuals during autumn migration (September to December) and 627,816 during spring migration (January to February) (Furness, 2015).

1706. Furness (2015) estimated that 20% of the Troup, Pennan and Lion's Heads SPA breeding adults are present within the UK Western Waters and Channel BDMPS during the autumn migration period, which is 5,994 birds. During the spring migration period 30% of the population is estimated to be present, which is 8,992 birds. This represents 0.66% and 1.43% of the BDMPS population for the autumn and spring periods respectively. During autumn migration and spring migration, 0.66%, and 1.43% of impacts are therefore considered to affect birds from the SPA.

## Potential effects on the qualifying feature from the Project-alone

1707. The kittiwake qualifying feature of the Troup, Pennan and Lion’s Heads SPA has been screened into the Appropriate Assessment due to the potential risk of collision.

### Operation and maintenance phase collision risk

1708. Information for collision risk on breeding adult kittiwakes belonging to the Troup, Pennan and Lion’s Heads SPA population is presented in **Table 8.106**. Collision estimates, calculated using the sCRM, are presented by biological season. A summary of the annual outputs and the corresponding increase in the annual baseline mortality rate is also presented. Parameters used in the sCRM were agreed with Natural England during the ETG process and are described in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES.

1709. Based on the mean collision rates, the annual total of breeding adult kittiwakes from Troup, Pennan and Lion’s Heads SPA at risk of collision as a result of the Project is less than one bird (0.06). This would increase the existing mortality of the SPA breeding population by 0.00%.

*Table 8.106 Predicted seasonal and annual collision mortality (Stochastic model Option 2, avoidance rate 0.993 (±0.0003)), for breeding adult kittiwakes at the windfarm site, apportioned to Troup, Pennan and Lion's Heads SPA, with corresponding increases to baseline mortality of the population*

| Period | Breeding Season<br>Mar-Aug | Autumn Migration<br>Sep-Dec | Non-breeding/winter<br>- | Spring Migration<br>Jan-Feb | Annual<br>Jan-Dec |
|---|---|---|---|---|---|
| Total collisions[1] (mean and 95% CIs) | 15.36<br>(4.17-33.87) | 8.49<br>(2.20-18.85) | - | 0.63<br>(0.00-1.46) | 24.48<br>(6.37-54.17) |
| % apportioned to the SPA | 0.00% | 0.66% | - | 1.43% | - |
| Total SPA collisions (mean and 95% CIs) | 0.00<br>(0.00-0.00) | 0.06<br>(0.01-0.12) | - | 0.01<br>(0.00-0.02) | 0.06<br>(0.01-0.14) |
| Mortality increase[2] (mean and 95% CIs) | 0.00%<br>(0.00-0.00%) | 0.00%<br>(0.00-0.00%) | - | 0.00%<br>(0.00-0.00%) | 0.00%<br>(0.00-0.00%) |

[1] Breeding season collision values reduced to 94.1% of modelled value to reflect proportion of adult birds recorded during site surveys
[2] Assuming predicted annual SPA mortality of 3,100 birds (21,232 x 0.146)

1710. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that no detectable changes in mortality rates would occur on this population for the mean monthly collision estimates for the Project, or for the upper 95% CI collision estimate.

1711. **It is concluded that based on predicted kittiwake mortality due to collision at the windfarm site there is no potential for the Project to have an adverse effect on the integrity of the Troup, Pennan and Lion's Heads SPA.**

1712. The confidence in the assessment is high. The evidence used to define the CRM input parameters presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** as part of the ES submitted alongside this RIAA is of high applicability and quality. Whilst there is uncertainty around some of the input parameters (e.g. avoidance rate), the rates selected are considered to be sufficiently precautionary based on expert opinion to provide confidence that collision rates are not underestimated.

### Potential effects on the qualifying feature in-combination with other projects

1713. As the Project would have no measurable effect on kittiwake populations from the Troup, Pennan and Lion's Heads SPA, there would be no contribution to any in-combination effects on this feature**. Therefore, it is concluded that there would be no adverse effect on the integrity of Troup, Pennan and Lion's Heads SPA, when assessed in-combination with other plans or projects**.

# 8.42 Isles of Scilly SPA

1714. Isles of Scilly SPA is located approximately 459km (straight line distance) or 478km (across sea) from the windfarm site.

## 8.42.1 Description of designation

1715. The Isles of Scilly is an archipelago of over 200 low-lying granite islands and rocks situated in the South-West Approaches, 45km south-west of Land's End. The SPA supports a breeding seabird assemblage of European importance. The isolated nature of the islands with their low levels of disturbance and predation, makes them particularly suitable for nesting seabirds including European storm-petrel and lesser black-backed gull.

## 8.42.2 Conservation objectives

1716. The SPA's conservation objectives are to ensure that, subject to natural change, the integrity of the site is maintained or restored as appropriate, and that the site contributes to achieving the aims of the Wild Birds Directive, by maintaining or restoring:

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

## 8.42.3 Assessment

1717. The qualifying features of the Isles of Scilly SPA screened into the Appropriate Assessment are listed in **Table 5.2**. These are breeding shag, breeding lesser black-backed gull, breeding great black-backed gull, and the seabird assemblage.

### 8.42.3.1 Shag

#### Status

1718. The Isles of Scilly SPA shag population was cited as 2,028 individuals in 2015-16 (Natural England, 2020e). This is used as the reference population for the assessment since more recent counts on the SMP database (JNCC, 2023a) do not encompass all of the SPA breeding sites.

1719. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.142 (1 – 0.858; Horswill and Robinson, 2015), the expected annual mortality from the SPA population would be 288 breeding adults.

### Functional linkage and seasonal apportionment of potential effects

1720. The mean maximum foraging range of shag is 13.2km (±10.5km) and the maximum foraging range is 46km (Woodward *et al*., 2019). The Project is located approximately 478km from Isles of Scilly SPA, which means that the Project is beyond the maximum foraging range for shags from the SPA. No impacts during the breeding season from the Project are therefore apportioned to shags breeding at this SPA.

1721. Outside the breeding season, breeding shags from the SPA are not tied to the colony and therefore have the potential to mix with birds of all ages from breeding colonies in the UK and beyond. However, Furness (2015) stated that adult shags show only limited migration, with evidence to suggest that the majority of adults move less than 50km from their breeding colony. Given the distance of the windfarm site from the SPA (i.e. c.478km) and the low numbers of birds recorded within the Project area (mean peak density 0.02 birds/km$^2$ / <4 birds within the windfarm site and 2km buffer during the non-breeding period), it is concluded that it is very unlikely that breeding adult shags from the Isles of Scilly SPA will occur at the windfarm site. Accordingly, no impacts during the non-breeding season from the Project are apportioned to shags breeding at this SPA.

### Potential effects on the qualifying feature from the Project-alone

1722. No effects on shags from Isles of Scilly SPA are predicted. **Therefore, it is concluded that there would be no adverse effect on the integrity of the Isles of Scilly SPA for the Project-alone**.

### Potential effects on the qualifying feature in-combination with other projects

1723. As the Project would have no measurable effect on shag populations from the Isles of Scilly SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of Isles of Scilly SPA, when assessed in-combination with other plans or projects**.

#### 8.42.3.2 Lesser black-backed gull

### Status

1724. The Isles of Scilly SPA lesser black-backed gull population was cited as 4,922 individuals in 2015-16 (Natural England 2020d). The most recent complete

SMP count was 2,793 pairs (AON), or 5,586 breeding adults, in 2015 (JNCC, 2023a); this is used as the reference population for the assessment.

1725. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.115 (1 – 0.885; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 642 breeding adults.

### Functional linkage and seasonal apportionment of potential effects

1726. The mean maximum foraging range of lesser black-backed gull is 127km (±109km) and the maximum foraging range is 533km (Woodward *et al*., 2019). The Project is located approximately 459km from Isles of Scilly SPA, which means that the Project is beyond the mean maximum foraging range +1SD of breeding lesser black-backed gulls from the SPA, but within the maximum foraging range. The maximum foraging range is a poor indicator of typical foraging behaviour. It would be expected that few birds or foraging trips will occur at this distance from the colony, and even fewer with any regularity. No impacts during the breeding season are therefore apportioned to birds breeding at this colony.

1727. Outside the breeding season, breeding lesser black-backed gulls from the SPA are assumed to range widely and to mix with lesser black-backed gulls of all ages from breeding colonies in the UK and beyond. The relevant non-breeding season reference population is the UK Western Waters BDMPS, consisting of 163,304 individuals during spring and autumn migration (March and September to October) and 41,159 during winter (November to February) (Furness, 2015).

1728. Furness (2015) estimated that 90% of the Isles of Scilly SPA breeding adults are present within the UK Western Waters and Channel BDMPS during the autumn and spring migration periods, which is 6,120 birds. During the winter period 20% of the population is estimated to be present, which is 1,360 birds. This represents 3.75% of the BDMPS population for the autumn and spring periods, and 3.30% during the winter period. Impacts to birds from the SPA during these periods are therefore apportioned accordingly.

### Potential effects on the qualifying feature from the Project-alone

1729. The lesser black-backed gull qualifying feature of the Isles of Scilly SPA has been screened into the Appropriate Assessment due to the potential risk of collision.

#### Operation and maintenance phase collision risk

1730. Information for collision risk on breeding adult lesser black-backed gulls belonging to the Isles of Scilly SPA population is presented in **Table 8.107**.

Collision estimates, calculated using the sCRM, are presented by biological season. A summary of the annual outputs and the corresponding increase in the annual baseline mortality rate is also presented. Parameters used in the sCRM were agreed with Natural England during the ETG process and are described in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES.

1731. Based on the mean collision rates, the annual total of breeding adult lesser black-backed gulls from Isles of Scilly SPA at risk of collision as a result of the Project is less than one bird (0.06). This would increase the existing mortality of the SPA breeding population by 0.01%.

*Table 8.107 Predicted seasonal and annual collision mortality (Stochastic model Option 2, avoidance rate 0.994 (±0.0004)), for breeding adult lesser black-backed gulls at the windfarm site, apportioned to Isles of Scilly SPA, with corresponding increases to baseline mortality of the population*

| | ***Breeding Season*** | ***Autumn Migration*** | ***Non-breeding/winter*** | ***Spring Migration*** | ***Annual*** |
|---|---|---|---|---|---|
| **Period** | **Apr-Aug** | **Sep-Oct** | **Nov-Feb** | **Mar** | **Jan-Dec** |
| Total collisions[1] (mean and 95% CIs) | 1.44 (0.00-4.53) | 1.25 (0.00-5.63) | 0.15 (0.00-0.80) | 0.15 (0.00-0.94) | 2.98 (0.00-11.90) |
| % apportioned to the SPA | 0.00% | 3.75% | 3.30% | 3.75% | - |
| Total SPA collisions (mean and 95% CIs) | 0.00 (0.00-0.00) | 0.05 (0.00-0.21) | 0.00 (0.00-0.03) | 0.01 (0.00-0.04) | 0.06 (0.00-0.27) |
| Mortality increase[2] (mean and 95% CIs) | 0.00% (0.00-0.00%) | 0.00% (0.00-0.02%) | 0.00% (0.00-0.00%) | 0.00% (0.00-0.00%) | 0.01% (0.00-0.03%) |

[1] Breeding season collision values reduced to 71.9% of modelled value to reflect proportion of adult birds recorded during site surveys
[2] Assuming predicted annual SPA mortality of 642 birds (5,586 x 0.115)

1732. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that no detectable changes in mortality rates would occur on this population for the mean monthly collision estimates for the Project, or for the upper 95% CI collision estimate.

1733. **It is concluded that based on predicted lesser black-backed gull mortality due to collision at the windfarm site there is no potential for the Project to have an adverse effect on the integrity of the Isles of Scilly SPA.**

1734. The confidence in the assessment is high. The evidence used to define the CRM input parameters presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** as part of the ES submitted alongside this RIAA is of high applicability and quality. Whilst there is uncertainty around some of the input parameters (e.g. avoidance rate), the rates selected are considered to be sufficiently precautionary based on expert opinion to provide confidence that collision rates are not underestimated.

#### Potential effects on the qualifying feature in-combination with other projects

1735. As the Project would have no measurable effect on lesser black-backed gull populations from the Isles of Scilly SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of Isles of Scilly SPA, when assessed in-combination with other plans or projects**.

### 8.42.3.3 Great black-backed gull

#### Status

1736. The Isles of Scilly SPA great black-backed gull population was cited as 1,882 individuals in 2015-16 (Natural England 2020d). The most recent complete SMP count was 905 pairs (AON), or 1,810 breeding adults, in 2015 (JNCC 2023); this is used as the reference population for the assessment.

1737. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.07 (1 – 0.930; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 127 breeding adults.

#### Functional linkage and seasonal apportionment of potential effects

1738. The mean maximum foraging range of great black-backed gull is 73km, as is the maximum foraging range (Woodward *et al*., 2019). The Project is located approximately 459km from Isles of Scilly SPA, which means that the Project is beyond the maximum foraging range for great black-backed gulls from the

SPA. No impacts during the breeding season from the Project are therefore apportioned to great black-backed gulls breeding at this SPA.

1739. Outside the breeding season, breeding great black-backed gulls from the SPA are assumed to range widely and to mix with great black-backed gulls of all ages from breeding colonies in the UK and beyond. The relevant non-breeding season reference population is the UK south-west and Channel waters BDMPS, consisting of 17,742 individuals during the non-breeding season (September to March) (Furness, 2015).

1740. Furness (2015) estimated that 90% of the Isles of Scilly SPA breeding adults are present within the UK south-west and Channel waters BDMPS during the non-breeding season, which is 1,622 birds. This represents 9.14% of the BDMPS population; impacts to birds from the SPA during this period is therefore apportioned accordingly.

### Potential effects on the qualifying feature from the Project-alone

1741. The great black-backed gull qualifying feature of the Isles of Scilly SPA has been screened into the Appropriate Assessment due to the potential risk of collision.

#### Operation and maintenance phase collision risk

1742. Information for collision risk on breeding adult great black-backed gulls belonging to the Isles of Scilly SPA population is presented in **Table 8.108**. Collision estimates, calculated using the sCRM, are presented by biological season. A summary of the annual outputs and the corresponding increase in the annual baseline mortality rate is also presented. Parameters used in the sCRM were agreed with Natural England during the ETG process and are described in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES.

1743. Based on the mean collision rates, the annual total of breeding adult great black-backed gulls from Isles of Scilly SPA at risk of collision as a result of the Project is less than one bird (0.10). This would increase the existing mortality of the SPA breeding population by 0.08%.

*Table 8.108 Predicted seasonal and annual collision mortality (Stochastic model Option 2, avoidance rate 0.994 (±0.0004)), for breeding adult great black-backed gulls at the windfarm site, apportioned to Isles of Scilly SPA, with corresponding increases to baseline mortality of the population*

| Period | Breeding Season<br>Apr-Aug | Autumn Migration<br>n/a | Non-breeding/winter<br>Sep-Mar | Spring Migration<br>n/a | Annual<br>Jan-Dec |
|---|---|---|---|---|---|
| Total collisions (mean and 95% CIs) | 0.66<br>(0.00-4.11) | - | 1.09<br>(0.00-5.24) | - | 1.09<br>(0.00-5.24) |
| % apportioned to the SPA | 0.00% | - | 9.14% | - | - |
| Total SPA collisions (mean and 95% CIs) | 0.00<br>(0.00-0.00) | - | 0.10<br>(0.00-0.48) | - | 0.10<br>(0.00-0.48) |
| Mortality increase[1] (mean and 95% CIs) | 0.00%<br>(0.00-0.00%) | - | 0.08%<br>(0.00-0.38%) | - | 0.08%<br>(0.00-0.38%) |
| [1] Assuming predicted annual SPA mortality of 127 birds (1,810 x 0.070) | | | | | |

1744. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that no detectable changes in mortality rates would occur on this population for the mean monthly collision estimates for the Project, or for the upper 95% CI collision estimate.

1745. **It is concluded that based on predicted great black-backed gull mortality due to collision at the windfarm site there is no potential for the Project to have an adverse effect on the integrity of the Isles of Scilly SPA.**

1746. The confidence in the assessment is high. The evidence used to define the CRM input parameters presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** as part of the ES submitted alongside this RIAA is of high applicability and quality. Whilst there is uncertainty around some of the input parameters (e.g. avoidance rate), the rates selected are considered to be sufficiently precautionary based on expert opinion to provide confidence that collision rates are not underestimated.

### Potential effects on the qualifying feature in-combination with other projects

1747. As the Project would have no measurable effect on great black-backed gull populations from the Isles of Scilly SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of Isles of Scilly SPA, when assessed in-combination with other plans or projects**.

## 8.43 East Caithness Cliffs SPA

1748. East Caithness Cliffs SPA is located approximately 474km from the windfarm site.

### 8.43.1 Description of designation

1749. East Caithness Cliffs SPA is of high nature conservation and scientific importance within Britain and Europe for supporting very large populations of breeding seabirds. It includes most of the sea cliff areas between Wick and Helmsdale on the north-east coast of the Scottish mainland. The seaward elements of the SPA extend 2km into the marine environment and includes the seabed, water column and surface. Seabirds included within the designation (including great black-backed gull, herring gull, kittiwake, fulmar, guillemot and razorbill) feed both inside and outside the SPA in nearby waters, as well as more distantly in the wider North Sea.

### 8.43.2 Conservation objectives

1750. The overarching conservation objectives for the site are:

- To avoid deterioration of the habitats of the qualifying species or significant disturbance to the qualifying species, thus ensuring that the integrity of the site is maintained
- To ensure for the qualifying species that the following are maintained in the long term:
    - Population of the species as a viable component of the site
    - Distribution of the species within site
    - Distribution and extent of habitats supporting the species
    - Structure, function and supporting processes of habitats supporting the species
    - No significant disturbance of the species

### 8.43.3 Assessment

1751. The qualifying features of East Caithness Cliffs SPA screened into the Appropriate Assessment are listed in **Table 5.2**. These are breeding fulmar and breeding kittiwake. The seabird assemblage has also been screened in for these species.

#### 8.43.3.1 Fulmar

**Status**

1752. The East Caithness Cliffs SPA breeding fulmar population is cited as 15,000 pairs in 1996 (Furness, 2015, Stroud *et al*., 2016). Furness (2015) gave the breeding population of 14,202 pairs in 1999. The most recent count (2015) is 13,714 AOS, or 27,428 breeding adults (JNCC, 2023a); this is used as the reference population for the assessment.

1753. Based on the most recent SPA population of breeding adults, and an annual breeding adult baseline mortality rate of 0.064 (1 – 0.936, Horswill and Robinson 2015), 1,755 breeding adults from the SPA population would be expected to die each year.

**Functional linkage and seasonal apportionment of potential effects**

1754. The mean maximum foraging range of fulmar is 542.3km (±657.9km) and the maximum foraging range is 2,736km (Woodward *et al*., 2019). The Project is located approximately 474km from East Caithness Cliffs SPA, which means that the Project is within the mean maximum foraging range of fulmars breeding at this SPA.

**Potential effects on the qualifying feature from the Project-alone**

1755. Fulmar are considered to have low vulnerability to both collision and displacement impacts from offshore windfarms (Bradbury *et al*., 2014). This species was recorded infrequently within the windfarm site (in only five of 24 surveys, all during the breeding season (January to August; Furness, 2015)) and in low numbers, with densities in four of the five months less than 0.1 birds/km$^2$. A peak density of 0.23 birds/km$^2$ was recorded in May 2022.

1756. On the basis of the low vulnerability of fulmars to potential impacts, and the low frequency and abundance of birds present at the windfarm site, it is concluded that any effects on breeding populations at East Caithness Cliffs SPA are very unlikely, both during and outside of the breeding season.

1757. **It is therefore concluded that there would be no measurable effects on fulmar due to the project alone, and no adverse effect on the integrity of the East Caithness Cliffs SPA is predicted.**

**Potential effects on the qualifying feature in-combination with other projects**

1758. As the Project would have no measurable effect on fulmar populations from the East Caithness Cliffs SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of East Caithness Cliffs SPA, when assessed in-combination with other plans or projects.**

#### 8.43.3.2 Kittiwake

##### Status

1759. The East Caithness Cliffs SPA breeding kittiwake population at classification (1994) was cited as 32,500 pairs, or 65,000 breeding adults, between 1985 and 1987 (SNH, 2017). Furness (2015) gave a breeding population of 40,410 breeding pairs or 80,820 adults for 1999. The most recent available count is 24,460 pairs, or 48,920 breeding adults, in 2015 (JNCC, 2022); this is used as the reference population for the assessment.

1760. Based on the most recent SPA population of breeding adults and an annual baseline adult mortality rate of 0.146 (1 – 0.854, Horswill and Robinson (2015)), 7,142 breeding adults from this SPA population would be expected to die each year.

##### Functional linkage and seasonal apportionment of potential effects

1761. The mean maximum foraging range of kittiwake is 156.1km (±144.5km) and the maximum foraging range is 770km (Woodward *et al*., 2019). The Project is located approximately 474km from East Caithness Cliffs SPA, which means that the Project is beyond the mean maximum foraging range +1SD of kittiwakes breeding at this SPA, but within the maximum foraging range. The maximum foraging range is a poor indicator of typical foraging behaviour. It would be expected that few birds or foraging trips will occur at this distance from the colony, and even fewer with any regularity. No impacts during the breeding season are therefore apportioned to birds breeding at this colony.

1762. Outside the breeding season, breeding kittiwakes from the SPA are assumed to range widely and to mix with kittiwakes of all ages from breeding colonies in the UK and beyond. The relevant non-breeding season reference population is the UK Western Waters and Channel BDMPS, consisting of 911,586 individuals during autumn migration (September to December) and 627,816 during spring migration (January to February) (Furness, 2015).

1763. Furness (2015) estimated that 20% of the East Caithness Cliffs SPA breeding adults are present within the UK Western Waters and Channel BDMPS during the autumn migration period, which is 16,164 birds. During the spring migration period 30% of the population is estimated to be present, which is 24,246 birds. This represents 1.77% and 3.86% of the BDMPS population for the autumn and spring periods respectively. During autumn migration and spring migration, 1.77%, and 3.86% of impacts are therefore considered to affect birds from the SPA.

## Potential effects on the qualifying feature from the Project-alone

1764. The kittiwake qualifying feature of the East Caithness Cliffs SPA has been screened into the Appropriate Assessment due to the potential risk of collision.

### Operation and maintenance phase collision risk

1765. Information for collision risk on breeding adult kittiwakes belonging to the East Caithness Cliffs SPA population is presented in **Table 8.109**. Collision estimates, calculated using the sCRM, are presented by biological season. A summary of the annual outputs and the corresponding increase in the annual baseline mortality rate is also presented. Parameters used in the sCRM were agreed with Natural England during the ETG process and are described in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES.

1766. Based on the mean collision rates, the annual total of breeding adult kittiwakes from East Caithness Cliffs SPA at risk of collision as a result of the Project is less than one bird (0.17). This would increase the existing mortality of the SPA breeding population by 0.00%.

*Table 8.109 Predicted seasonal and annual collision mortality (Stochastic model Option 2, avoidance rate 0.993 (±0.0003)), for breeding adult kittiwakes at the windfarm site, apportioned to East Caithness Cliffs SPA, with corresponding increases to baseline mortality of the population*

| Period | Breeding Season<br>Mar-Aug | Autumn Migration<br>Sep-Dec | Non-breeding/winter<br>- | Spring Migration<br>Jan-Feb | Annual<br>Jan-Dec |
|---|---|---|---|---|---|
| Total collisions[1] (mean and 95% CIs) | 15.36 (4.17-33.87) | 8.49 (2.20-18.85) | - | 0.63 (0.00-1.46) | 24.48 (6.37-54.17) |
| % apportioned to the SPA | 0.00% | 1.77% | - | 3.86% | - |
| Total SPA collisions (mean and 95% CIs) | 0.00 (0.00-0.00) | 0.15 (0.04-0.33) | - | 0.02 (0.00-0.06) | 0.17 (0.04-0.39) |
| Mortality increase[2] (mean and 95% CIs) | 0.00% (0.00-0.00%) | 0.00% (0.00-0.00%) | - | 0.00% (0.00-0.00%) | 0.00% (0.00-0.01%) |

[1] Breeding season collision values reduced to 94.1% of modelled value to reflect proportion of adult birds recorded during site surveys
[2] Assuming predicted annual SPA mortality of 7,142 birds (48,920 x 0.146)

1767. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that no detectable changes in mortality rates would occur on this population for the mean monthly collision estimates for the Project, or for the upper 95% CI collision estimate.

1768. **It is concluded that based on predicted kittiwake mortality due to collision at the windfarm site there is no potential for the Project to have an adverse effect on the integrity of the East Caithness Cliffs SPA**.

1769. The confidence in the assessment is high. The evidence used to define the CRM input parameters presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** as part of the ES submitted alongside this RIAA is of high applicability and quality. Whilst there is uncertainty around some of the input parameters (e.g. avoidance rate), the rates selected are considered to be sufficiently precautionary based on expert opinion to provide confidence that collision rates are not underestimated.

### Potential effects on the qualifying feature in-combination with other projects

1770. As the Project would have no measurable effect on kittiwake populations from the East Caithness Cliffs SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of East Caithness Cliffs SPA, when assessed in-combination with other plans or projects**.

## 8.44 Shiant Isles SPA

1771. Shiant Isles SPA is located approximately 474km from the windfarm site.

### 8.44.1 Description of designation

1772. The four islands that comprise the Shiant Isles SPA, with their skerries, are situated 6km east of Harris in the Western Isles. The boundary of the SPA overlaps with the boundary of Shiant Islands SSSI, and the seaward elements extend approximately 2km into the marine environment to include the seabed, water column and surface. Qualifying seabird species of the SPA comprise fulmar, shag, kittiwake, guillemot, razorbill and puffin.

### 8.44.2 Conservation objectives

1773. The overarching conservation objectives for the site are:

- To avoid deterioration of the habitats of the qualifying species or significant disturbance to the qualifying species, thus ensuring that the integrity of the site is maintained
- To ensure for the qualifying species that the following are maintained in the long term:
  - Population of the species as a viable component of the site
  - Distribution of the species within site
  - Distribution and extent of habitats supporting the species
  - Structure, function and supporting processes of habitats supporting the species
  - No significant disturbance of the species

### 8.44.3 Assessment

1774. The qualifying features of the Shiant Isles SPA screened into the Appropriate Assessment are listed in **Table 5.2.** These are breeding fulmar, breeding guillemot, breeding razorbill and breeding puffin. The seabird assemblage has also been screened in for these species.

#### 8.44.3.1 Fulmar

#### Status

1775. The Shiant Isles SPA breeding fulmar population is cited as 6,820 pairs, or 13,640 breeding adults, in 1992 (Furness, 2015, Stroud *et al*., 2016). Furness (2015) gave the breeding population of 4,387 pairs, or 8,774 breeding adults, in 1999. The most recent count is 1,506 pairs (AON), or 3,012 breeding adults, in 2015 (JNCC, 2023a); this is used as the reference population for the assessment.

1776. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.064 (1 – 0.936, Horswill and Robinson 2015) the expected annual mortality from the SPA population would be 193 breeding adults.

### Functional linkage and seasonal apportionment of potential effects

1777. The mean maximum foraging range of fulmar is 542.3km (±657.9km) and the maximum foraging range is 2,736km (Woodward *et al*., 2019). The Project is located approximately 474km from Shiant Isles SPA, which means that the Project is within the mean maximum foraging range of fulmars breeding at this SPA.

### Potential effects on the qualifying feature from the Project-alone

1778. Fulmar are considered to have low vulnerability to both collision and displacement impacts from offshore windfarms (Bradbury *et al*., 2014). This species was recorded infrequently within the windfarm site (in only five of 24 surveys, all during the breeding season (January to August; Furness, 2015)) and in low numbers, with densities in four of the five months less than 0.1 birds/km$^2$. A peak density of 0.23 birds/km$^2$ was recorded in May 2022.

1779. On the basis of the low vulnerability of fulmars to potential impacts, and the low frequency and abundance of birds present at the windfarm site, it is concluded that any effects on breeding populations at Shiant Isles SPA are very unlikely, both during and outside of the breeding season.

1780. **It is therefore concluded that there would be no measurable effects on fulmar due to the project alone, and no adverse effect on the integrity of the Shiant Isles SPA is predicted.**

### Potential effects on the qualifying feature in-combination with other projects

1781. As the Project would have no measurable effect on fulmar populations from the Shiant Isles SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of Shiant Isles SPA, when assessed in-combination with other plans or projects.**

#### 8.44.3.2 Guillemot

##### Status

1782. The Shiant Isles SPA breeding guillemot population was cited as 12,315 pairs, or 24,630 breeding adults, in 1992 (Furness, 2015, Stroud *et al*., 2016). Furness (2015) gave the breeding population of 5,148 pairs, or 10,296 breeding adults, in 2008. The most recent count is 9,054 individuals in 2015 (JNCC, 2023a); this is used as the reference population for the assessment.

1783. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.061 (1 – 0.939; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 553 breeding adults.

##### Functional linkage and seasonal apportionment of potential effects

1784. The mean maximum foraging range of guillemot is 73.2km (±80.5km) and the maximum foraging range is 338km (Woodward *et al*., 2019). The Project is located approximately 474km from Shiant Isles SPA, which means that the Project is beyond the maximum foraging range of guillemots from the SPA. No impacts during the breeding season from the Project are therefore apportioned to guillemots breeding at this SPA.

1785. Outside the breeding season, breeding guillemots from the SPA are assumed to range widely and to mix with guillemots of all ages from breeding colonies in the UK and beyond. The relevant non-breeding season reference population is the UK Western Waters BDMPS, consisting of 1,139,220 individuals (August to February) (Furness, 2015).

1786. Furness (2015) estimated that 95% of the Shiant Isles SPA breeding adults are present within the UK Western Waters BDMPS during the non-breeding season, which is 9,781 birds. This represents 0.9% of the BDMPS population for this period (1,139,220). It is therefore assumed that 0.9% of guillemots present at the Project site are breeding adults from Shiant Isles SPA.

##### Potential effects on the qualifying feature

**Operation and maintenance phase disturbance/displacement/barrier effects**

*Project-alone*

1787. The mean peak abundance of guillemots present within the windfarm site and 2km buffer during the non-breeding season was 8,315 (6,085-12,047) individuals (refer to **Appendix 12.1** of the ES). Of these, 75 birds (55-108) were likely to be breeding adults from the Shiant Isles SPA.

1788. **Table 8.110** sets out the predicted impacts on guillemots from Shiant Isles SPA during the non-breeding season. Displacement rates of 30% to 70% are considered for this species, along with a range of mortality rates of 1% to 10% of displaced birds (UK SNCBs, 2017).

1789. The available evidence suggests that the upper ranges of these displacement and mortality rates may be excessively precautionary. Whilst it is true that guillemots and razorbills tend to be displaced by OWFs they do not avoid them completely, and displacement rates vary between sites. MacArthur Green (2019b) concluded that, on average, densities within OWFs were around half of the density found in the habitats around the OWF. At some OWFs there was also displacement of birds from a buffer zone surrounding it. The size of the buffer zone varied between OWFs and was generally less than 2km, with auk density increasing across the buffer zone as distance from turbines increased, up to the density in the wider area. Another recent review (APEM, 2022) found that the current displacement rates suggested by the SNCBs for guillemot and razorbill (30% to 70%) did not account for the quality of or confidence in the studies which were used to inform this position, and that studies where no significant effects were recorded were not accounted for during the provision of the advice. APEM (2022) suggested that in the case of Hornsea Project Four, an evidence-based displacement rate of 50% was appropriate. However, the study also recognised that larger displacement effects are possible.

1790. Mortality due to displacement could arise due to increased energy costs and/or decreased energy intake, if displacement results in increased flying time to avoid OWFs, and/or increased bird densities and competition for prey in areas of unimpacted habitat outside the OWF. Given that UK OWFs in the Irish and Celtic Seas represent a very small proportion of the available foraging habitat for guillemots and razorbills within UK Western Waters, the increase in density of auks outside OWFs due to displacement is likely to be negligible (MacArthur Green, 2019b). It is considered unlikely that the mortality rate due to displacement would be as high as the empirically estimated 6.1% annual mortality for adult guillemots that occurs due to the combination of 'natural' factors and anthropogenic activities (Horswill and Robinson, 2015). Indeed, it may be much lower; MacArthur Green (2019b) recommended precautionary rates of 50% displacement for guillemot and 1% mortality of displaced birds based on a review of available evidence. Modelling undertaken by APEM (2022) found that in the case of Hornsea Project Four, a mortality rate of 1% for displaced auks was considered to be precautionary, and that this may be an overestimate given that Hornsea Project Four is located towards the upper end of the mean maximum foraging range for guillemot from the closest designated sites. It is also noted that in the recent decision on Hornsea Project Four, the SoS determined that a rate of 70% displacement/2% mortality was appropriate to inform the assessment of effects on guillemots from Flamborough and Filey Coast SPA (DESNZ, 2023b).

1791. The full ranges of recommended displacement and mortality effects are considered by the assessment, along with evidence-based displacement and mortality rates of 50% and 1%, respectively (APEM, 2022; MacArthur Green, 2019b).

*Table 8.110 Guillemot – predicted operation and maintenance phase displacement and mortality from Shiant Isles SPA*

| Mean peak abundance estimate type | Mean peak abundance estimate | Number of Shiant Isles SPA breeding adults present (non-breeding season)[1] | Annual mortality range[2] | Annual baseline mortality increase range[3] |
|---|---|---|---|---|
| Upper 95% CI | 12,047 | 108 | 0-8 | 0.06-1.37% |
| Mean | 8,315 | 75 | 0-5 | 0.04-0.95% |
| Lower 95% CI | 6,085 | 55 | 0-4 | 0.03-0.69% |

[1] Assumes 0.9% of birds present during the non-breeding season are Shiant Isles SPA breeding adults

[2] Assumes displacement rates of 30-70% and mortality rates of 1-10%

[3] Background mortality rate of 6.1% (Horswill and Robinson, 2015)

1792. Based on the mean peak abundances, the annual total of guillemots from the Shiant Isles SPA at risk of displacement is 75 birds (**Table 8.110**). At displacement rates of 30% to 70%, and mortality rates of 1% to 10% for displaced birds, 0 to 5 SPA breeding adults would be predicted to die each year due to displacement from the Project.

1793. Assuming a displacement rate of 70% and a mortality rate of 10% of displaced birds, annual mortality within the SPA breeding adult population would increase by 0.95%. Using an evidence-based displacement rate of 50%, and a mortality rate for displaced birds of 1%, annual mortality in the population would instead increase by 0.07% (<1 bird).

1794. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that no detectable changes in mortality rates would occur under any combination of displacement and mortality rates when the mean peak abundance estimate assessments are considered.

1795. Increases of over 1% are predicted if the upper 95% CIs for mean peak abundances are used as inputs to the assessment alongside a 10% mortality rate for displaced birds. The probability of this occurring is extremely small for two reasons. Firstly, the upper 95% CI for the mean peak abundances are highly unlikely to occur regularly at the windfarm site. Secondly, mortality rates for displaced birds of 10% are much higher than evidence suggests will actually be the case, and use of the evidence-based displacement (50%) and mortality rate (1%) (and also 70%/2%) would again result in a mortality increase of significantly less than 1%.

1796. **It is concluded that predicted guillemot mortality due to operational phase displacement for the project alone would not adversely affect the integrity of the Shiant Isles SPA.**

1797. The confidence in the assessment is high for several reasons. Firstly, the evidence used to inform the evidence-based displacement rates is of high applicability and quality (based on the criteria discussed in **Chapter 12 Offshore Ornithology** of the ES). Whilst there is limited available evidence to inform mortality rates, 1% is considered to be sufficiently precautionary based on expert opinion. This species is not regarded as being highly specialised in its habitat requirements (Bradbury *et al*., 2014; Furness and Wade, 2012; Garthe and Hüppop, 2004), and it is therefore anticipated that displaced birds will find alternative habitat in the vast majority of cases. Finally, the conclusion of the assessment is the same irrespective of whether the mean or 95% upper CI mean peak abundances are used to calculate potential mortality and increases in the baseline mortality rate of the background population, provided the evidence-based displacement and mortality rates are used.

*In-combination*

1798. The in-combination assessment for guillemots from Shiant Isles SPA has been undertaken in accordance with the approach presented in **Section 8.1**. The total population apportioned to Shiant Isles SPA at risk of displacement is estimated to be 445 breeding adults (**Appendix 12.1** of the ES). Annual in-combination displacement and mortality rates for birds from Shiant Isles SPA are presented in **Table 8.111**.

1799. Assuming a displacement rate of 70% and a mortality rate of 10% of displaced birds, 31 breeding adult SPA birds would be lost to displacement annually. Including additional apportioned mortality from the Morlais and Holyhead tidal projects (total 0.28 birds), this would increase the existing mortality within the SPA population (552 breeding adult birds per year) by 5.70%. Using an evidence-based displacement rate of 50%, and a mortality rate for displaced birds of 1%, the annual in-combination displacement mortality would be two birds. This would increase the existing mortality within this population by 0.45%. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that detectable changes in mortality rates will not occur due to the level or mortality predicted if the more realistic rates for mortality are used.

1800. **It is concluded that predicted guillemot mortality due to of operational phase displacement impacts of the Project in-combination with other projects, would not adversely affect the integrity of Shiant Isles SPA.**

Table 8.111 In-combination year-round displacement matrix for guillemot from Shiant Isles SPA

| Annual | Mortality | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Displacement | 1% | 2% | 3% | 4% | 5% | 10% | 20% | 30% | 50% | 80% | 100% |
| 10% | 0 | 1 | 1 | 2 | 2 | 4 | 9 | 13 | 22 | 36 | 45 |
| 20% | 1 | 2 | 3 | 4 | 4 | 9 | 18 | 27 | 45 | 71 | 89 |
| 30% | 1 | 3 | 4 | 5 | 7 | 13 | 27 | 40 | 67 | 107 | 134 |
| 40% | 2 | 4 | 5 | 7 | 9 | 18 | 36 | 53 | 89 | 143 | 178 |
| 50% | 2 | 4 | 7 | 9 | 11 | 22 | 45 | 67 | 111 | 178 | 223 |
| 60% | 3 | 5 | 8 | 11 | 13 | 27 | 53 | 80 | 134 | 214 | 267 |
| 70% | 3 | 6 | 9 | 12 | 16 | 31 | 62 | 94 | 156 | 249 | 312 |
| 80% | 4 | 7 | 11 | 14 | 18 | 36 | 71 | 107 | 178 | 285 | 356 |
| 90% | 4 | 8 | 12 | 16 | 20 | 40 | 80 | 120 | 200 | 321 | 401 |
| 100% | 4 | 9 | 13 | 18 | 22 | 45 | 89 | 134 | 223 | 356 | 445 |

Note: The cells show the number of birds predicted to die (rounded to the nearest integer) at a given rate of displacement and mortality. Highlighted cells are considered to be the most realistic scenario, in accordance with SNCB advice (SNCBs, 2022).

#### 8.44.3.3 Razorbill

##### Status

1801. The Shiant Isles SPA breeding razorbill population was cited as 7,337 pairs, or 14,674 breeding adults in 1986 (Furness, 2015, Stroud *et al*., 2016). Furness (2015) gave the breeding population of 4,248 pairs, or 8,496 breeding adults, in 2008. The most recent count is 8,029 individuals in 2015 (JNCC, 2023a); this is used as the reference population for the assessment.

1802. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.105 (1 – 0.895; Horswill and Robinson, 2015), the expected annual mortality from the SPA population would be 843 breeding adults.

##### Functional linkage and seasonal apportionment of potential effects

1803. The mean maximum foraging range of razorbill is 88.7km (±75.9km) and the maximum foraging range is 313km (Woodward *et al*., 2019). The Project is located approximately 474km from Shiant Isles SPA, which means that the Project is beyond the maximum foraging range for razorbills from the SPA. No impacts during the breeding season from the Project are therefore apportioned to guillemots breeding at this SPA.

1804. Outside the breeding season, breeding razorbills from the SPA are assumed to range widely and to mix with razorbills of all ages from breeding colonies in the UK and beyond. The relevant non-breeding season reference population is the UK Western Waters BDMPS, consisting of 606,914 individuals during autumn and spring passage periods (August to October and January to March), and 341,422 individuals during winter (November and December) (Furness, 2015). During autumn and spring migration, 98% of the SPA breeding adults (8,326 individuals based on the 2008 population estimate) are assumed to be present in the BDMPS, representing 1.4% of the BDMPS population (606,914 individuals of all ages). During the winter season, 40% of the SPA breeding adults (3,398 individuals based on the 2008 population estimate) are assumed to be present in the BDMPS, representing 1.0% of the BDMPS population (341,422 individuals of all ages). These percentages (i.e. 1.4% and 1.0%) are the proportions of birds present at the windfarm site that are presumed to originate from the Shiant Isles SPA during the relevant seasons.

## Potential effects on the qualifying feature

### Operation and maintenance phase disturbance/displacement/barrier effects

*Project-alone*

1805. The year-round mean peak abundance of razorbills present within the windfarm site and 2km buffer was 1,979 (703-3,552) individuals (refer to **Appendix 12.1** of the ES). Of these, 51 birds (21-86) were likely to be breeding adults from the Shiant Isles SPA.

1806. **Table 8.112** sets out the predicted impacts on razorbills from Shiant Isles SPA. Displacement rates of 30% to 70% are considered for this species, along with a range of mortality rates of 1% to 10% of displaced birds (UK SNCBs, 2017).

1807. The available evidence suggests that the upper ranges of these displacement and mortality rates may be excessively precautionary. Whilst it is true that guillemots and razorbills tend to be displaced by OWFs they do not avoid them completely, and displacement rates vary between sites. MacArthur Green (2019b) concluded that, on average, it was concluded that densities within OWFs are around half of the density found in the habitats around the OWF. At some OWFs there was also displacement of birds from a buffer zone surrounding it. The size of the buffer zone varied between OWFs and was generally less than 2km, with auk density increasing across the buffer zone as distance from turbines increased, up to the density in the wider area. Another recent review (APEM, 2022) found that the current displacement rates suggested by the SNCBs for guillemot and razorbill (30% to 70%) did not account for the quality of or confidence in the studies which were used to inform this position, and that studies where no significant effects were recorded were not accounted for during the provision of the advice. APEM (2022) suggested that in the case of Hornsea Project Four, an evidence-based displacement rate of 50% was appropriate. However, the study also recognised that larger displacement effects are possible.

1808. Mortality due to displacement could arise due to increased energy costs and/or decreased energy intake, if displacement results in increased flying time to avoid OWFs, and/or increased bird densities and competition for prey in areas of unimpacted habitat outside the OWF. Given that UK OWFs in the Irish and Celtic Seas represent a very small proportion of the available foraging habitat for guillemots and razorbills within UK Western Waters, the increase in density of auks outside OWFs due to displacement is likely to be negligible (MacArthur Green, 2019b). It is considered unlikely that the mortality rate due to displacement would be as high as the empirically estimated 10.5% annual mortality for adult razorbills that occurs due to the combination of ‘natural’ factors and anthropogenic activities (Horswill and Robinson, 2015). Indeed, it may be much lower; MacArthur Green (2019b) recommended

precautionary rates of 50% displacement for auks and 1% mortality of displaced birds based on a review of available evidence. Modelling undertaken by APEM (2022) found that in the case of Hornsea Project Four, a mortality rate of 1% for displaced auks was considered to be precautionary, and that this may be an overestimate given that Hornsea Project Four is located towards the upper end of the mean maximum foraging range for guillemot from the closest designated sites.

1809. The full ranges of recommended displacement and mortality effects are considered by the assessment, along with evidence-based displacement and mortality rates of 50% and 1%, respectively (APEM, 2022; MacArthur Green, 2019b).

*Table 8.112 Razorbill – predicted operation and maintenance phase displacement and mortality from Shiant Isles SPA*

| Mean peak abundance estimate type | Mean peak abundance estimate by season[1] | Number of Shiant Isles SPA breeding adults present by season)[2] | Annual mortality range[3] | Annual baseline mortality increase range[4] |
|---|---|---|---|---|
| Upper 95% CI | 605 (b)<br>1,070 (aut)<br>1,297 (win)<br>580 (spr)<br>3,552 (year round) | 0 (b)<br>15 (aut)<br>13 (win)<br>8 (spr)<br>36 (year round) | 0-3 (0) | 0.01-0.30% (0.02%) |
| Mean | 252 (b)<br>694 (aut)<br>651 (win)<br>381 (spr)<br>1,979 (year round) | 0 (b)<br>10 (aut)<br>7 (win)<br>5 (spr)<br>22 (year round) | 0-2 (0) | 0.01-0.18% (0.01%) |
| Lower 95% CI | 21 (b)<br>309 (aut)<br>159 (win)<br>214 (spr)<br>703 (year round) | 0 (b)<br>4 (aut)<br>2 (win)<br>3 (spr)<br>9 (year round) | 0-1 (0) | 0.00-0.07% (0.01%) |

[1] Breeding season = b, autumn migration season = aut, winter season = win, spring migration season = spr

[2] Assumes breeding adult apportioning of 0.0% (breeding season), 1.4% (spring and autumn migration) and 1.0% (winter) to Shiant Isles SPA.

[3] Assumes displacement rates of 30-70% and mortality rates of 1-10%. Evidence-based estimates assuming a 50% displacement rate and 1% mortality of displaced birds are presented in parentheses.

[4] Background mortality rate of 10.5% (Horswill and Robinson, 2015)

1810. Based on the mean peak abundances, the annual total of razorbills from the Shiant Isles SPA at risk of displacement is 22 birds (**Table 8.112**). At displacement rates of 30% to 70%, and mortality rates of 1% to 10% for displaced birds, 0 to 2 SPA breeding adults would be predicted to die each year due to displacement from the Project.

1811. Assuming a displacement rate of 70% and a mortality rate of 10% of displaced birds, annual mortality within the SPA breeding adult population would increase by 0.18%. Using an evidence-based displacement rate of 50%, and a mortality rate for displaced birds of 1%, annual mortality in the population would instead increase by 0.01% (<1 bird).

1812. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that no detectable changes in mortality rates would occur under almost any combination of displacement and mortality rates when the mean peak abundance estimate assessments are considered. This would be the case even when upper 95% CIs are considered.

1813. **It is concluded that predicted razorbill mortality due to operational phase displacement for the project alone would not adversely affect the integrity of the Shiant Isles SPA.**

1814. The confidence in the assessment is high for several reasons. Firstly, the evidence used to inform the evidence-based displacement rates is of high applicability and quality (based on the criteria discussed in **Chapter 12 Offshore Ornithology** of the ES). Whilst there is limited available evidence to inform mortality rates, 1% is considered to be sufficiently precautionary based on expert opinion. This species is not regarded as being highly specialised in its habitat requirements (Bradbury *et al*., 2014; Furness and Wade, 2012; Garthe and Hüppop, 2004), and it is therefore anticipated that displaced birds will find alternative habitat in the vast majority of cases. Finally, the conclusion of the assessment is the same irrespective of whether the mean or 95% upper CI mean peak abundances are used to calculate potential mortality and increases in the baseline mortality rate of the background population, provided the evidence-based displacement and mortality rates are used.

*In-combination*

1815. The in-combination assessment for razorbills from Shiant Isles SPA has been undertaken in accordance with the approach presented in **Section 8.1**. The total population apportioned to the SPA at risk of displacement is estimated to be 140 breeding adults (**Appendix 12.1** of the ES). Annual in-combination displacement and mortality rates for birds from Shiant Isles SPA are presented in **Table 8.113**.

1816. Assuming a displacement rate of 70% and a mortality rate of 10% of displaced birds, 10 breeding adult SPA birds would be lost to displacement annually. Including additional apportioned mortality from the Morlais and Holyhead tidal projects (total 0.21 birds), this would increase the existing mortality within the SPA population (843 breeding adult birds per year) by 1.19%. Using an evidence-based displacement rate of 50%, and a mortality rate for displaced birds of 1%, the annual in-combination mortality would be one bird. This would increase the existing mortality within this population by 0.11%. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that detectable changes in mortality rates would not occur due to the level or mortality predicted if the more realistic rates for mortality are used.

1817. **It is concluded that predicted razorbill mortality due to of operational phase displacement impacts of the Project in-combination with other projects, would not adversely affect the integrity of Shiant Isles SPA.**

*Table 8.113 In-combination year-round displacement matrix for razorbill from Shiant Isles SPA*

| Annual | Mortality | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| **Displacement** | **1%** | **2%** | **3%** | **4%** | **5%** | **10%** | **20%** | **30%** | **50%** | **80%** | **100%** |
| **10%** | 0 | 0 | 0 | 1 | 1 | 1 | 3 | 4 | 7 | 11 | 14 |
| **20%** | 0 | 1 | 1 | 1 | 1 | 3 | 6 | 8 | 14 | 22 | 28 |
| **30%** | 0 | 1 | 1 | 2 | 2 | 4 | 8 | 13 | 21 | 34 | 42 |
| **40%** | 1 | 1 | 2 | 2 | 3 | 6 | 11 | 17 | 28 | 45 | 56 |
| **50%** | 1 | 1 | 2 | 3 | 3 | 7 | 14 | 21 | 35 | 56 | 70 |
| **60%** | 1 | 2 | 3 | 3 | 4 | 8 | 17 | 25 | 42 | 67 | 84 |
| **70%** | 1 | 2 | 3 | 4 | 5 | 10 | 20 | 29 | 49 | 78 | 98 |
| **80%** | 1 | 2 | 3 | 4 | 6 | 11 | 22 | 34 | 56 | 90 | 112 |
| **90%** | 1 | 3 | 4 | 5 | 6 | 13 | 25 | 38 | 63 | 101 | 126 |
| **100%** | 1 | 3 | 4 | 6 | 7 | 14 | 28 | 42 | 70 | 112 | 140 |

Note: The cells show the number of birds predicted to die (rounded to the nearest integer) at a given rate of displacement and mortality. Highlighted cells are considered to be the most realistic scenario, in accordance with SNCB advice (SNCBs, 2022).

#### 8.44.3.4 Puffin

##### Status

1818. The Shiant Isles SPA breeding puffin population was cited as 76,100 breeding pairs, or 152,200 breeding adults, in 1970, and a breeding population of 65,170 pairs, or 130,340 adults, was given in 2000 (Furness, 2015). The most recent count is 64,695 pairs (apparently occupied burrows), or 129,390 breeding adults, in 2015 (JNCC, 2023a); this is used as the reference population for the assessment.

1819. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.094 (1 – 0.906; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 12,163 breeding adults.

##### Functional linkage and seasonal apportionment of potential effects

1820. The mean maximum foraging range of puffin is 137.1km (±128.3km) and the maximum foraging range is 383km (Woodward *et al*., 2019). The Project is located approximately 474km from Shiant Isles SPA, which means that the Project is beyond the maximum foraging range for puffins from the SPA. No impacts during the breeding season from the Project are therefore apportioned to puffins breeding at this SPA.

1821. Outside of the breeding season, breeding puffins, including those from the Shiant Isles SPA, are not constrained by requirements to visit nests to incubate eggs or provision chicks. At this time, they are assumed to range more widely and to mix with puffins of all age classes from breeding colonies in the UK and further afield. The background population during these seasons is the UK Western Waters BDMPS. This consists of 304,557 individuals during the non-breeding season (August to March).

1822. Furness (2015) estimated that 18% of the Shiant Isles SPA breeding adults (130,340) are present within the UK Western Waters BDMPS during the non-breeding season, which is 23,461 birds. This represents 7.7% of the BDMPS population for this period (304,557). It is therefore assumed that 7.7% of puffins present at the Project site are breeding adults from Shiant Isles SPA.

##### Potential effects on the qualifying feature

**Operation and maintenance phase disturbance/displacement/barrier effects**

*Project-alone*

1823. The mean peak abundance of puffins present within the windfarm site and 2km buffer during the non-breeding season was 19.7 (1.9-50.8) individuals (refer to **Appendix 12.1** of the ES). Of these, less than two birds (1.5 (0.1-3.9)) were likely to be a breeding adult from Shiant Isles SPA.

1824. **Table 8.114** sets out the predicted impacts on puffins from Shiant Isles SPA during the non-breeding season. This estimates that there would be no measurable increase in mortality from the SPA population, assuming a displacement rate of 30-70% and a mortality of 1-10% for displaced birds.

*Table 8.114 Puffin – predicted operation and maintenance phase displacement and mortality from Shiant Isles SPA*

| Mean peak abundance estimate type | Mean peak abundance estimate | Number of Shiant Isles SPA breeding adults present (non-breeding season)[1] | Annual mortality range[2] | Annual baseline mortality increase range[3] |
|---|---|---|---|---|
| Upper 95% CI | 50.8 | 3.9 | 0-0 | 0.00-0.00% |
| Mean | 19.7 | 1.5 | 0-0 | 0.00-0.00% |
| Lower 95% CI | 1.9 | 0.1 | 0-0 | 0.00-0.00% |

[1] Assumes 7.7% of birds present during the non-breeding season are Shiant Isles SPA breeding adults

[2] Assumes displacement rates of 30-70% and mortality rates of 1-10%

[3] Background mortality rate of 9.4% (Horswill and Robinson, 2015)

1825. **It is concluded that predicted puffin mortality due to operational phase displacement would not adversely affect the integrity of the Shiant Isles SPA.**

1826. The confidence in the assessment is high for several reasons. Firstly, the evidence used to inform the evidence-based displacement rates is of high applicability and quality (based on the criteria discussed in **Chapter 12 Offshore Ornithology** of the ES). Whilst there is limited available evidence to inform mortality rates, 1% is considered to be sufficiently precautionary based on expert opinion. This species is not regarded as being highly specialised in its habitat requirements (Bradbury *et al*., 2014; Furness and Wade, 2012; Garthe and Hüppop, 2004), and it is therefore anticipated that displaced birds will find alternative habitat in the vast majority of cases. Finally, the conclusion of the assessment is the same irrespective of whether the mean or 95% upper CI mean peak abundances are used to calculate potential mortality and increases in the baseline mortality rate of the background population.

### *In-combination*

1827. As the Project would have no measurable effect on puffin populations from the Shiant Isles SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that predicted puffin mortality due to operational phase displacement due to the Project in-combination with other plans or projects would not adversely affect the integrity of the Shiant Isles SPA.**

## 8.45 Handa SPA

1828. Handa SPA is located approximately 509km from the windfarm site.

### 8.45.1 Description of designation

1829. Handa SPA consists of an island surrounded by high sea-cliffs and adjacent coastal waters lying a short distance from the west coast of Sutherland. It provides a strategic nesting locality for seabirds that feed in the productive waters of the northern Minch, outside the SPA. Most of the island is vegetated with sub-maritime grasslands and heaths. The SPA's principal ornithological importance is for its breeding seabirds. The boundary of the SPA overlaps with the boundary of Handa Island SSSI, and the seaward elements extend approximately 2km into the marine environment to include the seabed, water column and surface. The qualifying seabird species for the SPA comprise fulmar, great skua, kittiwake, guillemot and razorbill.

### 8.45.2 Conservation objectives

1830. The overarching conservation objectives for the site are:

- To avoid deterioration of the habitats of the qualifying species or significant disturbance to the qualifying species, thus ensuring that the integrity of the site is maintained
- To ensure for the qualifying species that the following are maintained in the long term:
  - Population of the species as a viable component of the site
  - Distribution of the species within site
  - Distribution and extent of habitats supporting the species
  - Structure, function and supporting processes of habitats supporting the species
  - No significant disturbance of the species

### 8.45.3 Assessment

1831. The qualifying features of Handa SPA screened into the Appropriate Assessment are listed in **Table 5.2**. These are breeding fulmar, breeding great skua, breeding kittiwake, breeding guillemot and breeding razorbill. These species are also screened in as named components of the seabird assemblage.

#### 8.45.3.1 Fulmar

##### Status

1832. The Handa SPA breeding fulmar population was cited as 3,500 pairs, or 7,000 breeding adults in 1986, and a breeding population of 1,870 pairs, or 3,740 breeding adults, was given for 2012 (Furness, 2015). The most recent estimate is 854 pairs (AOS), or 1,708 breeding adults, in 2022 (JNCC, 2023a); this is used as the reference population for the assessment.

1833. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.064 (1 – 0.936; Horswill and Robinson, 2015), 109 breeding adults from the SPA population would be expected to die each year.

##### Functional linkage and seasonal apportionment of potential effects

1834. The mean maximum foraging range of fulmar is 542.3km (±657.9km) and the maximum foraging range is 2,736km (Woodward *et al*., 2019). The Project is located approximately 509km from Handa SPA, which means that the Project is within the mean maximum foraging range of fulmars breeding at this SPA.

##### Potential effects on the qualifying feature from the Project-alone

1835. Fulmar are considered to have low vulnerability to both collision and displacement impacts from offshore windfarms (Bradbury *et al*., 2014). This species was recorded infrequently within the windfarm site (in only five of 24 surveys, all during the breeding season (January to August; Furness, 2015)) and in low numbers, with densities in four of the five months less than 0.1 birds/km$^2$. A peak density of 0.23 birds/km$^2$ was recorded in May 2022.

1836. On the basis of the low vulnerability of fulmars to potential impacts, and the low frequency and abundance of birds present at the windfarm site, it is concluded that any effects on breeding populations at Handa SPA are very unlikely, both during and outside of the breeding season.

1837. **It is therefore concluded that there would be no measurable effects on fulmar due to the project alone, and no adverse effect on the integrity of the Handa SPA is predicted.**

##### Potential effects on the qualifying feature in-combination with other projects

1838. As the Project would have no measurable effect on fulmar populations from the Handa SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of Handa SPA, when assessed in-combination with other plans or projects.**

#### 8.45.3.2 Great skua

##### Status

1839. The Handa SPA breeding great skua population was cited as 110 pairs, or 220 breeding adults, in 1990 (Furness, 2015, Stroud *et al*., 2016). Furness (2015) gave a population of 135 pairs, or 270 breeding adults, in 2013. The most recent count is 73 pairs (apparently occupied territories; AOT), or 146 breeding adults, in 2022 (JNCC, 2023a). This is used as the reference population for the assessment.

1840. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.118 (1 – 0.882; Horswill and Robinson 2015), 17 breeding adults from the SPA population would be expected to die each year.

##### Functional linkage and seasonal apportionment of potential effects

1841. The mean maximum foraging range of great skua is 443.3km (±487.9km) and the maximum foraging range is 1,003km (Woodward *et al*., 2019). The Project is located approximately 509km from Handa SPA, which means that the project is beyond the mean maximum foraging range of great skuas breeding at this SPA, but within the maximum foraging range +1SD, and the maximum foraging range.

##### Potential effects on the qualifying feature from the Project-alone

**Operation and maintenance phase collision risk**

1842. The great skua qualifying feature of the Handa SPA has been screened into the Appropriate Assessment due to the potential risk of collision. However, this species was not recorded within the windfarm site during site surveys, and therefore there would be no risk that collision mortality would affect great skua populations from Handa SPA. It is noted that a separate assessment of collision risk to migrant great skuas has also been undertaken, as set out in Chapter 12 of the ES. This also predicted negligible annual mortality for this species (0.03 birds), which would equate to no measurable increase in mortality apportioned to populations from Handa SPA. **It is concluded that there would be no adverse effect on the integrity of Handa SPA**.

1843. The confidence in the assessment is high. As both the surveys of the Project site and separate migrant collision risk assessment indicate that there would be negligible collision impacts on this species, it is considered extremely unlikely that there would be any effects on populations from the SPA.

### Potential effects on the qualifying feature in-combination with other projects

1844. As the Project would have no measurable effect on great skua populations from the Handa SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of Handa SPA, when assessed in-combination with other plans or projects**.

### 8.45.3.3 Kittiwake

### Status

1845. The Handa SPA breeding kittiwake population was cited as 7,420 pairs, or 14,840 breeding adults, in 1990 (Furness, 2015, Stroud *et al*., 2016). Furness (2015) gave a population of 1,872 pairs, or 3,744 breeding adults, in 2013. The most recent count is 2,575 pairs (AON), or 5,150 breeding adults, in 2018 (JNCC, 2023a); this is used as the reference population for the assessment.

1846. Based on the most recent SPA population of assumed breeding adults and an annual baseline adult mortality rate of 0.146 (1 – 0.854; Horswill and Robinson 2015), 752 breeding adults from the SPA population would be expected to die each year.

### Functional linkage and seasonal apportionment of potential effects

1847. The mean maximum foraging range of kittiwake is 156.1km (±144.5km) and the maximum foraging range is 770km (Woodward *et al*., 2019). The Project is approximately 509km from Handa SPA, which means that the Project is beyond the mean maximum foraging range +1SD of kittiwakes breeding at this SPA, but within the maximum foraging range. The maximum foraging range is a poor indicator of typical foraging behaviour. It would be expected that few birds or foraging trips will occur at this distance from the colony, and even fewer with any regularity. No impacts during the breeding season are therefore apportioned to birds breeding at this colony.

1848. Outside the breeding season, breeding kittiwakes from the SPA are assumed to range widely and to mix with kittiwakes of all ages from breeding colonies in the UK and beyond. The relevant non-breeding season reference population is the UK Western Waters and Channel BDMPS, consisting of 911,586 individuals during autumn migration (September to December) and 627,816 during spring migration (January to February) (Furness, 2015).

1849. Furness (2015) estimates that 60% of the Handa SPA breeding adults are present within the UK Western Waters and Channel BDMPS during the autumn migration period, which is 2,246 birds. During the spring migration period 80% of the population is estimated to be present, which is 2,995 birds. This represents 0.25% and 0.48% of the BDMPS population for the autumn and spring periods respectively. During autumn migration and spring migration, 0.25%, and 0.48% of impacts are therefore considered to affect birds from the SPA.

### Potential effects on the qualifying feature from the Project-alone

1850. The kittiwake qualifying feature of the Handa SPA has been screened into the Appropriate Assessment due to the potential risk of collision.

#### Operation and maintenance phase collision risk

1851. Information for collision risk on breeding adult kittiwakes belonging to the Handa SPA population is presented in **Table 8.115**. Collision estimates, calculated using the sCRM, are presented by biological season. A summary of the annual outputs and the corresponding increase in the annual baseline mortality rate is also presented. Parameters used in the sCRM were agreed with Natural England during the ETG process and are described in **Chapter 12 Offshore Ornithology** and **Appendix 12.1 o**f the ES.

1852. Based on the mean collision rates, the annual total of breeding adult kittiwakes from Handa SPA at risk of collision as a result of the Project is less than one bird (0.02). This would increase the existing mortality of the SPA breeding population by 0.00%.

*Table 8.115 Predicted seasonal and annual collision mortality (Stochastic model Option 2, avoidance rate 0.993 (±0.0003)), for breeding adult kittiwakes at the windfarm site, apportioned to Handa SPA, with corresponding increases to baseline mortality of the population*

| Period | Breeding Season<br>Mar-Aug | Autumn Migration<br>Sep-Dec | Non-breeding/winter<br>- | Spring Migration<br>Jan-Feb | Annual<br>Jan-Dec |
|---|---|---|---|---|---|
| Total collisions[1] (mean and 95% CIs) | 15.36<br>(4.17-33.87) | 8.49<br>(2.20-18.85) | - | 0.63<br>(0.00-1.46) | 24.48<br>(6.37-54.17) |
| % apportioned to the SPA | 0.00% | 0.25% | - | 0.48% | - |
| Total SPA collisions (mean and 95% CIs) | 0.00<br>(0.00-0.00) | 0.02<br>(0.01-0.05) | - | 0.00<br>(0.00-0.01) | 0.02<br>(0.01-0.05) |
| Mortality increase[2] (mean and 95% CIs) | 0.00%<br>(0.00-0.00%) | 0.00%<br>(0.00-0.01%) | - | 0.00%<br>(0.00-0.00%) | 0.00%<br>(0.00-0.01%) |

[1] Breeding season collision values reduced to 94.1% of modelled value to reflect proportion of adult birds recorded during site surveys
[2] Assuming predicted annual SPA mortality of 752 birds (5,150 x 0.146)

1853. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that no detectable changes in mortality rates would occur on this population for the mean monthly collision estimates for the Project, or for the upper 95% CI collision estimate.

1854. **It is concluded that based on predicted kittiwake mortality due to collision at the windfarm site there is no potential for the Project to have an adverse effect on the integrity of the Handa SPA.**

1855. The confidence in the assessment is high. The evidence used to define the CRM input parameters presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** as part of the ES submitted alongside this RIAA is of high applicability and quality. Whilst there is uncertainty around some of the input parameters (e.g. avoidance rate), the rates selected are considered to be sufficiently precautionary based on expert opinion to provide confidence that collision rates are not underestimated.

### Potential effects on the qualifying feature in-combination with other projects

1856. As the Project would have no measurable effect on kittiwake populations from the Handa SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of Handa SPA, when assessed in-combination with other plans or projects**.

### 8.45.3.4 Guillemot

### Status

1857. The Handa SPA breeding guillemot population was cited as 76,105 pairs, or 152,210 breeding adults in 1994, and a breeding population of 37,993 pairs, or 75,986 breeding adults, was given for 2012 (Furness, 2015). The most recent count (2018) is 68,524 individuals (JNCC, 2023a); this is used as the reference population for the assessment.

1858. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.061 (1 – 0.939, Horswill and Robinson 2015), 4,180 breeding adults from the SPA population would be expected to die each year.

### Functional linkage and seasonal apportionment of potential effects

1859. The mean maximum foraging range of guillemot is 73.2km (±80.5km) and the maximum foraging range is 338km (Woodward *et al*., 2019). The Project is located approximately 509km from Handa SPA, which means that the Project is beyond the maximum foraging range for guillemots from the SPA. No impacts during the breeding season from the Project are therefore apportioned to guillemots breeding at this SPA.

1860. Outside the breeding season, breeding guillemots from the SPA are assumed to range widely and to mix with guillemots of all ages from breeding colonies in the UK and beyond. The relevant non-breeding season reference population is the UK Western Waters BDMPS, consisting of 1,139,220 individuals (August to February) (Furness, 2015).

1861. Furness (2015) estimates that 95% of the Handa SPA breeding adults are present within the UK Western Waters BDMPS during the non-breeding season, which is 72,187 birds. This represents 6.3% of the BDMPS population for this period (1,139,220). It is therefore assumed that 6.3% of guillemots present at the Project site are breeding adults from Handa SPA.

## Potential effects on the qualifying feature

### Operation and maintenance phase disturbance/displacement/barrier effects

*Project-alone*

1862. The mean peak abundance of guillemots present within the windfarm site and 2km buffer during the non-breeding season is 8,315 (6,085-12,047) individuals (refer to **Appendix 12.1** of the ES). Of these, 524 birds (383-759) are likely to be breeding adults from the Handa SPA.

1863. **Table 8.116** sets out the predicted impacts on guillemots from Handa SPA during the non-breeding season. Displacement rates of 30% to 70% are considered for this species, along with a range of mortality rates of 1% to 10% of displaced birds (UK SNCBs, 2017).

1864. The available evidence suggests that the upper ranges of these displacement and mortality rates may be excessively precautionary. Whilst it is true that guillemots and razorbills tend to be displaced by OWFs they do not avoid them completely, and displacement rates vary between sites. MacArthur Green (2019b) concluded that, on average, densities within OWFs were around half of the density found in the habitats around the OWF. At some OWFs there was also displacement of birds from a buffer zone surrounding it. The size of the buffer zone varied between OWFs and was generally less than 2km, with auk density increasing across the buffer zone as distance from turbines increased, up to the density in the wider area. Another recent review (APEM, 2022) found that the current displacement rates suggested by the SNCBs for guillemot and razorbill (30% to 70%) did not account for the quality of or confidence in the studies which were used to inform this position, and that studies where no significant effects were recorded were not accounted for during the provision of the advice. APEM (2022) suggested that in the case of Hornsea Project Four, an evidence-based displacement rate of 50% was appropriate. However, the study also recognised that larger displacement effects are possible.

1865. Mortality due to displacement could arise due to increased energy costs and/or decreased energy intake, if displacement results in increased flying time to avoid OWFs, and/or increased bird densities and competition for prey in areas of unimpacted habitat outside the OWF. Given that UK OWFs in the Irish and Celtic Seas represent a very small proportion of the available foraging habitat for guillemots and razorbills within UK Western Waters, the increase in density of auks outside OWFs due to displacement is likely to be negligible (MacArthur Green, 2019b).

1866. It is considered unlikely that the mortality rate due to displacement would be as high as the empirically estimated 6.1% annual mortality for adult guillemots that occurs due to the combination of ‘natural’ factors and anthropogenic activities (Horswill and Robinson, 2015). Indeed, it may be much lower; MacArthur Green (2019b) recommended precautionary rates of 50% displacement for guillemot and 1% mortality of displaced birds based on a review of available evidence.

1867. Modelling undertaken by APEM (2022) found that in the case of Hornsea Project Four, a mortality rate of 1% for displaced auks was considered to be precautionary, and that this may be an overestimate given that Hornsea Project Four is located towards the upper end of the mean maximum foraging range for guillemot from the closest designated sites. It is also noted that in the recent decision on Hornsea Project Four, the SoS determined that a rate of 70% displacement/2% mortality was appropriate to inform the assessment of effects on guillemots from Flamborough and Filey Coast SPA (DESNZ, 2023b).

1868. The full ranges of recommended displacement and mortality effects are considered by the assessment, along with evidence-based displacement and mortality rates of 50% and 1%, respectively (APEM, 2022; MacArthur Green, 2019b).

*Table 8.116 Guillemot – predicted operation and maintenance phase displacement and mortality from Handa SPA*

| Mean peak abundance estimate type | Mean peak abundance estimate | Number of Handa SPA breeding adults present (non-breeding season)[1] | Annual mortality range[2] | Annual baseline mortality increase range[3] |
|---|---|---|---|---|
| Upper 95% CI | 12,047 | 759 | 2-53 | 0.05-1.27% |
| Mean | 8,315 | 524 | 2-37 | 0.04-0.88% |

| Mean peak abundance estimate type | Mean peak abundance estimate | Number of Handa SPA breeding adults present (non-breeding season)[1] | Annual mortality range[2] | Annual baseline mortality increase range[3] |
|---|---|---|---|---|
| Lower 95% CI | 6,085 | 383 | 1-27 | 0.03-0.64% |
| [1] Assumes 6.3% of birds present during the non-breeding season are Handa SPA breeding adults<br>[2] Assumes displacement rates of 30-70% and mortality rates of 1-10%<br>[3] Background mortality rate of 6.1% (Horswill and Robinson, 2015) | | | | |

1869. Based on the mean peak abundances, the annual total of guillemots from the Handa SPA at risk of displacement is 524 birds (**Table 8.116**). At displacement rates of 30% to 70%, and mortality rates of 1% to 10% for displaced birds, 2 to 37 SPA breeding adults would be predicted to die each year due to displacement from the Project.

1870. Assuming a displacement rate of 70% and a mortality rate of 10% of displaced birds, annual mortality within the SPA breeding adult population would increase by 0.88%. Using an evidence-based displacement rate of 50%, and a mortality rate for displaced birds of 1%, annual mortality in the population would instead increase by 0.06% (3 birds).

1871. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that no detectable changes in mortality rates would occur under any combination of displacement and mortality rates when the mean peak abundance estimate assessments are considered.

1872. Increases of over 1% are predicted if the upper 95% CIs for mean peak abundances are used as inputs to the assessment alongside a 10% mortality rate for displaced birds. The probability of this occurring is extremely small for two reasons. Firstly, the upper 95% CI for the mean peak abundances are highly unlikely to occur regularly at the windfarm site. Secondly, mortality rates for displaced birds of 10% are much higher than evidence suggested will actually be the case, and use of the evidence-based displacement (50%) and mortality rate (1%) (and also 70%/2%) would again result in a mortality increase of significantly less than 1%.

1873. **It is concluded that predicted guillemot mortality due to operational phase displacement for the project alone would not adversely affect the integrity of the Handa SPA.**

1874. The confidence in the assessment is high for several reasons. Firstly, the evidence used to inform the evidence-based displacement rates is of high applicability and quality (based on the criteria discussed in **Chapter 12**

**Offshore Ornithology** of the ES). Whilst there is limited available evidence to inform mortality rates, 1% is considered to be sufficiently precautionary based on expert opinion. This species is not regarded as being highly specialised in its habitat requirements (Bradbury *et al*., 2014; Furness and Wade, 2012; Garthe and Hüppop, 2004), and it is therefore anticipated that displaced birds will find alternative habitat in the vast majority of cases. Finally, the conclusion of the assessment is the same irrespective of whether the mean or 95% upper CI mean peak abundances are used to calculate potential mortality and increases in the baseline mortality rate of the background population, provided the evidence-based displacement and mortality rates are used.

*In-combination*

1875. The in-combination assessment for guillemots from Handa SPA has been undertaken in accordance with the approach presented in **Section 8.1**. The total population apportioned to Handa SPA at risk of displacement is estimated to be 3,123 breeding adults (**Appendix 12.1** of the ES). Annual in-combination displacement and mortality rates for birds from Handa SPA are presented in **Table 8.117**.

1876. Assuming a displacement rate of 70% and a mortality rate of 10% of displaced birds, 219 breeding adult SPA birds would be lost to displacement annually. Including additional apportioned mortality from the Morlais and Holyhead tidal projects (total 1.98 birds), this would increase the existing mortality within the SPA population (4,180 breeding adult birds per year) by 5.28%. Using an evidence-based displacement rate of 50%, and a mortality rate for displaced birds of 1%, the annual in-combination displacement mortality would be 16 birds. This would increase the existing mortality within this population by 0.42%. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that detectable changes in mortality rates will not occur due to the level or mortality predicted if the more realistic rates for mortality are used.

1877. **It is concluded that predicted guillemot mortality due to of operational phase displacement impacts of the Project in-combination with other projects, would not adversely affect the integrity of Handa SPA.**

*Table 8.117 In-combination year-round displacement matrix for guillemot from Handa SPA*

| **Annual** | **Mortality** | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| **Displacement** | 1% | 2% | 3% | 4% | 5% | 10% | 20% | 30% | **50%** | **80%** | **100%** |
| **10%** | 3 | 6 | 9 | 12 | 16 | 31 | 62 | 94 | 156 | 250 | 312 |
| **20%** | 6 | 12 | 19 | 25 | 31 | 62 | 125 | 187 | 312 | 500 | 625 |
| **30%** | **9** | **19** | **28** | **37** | **47** | **94** | 187 | 281 | 468 | 750 | 937 |
| **40%** | **12** | **25** | **37** | **50** | **62** | **125** | 250 | 375 | 625 | 999 | 1249 |
| **50%** | **16** | **31** | **47** | **62** | **78** | **156** | 312 | 468 | 781 | 1249 | 1561 |
| **60%** | **19** | **37** | **56** | **75** | **94** | **187** | 375 | 562 | 937 | 1499 | 1874 |
| **70%** | **22** | **44** | **66** | **87** | **109** | **219** | 437 | 656 | 1093 | 1749 | 2186 |
| **80%** | 25 | 50 | 75 | 100 | 125 | 250 | 500 | 750 | 1249 | 1999 | 2498 |
| **90%** | 28 | 56 | 84 | 112 | 141 | 281 | 562 | 843 | 1405 | 2249 | 2811 |
| **100%** | 31 | 62 | 94 | 125 | 156 | 312 | 625 | 937 | 1561 | 2498 | 3123 |

Note: The cells show the number of birds predicted to die (rounded to the nearest integer) at a given rate of displacement and mortality. Highlighted cells are considered to be the most realistic scenario, in accordance with SNCB advice (SNCBs, 2022).

#### 8.45.3.5 Razorbill

##### Status

1878. The Handa SPA breeding razorbill population was cited as 10,432 pairs, or 20,864 breeding adults, in 1997 (Furness, 2015, Stroud *et al*., 2016). Furness (2015) gave a breeding population of 5,156 pairs, or 10,312 breeding adults in 2010. The most recent count (2019) is 8,207 individuals (JNCC, 2023a); this is used as the reference population for the assessment.

1879. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.105 (1 – 0.895; Horswill and Robinson 2015), 862 breeding adults from the SPA population would be expected to die each year.

##### Functional linkage and seasonal apportionment of potential effects

1880. The mean maximum foraging range of razorbill is 88.7km (±75.9km) and the maximum foraging range is 313km (Woodward *et al*., 2019). The Project is located approximately 509km from Handa SPA, which means that the Project is beyond the maximum foraging range for razorbills from the SPA. No impacts during the breeding season from the Project are therefore apportioned to guillemots breeding at this SPA.

1881. Outside the breeding season, breeding razorbills from the SPA are assumed to range widely and to mix with razorbills of all ages from breeding colonies in the UK and beyond. The relevant non-breeding season reference population is the UK Western Waters BDMPS, consisting of 606,914 individuals during autumn and spring passage periods (August to October and January to March), and 341,422 individuals during winter (November and December) (Furness, 2015). During autumn and spring migration, 98% of the SPA breeding adults (10,123 individuals based on the 2010 population estimate) are assumed to be present in the BDMPS, representing 1.7% of the BDMPS population (606,914 individuals of all ages). During the winter season, 40% of the SPA breeding adults (4,132 individuals based on the 2010 population estimate) are assumed to be present in the BDMPS, representing 1.2% of the BDMPS population (341,422 individuals of all ages). These percentages (i.e. 1.7% and 1.2%) are the proportions of birds present at the windfarm site that are presumed to originate from the Handa SPA during the relevant seasons.

## Potential effects on the qualifying feature

### Operation and maintenance phase disturbance/displacement/barrier effects

*Project-alone*

1882. The year-round mean peak abundance of razorbills present within the windfarm site and 2km buffer was 1,979 (703-3,552) individuals (refer to **Appendix 12.1** of the ES). Of these, 26 birds (11-44) were likely to be breeding adults from the Handa SPA.

1883. **Table 8.118** sets out the predicted impacts on razorbills from Handa SPA. Displacement rates of 30% to 70% are considered for this species, along with a range of mortality rates of 1% to 10% of displaced birds (UK SNCBs, 2017).

1884. The available evidence suggests that the upper ranges of these displacement and mortality rates may be excessively precautionary. Whilst it is true that guillemots and razorbills tend to be displaced by OWFs they do not avoid them completely, and displacement rates vary between sites. MacArthur Green (2019b) concluded that, on average, densities within OWFs were around half of the density found in the habitats around the OWF. At some OWFs there was also displacement of birds from a buffer zone surrounding it. The size of the buffer zone varied between OWFs and was generally less than 2km, with auk density increasing across the buffer zone as distance from turbines increased, up to the density in the wider area. Another recent review (APEM, 2022) found that the current displacement rates suggested by the SNCBs for guillemot and razorbill (30% to 70%) did not account for the quality of or confidence in the studies which were used to inform this position, and that studies where no significant effects were recorded were not accounted for during the provision of the advice. APEM (2022) suggested that in the case of Hornsea Project Four, an evidence-based displacement rate of 50% was appropriate. However, the study also recognised that larger displacement effects are possible.

1885. Mortality due to displacement could arise due to increased energy costs and/or decreased energy intake, if displacement results in increased flying time to avoid OWFs, and/or increased bird densities and competition for prey in areas of unimpacted habitat outside the OWF. Given that UK OWFs in the Irish and Celtic Seas represent a very small proportion of the available foraging habitat for guillemots and razorbills within UK Western Waters, the increase in density of auks outside OWFs due to displacement is likely to be negligible (MacArthur Green, 2019b). It is considered unlikely that the mortality rate due to displacement would be as high as the empirically estimated 10.5% annual mortality for adult razorbills that occurs due to the combination of 'natural' factors and anthropogenic activities (Horswill and Robinson, 2015). Indeed, it may be much lower; MacArthur Green (2019b) recommended precautionary rates of 50% displacement for auks and 1% mortality of displaced birds based on a review of available evidence. Modelling undertaken by APEM (2022) found that in the case of Hornsea Project Four, a mortality rate of 1% for displaced auks was considered to be precautionary, and that this may be an overestimate given that Hornsea Project Four is located towards the upper end of the mean maximum foraging range for guillemot from the closest designated sites.

1886. The full ranges of recommended displacement and mortality effects are considered by the assessment, along with evidence-based displacement and mortality rates of 50% and 1%, respectively (APEM, 2022; MacArthur Green, 2019b).

*Table 8.118 Razorbill – predicted operation and maintenance phase displacement and mortality from Handa SPA*

| Mean peak abundance estimate type | Mean peak abundance estimate by season[1] | Number of Handa SPA breeding adults present by season)[2] | Annual mortality range[3] | Annual baseline mortality increase range[4] |
|---|---|---|---|---|
| Upper 95% CI | 605 (b)<br>1,070 (aut)<br>1,297 (win)<br>580 (spr)<br>3,552 (year round) | 0 (b)<br>18 (aut)<br>16 (win)<br>10 (spr)<br>44 (year round) | 0-3 (0) | 0.02-0.35% (0.03%) |
| Mean | 252 (b)<br>694 (aut)<br>651 (win)<br>381 (spr)<br>1,979 (year round) | 0 (b)<br>12 (aut)<br>8 (win)<br>6 (spr)<br>26 (year round) | 0-4 (0) | 0.01-0.21% (0.02%) |
| Lower 95% CI | 21 (b)<br>309 (aut)<br>159 (win)<br>214 (spr)<br>703 (year round) | 0 (b)<br>5 (aut)<br>2 (win)<br>4 (spr)<br>11 (year round) | 0-1 (0) | 0.00-0.09% (0.01%) |

[1] Breeding season = b, autumn migration season = aut, winter season = win, spring migration season = spr

[2] Assumes breeding adult apportioning of 0.0% (breeding season), 1.7% (spring and autumn migration) and 1.2% (winter) to Handa SPA.

[3] Assumes displacement rates of 30-70% and mortality rates of 1-10%. Evidence-based estimates assuming a 50% displacement rate and 1% mortality of displaced birds are presented in parentheses.

[4] Background mortality rate of 10.5% (Horswill and Robinson, 2015)

1887. Based on the mean peak abundances, the annual total of razorbills from the Handa SPA at risk of displacement is 26 birds (**Table 8.118**). At displacement rates of 30% to 70%, and mortality rates of 1% to 10% for displaced birds, 0 to 4 SPA breeding adults would be predicted to die each year due to displacement from the Project.

1888. Assuming a displacement rate of 70% and a mortality rate of 10% of displaced birds, annual mortality within the SPA breeding adult population would increase by 0.21%. Using an evidence-based displacement rate of 50%, and a mortality rate for displaced birds of 1%, annual mortality in the population would instead increase by 0.02% (<1 bird).

1889. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that no detectable changes in mortality rates would occur under almost any combination of displacement and mortality rates when the mean peak abundance estimate assessments are considered. This would be the case even when upper 95% CIs are considered.

1890. **It is concluded that predicted razorbill mortality due to operational phase displacement for the project alone would not adversely affect the integrity of the Handa SPA.**

1891. The confidence in the assessment is high for several reasons. Firstly, the evidence used to inform the evidence-based displacement rates is of high applicability and quality (based on the criteria discussed in **Chapter 12 Offshore Ornithology** of the ES). Whilst there is limited available evidence to inform mortality rates, 1% is considered to be sufficiently precautionary based on expert opinion. This species is not regarded as being highly specialised in its habitat requirements (Bradbury *et al*., 2014; Furness and Wade, 2012; Garthe and Hüppop, 2004), and it is therefore anticipated that displaced birds will find alternative habitat in the vast majority of cases. Finally, the conclusion of the assessment is the same irrespective of whether the mean or 95% upper CI mean peak abundances are used to calculate potential mortality and increases in the baseline mortality rate of the background population, provided the evidence-based displacement and mortality rates are used.

*In-combination*

1892. The in-combination assessment for razorbills from Handa SPA has been undertaken in accordance with the approach presented in **Section 8.1**. The total population apportioned to Handa SPA at risk of displacement is estimated to be 181 breeding adults (**Appendix 12.1** of the ES). Annual in-combination displacement and mortality rates for birds from Handa SPA are presented in **Table 8.119**.

1893. Assuming a displacement rate of 70% and a mortality rate of 10% of displaced birds, 13 breeding adult SPA birds would be lost to displacement annually. Including additional apportioned mortality from the Morlais and Holyhead tidal projects (total 0.25 birds), this would increase the existing mortality within the SPA population (862 breeding adult birds per year) by 1.50%. Using an evidence-based displacement rate of 50%, and a mortality rate for displaced birds of 1%, the annual in-combination mortality would be one bird. This would increase the existing mortality within this population by 0.13%. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that detectable changes in mortality rates will not occur due to the level or mortality predicted if the more realistic rates for mortality are used.

1894. **It is concluded that predicted razorbill mortality due to of operational phase displacement impacts of the Project in-combination with other projects, would not adversely affect the integrity of Handa SPA.**

*Table 8.119 In-combination year-round displacement matrix for razorbill from Handa SPA*

| Annual | Mortality | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| **Displacement** | **1%** | **2%** | **3%** | **4%** | **5%** | **10%** | **20%** | **30%** | **50%** | **80%** | **100%** |
| **10%** | 0 | 0 | 1 | 1 | 1 | 2 | 4 | 5 | 9 | 15 | 18 |
| **20%** | 0 | 1 | 1 | 1 | 2 | 4 | 7 | 11 | 18 | 29 | 36 |
| **30%** | 1 | 1 | 2 | 2 | 3 | 5 | 11 | 16 | 27 | 44 | 54 |
| **40%** | 1 | 1 | 2 | 3 | 4 | 7 | 15 | 22 | 36 | 58 | 73 |
| **50%** | 1 | 2 | 3 | 4 | 5 | 9 | 18 | 27 | 45 | 73 | 91 |
| **60%** | 1 | 2 | 3 | 4 | 5 | 11 | 22 | 33 | 54 | 87 | 109 |
| **70%** | 1 | 3 | 4 | 5 | 6 | 13 | 25 | 38 | 63 | 102 | 127 |
| **80%** | 1 | 3 | 4 | 6 | 7 | 15 | 29 | 44 | 73 | 116 | 145 |
| **90%** | 2 | 3 | 5 | 7 | 8 | 16 | 33 | 49 | 82 | 131 | 163 |
| **100%** | 2 | 4 | 5 | 7 | 9 | 18 | 36 | 54 | 91 | 145 | 181 |

Note: The cells show the number of birds predicted to die (rounded to the nearest integer) at a given rate of displacement and mortality. Highlighted cells are considered to be the most realistic scenario, in accordance with SNCB advice (SNCBs, 2022).

## 8.46 North Caithness Cliffs SPA

1895. North Caithness Cliffs SPA is located approximately 524km from the windfarm site.

### 8.46.1 Description of designation

1896. North Caithness Cliffs SPA is of special nature conservation and scientific importance within Britain and Europe for supporting very large populations of breeding seabirds. The seaward extension of the SPA extends 2km into the marine environment and includes the seabed, water column and surface. Seabirds included within the designation feed both inside and outside the SPA in nearby waters, as well as more distantly in the wider North Sea. Qualifying seabird species of the SPA comprise fulmar, kittiwake, guillemot, razorbill and puffin.

### 8.46.2 Conservation objectives

1897. The overarching conservation objectives for the site are:

- To avoid deterioration of the habitats of the qualifying species or significant disturbance to the qualifying species, thus ensuring that the integrity of the site is maintained
- To ensure for the qualifying species that the following are maintained in the long term:
    - Population of the species as a viable component of the site
    - Distribution of the species within site
    - Distribution and extent of habitats supporting the species
    - Structure, function and supporting processes of habitats supporting the species
    - No significant disturbance of the species

### 8.46.3 Assessment

1898. The qualifying features of North Caithness Cliffs SPA screened into the Appropriate Assessment are listed in **Table 5.2**. These are breeding fulmar and breeding kittiwake. The seabird assemblage has also been screened in for these species.

#### 8.46.3.1 Fulmar

##### Status

1899. The North Caithness Cliffs SPA breeding fulmar population was cited as 16,310 pairs, or 32,620 breeding adults, in 1996 (Stroud *et al*., 2016). Furness (2015) gave a breeding population of 14,250 pairs, or 28,500 breeding adults, in 2000. The most recent count is 8,619 pairs (AOS), or 17,238 breeding adults, in 2016 (JNCC, 2023a); this is used as the reference population for the assessment.

1900. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.064 (1 – 0.936; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 1103 breeding adults.

##### Functional linkage and seasonal apportionment of potential effects

1901. The mean maximum foraging range of fulmar is 542.3km (±657.9km) and the maximum foraging range is 2,736km (Woodward *et al*., 2019). The Project is located approximately 524km from North Caithness Cliffs SPA, which means that the Project is within the mean maximum foraging range of fulmars breeding at this SPA.

##### Potential effects on the qualifying feature from the Project-alone

1902. Fulmar are considered to have low vulnerability to both collision and displacement impacts from offshore windfarms (Bradbury *et al*., 2014). This species was recorded infrequently within the windfarm site (in only five of 24 surveys, all during the breeding season (January to August; Furness, 2015)) and in low numbers, with densities in four of the five months less than 0.1 birds/km$^2$. A peak density of 0.23 birds/km$^2$ was recorded in May 2022.

1903. On the basis of the low vulnerability of fulmars to potential impacts, and the low frequency and abundance of birds present at the windfarm site, it is concluded that any effects on breeding populations at North Caithness Cliffs SPA are very unlikely, both during and outside of the breeding season.

1904. **It is therefore concluded that there would be no measurable effects on fulmar due to the project alone, and no adverse effect on the integrity of the North Caithness Cliffs SPA is predicted.**

##### Potential effects on the qualifying feature in-combination with other projects

1905. As the Project would have no measurable effect on fulmar populations from the North Caithness Cliffs SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there**

**would be no adverse effect on the integrity of North Caithness Cliffs SPA, when assessed in-combination with other plans or projects.**

### 8.46.3.2 Kittiwake

#### Status

1906. The North Caithness Cliffs SPA breeding kittiwake population was cited as 15,650 pairs, or 31,300 breeding adults, in 1996 (Furness, 2015, Stroud *et al*., 2016). Furness (2015) gave a breeding population of 10,150 pairs, or 20,300 breeding adults, in 2000. The most recent count is 3,778 pairs (AON), or 7,556 breeding adults, in 2016 (JNCC, 2023a). However, it is not clear if this encompasses all of the SPA breeding sites. Therefore, the most recent population is assumed to be 20,300 breeding adults; this is used as the reference population for the assessment.

1907. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.146 (1 – 0.854; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 2,964 breeding adults.

#### Functional linkage and seasonal apportionment of potential effects

1908. The mean maximum foraging range of kittiwake is 156.1km (±144.5km) and the maximum foraging range is 770km (Woodward *et al*., 2019). The Project is approximately 524km from North Caithness Cliffs SPA, which means that the Project is beyond the mean maximum foraging range +1SD of kittiwakes breeding at this SPA, but within the maximum foraging range. The maximum foraging range is a poor indicator of typical foraging behaviour. It would be expected that few birds or foraging trips will occur at this distance from the colony, and even fewer with any regularity. No impacts during the breeding season are therefore apportioned to birds breeding at this colony.

1909. Outside the breeding season, breeding kittiwakes from the SPA are assumed to range widely and to mix with kittiwakes of all ages from breeding colonies in the UK and beyond. The relevant non-breeding season reference population is the UK Western Waters and Channel BDMPS, consisting of 911,586 individuals during autumn migration (September to December) and 627,816 during spring migration (January to February) (Furness, 2015).

1910. Furness (2015) estimated that 20% of the North Caithness Cliffs SPA breeding adults are present within the UK Western Waters and Channel BDMPS during the autumn migration period, which is 4,060 birds. During the spring migration period 30% of the population is estimated to be present, which is 6,090 birds. This represents 0.45% and 0.97% of the BDMPS population for the autumn and spring periods respectively. During autumn migration and spring migration, 0.45%, and 0.97% of impacts are therefore considered to affect birds from the SPA.

## Potential effects on the qualifying feature from the Project-alone

1911. The kittiwake qualifying feature of the North Caithness Cliffs SPA has been screened into the Appropriate Assessment due to the potential risk of collision.

### Operation and maintenance phase collision risk

1912. Information for collision risk on breeding adult kittiwakes belonging to the North Caithness Cliffs SPA population is presented in **Table 8.120**. Collision estimates, calculated using the sCRM, are presented by biological season. A summary of the annual outputs and the corresponding increase in the annual baseline mortality rate is also presented. Parameters used in the sCRM were agreed with Natural England during the ETG process and are described in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES.

1913. Based on the mean collision rates, the annual total of breeding adult kittiwakes from North Caithness Cliffs SPA at risk of collision as a result of the Project is less than one bird (0.04). This would increase the existing mortality of the SPA breeding population by 0.00%.

*Table 8.120 Predicted seasonal and annual collision mortality (Stochastic model Option 2, avoidance rate 0.993 (±0.0003)), for breeding adult kittiwakes at the windfarm site, apportioned to North Caithness Cliffs SPA, with corresponding increases to baseline mortality of the population*

| Period | Breeding Season<br>Mar-Aug | Autumn Migration<br>Sep-Dec | Non-breeding/winter<br>- | Spring Migration<br>Jan-Feb | Annual<br>Jan-Dec |
|---|---|---|---|---|---|
| Total collisions[1] (mean and 95% CIs) | 15.36<br>(4.17-33.87) | 8.49<br>(2.20-18.85) | - | 0.63<br>(0.00-1.46) | 24.48<br>(6.37-54.17) |
| % apportioned to the SPA | 0.00% | 0.45% | - | 0.97% | - |
| Total SPA collisions (mean and 95% CIs) | 0.00<br>(0.00-0.00) | 0.04<br>(0.01-0.08) | - | 0.01<br>(0.00-0.01) | 0.04<br>(0.01-0.10) |
| Mortality increase[2] (mean and 95% CIs) | 0.00%<br>(0.00-0.00%) | 0.00%<br>(0.00-0.00%) | - | 0.00%<br>(0.00-0.00%) | 0.00%<br>(0.00-0.00%) |

[1] Breeding season collision values reduced to 94.1% of modelled value to reflect proportion of adult birds recorded during site surveys
[2] Assuming predicted annual SPA mortality of 2964 birds (20,300 x 0.146)

1914. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that no detectable changes in mortality rates would occur on this population for the mean monthly collision estimates for the Project, or for the upper 95% CI collision estimate.

1915. **It is concluded that based on predicted kittiwake mortality due to collision at the windfarm site there is no potential for the Project to have an adverse effect on the integrity of the North Caithness Cliffs SPA.**

1916. The confidence in the assessment is high. The evidence used to define the CRM input parameters presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** as part of the ES submitted alongside this RIAA is of high applicability and quality. Whilst there is uncertainty around some of the input parameters (e.g. avoidance rate), the rates selected are considered to be sufficiently precautionary based on expert opinion to provide confidence that collision rates are not underestimated.

### Potential effects on the qualifying feature in-combination with other projects

1917. As the Project would have no measurable effect on kittiwake populations from the North Caithness Cliffs SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of North Caithness Cliffs SPA, when assessed in-combination with other plans or projects**.

## 8.47 St Kilda SPA

1918. St Kilda SPA is located approximately 526km from the windfarm site.

### 8.47.1 Description of designation

1919. St Kilda is a group of remote islands lying in the North Atlantic about 70 km west of North Uist in the Outer Hebrides. The islands are steep, with precipitous cliffs reaching 430m on Hirta and 380m on Soay and Boreray. The vegetation is strongly influenced by sea spray and the presence of seabirds and livestock. Inland on Hirta, species-poor acidic grassland and sub-maritime heaths occupy extensive areas. The islands provide a strategic nesting locality for seabirds that feed in the rich waters to the west of Scotland. The total population of seabirds exceeds 600,000 individuals, making this one of the largest concentrations in the North Atlantic and the largest in the UK. The boundary of the SPA overlaps with the boundary of St. Kilda SSSI, and the seaward elements extend approximately 4km into the marine environment to include the seabed, water column and surface.

### 8.47.2 Conservation objectives

1920. The overarching conservation objectives for the site are:

- To avoid deterioration of the habitats of the qualifying species or significant disturbance to the qualifying species, thus ensuring that the integrity of the site is maintained
- To ensure for the qualifying species that the following are maintained in the long term:
  - Population of the species as a viable component of the site
  - Distribution of the species within site
  - Distribution and extent of habitats supporting the species
  - Structure, function and supporting processes of habitats supporting the species
  - No significant disturbance of the species

### 8.47.3 Assessment

1921. The qualifying features of St Kilda SPA screened into the Appropriate Assessment are listed in **Table 5.2**. These are fulmar, Manx shearwater, Leach's storm-petrel, great skua, guillemot, puffin and gannet. The seabird assemblage has also been screened in for these species.

#### 8.47.3.1 Fulmar

##### Status

1922. The St Kilda SPA breeding fulmar population was cited as 62,800 pairs, or 125,600 breeding adults in 1992 (Furness, 2015, Stroud *et al*., 2016). Furness (2015) gave the breeding population of 66,055 pairs, or 132,110 breeding adults, in 1999. The most recent counts (covering 2015 and 2016) identify 29,186 AOS, or 58,372 breeding adults (JNCC, 2022); however, it is not clear if this encompasses all the SPA breeding sites. Therefore, the most recent population is assumed to be 132,110 breeding adults; this is used as the reference population for the assessment.

1923. Based on the most recent SPA population of breeding adults, and an annual breeding adult baseline mortality rate of 0.064 (1 – 0.936; Horswill and Robinson 2015), 8,455 breeding adults from the SPA population would be expected to die each year.

##### Functional linkage and seasonal apportionment of potential effects

1924. The mean maximum foraging range of fulmar is 542.3km (±657.9km) and the maximum foraging range is 2,736km (Woodward *et al*., 2019). The Project is located approximately 526km from St Kilda SPA, which means that the Project is within the mean maximum foraging range of fulmars breeding at this SPA.

##### Potential effects on the qualifying feature from the Project-alone

1925. Fulmar are considered to have low vulnerability to both collision and displacement impacts from offshore windfarms (Bradbury *et al*., 2014). This species was recorded infrequently within the windfarm site (in only five of 24 surveys, all during the breeding season (January to August; Furness, 2015)) and in low numbers, with densities in four of the five months less than 0.1 birds/km$^2$. A peak density of 0.23 birds/km$^2$ was recorded in May 2022.

1926. On the basis of the low vulnerability of fulmars to potential impacts, and the low frequency and abundance of birds present at the windfarm site, it is concluded that any effects on breeding populations at St Kilda SPA are very unlikely, both during and outside of the breeding season.

1927. **It is therefore concluded that there would be no measurable effects on fulmar due to the project alone, and no adverse effect on the integrity of the St Kilda SPA is predicted.**

##### Potential effects on the qualifying feature in-combination with other projects

1928. As the Project would have no measurable effect on fulmar populations from the St Kilda SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse**

**effect on the integrity of St Kilda SPA, when assessed in-combination with other plans or projects.**

#### 8.47.3.2 Manx shearwater

##### Status

1929. The St Kilda SPA breeding Manx shearwater population at classification was cited as <5,000 pairs, or <10,000 breeding adults, in 2001, and a breeding population of 4,802 pairs, or 9,604 breeding adults, was counted in 1999 (Furness, 2015). There are no recent (post-2000) counts on the Seabird Monitoring Programme database (JNCC, 2033). Therefore, the most recent accurate population estimate is taken to be 9,604 breeding adults, this is used as the reference population for the assessment.

1930. Based on the most recent SPA population of breeding adults, and an annual breeding adult baseline mortality rate of 0.13 (1 – 0.870; Horswill and Robinson 2015), 1,249 breeding adults from the SPA population would be expected to die each year.

##### Functional linkage and seasonal apportionment of potential effects

1931. The mean maximum foraging range of Manx shearwater is 1,346.8km (±1018.7km) and the maximum foraging range is 2890km. The Project is located approximately 526km from St Kilda SPA, which means that the Project is within the mean maximum foraging range of Manx shearwaters breeding at this SPA.

1932. A number of SPA and non-SPA Manx shearwater colonies are located in and around the UK Western Waters BDMPS area, all of which are within the mean maximum foraging range of this species. For a review of these sites see **Section 8.21.3.1**.

1933. The NatureScot apportioning tool (NatureScot, 2018) has been used to estimate the proportion of Manx shearwaters from each of the relevant SPAs present at the windfarm site during the breeding season. The apportioning to SPA and non-SPA colonies is set out in **Table 8.121**; refer also to **Appendix 12.1** of the ES for further information on the apportioning approach and results. Accordingly, 0.20% of impacts at the windfarm site during the breeding season are apportioned to St Kilda SPA.

*Table 8.121 Manx shearwater breeding season apportioning*

| Site | Apportioning rate |
|---|---|
| Glannau Aberdaron ac Ynys Enlli/Aberdaron Coast and Bardsey Island SPA | 8.63% |

| Site | Apportioning rate |
|---|---|
| Copeland Islands SPA | 2.22% |
| Skomer, Skokholm and the Seas off Pembrokeshire / Sgomer, Sgogwm a Moroedd Penfro SPA | 76.54% |
| Rum SPA | 8.44% |
| **St Kilda SPA** | **0.20%** |
| Cruagh Island SPA (transboundary site) | 0.10% |
| Blasket Islands SPA (transboundary site) | 0.61% |
| Deenish Island and Scariff Island SPA (transboundary site) | 0.08% |
| Puffin Island SPA (transboundary site) | 0.22% |
| Skelligs SPA (transboundary site) | 0.03% |
| Non-SPA colonies | 2.91% |

1934. During the pre- and post-breeding periods, breeding Manx shearwaters from the St Kilda SPA migrate through UK waters. The relevant reference population is considered to be the UK Western Waters BDMPS. This consists of 1,580,895 individuals during the post-breeding (August-early October) and return migration (late march-May) periods.

1935. Estimates of the proportion of Manx shearwaters present at the windfarm site which originate from the St Kilda SPA during the post-breeding and return migration periods (and therefore the proportion of predicted mortalities from the SPA population) are based on the SPA population (i.e. 240,000 adults) as a proportion of the UK Western Waters BDMPS during the relevant season. During the post-breeding and return migration periods, 0.6% of impacts are considered to affect birds from the SPA (Furness, 2015).

## Potential effects on the qualifying feature from the Project-alone

1936. The Manx shearwater qualifying feature of the St Kilda SPA has been screened into the assessment due to the potential risk of disturbance, displacement and barrier effects during the construction and decommissioning, and operation and maintenance phases of the Project.

### Construction and decommissioning phase disturbance/displacement/barrier effects

1937. Effects during the construction and decommissioning phases of the Project are considered unlikely, given the transient presence of the species and low susceptibility to disturbance related impacts; refer to **Paragraph 1939** below. However, in accordance with feedback received from Natural England and NRW, a precautionary estimation of construction and decommissioning phase disturbance, displacement and barrier effects has been undertaken assuming 50% of the operational phase effect.

1938. Applying 50% reduction to the operational values presented in **Table 8.122**, and based on mean density, predicted mortality would be between zero and one bird (30-70% displacement and 1-10% mortality of displaced birds). Using realistic values (i.e. mean density, 50% displacement and 1% mortality), there would be an annual increase in mortality of less than one (0.1) birds, which is equivalent to a 0.01% increase in background mortality for the SPA population. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. Accordingly, no significant effects on Manx shearwater are predicted during the construction and decommissioning phases, and **it is concluded that there would be no adverse effect on the integrity of St Kilda SPA**.

**Operation and maintenance phase disturbance/displacement/barrier effects**

1939. Manx shearwater are generally considered to have a low susceptibility to disturbance and displacement (Furness *et al.*, 2013). See **Section 8.21.3.1** for summary of effects from Dierschke *et al*., (2016) and Bradbury *et al.,* (2014).

1940. Displacement effects for Manx shearwater for the Project were assessed during the breeding, autumn migration and spring migration periods, based on an unapportioned peak mean population of 4,705, 2,650 and 1,617 birds respectively, calculated for the windfarm site and a 2km buffer, in line with recommendations within the SNCB guidance (SNCBs, 2017). The displacement matrices used to calculate potential impacts are presented in **Appendix 12.1** of the ES and summarised in **Table 8.122.** The application of the same displacement rate to the OWF and the 2km buffer, to determine the total number of birds subject to displacement, is precautionary, as in reality the displacement rate is likely to fall with distance from the windfarm site.

1941. A displacement rate of 30-70% and mortality rate of 1-10% has been presented. Given that 10% would represent a rate close to the expected ‘natural’ annual mortality (0.13), this rate is considered very unlikely. Accordingly, a 1% mortality rate is considered to be most appropriate, with the upper end of this range likely to be precautionary. Given the very extensive foraging range of this species (Woodward *et al.*, 2019), there may be no mortality costs to displacement from the relatively very small footprints of OWFs.

*Table 8.122 Manx shearwater – predicted operation and maintenance phase displacement and mortality from St Kilda SPA*

| Mean peak abundance estimate type | Mean peak abundance estimate | Number of SPA breeding adults present by season[1] | Annual mortality range[2] | Annual baseline mortality increase range[3] |
|---|---|---|---|---|
| Upper 95% CI | 10,010 (breeding)<br>4,447 (autumn)<br>4,711 (spring)<br>19,168 (year round) | 20 (breeding)<br>27 (autumn)<br>29 (spring)<br>76 (year round) | 0-2 | 0.02-0.42% |
| Mean | 4,705 (breeding)<br>2,650 (autumn)<br>1,617 (spring)<br>8,972 (year round) | 9 (breeding)<br>16 (autumn)<br>10 (spring)<br>35 (year round) | 0-2 | 0.01-0.20% |
| Lower 95% CI | 783 (breeding)<br>1,308 (autumn)<br>0 (spring)<br>2,092 (year round) | 2 (breeding)<br>8 (autumn)<br>0 (spring)<br>10 (year round) | 0-1 | 0.00-0.05% |

[1] During the breeding season, assumes 0.2% of recorded birds are adults from the SPA population (11,806), and 0.6% during the autumn and spring migration periods

[2] Assumes displacement rates of 30-70% and mortality rates of 1-10%

[3] Background population is St Kilda SPA breeding adults (9,604 individuals), adult age class annual mortality rate of 13% (Horswill and Robinson, 2015)

1942. Using realistic values (i.e. mean density, 50% displacement and 1% mortality), there would be an annual increase in mortality of <1 (0.18) bird, representing a 0.01% increase in mortality rate. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. Accordingly, no significant effects on Manx shearwater are predicted during the operation and maintenance phase, and **it is concluded that there is no potential for the Project to have an adverse effect on the integrity of St Kilda SPA**.

1943. A review of the potential effects of artificial light on Manx shearwaters is presented in Section 12.6.3.1 of **Chapter 12 Offshore Ornithology**. This concludes that lighting associated with the Project is very unlikely to significantly affect disturbance and displacement effects on Manx shearwater, and therefore the conclusions of the assessment are unchanged.

1944. The confidence in the assessment is high. The evidence used to set the displacement rates presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES is of high applicability and quality. Whilst there is limited available evidence to inform mortality rates, 1% is considered to be sufficiently precautionary based on expert opinion.

### Potential effects in-combination with other projects

1945. No in-combination effects are predicted during the construction and decommissioning phases. This is because it is unlikely that there would be significant temporal and/or spatial overlap with other plans or projects, and due to the negligible effects predicted from the project alone.

1946. During the operation and maintenance phase, the in-combination assessment for Manx shearwaters from St Kilda SPA has been undertaken in accordance with the approach presented in **Section 8.1**. The total population apportioned to the SPA at risk of displacement is estimated to be 53 breeding adults (**Appendix 12.1** of the ES). Annual in-combination displacement and mortality rates for birds from St Kilda SPA are presented in **Table 8.47**.

1947. Assuming a displacement rate of 70% and a mortality rate of 10% of displaced birds, 4 breeding adult SPA birds would be lost to displacement annually. This would increase the existing mortality within the SPA population (1,249 breeding adult birds per year) by 0.30%. Using a realistic displacement rate of 50%, and a mortality rate for displaced birds of 1%, the annual in-combination mortality would be <1 bird. This would increase the existing mortality within this population by 0.02%. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that detectable changes in mortality rates will not occur due to the level or mortality predicted if the more realistic rates for mortality are used.

1948. **It is concluded that predicted Manx shearwater mortality due to of operational phase displacement impacts of the Project in-combination with other projects, would not adversely affect the integrity of St Kilda SPA.** This accords with the conclusions of the Round 4 offshore wind leasing HRA (NIRAS, 2021), which concluded no effect on site integrity (for all SPAs) on the basis of the low vulnerability to OWFs and low density of this species within Round 4 areas.

*Table 8.123 In-combination year-round displacement matrix for Manx shearwater from St Kilda SPA*

| Annual | Mortality | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| **Displacement** | 1% | 2% | 3% | 4% | 5% | 10% | 20% | 30% | **50%** | **80%** | **100%** |
| **10%** | 0 | 0 | 0 | 0 | 0 | 1 | 1 | 2 | 3 | 4 | 5 |
| **20%** | 0 | 0 | 0 | 0 | 1 | 1 | 2 | 3 | 5 | 8 | 11 |
| **30%** | **0** | **0** | **0** | **1** | **1** | **2** | 3 | 5 | 8 | 13 | 16 |
| **40%** | **0** | **0** | **1** | **1** | **1** | **2** | 4 | 6 | 11 | 17 | 21 |
| **50%** | **0** | **1** | **1** | **1** | **1** | **3** | 5 | 8 | 13 | 21 | 27 |
| **60%** | **0** | **1** | **1** | **1** | **2** | **3** | 6 | 10 | 16 | 25 | 32 |
| **70%** | **0** | **1** | **1** | **1** | **2** | **4** | 7 | 11 | 19 | 30 | 37 |
| **80%** | 0 | 1 | 1 | 2 | 2 | 4 | 8 | 13 | 21 | 34 | 42 |
| **90%** | 0 | 1 | 1 | 2 | 2 | 5 | 10 | 14 | 24 | 38 | 48 |
| **100%** | 1 | 1 | 2 | 2 | 3 | 5 | 11 | 16 | 27 | 42 | 53 |

Note: The cells show the number of birds predicted to die (rounded to the nearest integer) at a given rate of displacement and mortality. Highlighted cells are considered to be the most realistic scenario, in accordance with SNCB advice (SNCBs, 2022).

#### 8.47.3.3 Leach's storm-petrel

##### Status

1949. The St Kilda SPA breeding Leach's storm-petrel population at classification (1992) was cited as 5,000 pairs, or 10,000 breeding adults (SNH 2009c). Stroud *et al.,* (2016) gave a population of 45,433 pairs, or 90,866 breeding adults, for the period 1999 – 2000. This is used as the reference population for the assessment.

##### Functional linkage and seasonal apportionment of potential effects

1950. The mean foraging range of Leach's storm-petrel is 657km (Woodward *et al.*, 2019); estimates for maximum and mean maximum foraging ranges are not available. The Project is located approximately 526km from St Kilda SPA, which means that the Project is within the mean foraging range of Leach's storm-petrels breeding at this SPA.

##### Potential effects on the qualifying feature from the Project-alone

1951. Leach's storm-petrel was not recorded during baseline surveys of the windfarm site (including buffer areas). It is therefore concluded that this species does not occur regularly in this area. It is noted that Leach's storm-petrel is considered to have low vulnerability to collision risk and very low vulnerability to displacement impacts (Bradbury *et al*., 2014), and therefore the risk of significant effects would be low, even if this species occurred at the windfarm site.

1952. **It is therefore concluded that there would be no measurable effects on Leach's storm-petrel due to the project alone, and no adverse effect on the integrity of the St Kilda SPA is predicted**.

##### Potential effects on the qualifying feature in-combination with other projects

1953. As the Project would have no measurable effect on Leach's storm-petrel populations from the St Kilda SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of St Kilda SPA, when assessed in-combination with other plans or projects.**

#### 8.47.3.4 Great skua

##### Status

1954. The St Kilda SPA breeding great skua population at classification (1995) was cited at 270 pairs, or 540 breeding adults (Furness, 2015, Stroud *et al*., 2016). Furness (2015) gave the breeding population of 151 pairs, or 302 breeding adults, in 2012. The most recent counts (covering 2019 and 2022) identified 94 pairs (AOT), or 188 breeding adults (JNCC, 2023a); this is used as the reference population for the assessment.

1955. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.118 (1 – 0.882; Horswill and Robinson 2015), 22 breeding adults from the SPA population would be expected to die each year.

## Functional linkage and seasonal apportionment of potential effects

1956. The mean maximum foraging range of great skua is 443.3km (±487.9km) and the maximum foraging range is 1,003km (Woodward *et al*., 2019). The Project is located approximately 526km from St Kilda SPA, which means that the Project is beyond the mean maximum foraging range of great skuas breeding at this SPA, but within the maximum foraging range +1SD, and the maximum foraging range.

## Potential effects on the qualifying feature from the Project-alone

### Operation and maintenance phase collision risk

1957. The great skua qualifying feature of the St Kilda SPA has been screened into the Appropriate Assessment due to the potential risk of collision. However, this species was not recorded within the windfarm site during site surveys, and therefore there would be no risk that collision mortality would affect great skua populations from St Kilda SPA. It is noted that a separate assessment of collision risk to migrant great skuas has also been undertaken, as set out in Chapter 12 of the ES. This also predicted negligible annual mortality for this species (0.03 birds), which would equate to no measurable increase in mortality apportioned to populations from St Kilda SPA. **It is concluded that there would be no adverse effect on the integrity of St Kilda SPA**.

1958. The confidence in the assessment is high. As both the surveys of the Project site and separate migrant collision risk assessment indicate that there would be negligible collision impacts on this species, it is considered extremely unlikely that there would be any effects on populations from the SPA.

## Potential effects on the qualifying feature in-combination with other projects

1959. As the Project would have no measurable effect on great skua populations from the St Kilda SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of St Kilda SPA, when assessed in-combination with other plans or projects.**

### 8.47.3.5 Guillemot

#### Status

1960. The St Kilda SPA breeding guillemot population was cited as 15,209 pairs, or 30,418 breeding adults, in 1992 (Furness, 2015, Stroud *et al*., 2016). Furness (2015) gave the breeding population of 15,700 pairs, or 31,400 breeding adults, in 1999. The most recent counts (covering 2015 and 2016) identified 10,303 individuals (JNCC, 2022); however, it is not clear if this encompasses all of the SPA breeding sites. Therefore, the most recent population is assumed to be 31,400 breeding adults; this is used as the reference population for the assessment.

1961. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.061 (1 – 0.939; Horswill and Robinson 2015), 1,915 breeding adults from the SPA population would be expected to die each year.

#### Functional linkage and seasonal apportionment of potential effects

1962. The mean maximum foraging range of guillemot is 73.2km (±80.5km) and the maximum foraging range is 338km (Woodward *et al*., 2019). The Project is located approximately 526km from St Kilda SPA, which means that the Project is beyond the maximum foraging range for guillemots from the SPA. No impacts during the breeding season from the Project are therefore apportioned to guillemots breeding at this SPA.

1963. Outside the breeding season, breeding guillemots from the SPA are assumed to range widely and to mix with guillemots of all ages from breeding colonies in the UK and beyond. The relevant non-breeding season reference population is the UK Western Waters BDMPS, consisting of 1,139,220 individuals (August to February) (Furness, 2015).

1964. Furness (2015) estimated that 95% of the St Kilda SPA breeding adults are present within the UK Western Waters BDMPS during the non-breeding season, which is 29,830 birds. This represents 2.6% of the BDMPS population for this period (1,139,220). It is therefore assumed that 2.6% of guillemots present at the Project site are breeding adults from St Kilda SPA.

#### Potential effects on the qualifying feature

**Operation and maintenance phase disturbance/displacement/barrier effects**

*Project-alone*

1965. The mean peak abundance of guillemots present within the windfarm site and 2km buffer during the non-breeding season is 8,315 (6,085-12,047) individuals (refer to **Appendix 12.1** of the ES). Of these, 216 birds (158-313) are likely to be breeding adults from the St Kilda SPA.

1966. **Table 8.124** sets out the predicted impacts on guillemots from St Kilda SPA during the non-breeding season. Displacement rates of 30% to 70% are considered for this species, along with a range of mortality rates of 1% to 10% of displaced birds (UK SNCBs, 2017).

1967. The available evidence suggests that the upper ranges of these displacement and mortality rates may be excessively precautionary. Whilst it is true that guillemots and razorbills tend to be displaced by OWFs they do not avoid them completely, and displacement rates vary between sites. MacArthur Green (2019b) concluded that, on average, densities within OWFs were around half of the density found in the habitats around the OWF. At some OWFs there was also displacement of birds from a buffer zone surrounding it. The size of the buffer zone varied between OWFs and was generally less than 2km, with auk density increasing across the buffer zone as distance from turbines increased, up to the density in the wider area. Another recent review (APEM, 2022) found that the current displacement rates suggested by the SNCBs for guillemot and razorbill (30% to 70%) did not account for the quality of or confidence in the studies which were used to inform this position, and that studies where no significant effects were recorded were not accounted for during the provision of the advice. APEM (2022) suggested that in the case of Hornsea Project Four, an evidence-based displacement rate of 50% was appropriate. However, the study also recognised that larger displacement effects were possible.

1968. Mortality due to displacement could arise due to increased energy costs and/or decreased energy intake, if displacement results in increased flying time to avoid OWFs, and/or increased bird densities and competition for prey in areas of unimpacted habitat outside the OWF. Given that UK OWFs in the Irish and Celtic Seas represent a very small proportion of the available foraging habitat for guillemots and razorbills within UK Western Waters, the increase in density of auks outside OWFs due to displacement is likely to be negligible (MacArthur Green, 2019b). It is considered unlikely that the mortality rate due to displacement would be as high as the empirically estimated 6.1% annual mortality for adult guillemots that occurs due to the combination of 'natural' factors and anthropogenic activities (Horswill and Robinson, 2015). Indeed, it may be much lower; MacArthur Green (2019b) recommended precautionary rates of 50% displacement for guillemot and 1% mortality of displaced birds based on a review of available evidence. Modelling undertaken by APEM (2022) found that in the case of Hornsea Project Four, a mortality rate of 1% for displaced auks was considered to be precautionary, and that this may be an overestimate given that Hornsea Project Four is located towards the upper end of the mean maximum foraging range for guillemot from the closest designated sites. It is also noted that in the recent decision on Hornsea Project Four, the SoS determined that a rate of 70% displacement/2% mortality was appropriate to inform the assessment of effects on guillemots from Flamborough and Filey Coast SPA (DESNZ, 2023b).

1969. The full ranges of recommended displacement and mortality effects are considered by the assessment, along with evidence-based displacement and mortality rates of 50% and 1%, respectively (APEM, 2022; MacArthur Green, 2019b).

*Table 8.124 Guillemot – predicted operation and maintenance phase displacement and mortality* from *St Kilda SPA*

| Mean peak abundance estimate type | Mean peak abundance estimate | Number of St Kilda SPA breeding adults present (non-breeding season)[1] | Annual mortality range[2] | Annual baseline mortality increase range[3] |
|---|---|---|---|---|
| Upper 95% CI | 12,047 | 313 | 1-22 | 0.05-1.14% |
| Mean | 8,315 | 216 | 1-15 | 0.03-0.79% |
| Lower 95% CI | 6,085 | 158 | 0-11 | 0.02-0.58% |

[1] Assumes 2.6% of birds present during the non-breeding season are St Kilda SPA breeding adults
[2] Assumes displacement rates of 30-70% and mortality rates of 1-10%
[3] Background mortality rate of 6.1% (Horswill and Robinson, 2015)

1970. Based on the mean peak abundances, the annual total of guillemots from the St Kilda SPA at risk of displacement is 216 birds (**Table 8.124**). At displacement rates of 30% to 70%, and mortality rates of 1% to 10% for displaced birds, 1 to 15 SPA breeding adults would be predicted to die each year due to displacement from the Project.

1971. Assuming a displacement rate of 70% and a mortality rate of 10% of displaced birds, annual mortality within the SPA breeding adult population would increase by 0.79%. Using an evidence-based displacement rate of 50%, and a mortality rate for displaced birds of 1%, annual mortality in the population would instead increase by 0.06% (1 bird).

1972. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that no detectable changes in mortality rates would occur under any combination of displacement and mortality rates when the mean peak abundance estimate assessments are considered.

1973. Increases of over 1% are predicted if the upper 95% CIs for mean peak abundances are used as inputs to the assessment alongside a 10% mortality rate for displaced birds. The probability of this occurring is extremely small for two reasons. Firstly, the upper 95% CI for the mean peak abundances are highly unlikely to occur regularly at the windfarm site. Secondly, mortality rates for displaced birds of 10% are much higher than evidence suggests will actually be the case, and use of the evidence-based displacement (50%) and mortality rate (1%) (and also 70%/2%) would again result in a mortality increase of significantly less than 1%.

1974. **It is concluded that predicted guillemot mortality due to operational phase displacement for the project alone would not adversely affect the integrity of the St Kilda SPA.**

1975. The confidence in the assessment is high for several reasons. Firstly, the evidence used to inform the evidence-based displacement rates is of high applicability and quality (based on the criteria discussed in **Chapter 12 Offshore Ornithology** of the ES). Whilst there was limited available evidence to inform mortality rates, 1% is considered to be sufficiently precautionary based on expert opinion. This species is not regarded as being highly specialised in its habitat requirements (Bradbury *et al*., 2014; Furness and Wade, 2012; Garthe and Hüppop, 2004), and it is therefore anticipated that displaced birds will find alternative habitat in the vast majority of cases. Finally, the conclusion of the assessment is the same irrespective of whether the mean or 95% upper CI mean peak abundances are used to calculate potential mortality and increases in the baseline mortality rate of the background population, provided the evidence-based displacement and mortality rates are used.

*In-combination*

1976. The in-combination assessment for guillemots from St Kilda SPA has been undertaken in accordance with the approach presented in **Section 8.1**. The total population apportioned to St Kilda SPA at risk of displacement is estimated to be 1,287 breeding adults (**Appendix 12.1** of the ES). Annual in-combination displacement and mortality rates for birds from St Kilda SPA are presented in **Table 8.125.**

1977. Assuming a displacement rate of 70% and a mortality rate of 10% of displaced birds, 90 breeding adult SPA birds would be lost to displacement annually. Including additional apportioned mortality from the Morlais and Holyhead tidal projects (total 0.82 birds), this would increase the existing mortality within the SPA population (1,915 breeding adult birds per year) by 4.75%. Using an evidence-based displacement rate of 50%, and a mortality rate for displaced birds of 1%, the annual in-combination displacement mortality would be six birds. This would increase the existing mortality within this population by

0.38%. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that detectable changes in mortality rates will not occur due to the level or mortality predicted if the more realistic rates for mortality are used.

1978. **It is concluded that predicted guillemot mortality due to of operational phase displacement impacts of the Project in-combination with other projects, would not adversely affect the integrity of St Kilda SPA.**

*Table 8.125 In-combination year-round displacement matrix for guillemot from St Kilda SPA*

| **Annual** | **Mortality** | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| **Displacement** | 1% | **2%** | **3%** | **4%** | **5%** | **10%** | **20%** | **30%** | **50%** | **80%** | **100%** |
| **10%** | 1 | 3 | 4 | 5 | 6 | 13 | 26 | 39 | 64 | 103 | 129 |
| **20%** | 3 | 5 | 8 | 10 | 13 | 26 | 51 | 77 | 129 | 206 | 257 |
| **30%** | **4** | **8** | **12** | **15** | **19** | **39** | 77 | 116 | 193 | 309 | 386 |
| **40%** | **5** | **10** | **15** | **21** | **26** | **51** | 103 | 154 | 257 | 412 | 515 |
| **50%** | **6** | **13** | **19** | **26** | **32** | **64** | 129 | 193 | 322 | 515 | 644 |
| **60%** | **8** | **15** | **23** | **31** | **39** | **77** | 154 | 232 | 386 | 618 | 772 |
| **70%** | **9** | **18** | **27** | **36** | **45** | **90** | 180 | 270 | 451 | 721 | 901 |
| **80%** | 10 | 21 | 31 | 41 | 51 | 103 | 206 | 309 | 515 | 824 | 1030 |
| **90%** | 12 | 23 | 35 | 46 | 58 | 116 | 232 | 348 | 579 | 927 | 1159 |
| **100%** | 13 | 26 | 39 | 51 | 64 | 129 | 257 | 386 | 644 | 1030 | 1287 |

Note: The cells show the number of birds predicted to die (rounded to the nearest integer) at a given rate of displacement and mortality. Highlighted cells are considered to be the most realistic scenario, in accordance with SNCB advice (SNCBs, 2022).

#### 8.47.3.6 Puffin

##### Status

1979. The St Kilda SPA breeding puffin population at classification was cited as 155,000 pairs, or 310,000 breeding adults in 1989 (Furness, 2015, Stroud *et al*., 2016). Furness (2015) gave a breeding population of 142,264 breeding pairs, or 284,528 breeding adults, in 2000. The most recent counts (2018) identified 34,753 apparently occupied burrows, or 69,506 breeding adults, however it was not clear if this encompassed all of the SPA breeding sites. Therefore, the most recent population is assumed to be 284,528 breeding adults; this is used as the reference population for the assessment.

1980. Based on the most recent SPA population of breeding adults, and an annual breeding adult baseline mortality rate of 0.094 (1 – 0.906; Horswill and Robinson 2015), 26,746 breeding adults from the SPA population would be expected to die each year.

##### Functional linkage and seasonal apportionment of potential effects

1981. The mean maximum foraging range of puffin is 137.1km (±128.3km) and the maximum foraging range is 383km (Woodward *et al*., 2019). The Project is located approximately 526km from St Kilda SPA, which means that the Project is beyond the maximum foraging range for puffins from the SPA. No impacts during the breeding season from the Project are therefore apportioned to puffins breeding at this SPA.

1982. Outside of the breeding season, puffins, including those from the St Kilda SPA, are not constrained by requirements to visit nests to incubate eggs or provision chicks. At this time, they are assumed to range more widely and to mix with puffins of all age classes from breeding colonies in the UK and further afield. The background population during these seasons is the UK Western Waters BDMPS. This consists of 304,557 individuals during the non-breeding season (August to March).

1983. Furness (2015) estimates that 18% of the St Kilda SPA breeding adults (284,528) are present within the UK Western Waters BDMPS during the non-breeding season, which is 51,215 birds. This represents 16.8% of the BDMPS population for this period (304,557). It is therefore assumed that 16.8% of puffins present at the Project site are breeding adults from St Kilda SPA.

## Potential effects on the qualifying feature

### Operation and maintenance phase disturbance/displacement/barrier effects

*Project-alone*

1984. The mean peak abundance of puffins present within the windfarm site and 2km buffer during the non-breeding season was 19.7 (1.9-50.8) individuals (refer to **Appendix 12.1** of the ES). Of these, approximately three birds (3.3 (0.3-8.5)) were likely to be a breeding adults from St Kilda SPA.

1985. **Table 8.126** sets out the predicted impacts on puffins from St Kilda SPA during the non-breeding season. This estimates that there would be no measurable increase in mortality from the SPA population, assuming a displacement rate of 30-70% and a mortality of 1-10% for displaced birds.

*Table 8.126 Puffin – predicted operation and maintenance phase displacement and mortality from St Kilda SPA*

| Mean peak abundance estimate type | Mean peak abundance estimate | Number of St Kilda SPA breeding adults present (non-breeding season)[1] | Annual mortality range[2] | Annual baseline mortality increase range[3] |
|---|---|---|---|---|
| Upper 95% CI | 50.8 | 8.5 | 0-1 | 0.00-0.00% |
| Mean | 19.7 | 3.3 | 0-0 | 0.00-0.00% |
| Lower 95% CI | 1.9 | 0.3 | 0-0 | 0.00-0.00% |

[1] Assumes 16.8% of birds present during the non-breeding season are St Kilda SPA breeding adults
[2] Assumes displacement rates of 30-70% and mortality rates of 1-10%
[3] Background mortality rate of 9.4% (Horswill and Robinson, 2015)

1986. **It is concluded that predicted puffin mortality due to operational phase displacement would not adversely affect the integrity of the St Kilda SPA.**

1987. The confidence in the assessment is high for several reasons. Firstly, the evidence used to inform the evidence-based displacement rates is of high applicability and quality (based on the criteria discussed in **Chapter 12 Offshore Ornithology** of the ES). Whilst there is limited available evidence to inform mortality rates, 1% is considered to be sufficiently precautionary based on expert opinion. This species is not regarded as being highly specialised in its habitat requirements (Bradbury *et al*., 2014; Furness and Wade, 2012; Garthe and Hüppop, 2004), and it is therefore anticipated that displaced birds will find alternative habitat in the vast majority of cases. Finally, the conclusion of the assessment is the same irrespective of whether the mean or 95% upper

CI mean peak abundances are used to calculate potential mortality and increases in the baseline mortality rate of the background population.

*In-combination*

1988. As the Project would have no measurable effect on puffin populations from the St Kilda SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that predicted puffin mortality due to operational phase displacement due to the Project in-combination with other plans or projects would not adversely affect the integrity of the St Kilda SPA.**

#### 8.47.3.7 Gannet

##### Status

1989. The St Kilda SPA breeding gannet population at classification was cited as 50,050 pairs, or 100,100 breeding adults, in 1985, and the breeding population was given as 59,622 pairs, or 119,244 breeding adults, in 2004 (Furness, 2015). The most recent count is 60,290 AOS, or 120,580 breeding adults, in 2013 (JNCC, 2023a); this is used as the reference population for the assessment.

1990. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.081 (1 – 0.919; Horswill and Robinson 2015), 9,767 breeding adults from the SPA population would be expected to die each year.

##### Functional linkage and seasonal apportionment of potential effects

1991. The mean maximum foraging range of gannet is 315.2km (±194.2km), and the maximum foraging range is 709km (Woodward *et al*., 2019). The Project is located approximately 526km from St Kilda SPA, which means that the Project is beyond the mean maximum foraging range +1SD for gannets from the SPA, but within the maximum foraging range. The maximum foraging range is a poor indicator of typical foraging behaviour. It would be expected that few birds or foraging trips will occur at this distance from the colony, and even fewer with any regularity. No impacts during the breeding season are therefore apportioned to birds breeding at this colony.

1992. Outside the breeding season breeding gannets, including those from the St Kilda SPA, are not constrained by requirements to visit nests to incubate eggs or provision chicks. At this time, they are assumed to range more widely and to mix with gannets of all age classes from breeding colonies in the UK and further afield. The background population during these seasons is the UK Western Waters BDMPS. This consists of 545,954 individuals during autumn migration (September to November), and 661,888 individuals during spring migration (December to March) (Furness, 2015).

1993. Furness (2015) estimated that 90% of the St Kilda SPA breeding adults (119,244) are present within the UK Western Waters BDMPS during the non-breeding season, which is 107,320 birds, but that all of the SPA population (i.e. 119,244 birds) is present during spring migration. Estimates of the proportion of gannets present at the windfarm site which originate from the St Kilda SPA during the non-breeding season (and therefore the proportion of predicted mortalities from the SPA population) are based on these population estimates as a proportion of the UK Western Waters BDMPS during the relevant season. During autumn migration and spring migration, 19.7%, and 18.1% of impacts are considered to affect birds from the SPA respectively (Furness, 2015).

## Potential effects on the qualifying feature from the Project-alone

1994. The gannet qualifying feature of the St Kilda SPA has been screened into the assessment due to the potential risk of collision and operational phase displacement/barrier effects during the operation and maintenance phase of the Project.

### Operation and maintenance phase displacement/barrier effects

1995. Displacement effects for gannet for the Project were assessed during the autumn and spring migration periods, based on an unapportioned peak mean population of 124 and eight birds respectively, calculated for the windfarm site and a 2km buffer, in line with recommendations within the SNCB guidance (SNCB 2017). As set out above, no gannets present at the windfarm site have been apportioned to St Kilda SPA during the breeding season. The displacement matrices used to calculate potential impacts are presented in **Appendix 12.1** of the ES, and summarised in **Table 8.127**. The inclusion of all birds within the 2km buffer, to determine the total number of birds subject to displacement, is precautionary, as in reality the avoidance rate is likely to fall with distance from the windfarm site.

1996. A displacement rate of 60-80% and mortality rate of 1% has been presented. A maximum 1% mortality value has been selected firstly because gannet are known to possess high habitat flexibility (Furness and Wade, 2012). This suggests that displaced birds will readily find alternative habitats including foraging areas. Secondly, no evidence of displacement-induced mortality has been identified, which means there is limited justification for setting predicted mortality rates at a higher level. Given the extensive foraging range of this species (Woodward *et al.*, 2019), there may be no mortality costs to displacement from the relatively small footprints of OWFs.

*Table 8.127 Gannet – predicted operation and maintenance phase displacement and mortality from St Kilda SPA*

| Mean peak abundance estimate type | Mean peak abundance estimate | Number of SPA breeding adults present by season[1] | Annual mortality range[2] | Annual baseline mortality increase range[3] |
|---|---|---|---|---|
| Upper 95% CI | 809 (breeding)<br>189 (autumn)<br>16 (spring)<br>1,014 (year round) | 0 (breeding)<br>37 (autumn)<br>3 (spring)<br>40 (year round) | 0-0 | 0.00-0.00% |
| Mean | 541 (breeding)<br>124 (autumn)<br>8 (spring)<br>673 (year round) | 0 (breeding)<br>24 (autumn)<br>1 (spring)<br>26 (year round) | 0-0 | 0.00-0.00% |
| Lower 95% CI | 160 (breeding)<br>0 (autumn)<br>0 (spring)<br>160 (year round) | 0 (breeding)<br>0 (autumn)<br>0 (spring)<br>0 (year round) | 0-0 | 0.00-0.00% |

[1] 19.7% and 18.1% of birds are assumed to be breeding adults from the SPA population during the autumn and spring migration periods respectively.
[2] Assumes displacement rates of 60-80% and mortality rate of 1%
[3] Background population is St Kilda SPA breeding adults (72,022 individuals), adult age class annual mortality rate of 8.1% (Horswill and Robinson, 2015)

1997. Using the maximum potential mortality value, there would be no measurable increase in gannet mortality. Accordingly, no significant effects on gannet are predicted during the operation and maintenance phase, and **it is concluded that there would be no potential for the Project to have an adverse effect on the integrity of St Kilda SPA.**

1998. The confidence in the assessment is high for several reasons. Firstly, the evidence used to set the displacement rates presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES is of high applicability and quality. Whilst there is limited available evidence to inform mortality rates, 1% is considered to be sufficiently precautionary based on expert opinion. Finally, the conclusion of the assessment is the same irrespective of whether the mean or 95% upper CI mean peak abundances are used to calculate potential mortality and increases in the baseline mortality rate of the background population.

**Operation and maintenance phase collision risk**

1999. Information to support the Appropriate Assessment for collision risk on breeding adult gannets belonging to the St Kilda SPA population is presented in **Table 8.128**. Collision estimates, calculated using the sCRM, are presented by biological season. A summary of the annual outputs and the corresponding increase in the annual baseline mortality rate is also presented. Parameters used in the sCRM were agreed with Natural England during the ETG process and are described in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES. In accordance with Natural England advice, a 70% macro-avoidance correction was applied to gannet abundance data used in the sCRM.

2000. Based on the mean collision rates, no breeding adult gannets from St Kilda SPA are considered at risk of collision as a result of the Project. Therefore, there would be no measurable increase the existing mortality of the SPA breeding population.

*Table 8.128 Predicted seasonal and annual collision mortality (Stochastic model Option 2, avoidance rate 0.993 (±0.0003), plus 70% macro-avoidance) for breeding adult gannets at the windfarm site, apportioned to St Kilda SPA, with corresponding increases to baseline mortality of the population*

| Period | Breeding Season<br>Apr-Aug | Autumn Migration<br>Sep-Nov | Non-breeding/winter<br>- | Spring Migration<br>Dec-Mar | Annual<br>Jan-Dec |
|---|---|---|---|---|---|
| Total collisions[1] (mean and 95% CIs) | 0.83<br>(0.00-3.35) | 0.14<br>(0.00-0.74) | - | 0.00 | 0.97<br>(0.00-4.10) |
| % apportioned to the SPA | 0.0% | 19.7% | - | 18.1% | - |
| Total SPA collisions (mean and 95% CIs) | 0.00<br>(0.00-0.00) | 0.03<br>(0.00-0.15) | - | 0.00<br>(0.00-0.00) | 0.03<br>(0.00-0.15) |
| Mortality increase[2] (mean and 95% Cis) | 0.00%<br>(0.00-0.00%) | 0.00%<br>(0.00-0.01%) | - | 0.00%<br>(0.00-0.00%) | 0.00%<br>(0.00-0.01%) |

[1] Breeding season collision values reduced to 73.8% of modelled value to reflect proportion of adult birds recorded during site surveys

[2] Assuming predicted annual SPA mortality of 9,767 birds (120,580 x 0.081)

2001. Accordingly, no significant effects on gannet are predicted during the operation and maintenance phase, and **it is concluded that there would be no potential for the Project to have an adverse effect on the integrity of St Kilda SPA**. Comments received from RSPB during the ETG process, indicating that they do not accept the 70% macro-avoidance rate for collision risk recommended by Natural England, are noted. However, even in the absence of this correction factor, the net increase in mortality would be unchanged (i.e. no measurable change).

2002. The confidence in the assessment is high. The evidence used to define the CRM input parameters presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES submitted alongside this RIAA is of high applicability and quality. Whilst there is uncertainty around some of the input parameters (e.g. avoidance rate), the rates selected are considered to be sufficiently precautionary based on expert opinion to provide confidence that collision rates are not underestimated.

**Combined displacement/barrier effects and collision risk**

2003. As no measurable increase in mortality is predicted for both displacement and collision risk, the mean combined displacement and collision rates for breeding adult gannet from the St Kilda SPA would be zero. Therefore, there would be no net increase in existing mortality rates.

2004. **It is concluded that based on predicted gannet mortality due to the combined effects of operational phase displacement and collision there is no potential for the Project-alone or in-combination to have an adverse effect on the integrity of the St Kilda SPA.**

2005. The confidence in the assessment is high, for the reasons provided in the individual displacement and collision assessments.

### Potential effects in-combination with other projects

2006. As no measurable effects of displacement/barrier and collision on gannet are predicted as a result of the Project-alone, there would be no contribution to other plans or projects in-combination. **It is therefore concluded that there would be no potential for the Project-alone or in-combination to have an adverse effect on the integrity of St Kilda SPA**.

## 8.48 Cape Wrath SPA

2007. Cape Wrath SPA is located approximately 530km from the windfarm site.

### 8.48.1 Description of designation

2008. Cape Wrath SPA covers two stretches of cliffs around Cape Wrath headland in north-west Scotland. These cliffs support large colonies of breeding seabirds. The boundary of the SPA overlaps with the boundary of Cape Wrath SSSI, and the seaward elements extend approximately 2km into the marine environment to include the seabed, water column and surface.

### 8.48.2 Conservation objectives

2009. The overarching conservation objectives for the site are:

- To avoid deterioration of the habitats of the qualifying species or significant disturbance to the qualifying species, thus ensuring that the integrity of the site is maintained
- To ensure for the qualifying species that the following are maintained in the long term:
    - Population of the species as a viable component of the site
    - Distribution of the species within site
    - Distribution and extent of habitats supporting the species
    - Structure, function and supporting processes of habitats supporting the species
    - No significant disturbance of the species

### 8.48.3 Assessment

2010. The qualifying features of Cape Wrath SPA screened into the Appropriate Assessment are listed in **Table 5.2**. These are breeding fulmar, breeding kittiwake, breeding guillemot and breeding razorbill. The seabird assemblage has also been screened in for these species.

#### 8.48.3.1 Fulmar

#### Status

2011. The Cape Wrath SPA breeding fulmar population was 2,300 pairs, or 4,600 breeding adults, in 1996 (Furness, 2015, Stroud *et al*., 2016). Furness (2015) gave the breeding population of 2,115 pairs, or 4,230 breeding adults in 2000.

The most recent count is 1,477 pairs (AOS), or 2,954 breeding adults, in 2017 (JNCC, 2023a); this is used as the reference population for the assessment.

2012. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.064 (1 – 0.936; Horswill and Robinson, 2015), the expected annual mortality from the SPA population would be 189 breeding adults.

### Functional linkage and seasonal apportionment of potential effects

2013. The mean maximum foraging range of fulmar is 542.3km (±657.9km) and the maximum foraging range is 2,736km (Woodward *et al*., 2019). The Project is located approximately 530km from Cape Wrath SPA, which means that the Project is within the mean maximum foraging range of fulmars breeding at this SPA.

### Potential effects on the qualifying feature from the Project-alone

2014. Fulmar are considered to have low vulnerability to both collision and displacement impacts from offshore windfarms (Bradbury *et al*., 2014). This species was recorded infrequently within the windfarm site (in only five of 24 surveys, all during the breeding season (January to August; Furness, 2015)) and in low numbers, with densities in four of the five months less than 0.1 birds/km$^2$. A peak density of 0.23 birds/km$^2$ was recorded in May 2022.

2015. On the basis of the low vulnerability of fulmars to potential impacts, and the low frequency and abundance of birds present at the windfarm site, it is concluded that any effects on breeding populations at Cape Wrath SPA are very unlikely, both during and outside of the breeding season.

2016. **It is therefore concluded that there would be no measurable effects on fulmar due to the project alone, and no adverse effect on the integrity of the Cape Wrath SPA is predicted.**

### Potential effects on the qualifying feature in-combination with other projects

2017. As the Project would have no measurable effect on fulmar populations from the Cape Wrath SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of Cape Wrath SPA, when assessed in-combination with other plans or projects.**

## 8.48.3.2 Kittiwake

### Status

2018. The Cape Wrath SPA breeding kittiwake population was cited as 9,660 pairs, or 19,320 breeding adults, in 1996 (Furness, 2015, Stroud *et al*., 2016).

Furness (2015) gave the breeding population of 10,344 pairs, or 20,688 breeding adults, in 2000. The most recent count is 3,622 pairs (AON), or 7,244 breeding adults, in 2017 (JNCC, 2023a); this is used as the reference population for the assessment.

2019. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.146 (1 – 0.854; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 1,058 breeding adults.

### Functional linkage and seasonal apportionment of potential effects

2020. The mean maximum foraging range of kittiwake is 156.1km (±144.5km) and the maximum foraging range is 770km (Woodward *et al*., 2019). The Project is located approximately 530km from Cape Wrath SPA, which means that the Project is beyond the mean maximum foraging range +1SD of kittiwakes breeding at this SPA, but within the maximum foraging range. The maximum foraging range is a poor indicator of typical foraging behaviour. It would be expected that few birds or foraging trips will occur at this distance from the colony, and even fewer with any regularity. No impacts during the breeding season are therefore apportioned to birds breeding at this colony.

2021. Outside the breeding season, breeding kittiwakes from the SPA are assumed to range widely and to mix with kittiwakes of all ages from breeding colonies in the UK and beyond. The relevant non-breeding season reference population is the UK Western Waters and Channel BDMPS, consisting of 911,586 individuals during autumn migration (September to December) and 627,816 during spring migration (January to February) (Furness, 2015).

2022. Furness (2015) estimated that 60% of the Cape Wrath SPA breeding adults are present within the UK Western Waters and Channel BDMPS during the autumn migration period, which is 12,413 birds. During the spring migration period 80% of the population is estimated to be present, which is 16,550 birds. This represents 1.36% and 2.64% of the BDMPS population for the autumn and spring periods respectively. During autumn migration and spring migration, 1.36%, and 2.64% of impacts are therefore considered to affect birds from the SPA.

### Potential effects on the qualifying feature from the Project-alone

2023. The kittiwake qualifying feature of the Cape Wrath SPA has been screened into the Appropriate Assessment due to the potential risk of collision.

#### Operation and maintenance phase collision risk

2024. Information for collision risk on breeding adult kittiwakes belonging to the Cape Wrath SPA population is presented in **Table 8.129**. Collision estimates, calculated using the sCRM, are presented by biological season. A summary of the annual outputs and the corresponding increase in the annual baseline mortality rate is also presented. Parameters used in the sCRM were agreed with Natural England during the ETG process and are described in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES.

2025. Based on the mean collision rates, the annual total of breeding adult kittiwakes from Cape Wrath SPA at risk of collision as a result of the Project is less than one bird (0.13). This would increase the existing mortality of the SPA breeding population by 0.01%.

*Table 8.129 Predicted seasonal and annual collision mortality (Stochastic model Option 2, avoidance rate 0.993 (±0.0003)), for breeding adult kittiwakes at the windfarm site, apportioned to Cape Wrath SPA, with corresponding increases to baseline mortality of the population*

| Period | Breeding Season<br>Mar-Aug | Autumn Migration<br>Sep-Dec | Non-breeding/winter<br>- | Spring Migration<br>Jan-Feb | Annual<br>Jan-Dec |
|---|---|---|---|---|---|
| Total collisions[1] (mean and 95% CIs) | 15.36 (4.17-33.87) | 8.49 (2.20-18.85) | - | 0.63 (0.00-1.46) | 24.48 (6.37-54.17) |
| % apportioned to the SPA | 0.00% | 1.36% | - | 2.64% | - |
| Total SPA collisions (mean and 95% CIs) | 0.00 (0.00-0.00) | 0.12 (0.03-0.26) | - | 0.02 (0.00-0.04) | 0.13 (0.03-0.30) |
| Mortality increase[2] (mean and 95% CIs) | 0.00% (0.00-0.00%) | 0.01% (0.00-0.03%) | - | 0.00% (0.00-0.00%) | 0.01% (0.00-0.03%) |

[1] Breeding season collision values reduced to 94.1% of modelled value to reflect proportion of adult birds recorded during site surveys
[2] Assuming predicted annual SPA mortality of 1,058 birds (7,244 x 0.146)

2026. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that no detectable changes in mortality rates would occur on this population for the mean monthly collision estimates for the Project, or for the upper 95% CI collision estimate.

2027. **It is concluded that based on predicted kittiwake mortality due to collision at the windfarm site there is no potential for the Project to have an adverse effect on the integrity of the Cape Wrath SPA**.

2028. The confidence in the assessment is high. The evidence used to define the CRM input parameters presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** as part of the ES submitted alongside this RIAA is of high applicability and quality. Whilst there is uncertainty around some of the input parameters (e.g. avoidance rate), the rates selected are considered to be sufficiently precautionary based on expert opinion to provide confidence that collision rates are not underestimated.

### Potential effects on the qualifying feature in-combination with other projects

2029. As the Project would have no measurable effect on kittiwake populations from the Cape Wrath SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of Cape Wrath SPA, when assessed in-combination with other plans or projects**.

### 8.48.3.3 Guillemot

### Status

2030. The Cape Wrath SPA breeding guillemot population was cited as 9,159 pairs, or 18,318 breeding adults, in 1996 (Furness, 2015, Stroud *et al*., 2016). Furness (2015) gave the breeding population of 27,359 pairs, or 54,718 breeding pairs, in 2000. The most recent count (2017) is 38,109 individuals (JNCC, 2023a); this is used as the reference population for the assessment.

2031. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.061 (1 – 0.939; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 2,325 breeding adults.

### Functional linkage and seasonal apportionment of potential effects

2032. The mean maximum foraging range of guillemot is 73.2km (±80.5km) and the maximum foraging range is 338km (Woodward *et al*., 2019). The Project is located approximately 530km from Cape Wrath SPA, which means that the Project is beyond the maximum foraging range for guillemots from the SPA.

No impacts during the breeding season from the Project are therefore apportioned to guillemots breeding at this SPA.

2033. Outside the breeding season, breeding guillemots from the SPA are assumed to range widely and to mix with guillemots of all ages from breeding colonies in the UK and beyond. The relevant non-breeding season reference population is the UK Western Waters BDMPS, consisting of 1,139,220 individuals (August to February) (Furness, 2015).

2034. Furness (2015) estimated that 95% of the Cape Wrath SPA breeding adults are present within the UK Western Waters BDMPS during the non-breeding season, which is 51,982 birds. This represents 4.6% of the BDMPS population for this period (1,139,220). It is therefore assumed that 4.6% of guillemots present at the Project site are breeding adults from Cape Wrath Isles SPA.

## Potential effects on the qualifying feature

### Operation and maintenance phase disturbance / displacement / barrier effects

*Project-alone*

2035. The mean peak abundance of guillemots present within the windfarm site and 2km buffer during the non-breeding season was 8,315 (6,085-12,047) individuals (refer to **Appendix 12.1** of the ES). Of these, 382 birds (280-554) were likely to be breeding adults from the Cape Wrath SPA.

2036. **Table 8.130** sets out the predicted impacts on guillemots from Cape Wrath SPA during the non-breeding season. Displacement rates of 30% to 70% are considered for this species, along with a range of mortality rates of 1% to 10% of displaced birds (UK SNCBs, 2017).

2037. The available evidence suggests that the upper ranges of these displacement and mortality rates may be excessively precautionary. Whilst it is true that guillemots and razorbills tend to be displaced by OWFs they do not avoid them completely, and displacement rates vary between sites. MacArthur Green (2019b) concluded that, on average, densities within OWFs were around half of the density found in the habitats around the OWF. At some OWFs there was also displacement of birds from a buffer zone surrounding it. The size of the buffer zone varied between OWFs and was generally less than 2km, with auk density increasing across the buffer zone as distance from turbines increased, up to the density in the wider area. Another recent review (APEM, 2022) found that the current displacement rates suggested by the SNCBs for guillemot and razorbill (30% to 70%) did not account for the quality of or confidence in the studies which were used to inform this position, and that studies where no significant effects were recorded were not accounted for during the provision of the advice. APEM (2022) suggested that in the case of Hornsea Project Four, an evidence-based displacement rate of 50% was appropriate. However, the study also recognised that larger displacement effects were possible.

2038. Mortality due to displacement could arise due to increased energy costs and/or decreased energy intake, if displacement results in increased flying time to avoid OWFs, and/or increased bird densities and competition for prey in areas of unimpacted habitat outside the OWF. Given that UK OWFs in the Irish and Celtic Seas represent a very small proportion of the available foraging habitat for guillemots and razorbills within UK Western Waters, the increase in density of auks outside OWFs due to displacement is likely to be negligible (MacArthur Green, 2019b). It is considered unlikely that the mortality rate due to displacement would be as high as the empirically estimated 6.1% annual mortality for adult guillemots that occurs due to the combination of 'natural' factors and anthropogenic activities (Horswill and Robinson, 2015). Indeed, it may be much lower; MacArthur Green (2019b) recommended precautionary rates of 50% displacement for guillemot and 1% mortality of displaced birds based on a review of available evidence. Modelling undertaken by APEM (2022) found that in the case of Hornsea Project Four, a mortality rate of 1% for displaced auks was considered to be precautionary, and that this may be an overestimate given that Hornsea Project Four is located towards the upper end of the mean maximum foraging range for guillemot from the closest designated sites. It is also noted that in the recent decision on Hornsea Project Four, the SoS determined that a rate of 70% displacement/2% mortality was appropriate to inform the assessment of effects on guillemots from Flamborough and Filey Coast SPA (DESNZ, 2023b).

2039. The full ranges of recommended displacement and mortality effects are considered by the assessment, along with evidence-based displacement and mortality rates of 50% and 1%, respectively (APEM, 2022; MacArthur Green, 2019b).

*Table 8.130 Guillemot – predicted operation and maintenance phase displacement and mortality from Cape Wrath SPA*

| Mean peak abundance estimate type | Mean peak abundance estimate | Number of Cape Wrath SPA breeding adults present (non-breeding season)[1] | Annual mortality range[2] | Annual baseline mortality increase range[3] |
|---|---|---|---|---|
| Upper 95% CI | 12,047 | 554 | 2-39 | 0.07-1.67% |
| Mean | 8,315 | 382 | 1-27 | 0.05-1.15% |
| Lower 95% CI | 6,085 | 280 | 1-20 | 0.04-0.84% |

| Mean peak abundance estimate type | Mean peak abundance estimate | Number of Cape Wrath SPA breeding adults present (non-breeding season)[1] | Annual mortality range[2] | Annual baseline mortality increase range[3] |
|---|---|---|---|---|
| [1] Assumes 4.6% of birds present during the non-breeding season are Cape Wrath SPA breeding adults<br>[2] Assumes displacement rates of 30-70% and mortality rates of 1-10%<br>[3] Background mortality rate of 6.1% (Horswill and Robinson, 2015) | | | | |

2040. Based on the mean peak abundances, the annual total of guillemots from the Cape Wrath SPA at risk of displacement is 382 birds (**Table 8.130**). At displacement rates of 30% to 70%, and mortality rates of 1% to 10% for displaced birds, 1 to 27 SPA breeding adults would be predicted to die each year due to displacement from the Project.

2041. Assuming a displacement rate of 70% and a mortality rate of 10% of displaced birds, annual mortality within the SPA breeding adult population would increase by 1.15%. Using an evidence-based displacement rate of 50%, and a mortality rate for displaced birds of 1%, annual mortality in the population would instead increase by 0.08% (2 birds).

2042. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that no detectable changes in mortality rates would occur under any combination of displacement and mortality rates when the mean peak abundance estimate assessments are considered.

2043. Increases of over 1% are predicted if the upper 95% CIs for mean peak abundances are used as inputs to the assessment alongside a 10% mortality rate for displaced birds. The probability of this occurring is extremely small for two reasons. Firstly, the upper 95% CI for the mean peak abundances are highly unlikely to occur regularly at the windfarm site. Secondly, mortality rates for displaced birds of 10% are much higher than evidence suggested will actually be the case, and use of the evidence-based displacement (50%) and mortality rate (1%) (and also 70%/2%) would again result in a mortality increase of significantly less than 1%.

2044. **It is concluded that predicted guillemot mortality due to operational phase displacement for the project alone would not adversely affect the integrity of the Cape Wrath SPA.**

2045. The confidence in the assessment is high for several reasons. Firstly, the evidence used to inform the evidence-based displacement rates is of high applicability and quality (based on the criteria discussed in **Chapter 12**

**Offshore Ornithology** of the ES). Whilst there is limited available evidence to inform mortality rates, 1% is considered to be sufficiently precautionary based on expert opinion. This species is not regarded as being highly specialised in its habitat requirements (Bradbury *et al*., 2014; Furness and Wade, 2012; Garthe and Hüppop, 2004), and it is therefore anticipated that displaced birds will find alternative habitat in the vast majority of cases. Finally, the conclusion of the assessment is the same irrespective of whether the mean or 95% upper CI mean peak abundances are used to calculate potential mortality and increases in the baseline mortality rate of the background population, provided the evidence-based displacement and mortality rates are used.

*In-combination*

2046. The in-combination assessment for guillemots from Cape Wrath SPA has been undertaken in accordance with the approach presented in **Section 8.1**. The total population apportioned to Cape Wrath SPA at risk of displacement is estimated to be 2,282 breeding adults (**Appendix 12.1** of the ES). Annual in-combination displacement and mortality rates for birds from Cape Wrath SPA are presented in **Table 8.131**.

2047. Assuming a displacement rate of 70% and a mortality rate of 10% of displaced birds, 160 breeding adult SPA birds would be lost to displacement annually. Including additional apportioned mortality from the Morlais and Holyhead tidal projects (total 1.45 birds), this would increase the existing mortality within the SPA population (2,325 breeding adult birds per year) by 6.93%. Using an evidence-based displacement rate of 50%, and a mortality rate for displaced birds of 1%, the annual in-combination displacement mortality would be 11 birds. This would increase the existing mortality within this population by 0.55%. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that detectable changes in mortality rates will not occur due to the level or mortality predicted if the more realistic rates for mortality are used.

2048. **It is concluded that predicted guillemot mortality due to of operational phase displacement impacts of the Project in-combination with other projects, would not adversely affect the integrity of Cape Wrath SPA.**

*Table 8.131 In-combination year-round displacement matrix for guillemot from Cape Wrath SPA*

| Annual | Mortality | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Displacement | 1% | 2% | 3% | 4% | 5% | 10% | 20% | 30% | 50% | 80% | 100% |
| 10% | 2 | 5 | 7 | 9 | 11 | 23 | 46 | 68 | 114 | 183 | 228 |
| 20% | 5 | 9 | 14 | 18 | 23 | 46 | 91 | 137 | 228 | 365 | 456 |
| 30% | 7 | 14 | 21 | 27 | 34 | 68 | 137 | 205 | 342 | 548 | 685 |
| 40% | 9 | 18 | 27 | 37 | 46 | 91 | 183 | 274 | 456 | 730 | 913 |
| 50% | 11 | 23 | 34 | 46 | 57 | 114 | 228 | 342 | 571 | 913 | 1141 |
| 60% | 14 | 27 | 41 | 55 | 68 | 137 | 274 | 411 | 685 | 1095 | 1369 |
| 70% | 16 | 32 | 48 | 64 | 80 | 160 | 319 | 479 | 799 | 1278 | 1597 |
| 80% | 18 | 37 | 55 | 73 | 91 | 183 | 365 | 548 | 913 | 1461 | 1826 |
| 90% | 21 | 41 | 62 | 82 | 103 | 205 | 411 | 616 | 1027 | 1643 | 2054 |
| 100% | 23 | 46 | 68 | 91 | 114 | 228 | 456 | 685 | 1141 | 1826 | 2282 |

Note: The cells show the number of birds predicted to die (rounded to the nearest integer) at a given rate of displacement and mortality. Highlighted cells are considered to be the most realistic scenario, in accordance with SNCB advice (SNCBs, 2022).

#### 8.48.3.4 Razorbill

##### Status

2049. The Cape Wrath SPA breeding razorbill population was cited as 1,206 pairs, or 2,412 breeding adults, in 1996 (Furness, 2015, Stroud *et al*., 2016). Furness (2015) gave the breeding population of 2,090 pairs, or 4,180 breeding adults, in 2000. The most recent count is 3,246 individuals in 2017 (JNCC, 2023a); this is used as the reference population for the assessment.

2050. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.105 (1 – 0.895; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 341 breeding adults.

##### Functional linkage and seasonal apportionment of potential effects

2051. The mean maximum foraging range of razorbill is 88.7km (±75.9km) and the maximum foraging range is 313km (Woodward *et al*., 2019). The Project is located approximately 530km from Cape Wrath SPA, which means that the Project is beyond the maximum foraging range for razorbills from the SPA. No impacts during the breeding season from the Project are therefore apportioned to razorbill breeding at this SPA.

2052. Outside the breeding season, breeding razorbills from the SPA are assumed to range widely and to mix with razorbills of all ages from breeding colonies in the UK and beyond. The relevant non-breeding season reference population is the UK Western Waters BDMPS, consisting of 606,914 individuals during autumn and spring passage periods (August to October and January to March), and 341,422 individuals during winter (November and December) (Furness, 2015). During autumn and spring migration, 98% of the SPA breeding adults (4,096 individuals based on the 2000 population estimate) are assumed to be present in the BDMPS, representing 0.7% of the BDMPS population (606,914 individuals of all ages). During the winter season, 40% of the SPA breeding adults (1,672 individuals based on the 2000 population estimate) are assumed to be present in the BDMPS, representing 0.5% of the BDMPS population (341,422 individuals of all ages). These percentages (i.e. 0.7% and 0.5%) are the proportions of birds present at the windfarm site that are presumed to originate from the Cape Wrath SPA during the relevant seasons.

### Potential effects on the qualifying feature

#### Operation and maintenance phase disturbance/displacement/barrier effects

*Project-alone*

2053. The year-round mean peak abundance of razorbills present within the windfarm site and 2km buffer was 1,979 (703-3,552) individuals (refer to **Appendix 12.1** of the ES). Of these, 11 birds (4-18) were likely to be breeding adults from the Cape Wrath SPA.

2054. **Table 8.132** sets out the predicted impacts on razorbills from Cape Wrath SPA. Displacement rates of 30% to 70% are considered for this species, along with a range of mortality rates of 1% to 10% of displaced birds (UK SNCBs, 2017).

2055. The available evidence suggests that the upper ranges of these displacement and mortality rates may be excessively precautionary. Whilst it is true that guillemots and razorbills tend to be displaced by OWFs they do not avoid them completely, and displacement rates vary between sites. MacArthur Green (2019b) concluded that, on average, densities within OWFs were around half of the density found in the habitats around the OWF. At some OWFs there was also displacement of birds from a buffer zone surrounding it. The size of the buffer zone varied between OWFs and was generally less than 2km, with auk density increasing across the buffer zone as distance from turbines increased, up to the density in the wider area. Another recent review (APEM, 2022) found that the current displacement rates suggested by the SNCBs for guillemot and razorbill (30% to 70%) did not account for the quality of or confidence in the studies which were used to inform this position, and that studies where no significant effects were recorded were not accounted for during the provision of the advice. APEM (2022) suggested that in the case of Hornsea Project Four, an evidence-based displacement rate of 50% was appropriate. However, the study also recognised that larger displacement effects were possible.

2056. Mortality due to displacement could arise due to increased energy costs and/or decreased energy intake, if displacement results in increased flying time to avoid OWFs, and/or increased bird densities and competition for prey in areas of unimpacted habitat outside the OWF. Given that UK OWFs in the Irish and Celtic Seas represent a very small proportion of the available foraging habitat for guillemots and razorbills within UK Western Waters, the increase in density of auks outside OWFs due to displacement is likely to be negligible (MacArthur Green, 2019b). It is considered unlikely that the mortality rate due to displacement would be as high as the empirically estimated 10.5% annual mortality for adult razorbills that occurs due to the combination of ‘natural’ factors and anthropogenic activities (Horswill and Robinson, 2015). Indeed, it may be much lower; MacArthur Green (2019b) recommended

precautionary rates of 50% displacement for auks and 1% mortality of displaced birds based on a review of available evidence. Modelling undertaken by APEM (2022) found that in the case of Hornsea Project Four, a mortality rate of 1% for displaced auks was considered to be precautionary, and that this may be an overestimate given that Hornsea Project Four is located towards the upper end of the mean maximum foraging range for guillemot from the closest designated sites.

2057. The full ranges of recommended displacement and mortality effects are considered by the assessment, along with evidence-based displacement and mortality rates of 50% and 1%, respectively (APEM, 2022; MacArthur Green, 2019b).

*Table 8.132 Razorbill – predicted operation and maintenance phase displacement and mortality from Cape Wrath SPA*

| Mean peak abundance estimate type | Mean peak abundance estimate by season[1] | Number of Cape Wrath SPA breeding adults present by season)[2] | Annual mortality range[3] | Annual baseline mortality increase range[4] |
|---|---|---|---|---|
| Upper 95% CI | 605 (b)<br>1,070 (aut)<br>1,297 (win)<br>580 (spr)<br>3,552 (year round) | 0 (b)<br>7 (aut)<br>6 (win)<br>4 (spr)<br>18 (year round) | 0-1 (0) | 0.02-0.37% (0.03%) |
| Mean | 252 (b)<br>694 (aut)<br>651 (win)<br>381 (spr)<br>1,979 (year round) | 0 (b)<br>5 (aut)<br>3 (win)<br>3 (spr)<br>11 (year round) | 0-1 (0) | 0.01-0.22% (0.02%) |
| Lower 95% CI | 21 (b)<br>309 (aut)<br>159 (win)<br>214 (spr)<br>703 (year round) | 0 (b)<br>2 (aut)<br>1 (win)<br>1 (spr)<br>4 (year round) | 0-0 (0) | 0.00-0.09% (0.01%) |

[1] Breeding season = b, autumn migration season = aut, winter season = win, spring migration season = spr

[2] Assumes breeding adult apportioning of 0.0% (breeding season), 0.7% (spring and autumn migration) and 0.5% (winter) to Cape Wrath SPA.

[3] Assumes displacement rates of 30-70% and mortality rates of 1-10%. Evidence-based estimates assuming a 50% displacement rate and 1% mortality of displaced birds are presented in parentheses.

[4] Background mortality rate of 10.5% (Horswill and Robinson, 2015)

2058. Based on the mean peak abundances, the annual total of razorbills from the Cape Wrath SPA at risk of displacement is 11 birds (**Table 8.132**). At displacement rates of 30% to 70%, and mortality rates of 1% to 10% for displaced birds, 0 to 1 SPA breeding adults would be predicted to die each year due to displacement from the Project.

2059. Assuming a displacement rate of 70% and a mortality rate of 10% of displaced birds, annual mortality within the SPA breeding adult population would increase by 0.22%. Using an evidence-based displacement rate of 50%, and a mortality rate for displaced birds of 1%, annual mortality in the population would instead increase by 0.02% (<1 bird).

2060. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that no detectable changes in mortality rates would occur under almost any combination of displacement and mortality rates when the mean peak abundance estimate assessments are considered. This would be the case even when upper 95% CIs are considered.

2061. **It is concluded that predicted razorbill mortality due to operational phase displacement for the project alone would not adversely affect the integrity of the Cape Wrath SPA.**

2062. The confidence in the assessment is high for several reasons. Firstly, the evidence used to inform the evidence-based displacement rates is of high applicability and quality (based on the criteria discussed in **Chapter 12 Offshore Ornithology** of the ES). Whilst there is limited available evidence to inform mortality rates, 1% is considered to be sufficiently precautionary based on expert opinion. This species is not regarded as being highly specialised in its habitat requirements (Bradbury *et al*., 2014; Furness and Wade, 2012; Garthe and Hüppop, 2004), and it is therefore anticipated that displaced birds will find alternative habitat in the vast majority of cases. Finally, the conclusion of the assessment is the same irrespective of whether the mean or 95% upper CI mean peak abundances are used to calculate potential mortality and increases in the baseline mortality rate of the background population, provided the evidence-based displacement and mortality rates are used.

*In-combination*

2063. The in-combination assessment for razorbills from Cape Wrath SPA has been undertaken in accordance with the approach presented in **Section 8.1**. The total population apportioned to Cape Wrath SPA at risk of displacement is estimated to be 90 breeding adults (**Appendix 12.1** of the ES). Annual in-combination displacement and mortality rates for birds from Cape Wrath SPA are presented in **Table 8.133**.

2064. Assuming a displacement rate of 70% and a mortality rate of 10% of displaced birds, 6 breeding adult SPA birds would be lost to displacement annually. Including additional apportioned mortality from the Morlais and Holyhead tidal projects (total 0.10 birds), this would increase the existing mortality within the SPA population (341 breeding adult birds per year) by 1.89%. Using an evidence-based displacement rate of 50%, and a mortality rate for displaced birds of 1%, the annual in-combination mortality would be <1 bird. This would increase the existing mortality within this population by 0.16%. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that detectable changes in mortality rates will not occur due to the level or mortality predicted if the more realistic rates for mortality are used.

2065. **It is concluded that predicted razorbill mortality due to of operational phase displacement impacts of the Project in-combination with other projects, would not adversely affect the integrity of Cape Wrath SPA.**

*Table 8.133 In-combination year-round displacement matrix for razorbill from Cape Wrath SPA*

| **Annual** | **Mortality** | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| **Displacement** | **1%** | **2%** | **3%** | **4%** | **5%** | **10%** | **20%** | **30%** | **50%** | **80%** | **100%** |
| **10%** | 0 | 0 | 0 | 0 | 0 | 1 | 2 | 3 | 5 | 7 | 9 |
| **20%** | 0 | 0 | 1 | 1 | 1 | 2 | 4 | 5 | 9 | 14 | 18 |
| **30%** | **0** | **1** | **1** | **1** | **1** | **3** | 5 | 8 | 14 | 22 | 27 |
| **40%** | **0** | **1** | **1** | **1** | **2** | **4** | 7 | 11 | 18 | 29 | 36 |
| **50%** | **0** | **1** | **1** | **2** | **2** | **5** | 9 | 14 | 23 | 36 | 45 |
| **60%** | **1** | **1** | **2** | **2** | **3** | **5** | 11 | 16 | 27 | 43 | 54 |
| **70%** | **1** | **1** | **2** | **3** | **3** | **6** | 13 | 19 | 32 | 51 | 63 |
| **80%** | 1 | 1 | 2 | 3 | 4 | 7 | 14 | 22 | 36 | 58 | 72 |
| **90%** | 1 | 2 | 2 | 3 | 4 | 8 | 16 | 24 | 41 | 65 | 81 |
| **100%** | 1 | 2 | 3 | 4 | 5 | 9 | 18 | 27 | 45 | 72 | 90 |

Note: The cells show the number of birds predicted to die (rounded to the nearest integer) at a given rate of displacement and mortality. Highlighted cells are considered to be the most realistic scenario, in accordance with SNCB advice (SNCBs, 2022).

## 8.49 Flannan Isles SPA

2066. Flannan Isles SPA is located approximately 544km from the windfarm site.

### 8.49.1 Description of designation

2067. Flannan Isles SPA consists of a group of seven rocky islands, outlying skerries and adjacent coastal waters lying approximately 30km west of Lewis in the Outer Hebrides off the north-west coast of Scotland. The islands provide a strategically placed nesting locality for seabirds, which feed in the rich waters off the Western Isles. The vegetation of the islands is predominantly maritime grassland. The boundary of the Special Protection Area overlaps with the boundary of the Flannan Isles SSSI, and the seaward elements extend approximately 2km into the marine environment to include the seabed, water column and surface.

### 8.49.2 Conservation objectives

2068. The overarching conservation objectives for the site are:

- To avoid deterioration of the habitats of the qualifying species or significant disturbance to the qualifying species, thus ensuring that the integrity of the site is maintained
- To ensure for the qualifying species that the following are maintained in the long term:
  - Population of the species as a viable component of the site
  - Distribution of the species within site
  - Distribution and extent of habitats supporting the species
  - Structure, function and supporting processes of habitats supporting the species
  - No significant disturbance of the species

### 8.49.3 Assessment

2069. The qualifying features of the Flannan Isles SPA screened into the Appropriate Assessment are listed in **Table 5.2**. These are breeding fulmar, breeding Leach's storm-petrel, breeding guillemot and breeding puffin. The seabird assemblage has also been screened in for these species.

#### 8.49.3.1 Fulmar

##### Status

2070. The Flannan Isles SPA breeding fulmar population was cited as 4,700 pairs, or 9,400 breeding adults, in 1988 (Furness, 2015, Stroud *et al*., 2016). Furness (2015) gave the breeding population of 7,328 pairs, or 14,656 breeding adults, in 1998. The most recent count is 3,066 pairs (AOS), or 6,132 breeding adults, in 2015 (JNCC, 2023a); this is used as the reference population for the assessment.

2071. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.064 (1 – 0.936; Horswill and Robinson, 2015), the expected annual mortality from the SPA population would be 392 breeding adults.

##### Functional linkage and seasonal apportionment of potential effects

2072. The mean maximum foraging range of fulmar is 542.3km (±657.9km) and the maximum foraging range is 2,736km (Woodward *et al*., 2019). The Project is located approximately 544km from the Flannan Isles SPA, which means that the Project is just beyond the mean maximum foraging range of fulmars breeding at this SPA, but within the mean maximum foraging range +1SD, and the maximum foraging range.

##### Potential effects on the qualifying feature from the Project-alone

2073. Fulmar are considered to have low vulnerability to both collision and displacement impacts from offshore windfarms (Bradbury *et al*., 2014). This species was recorded infrequently within the windfarm site (in only five of 24 surveys, all during the breeding season (January to August; Furness, 2015)) and in low numbers, with densities in four of the five months less than 0.1 birds/km$^2$. A peak density of 0.23 birds/km$^2$ was recorded in May 2022.

2074. On the basis of the low vulnerability of fulmars to potential impacts, and the low frequency and abundance of birds present at the windfarm site, it is concluded that any effects on breeding populations at Flannan Isles SPA are very unlikely, both during and outside of the breeding season.

2075. **It is therefore concluded that there would be no measurable effects on fulmar due to the project alone, and no adverse effect on the integrity of the Flannan Isles SPA is predicted.**

##### Potential effects on the qualifying feature in-combination with other projects

2076. As the Project would have no measurable effect on fulmar populations from the Flannan Isles SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no**

**adverse effect on the integrity of Flannan Isles SPA, when assessed in-combination with other plans or projects.**

#### 8.49.3.2 Leach's storm-petrel

#### Status

2077. The Flannan Isles SPA breeding Leach's storm-petrel population at classification (1992) was cited as 100 – 1,000 pairs, or 200 - 2,000 breeding adults (SNH, 2009d). Stroud *et al.,* (2016) gave a population of 1,425 pairs, or 2,850 breeding adults, in 2001; in the absence of more recent SMP data, this is used as the reference population for the assessment.

#### Functional linkage and seasonal apportionment of potential effects

2078. The mean foraging range of Leach's storm-petrel is 657km (Woodward *et al*., 2019); estimates for maximum and mean maximum foraging ranges are not available. The Project is located approximately 544km from the Flannan Isles SPA, which means that the Project is within the mean foraging range of Leach's storm-petrels breeding at this SPA.

#### Potential effects on the qualifying feature from the Project-alone

2079. Leach's storm-petrel was not recorded during baseline surveys of the windfarm site (including buffer areas). It is therefore concluded that this species does not occur regularly in this area. It is noted that Leach's storm-petrel is considered to have low vulnerability to collision risk and very low vulnerability to displacement impacts (Bradbury *et al*., 2014), and therefore the risk of significant effects would be low, even if this species occurred at the windfarm site.

2080. **It is therefore concluded that there would be no measurable effects on Leach's storm-petrel due to the project alone, and no adverse effect on the integrity of the Flannan Isles SPA is predicted**.

#### Potential effects on the qualifying feature in-combination with other projects

2081. As the Project would have no measurable effect on Leach's storm-petrel populations from the Flannan Isles SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of Flannan Isles SPA, when assessed in-combination with other plans or projects.**

#### 8.49.3.3 Guillemot

##### Status

2082. The Flannan Isles SPA breeding guillemot population was cited as 14,693 pairs in 1992 (Furness, 2015, Stroud *et al*., 2016). Furness (2015) gave the breeding population of 9,807 pairs, or 19,614 breeding adults in 1998. The most recent count is 5,632 individuals in 2021 (JNCC, 2023a); this is used as the reference population for the assessment.

2083. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.061 (1 – 0.939; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 344 breeding adults.

##### Functional linkage and seasonal apportionment of potential effects

2084. The mean maximum foraging range of guillemot is 73.2km (±80.5km) and the maximum foraging range is 338km (Woodward *et al*., 2019). The Project is located approximately 544km from the Flannan Isles SPA, which means that the Project is beyond the maximum foraging range for guillemots from the SPA. No impacts during the breeding season from the Project are therefore apportioned to guillemots breeding at this SPA.

2085. Outside the breeding season, breeding guillemots from the SPA are assumed to range widely and to mix with guillemots of all ages from breeding colonies in the UK and beyond. The relevant non-breeding season reference population is the UK Western Waters BDMPS, consisting of 1,139,220 individuals (August to February) (Furness, 2015).

2086. Furness (2015) estimates that 95% of the Flannan Isles SPA breeding adults are present within the UK Western Waters BDMPS during the non-breeding season, which is 18,633 birds. This represents 1.6% of the BDMPS population for this period (1,139,220). It is therefore assumed that 1.6% of guillemots present at the Project site are breeding adults from Flannan Isles SPA.

##### Potential effects on the qualifying feature

**Operation and maintenance phase disturbance/displacement/barrier effects**

*Project-alone*

2087. The mean peak abundance of guillemots present within the windfarm site and 2km buffer during the non-breeding season was 8,315 (6,085-12,047) individuals (refer to **Appendix 12.1** of the ES). Of these, 133 birds (97-133) were likely to be breeding adults from the Flannan Isles SPA.

2088. **Table 8.134** sets out the predicted impacts on guillemots from Flannan Isles SPA during the non-breeding season. Displacement rates of 30% to 70% are considered for this species, along with a range of mortality rates of 1% to 10% of displaced birds (UK SNCBs, 2017).

2089. The available evidence suggests that the upper ranges of these displacement and mortality rates may be excessively precautionary. Whilst it is true that guillemots and razorbills tend to be displaced by OWFs they do not avoid them completely, and displacement rates vary between sites. MacArthur Green (2019b) concluded that, on average, densities within OWFs were around half of the density found in the habitats around the OWF. At some OWFs there was also displacement of birds from a buffer zone surrounding it. The size of the buffer zone varied between OWFs and was generally less than 2km, with auk density increasing across the buffer zone as distance from turbines increased, up to the density in the wider area. Another recent review (APEM, 2022) found that the current displacement rates suggested by the SNCBs for guillemot and razorbill (30% to 70%) did not account for the quality of or confidence in the studies which were used to inform this position, and that studies where no significant effects were recorded were not accounted for during the provision of the advice. APEM (2022) suggested that in the case of Hornsea Project Four, an evidence-based displacement rate of 50% was appropriate. However, the study also recognised that larger displacement effects were possible.

2090. Mortality due to displacement could arise due to increased energy costs and/or decreased energy intake, if displacement results in increased flying time to avoid OWFs, and/or increased bird densities and competition for prey in areas of unimpacted habitat outside the OWF. Given that UK OWFs in the Irish and Celtic Seas represent a very small proportion of the available foraging habitat for guillemots and razorbills within UK Western Waters, the increase in density of auks outside OWFs due to displacement is likely to be negligible (MacArthur Green, 2019b). It is considered unlikely that the mortality rate due to displacement would be as high as the empirically estimated 6.1% annual mortality for adult guillemots that occurs due to the combination of 'natural' factors and anthropogenic activities (Horswill and Robinson, 2015). Indeed, it may be much lower; MacArthur Green (2019b) recommended precautionary rates of 50% displacement for guillemot and 1% mortality of displaced birds based on a review of available evidence. Modelling undertaken by APEM (2022) found that in the case of Hornsea Project Four, a mortality rate of 1% for displaced auks was considered to be precautionary, and that this may be an overestimate given that Hornsea Project Four is located towards the upper end of the mean maximum foraging range for guillemot from the closest designated sites. It is also noted that in the recent decision on Hornsea Project Four, the SoS determined that a rate of 70% displacement/2% mortality was appropriate to inform the assessment of effects on guillemots from Flamborough and Filey Coast SPA (DESNZ, 2023b).

2091. The full ranges of recommended displacement and mortality effects are considered by the assessment, along with evidence-based displacement and mortality rates of 50% and 1%, respectively (APEM, 2022; MacArthur Green, 2019b).

*Table 8.134 Guillemot – predicted operation and maintenance phase displacement and mortality from Flannan Isles SPA*

| Mean peak abundance estimate type | Mean peak abundance estimate | Number of Flannan Isles SPA breeding adults present (non-breeding season)[1] | Annual mortality range[2] | Annual baseline mortality increase range[3] |
|---|---|---|---|---|
| Upper 95% CI | 12,047 | 193 | 1-13 | 0.17-3.93% |
| Mean | 8,315 | 133 | 0-9 | 0.12-2.71% |
| Lower 95% CI | 6,085 | 97 | 0-7 | 0.09-1.98% |

[1] Assumes 1.6% of birds present during the non-breeding season are Flannan Isles SPA breeding adults
[2] Assumes displacement rates of 30-70% and mortality rates of 1-10%
[3] Background mortality rate of 6.1% (Horswill and Robinson, 2015)

2092. Based on the mean peak abundances, the annual total of guillemots from the Flannan Isles SPA at risk of displacement is 133 birds (**Table 8.134**). At displacement rates of 30% to 70%, and mortality rates of 1% to 10% for displaced birds, 0 to 9 SPA breeding adults would be predicted to die each year due to displacement from the Project.

2093. Assuming a displacement rate of 70% and a mortality rate of 10% of displaced birds, annual mortality within the SPA breeding adult population would increase by 2.71%. Using an evidence-based displacement rate of 50%, and a mortality rate for displaced birds of 1%, annual mortality in the population would instead increase by 0.19% (1 bird).

2094. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that no detectable changes in mortality rates would occur under almost any combination of displacement and mortality rates when the mean peak abundance estimate assessments are considered. Mortality rate increases of over 1% are predicted for mean peak abundance estimate assessments only when the higher displacement/mortality rates (>70%/4%) are considered. These displacement and mortality rates are much higher than evidence suggested will actually be the case. Use of the evidence-based displacement (50%) and mortality rate (1%) would result in a mortality increase of significantly less than 1%, as would a rate of 70%/2% agreed by the SoS in respect of Hornsea Project Four (DESNZ, 2023b).

2095. Increases of over 1% are also predicted if the upper 95% CIs for mean peak abundances are used as inputs to the assessment alongside higher displacement/mortality rates (>60%/3%). The probability of this occurring is extremely small for two reasons. Firstly, the upper 95% CI for the mean peak abundances are highly unlikely to occur regularly at the windfarm site. Secondly, mortality rates for displaced birds of 10% are much higher than evidence suggested will actually be the case, and use of the evidence-based displacement (50%) and mortality rate (1%) (and also 70%/2%) would again result in a mortality increase of significantly less than 1%.

2096. **It is concluded that predicted guillemot mortality due to operational phase displacement for the project alone would not adversely affect the integrity of the Flannan Isles SPA.**

2097. The confidence in the assessment is high for several reasons. Firstly, the evidence used to inform the evidence-based displacement rates is of high applicability and quality (based on the criteria discussed in **Chapter 12 Offshore Ornithology** of the ES). Whilst there is limited available evidence to inform mortality rates, 1% is considered to be sufficiently precautionary based on expert opinion. This species is not regarded as being highly specialised in its habitat requirements (Bradbury *et al*., 2014; Furness and Wade, 2012; Garthe and Hüppop, 2004), and it is therefore anticipated that displaced birds will find alternative habitat in the vast majority of cases. Finally, the conclusion of the assessment is the same irrespective of whether the mean or 95% upper CI mean peak abundances are used to calculate potential mortality and increases in the baseline mortality rate of the background population, provided the evidence-based displacement and mortality rates are used.

*In-combination*

2098. The in-combination assessment for guillemots from Flannan Isles SPA has been undertaken in accordance with the approach presented in **Section 8.1**. The total population apportioned to Flannan Isles SPA at risk of displacement is estimated to be 793 breeding adults (**Appendix 12.1** of the ES). Annual in-combination displacement and mortality rates for birds from Flannan Isles SPA are presented in **Table 8.135**.

2099. Assuming a displacement rate of 70% and a mortality rate of 10% of displaced birds, 51 breeding adult SPA birds would be lost to displacement annually. Including additional apportioned mortality from the Morlais and Holyhead tidal projects (total 0.50 birds), this would increase the existing mortality within the SPA population (344 breeding adult birds per year) by 16.30%. Using an evidence-based displacement rate of 50%, and a mortality rate for displaced birds of 1%, the annual in-combination displacement mortality would be 4 birds. This would increase the existing mortality within this population by 1.30%. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. Although marginally above this 1% threshold, it is considered very unlikely that this would actually have a measurable effect on the SPA population. This is because of the very small number of potentially impacted birds due to displacement (<4), and the recognition that, as guillemot is a dispersive rather than a fully migratory species, birds do not travel great distances from the breeding colony during the non-breeding season (MS-LOT, 2022), and therefore apportioning using the BDMPS is likely to significantly overestimate the numbers of birds from the SPA present at the Project site.

2100. **It is concluded that predicted guillemot mortality due to of operational phase displacement impacts of the Project in-combination with other projects, would not adversely affect the integrity of Flannan Isles SPA.**

*Table 8.135 In-combination year-round displacement matrix for guillemot from Flannan Isles SPA*

| Annual | Mortality | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| **Displacement** | **1%** | **2%** | **3%** | **4%** | **5%** | **10%** | **20%** | **30%** | **50%** | **80%** | **100%** |
| **10%** | 1 | 2 | 2 | 3 | 4 | 8 | 16 | 24 | 40 | 63 | 79 |
| **20%** | 2 | 3 | 5 | 6 | 8 | 16 | 32 | 48 | 79 | 127 | 159 |
| **30%** | **2** | **5** | **7** | **10** | **12** | **24** | 48 | 71 | 119 | 190 | 238 |
| **40%** | **3** | **6** | **10** | **13** | **16** | **32** | 63 | 95 | 159 | 254 | 317 |
| **50%** | **4** | **8** | **12** | **16** | **20** | **40** | 79 | 119 | 198 | 317 | 396 |
| **60%** | **5** | **10** | **14** | **19** | **24** | **48** | 95 | 143 | 238 | 380 | 476 |
| **70%** | **6** | **11** | **17** | **22** | **28** | **55** | 111 | 166 | 277 | 444 | 555 |
| **80%** | 6 | 13 | 19 | 25 | 32 | 63 | 127 | 190 | 317 | 507 | 634 |
| **90%** | 7 | 14 | 21 | 29 | 36 | 71 | 143 | 214 | 357 | 571 | 713 |
| **100%** | 8 | 16 | 24 | 32 | 40 | 79 | 159 | 238 | 396 | 634 | 793 |

Note: The cells show the number of birds predicted to die (rounded to the nearest integer) at a given rate of displacement and mortality. Highlighted cells are considered to be the most realistic scenario, in accordance with SNCB advice (SNCBs, 2022).

#### 8.49.3.4 Puffin

##### Status

2101. The Flannan Isles SPA breeding puffin population at classification was cited as 5,500 pairs, or 11,000 breeding adults, in 1992 (Furness, 2015), and a breeding population of 15,600 pairs, or 31,200 breeding adults, in 2001. The most recent count in 2021 comprises 1,742 individuals and 47,730 pairs (apparently occupied burrows), the equivalent of 97,202 breeding adults (JNCC, 2023a). This is used as the reference population for the assessment.

2102. Based on the most recent SPA population of breeding adults, and an annual breeding adult baseline mortality rate of 0.094 (1 – 0.906; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 9,137 breeding adults.

##### Functional linkage and seasonal apportionment of potential effects

2103. The mean maximum foraging range of puffin is 137.1km (±128.3km) and the maximum foraging range is 383km (Woodward *et al*., 2019). The Project is located approximately 544km from the Flannan Isles SPA, which means the Project is beyond the maximum foraging range for puffins from the SPA. No impacts during the breeding season from the Project are therefore apportioned to puffins breeding at this SPA.

2104. Outside of the breeding season, breeding puffins, including those from the Flannan Isles SPA, are not constrained by requirements to visit nests to incubate eggs or provision chicks. At this time, they are assumed to range more widely and to mix with puffins of all age classes from breeding colonies in the UK and further afield. The background population during these seasons is the UK Western Waters BDMPS. This consists of 304,557 individuals during the non-breeding season (August to March).

2105. Furness (2015) estimates that 18% of the Flannan Isles SPA breeding adults (31,200) are present within the UK Western Waters BDMPS during the non-breeding season, which is 5,616 birds. This represents 1.8% of the BDMPS population for this period (304,557). It is therefore assumed that 1.8% of puffins present at the Project site are breeding adults from Flannan Isles SPA.

##### Potential effects on the qualifying feature

**Operation and maintenance phase disturbance/displacement/barrier effects**

*Project-alone*

2106. The mean peak abundance of puffins present within the windfarm site and 2km buffer during the non-breeding season is 19.7 (1.9-50.8) individuals (refer to **Appendix 12.1** of the ES). Of these, less than one bird (0.35 (0.03-0.91)) is likely to be a breeding adult from Flannan Isles SPA.

2107. **Table 8.136** sets out the predicted impacts on puffins from Flannan Isles SPA during the non-breeding season. This estimates that there would be no measurable increase in mortality from the SPA population, assuming a displacement rate of 30-70% and a mortality of 1-10% for displaced birds.

*Table 8.136 Puffin – predicted operation and maintenance phase displacement and mortality from Flannan Isles SPA*

| Mean peak abundance estimate type | Mean peak abundance estimate | Number of Flannan Isles SPA breeding adults present (non-breeding season)[1] | Annual mortality range[2] | Annual baseline mortality increase range[3] |
|---|---|---|---|---|
| Upper 95% CI | 50.8 | 0.9 | 0-0 | 0.00-0.00% |
| Mean | 19.7 | 0.4 | 0-0 | 0.00-0.00% |
| Lower 95% CI | 1.9 | 0.03 | 0-0 | 0.00-0.00% |

[1] Assumes 1.8% of birds present during the non-breeding season are Flannan Isles SPA breeding adults

[2] Assumes displacement rates of 30-70% and mortality rates of 1-10%

[3] Background mortality rate of 9.4% (Horswill and Robinson, 2015)

2108. **It is concluded that predicted puffin mortality due to operational phase displacement would not adversely affect the integrity of the Flannan Isles SPA.**

2109. The confidence in the assessment is high for several reasons. Firstly, the evidence used to inform the evidence-based displacement rates is of high applicability and quality (based on the criteria discussed in **Chapter 12 Offshore Ornithology** of the ES). Whilst there is limited available evidence to inform mortality rates, 1% is considered to be sufficiently precautionary based on expert opinion. This species is not regarded as being highly specialised in its habitat requirements (Bradbury *et al*., 2014; Furness and Wade, 2012; Garthe and Hüppop, 2004), and it is therefore anticipated that displaced birds will find alternative habitat in the vast majority of cases. Finally, the conclusion of the assessment is the same irrespective of whether the mean or 95% upper CI mean peak abundances are used to calculate potential mortality and increases in the baseline mortality rate of the background population.

### *In-combination*

2110. As the Project would have no measurable effect on puffin populations from the Flannan Isles SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that predicted puffin mortality due to operational phase displacement due to the Project in-combination with other plans or projects would not adversely affect the integrity of the Flannan Isles SPA.**

## 8.50 Hoy SPA

2111. Hoy SPA is located approximately 546km from the windfarm site.

### 8.50.1 Description of designation

2112. Hoy is a mountainous island at the south-western end of the Orkney archipelago. The SPA covers the northern and western two-thirds of Hoy and adjacent coastal waters. The upland areas and the high sea cliffs support an important assemblage of breeding moorland birds and seabirds. The seaward elements of the SPA extend approximately 2km into the marine environment and includes the seabed, water column and surface. Seabirds included within the designation feed both inside and outside the SPA in nearby waters, as well as more distantly in the wider North Sea.

### 8.50.2 Conservation objectives

2113. The overarching conservation objectives for the site are:

- To avoid deterioration of the habitats of the qualifying species or significant disturbance to the qualifying species, thus ensuring that the integrity of the site is maintained
- To ensure for the qualifying species that the following are maintained in the long term:
  - Population of the species as a viable component of the site
  - Distribution of the species within site
  - Distribution and extent of habitats supporting the species
  - Structure, function and supporting processes of habitats supporting the species
  - No significant disturbance of the species

### 8.50.3 Assessment

2114. The qualifying features of Hoy SPA screened into the Appropriate Assessment are listed in **Table 5.2**. These are breeding red-throated diver, breeding fulmar, and breeding great skua. The seabird assemblage has also been screened in for these species.

#### 8.50.3.1 Red-throated diver

##### Status

2115. The Hoy SPA breeding red-throated diver population was cited as 56 pairs, or 112 breeding adults, in 1994 (Furness, 2015, Stroud *et al*., 2016). Furness (2015) gave a breeding population of 60 pairs, or 120 breeding adults, in 2007; this is used as the reference population for the assessment.

2116. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.16 (1-0.840; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 19 breeding adults.

##### Functional linkage and seasonal apportionment of potential effects

2117. The mean maximum foraging range of red-throated diver is 9km (±0km), as is the maximum foraging range (Woodward *et al*., 2019). The Project is located approximately 546km from Hoy SPA, which means that the Project is beyond the maximum foraging range for red-throated divers from the SPA. No impacts during the breeding season from the Project are therefore apportioned to red-throated divers breeding at this SPA.

2118. Outside the breeding season, breeding red-throated divers from the SPA are assumed to range widely and to mix with red-throated divers of all ages from breeding colonies in the UK and further afield. The relevant background population during the autumn and spring migration seasons is the UK Western waters plus Channel BDMPS, consisting of 4,373 individuals during autumn and spring passage periods (September to November and February to April) (Furness, 2015). The relevant background population during the winter season is the NW England and Wales BDMPS, consisting of 1,657 individuals (Furness, 2015).

2119. During the spring and autumn migration seasons, Furness (2015) estimated that 5% of breeding adults from Hoy SPA (120 birds) are present within the UK Western waters plus Channel BDMPS, which is six birds. This represents 0.1% of the BDMPS population for that period (4,373). During the winter period it is estimated that 2% of breeding adults from Hoy SPA (120 birds) are present within the NW England and Wales BDMPS, which is two birds. This represents 0.1% of the BDMPS population for that period (1,657). These percentages (i.e. 0.1% and 0.1%) are the proportions of birds present at the windfarm site that are presumed to originate from the Hoy SPA during the relevant seasons.

## Potential effects on the qualifying feature

### Operation and maintenance phase disturbance/displacement/barrier effects

*Project-alone*

2120. The year-round mean peak abundance of red-throated divers present within the windfarm site and hybrid 10km buffer was 74 (0-240) individuals (refer to **Appendix 12.1** of the ES). Of these, less than one bird (0.07 birds (0-0.21)) was likely to be a breeding adult from the Hoy SPA.

2121. Red-throated divers have a very high sensitivity to disturbance and displacement from operational OWFs. The majority of birds present before OWFs are constructed are displaced by the operation of OWFs. It is expected (based on expert opinion), that this is due to a combination of anthropogenic activities (mainly vessel movements), as well as the presence of OWF infrastructure. A large body of work investigating the effects of displacement of red-throated divers due to operational OWFs exists (Dorsch *et al*., 2020; Elston *et al*., 2016; Gill *et al*., 2018; Hi Def Aerial Surveying, 2017; Irwin *et al*., 2019; MacArthur Green and Royal HaskoningDHV, 2021; McGovern *et al*., 2016; Mendel *et al*., 2019; Percival, 2014; Percival and Ford, 2017; Petersen *et al*., 2014, 2006; Vilela *et al*., 2020; Welcker and Nehls, 2016).

2122. There was a high degree of concordance of the available literature with respect to effects of operation of OWFs on red-throated diver distribution and abundance within OWFs. There was also a high degree of concordance that displacement effects extended beyond OWF boundaries. However, there was considerable variation with respect to the distance at which this effect remained detectable. Studies within the UK have ranged from no significant displacement effects being reported (McGovern *et al*., 2016), displacement effects being restricted to 1km to 2km of an OWF (Percival, 2014; Percival and Ford, 2017), to clear displacement effects across many years. These effects have been reported extending to 7km from OWFs (MacArthur Green and Royal HaskoningDHV, 2021), 9km from OWFs (Elston *et al*., 2016; Hi-Def Aerial Surveying, 2017), and beyond, though not all evidence was available to be referenced by this assessment. Studies from other countries have also recorded variable displacement distances, ranging from 1.5km to 2km (Welcker and Nehls, 2016) to 10km and beyond (Dorsch *et al*., 2020; Vilela *et al*., 2020). Displacement effects were detectable up to 20km from OWFs in one case.

2123. There was also concordance in the studies reviewed that displacement effects on red-throated diver due to operational OWFs occurred on a gradient, with the strongest effects observed either within, or close to OWFs. As the distance from the OWF increased, the magnitude of the effect reduced, until a distance was reached at which the effect was no longer detectable.

2124. No study to date has managed to provide insight into whether changes in red-throated diver distribution at any spatial scale have the potential to result in population level effects, either at local, regional, national or international levels. Red-throated divers have been noted to be capable of utilising a range of marine habitats and prey species (Dierschke *et al*., 2017; Guse *et al*., 2009; Kleinschmidt *et al*., 2016). Recent data from the Outer Thames Estuary SPA indicated that birds were much more commonly recorded in water depths of less than 20m (Irwin *et al*., 2019). During the non-breeding season, red-throated divers were mostly widely dispersed, at densities often less than four birds per $km^2$ (Dierschke *et al*., 2017), and were highly mobile (Dorsch *et al*., 2020; Duckworth *et al*., 2020). In some instances, home ranges of many thousands of square kilometres have been demonstrated (Nehls *et al*., 2018). This implies that following displacement, red-throated divers would be able to find alternative foraging sites, in some cases distant from the original area of displacement, which may already have been part of their existing non-breeding season range. It seems likely that in the vast majority of cases, mortality has not been a consequence of displacement.

2125. Displacement rates of 1.000, along with a range of mortality rates of 1% to 10% of displaced birds are considered for this species at this SPA (UK SNCBs, 2017). However, it is considered that there is a high possibility that displacement and mortality rates would be, based on recent research, substantially lower than this.

2126. **Table 8.137** sets out the predicted impacts on red-throated divers from Hoy SPA during the non-breeding season. A displacement rate of 1.000 is presented for this species, along with a range of mortality rates of 1% to 10% of displaced birds (UK SNCBs, 2017). An estimated annual mortality for the population is provided, along with the increase of existing mortality that would occur through such an impact. The displacement matrices used to calculate potential impacts are presented in **Appendix 12.1** of the ES.

*Table 8.137 Red-throated diver – predicted operation and maintenance phase displacement and mortality from Hoy SPA*

| Mean peak abundance estimate type | Mean peak abundance estimate by season[1] | Number of Hoy SPA breeding adults present by season)[2] | Annual mortality range[3] | Annual baseline mortality increase range[4] |
|---|---|---|---|---|
| Upper 95% CI | 35 (b)<br>23 (aut)<br>55 (win)<br>127 (spr)<br>240 (year round) | 0 (b)<br>0 (aut)<br>0 (win)<br>0 (spr)<br>0 (year round) | 0-0 | 0.01-0.11% |

| Mean peak abundance estimate type | Mean peak abundance estimate by season[1] | Number of Hoy SPA breeding adults present by season)[2] | Annual mortality range[3] | Annual baseline mortality increase range[4] |
|---|---|---|---|---|
| Mean | 8 (b)<br>10 (aut)<br>12 (win)<br>43 (spr)<br>74 (year round) | 0 (b)<br>0 (aut)<br>0 (win)<br>0 (spr)<br>0 (year round) | 0-0 | 0.00-0.03% |
| Lower 95% CI | 0 (b)<br>0 (aut)<br>0 (win)<br>0 (spr)<br>0 (year round) | 0 (b)<br>0 (aut)<br>0 (win)<br>0 (spr)<br>0 (year round) | 0-0 | 0.00-0.00% |

[1] Breeding season = b, autumn migration season = aut, winter season = win, spring migration season = spr

[2] Assumes breeding adult apportioning of 0.0% (breeding season), 0.1% (spring and autumn migration) and 0.1% (winter) to Hoy SPA.

[3] Assumes displacement rate of 100% and mortality rates of 1-10%.

[4] Background mortality rate of 16.0% (Horswill and Robinson, 2015)

2127. Based on the mean peak abundances, less than one red-throated diver from the Hoy SPA would be at risk of displacement (**Table 8.137**). At displacement rates of 1.000, and mortality rates of 1% to 10% for displaced birds, <0.1 SPA breeding adults would be predicted to die each year due to displacement from the Project.

2128. Assuming a displacement rate of 1.00 and a mortality rate of 10% of displaced birds, annual mortality within the SPA breeding adult population would increase by 0.03%. Using a more realistic mortality rate for displaced birds of 1%, annual mortality in the population would be less than 0.01%.

2129. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that no detectable changes in mortality rates would occur under any combination of displacement and mortality rates when the mean peak abundance estimate assessments are considered. This would be the case even when upper 95% CIs are considered.

2130. **It is concluded that predicted red-throated diver mortality due to operational phase displacement for the project alone would not adversely affect the integrity of the Hoy SPA.**

2131. The confidence in the assessment is high for several reasons. Firstly, the evidence used to inform the evidence-based displacement rates is of high

applicability and quality (based on the criteria discussed in **Chapter 12 Offshore Ornithology** of the ES). Whilst there was limited available evidence to inform mortality rates, 1% is considered to be sufficiently precautionary based on expert opinion. This species has been demonstrated to be highly mobile during the non-breeding season, and individuals frequently possessed very large home ranges during this time (Dorsch *et al*., 2020; Nehls *et al*., 2018). It is therefore anticipated that displaced birds will find alternative habitat in the vast majority of cases. Finally, the conclusion of the assessment is the same irrespective of whether 1% or 10% mortality, or the mean or 95% upper CI mean peak abundances, are used to calculate potential mortality and increases in the baseline mortality rate of the background population.

*In-combination*

2132. As the Project would have no measurable effect on red-throated diver populations from the Hoy SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of Hoy SPA, when assessed in-combination with other plans or projects.**

#### 8.50.3.2 Fulmar

### Status

2133. The Hoy SPA breeding fulmar population was cited as 35,000 pairs, or 70,000 breeding adults, in 2000 (Furness, 2015, Stroud *et al*., 2016). Furness (2015) gave a breeding population of 19,586 pairs, or 39,172 breeding adults, in 2007. The most recent count is 620 pairs (AOS) plus 19 individuals, giving a total of 1,259 assumed breeding adults, however it is not clear if this encompasses all of the SPA breeding sites. Therefore, the most recent population is assumed to be 39,172 breeding adults; this is used as the reference population for the assessment.

2134. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.064 (1 – 0.936; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 2,507 breeding adults.

### Functional linkage and seasonal apportionment of potential effects

2135. The mean maximum foraging range of fulmar is 542.3km (±657.9km) and the maximum foraging range is 2,736km (Woodward *et al*., 2019). The Project is located approximately 546km from Hoy SPA, which means that the Project is beyond the mean maximum foraging range of fulmars breeding at this SPA, but within the mean maximum foraging range +1SD, and the maximum foraging range.

### Potential effects on the qualifying feature from the Project-alone

2136. Fulmar are considered to have low vulnerability to both collision and displacement impacts from offshore windfarms (Bradbury *et al*., 2014). This species was recorded infrequently within the windfarm site (in only five of 24 surveys, all during the breeding season (January to August; Furness, 2015)) and in low numbers, with densities in four of the five months less than 0.1 birds/km$^2$. A peak density of 0.23 birds/km$^2$ was recorded in May 2022.

2137. On the basis of the low vulnerability of fulmars to potential impacts, and the low frequency and abundance of birds present at the windfarm site, it is concluded that any effects on breeding populations at Hoy SPA are very unlikely, both during and outside of the breeding season.

2138. **It is therefore concluded that there would be no measurable effects on fulmar due to the project alone, and no adverse effect on the integrity of the Hoy SPA is predicted.**

### Potential effects on the qualifying feature in-combination with other projects

2139. As the Project would have no measurable effect on fulmar populations from the Hoy SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of Hoy SPA, when assessed in-combination with other plans or projects.**

### 8.50.3.3 Great skua

### Status

2140. The Hoy SPA breeding great skua population was cited as 1,900 pairs, or 3,800 breeding adults, in 1992 (Furness, 2015, Stroud *et al*., 2016). Furness (2015) gave a breeding population of 1,346 pairs, or 2,692 breeding adults in 2010. The most recent complete SMP count from 2010 gives a revised population of 1,398 pairs (AOT), or 2,796 breeding adults; this is used as the reference population for the assessment.

2141. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.118 (1 – 0.882; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 330 breeding adults.

### Functional linkage and seasonal apportionment of potential effects

2142. The mean maximum foraging range of great skua is 443.3km (±487.9km) and the maximum foraging range is 1003km (Woodward *et al*., 2019). The Project is located approximately 546km from Hoy SPA, which means that the Project is beyond the mean maximum foraging range of great skuas breeding at this

SPA, but within the mean maximum foraging range +1SD, and the maximum foraging range.

**Operation and maintenance phase collision risk**

2143. The great skua qualifying feature of the Hoy SPA has been screened into the Appropriate Assessment due to the potential risk of collision. However, this species was not recorded within the windfarm site during site surveys, and therefore there would be no risk that collision mortality would affect great skua populations from Hoy SPA. It is noted that a separate assessment of collision risk to migrant great skuas has also been undertaken, as set out in Chapter 12 of the ES. This also predicted negligible annual mortality for this species (0.03 birds), which would equate to no measurable increase in mortality apportioned to populations from Hoy SPA. **It is concluded that there would be no adverse effect on the integrity of Hoy SPA**.

2144. The confidence in the assessment is high. As both the surveys of the Project site and separate migrant collision risk assessment indicate that there would be negligible collision impacts on this species, it is considered extremely unlikely that there would be any effects on populations from the SPA.

### Potential effects on the qualifying feature in-combination with other projects

2145. As the Project would have no measurable effect on great skua populations from the Hoy SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of Hoy SPA, when assessed in-combination with other plans or projects.**

# 8.51 Copinsay SPA

2146. Copinsay SPA is located approximately 562km from the windfarm site.

## 8.51.1 Description of designation

2147. Copinsay SPA lies 4km off the east coast of the Orkney mainland. It consists of the island of Copinsay and three islets (Corn Holm, Ward Holm and Black Holm). The three holms are vegetated, and a storm beach connects them to Copinsay at low water. The islands have good examples of unimproved sub-maritime grasslands and coastal inundation grasslands with several distinct vegetation zones. Copinsay is formed of Old Red Sandstone with the largely horizontal bedding planes providing ideal breeding ledges for seabirds (auks, Fulmar and Kittiwake), especially on the sheer cliffs of the south-east of Copinsay which reach to over 60m. Great Black-backed Gull also breed on the islands. The seabirds feed outside the SPA in the nearby waters, as well as more distantly.

## 8.51.2 Conservation objectives

2148. The overarching conservation objectives for the site are:

- To avoid deterioration of the habitats of the qualifying species or significant disturbance to the qualifying species, thus ensuring that the integrity of the site is maintained
- To ensure for the qualifying species that the following are maintained in the long term:
  - Population of the species as a viable component of the site
  - Distribution of the species within site
  - Distribution and extent of habitats supporting the species
  - Structure, function and supporting processes of habitats supporting the species
  - No significant disturbance of the species

## 8.51.3 Assessment

2149. One qualifying feature of Copinsay SPA has been screened into the Appropriate Assessment (**Table 5.2**): breeding fulmar. This species has also been screened in as a component of the seabird assemblage.

#### 8.51.3.1 Fulmar

##### Status

2150. The Copinsay SPA breeding fulmar population was cited as 1,615 pairs, or 3,230 breeding adults, in 1994 (Furness, 2015, Stroud *et al*., 2016). Furness (2015) gave a breeding population of 1,630 pairs, or 3,260 breeding adults, in 2008. The most recent complete count is 1,585 pairs (AOS), or 3,170 breeding adults, in 2015 (JNCC 2023); this is used as the reference population for the assessment.

2151. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.064 (1 – 0.936; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 203 breeding adults.

##### Functional linkage and seasonal apportionment of potential effects

2152. The mean maximum foraging range of fulmar is 542.3km (±657.9km) and the maximum foraging range is 2,736km (Woodward *et al*., 2019). The Project is located approximately 562km from Copinsay SPA, which means that the Project is beyond the mean maximum foraging range of fulmars breeding at this SPA, but within the mean maximum foraging range +1SD, and the maximum foraging range.

##### Potential effects on the qualifying feature from the Project-alone

2153. Fulmar are considered to have low vulnerability to both collision and displacement impacts from offshore windfarms (Bradbury *et al*., 2014). This species was recorded infrequently within the windfarm site (in only five of 24 surveys, all during the breeding season (January to August; Furness, 2015)) and in low numbers, with densities in four of the five months less than 0.1 birds/km$^2$. A peak density of 0.23 birds/km$^2$ was recorded in May 2022.

2154. On the basis of the low vulnerability of fulmars to potential impacts, and the low frequency and abundance of birds present at the windfarm site, it is concluded that any effects on breeding populations at Copinsay SPA are very unlikely, both during and outside of the breeding season.

2155. **It is therefore concluded that there would be no measurable effects on fulmar due to the project alone, and no adverse effect on the integrity of the Copinsay SPA is predicted.**

##### Potential effects on the qualifying feature in-combination with other projects

2156. As the Project would have no measurable effect on fulmar populations from the Copinsay SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no**

**adverse effect on the integrity of Copinsay SPA, when assessed in-combination with other plans or projects.**

## 8.52 Sule Skerry and Sule Stack SPA

2157. Sule Skerry and Sule Stack SPA is located 575km from the windfarm site.

### 8.52.1 Description of designation

2158. Sule Skerry and Sule Stack are isolated islets 60km west of Mainland Orkney. Sule Skerry is larger, low-lying and vegetated whereas Sule Stack is a higher, bare rock stack with no vascular plants. The boundary of the SPA overlaps with those of Sule Skerry SSSI and Sule Stack SSSI and the seaward elements extend approximately 2km into the marine environment to include the seabed, water column and surface. Qualifying seabird species of the SPA comprise gannet, Leach's petrel, storm petrel, guillemot, puffin and shag.

### 8.52.2 Conservation objectives

2159. The overarching conservation objectives for the site are:

- To avoid deterioration of the habitats of the qualifying species or significant disturbance to the qualifying species, thus ensuring that the integrity of the site is maintained
- To ensure for the qualifying species that the following are maintained in the long term:
  - Population of the species as a viable component of the site
  - Distribution of the species within site
  - Distribution and extent of habitats supporting the species
  - Structure, function and supporting processes of habitats supporting the species
  - No significant disturbance of the species

### 8.52.3 Assessment

2160. The qualifying features of Sule Skerry and Sule Stack SPA screened into the Appropriate Assessment are listed in **Table 5.2**. These are breeding Leach's storm-petrel, breeding gannet, breeding guillemot and breeding puffin. The seabird assemblage has also been screened in for these species.

#### 8.52.3.1 Leach's storm-petrel

#### Status

2161. The Sule Skerry and Sule Stack SPA breeding Leach's storm-petrel population at classification (1994) was cited as five pairs, or 10 breeding adults

(SNH, 2009e). Stroud *et al.,* (2016) gave a breeding population of zero pairs/breeding adults in 2001, however, on a precautionary basis, the population is assumed to be 10 breeding adults; this is used as the reference population for the assessment.

### Functional linkage and seasonal apportionment of potential effects

2162. The mean foraging range of Leach's storm-petrel is 657km (Woodward *et al*., 2019); estimates for maximum and mean maximum foraging ranges are not available. The Project is located approximately 575km from Sule Skerry and Sule Stack SPA, which means that the Project is within the mean foraging range of Leach's storm-petrels breeding at this SPA.

### Potential effects on the qualifying feature from the Project-alone

2163. Leach's storm-petrel was not recorded during baseline surveys of the windfarm site (including buffer areas). It is therefore concluded that this species does not occur regularly in this area. It is noted that Leach's storm-petrel is considered to have low vulnerability to collision risk and very low vulnerability to displacement impacts (Bradbury *et al*., 2014), and therefore the risk of significant effects would be low, even if this species occurred at the windfarm site.

2164. **It is therefore concluded that there would be no measurable effects on Leach's storm-petrel due to the project alone, and no adverse effect on the integrity of the Sule Skerry and Sule Stack SPA is predicted**.

### Potential effects on the qualifying feature in-combination with other projects

2165. As the Project would have no measurable effect on Leach's storm-petrel populations from the Sule Skerry and Sule Stack SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of Sule Skerry and Sule Stack SPA, when assessed in-combination with other plans or projects.**

### 8.52.3.2 Gannet

### Status

2166. The Sule Skerry and Sule Stack SPA breeding gannet population at classification (1994) was cited as 4,890 pairs, or 9,780 breeding adults (Furness, 2015, Stroud *et al*., 2016). Furness (2015) gave a breeding population of 4,675 pairs, or 9,350 breeding adults, in 2004. The most recent count is 4,515 AOS, or 9,030 breeding adults, in 2018 (JNCC, 2023a); this is used as the reference population for the assessment.

2167. Based on the most recent SPA population of breeding adults, and an annual breeding adult baseline mortality rate of 0.081 (1 – 0.919; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 731 breeding adults.

### Functional linkage and seasonal apportionment of potential effects

2168. The mean maximum foraging range of gannet is 315.2km (±194.2km), and the maximum foraging range is 709km (Woodward *et al*., 2019). The Project is located approximately 575km from Sule Skerry and Sule Stack SPA, which means that the Project is beyond the mean maximum foraging range +1SD for gannets from the SPA, but within the maximum foraging range. The maximum foraging range is a poor indicator of typical foraging behaviour. It would be expected that few birds or foraging trips will occur at this distance from the colony, and even fewer with any regularity. No impacts during the breeding season are therefore apportioned to birds breeding at this colony.

2169. Outside the breeding season breeding gannets, including those from the Sule Skerry and Sule Stack SPA, are not constrained by requirements to visit nests to incubate eggs or provision chicks. At this time, they are assumed to range more widely and to mix with gannets of all age classes from breeding colonies in the UK and further afield. The background population during these seasons is the UK Western Waters BDMPS. This consists of 545,954 individuals during autumn migration (September to November), and 661,888 individuals during spring migration (December to March) (Furness, 2015).

2170. Furness (2015) estimated that 90% of the Sule Skerry and Sule Stack SPA breeding adults (9,350) are present within the UK Western Waters BDMPS during the autumn migration period, which is 8,415 birds, but that all of the SPA population (i.e. 9,350 birds) is present during spring migration. Estimates of the proportion of gannets present at the windfarm site which originate from the Sule Skerry and Sule Stack SPA during the non-breeding season (and therefore the proportion of predicted mortalities from the SPA population) are based on these population estimates as a proportion of the UK Western Waters BDMPS during the relevant season. During autumn migration and spring migration, 1.5%, and 1.4% of impacts are considered to affect birds from the SPA respectively (Furness, 2015).

### Potential effects on the qualifying feature from the Project-alone

2171. The gannet qualifying feature of the Sule Skerry and Sule Stack SPA has been screened into the assessment due to the potential risk of collision and operational phase displacement/barrier effects during the operation and maintenance phase of the Project.

### Operation and maintenance phase displacement/barrier effects

2172. Displacement effects for gannet for the Project were assessed during the autumn and spring migration periods, based on an unapportioned peak mean population of 124 and eight birds respectively, calculated for the windfarm site and a 2km buffer, in line with recommendations within the SNCB guidance (SNCB, 2017). As set out above, no gannets present at the windfarm site have been apportioned to Sule Skerry and Sule Stack SPA during the breeding season. The displacement matrices used to calculate potential impacts are presented in **Appendix 12.1** of the ES and summarised in **Table 8.138**. The inclusion of all birds within the 2km buffer, to determine the total number of birds subject to displacement, is precautionary, as in reality the avoidance rate is likely to fall with distance from the windfarm site.

2173. A displacement rate of 60-80% and mortality rate of 1% has been presented. A maximum 1% mortality value has been selected firstly because gannet is known to possess high habitat flexibility (Furness and Wade, 2012). This suggests that displaced birds will readily find alternative habitats including foraging areas. Secondly, no evidence of displacement-induced mortality has been identified, which means there is limited justification for setting predicted mortality rates at a higher level. Given the extensive foraging range of this species (Woodward *et al.*, 2019), there may be no mortality costs to displacement from the relatively very small footprints of OWFs.

*Table 8.138 Gannet – predicted operation and maintenance phase displacement and mortality from Sule Skerry and Sule Stack SPA*

| Mean peak abundance estimate type | Mean peak abundance estimate | Number of SPA breeding adults present by season[1] | Annual mortality range[2] | Annual baseline mortality increase range[3] |
|---|---|---|---|---|
| Upper 95% CI | 809 (breeding)<br>189 (autumn)<br>16 (spring)<br>1,014 (year round) | 0 (breeding)<br>3 (autumn)<br>0 (spring)<br>3 (year round) | 0-0 | 0.00-0.00% |
| Mean | 541 (breeding)<br>124 (autumn)<br>8 (spring)<br>673 (year round) | 0 (breeding)<br>2 (autumn)<br>0 (spring)<br>2 (year round) | 0-0 | 0.00-0.00% |
| Lower 95% CI | 160 (breeding)<br>0 (autumn)<br>0 (spring)<br>160 (year round) | 0 (breeding)<br>0 (autumn)<br>0 (spring)<br>0 (year round) | 0-0 | 0.00-0.00% |

[1] 1.5% and 1.4% of birds are assumed to be breeding adults from the SPA population during the autumn and spring migration periods respectively.

[2] Assumes displacement rates of 60-80% and mortality rate of 1%

[3] Background population is Sule Skerry and Sule Stack SPA breeding adults (9,030 individuals), adult age class annual mortality rate of 8.1% (Horswill and Robinson, 2015)

2174. Using the maximum potential mortality value, there would be no measurable increase in gannet mortality. Accordingly, no significant effects on gannet are predicted during the operation and maintenance phase, and **it is concluded that there would be no potential for the Project to have an adverse effect on the integrity of Sule Skerry and Sule Stack SPA.**

2175. The confidence in the assessment is high for several reasons. Firstly, the evidence used to set the displacement rates presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES is of high applicability and quality. Whilst there is limited available evidence to inform mortality rates, 1% is considered to be sufficiently precautionary based on expert opinion. Finally, the conclusion of the assessment is the same irrespective of whether the mean or 95% upper CI mean peak abundances are used to calculate potential mortality and increases in the baseline mortality rate of the background population.

**Operation and maintenance phase collision risk**

2176. Information to support the Appropriate Assessment for collision risk on breeding adult gannets belonging to the Sule Skerry and Sule Stack SPA population is presented in **Table 8.139**. Collision estimates, calculated using the sCRM, are presented by biological season. A summary of the annual outputs and the corresponding increase in the annual baseline mortality rate is also presented. Parameters used in the sCRM were agreed with Natural England during the ETG process and are described in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES. In accordance with Natural England advice, a 70% macro-avoidance correction was applied to gannet abundance data used in the sCRM.

2177. Based on the mean collision rates, no breeding adult gannets from Sule Skerry and Sule Stack SPA are considered at risk of collision as a result of the Project. Therefore, there would be no measurable increase the existing mortality of the SPA breeding population.

*Table 8.139 Predicted seasonal and annual collision mortality (Stochastic model Option 2, avoidance rate 0.993 (±0.0003), plus 70% macro-avoidance) for breeding adult gannets at the windfarm site, apportioned to Sule Skerry and Sule Stack SPA, with corresponding increases to baseline mortality of the population*

| Period | Breeding Season Apr-Aug | Autumn Migration Sep-Nov | Non-breeding/winter - | Spring Migration Dec-Mar | Annual Jan-Dec |
|---|---|---|---|---|---|
| Total collisions[1] (mean and 95% Cis) | 0.83 (0.00-3.35) | 0.14 (0.00-0.74) | - | 0.00 | 0.97 (0.00-4.10) |
| % apportioned to the SPA | 0.0% | 1.5% | - | 1.4% | - |
| Total SPA collisions (mean and 95% Cis) | 0.00 (0.00-0.00) | 0.00 (0.00-0.01) | - | 0.00 (0.00-0.00) | 0.00 (0.00-0.01) |
| Mortality increase[2] (mean and 95% Cis) | 0.00% (0.00-0.00%) | 0.00% (0.00-0.00%) | - | 0.00% (0.00-0.00%) | 0.00% (0.00-0.00%) |

[1] Breeding season collision values reduced to 73.8% of modelled value to reflect proportion of adult birds recorded during site surveys
[2] Assuming predicted annual SPA mortality of 731 birds (9,030 x 0.081)

2178. Accordingly, no significant effects on gannet are predicted during the operation and maintenance phase, **and it is concluded that there would be no potential for the Project to have an adverse effect on the integrity of Sule Skerry and Sule Stack SPA.** Comments received from RSPB during the ETG process, indicating that they do not accept the 70% macro-avoidance rate for collision risk recommended by Natural England, are noted. However, even in the absence of this correction factor, the net increase in mortality would be unchanged (i.e. zero).

2179. The confidence in the assessment is high. The evidence used to define the CRM input parameters presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES submitted alongside this RIAA is of high applicability and quality. Whilst there is uncertainty around some of the input parameters (e.g. avoidance rate), the rates selected are considered to be sufficiently precautionary based on expert opinion to provide confidence that collision rates are not underestimated.

**Combined displacement/barrier effects and collision risk**

2180. As no measurable increase in mortality is predicted for both displacement and collision risk, the mean combined displacement and collision rates for breeding adult gannet from the Sule Skerry and Sule Stack SPA would be zero. Therefore, there would be no net increase in existing mortality rates.

2181. **It is concluded that based on predicted gannet mortality due to the combined effects of operational phase displacement and collision there is no potential for an adverse effect on the integrity of the Sule Skerry and Sule Stack SPA.**

2182. The confidence in the assessment is high, for the reasons provided in the individual displacement and collision assessments.

### Potential effects in-combination with other projects

2183. As no measurable effects of displacement/barrier and collision on gannet are predicted as a result of the Project-alone, there would be no contribution to other plans or projects in-combination. **It is therefore concluded that there would be no potential for the Project-alone or in-combination to have an adverse effect on the integrity of Sule Skerry and Sule Stack SPA**.

## 8.52.3.3 Guillemot

### Status

2184. The Sule Skerry and Sule Stack SPA breeding guillemot population was cited as 6,298 pairs, or 12,596 breeding adults, in 1986 (Furness, 2015, Stroud *et al*., 2016). Furness (2015) gave a breeding population of 7,633 pairs, or 15,266 breeding adults, in 1998. The most recent count is 10,068 individuals in 2018 (JNCC, 2023a); this is used as the reference population for the assessment.

2185. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.061 (1 – 0.939; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 614 breeding adults.

## Functional linkage and seasonal apportionment of potential effects

2186. The mean maximum foraging range of guillemot is 73.2km (±80.5km) and the maximum foraging range is 338km (Woodward *et al*., 2019). The Project is located approximately 575km from Sule Skerry and Sule Stack SPA, which means that the Project is beyond the maximum foraging range for guillemots from the SPA. No impacts during the breeding season from the Project are therefore apportioned to guillemots breeding at this SPA.

2187. Outside the breeding season, breeding guillemots from the SPA are assumed to range widely and to mix with guillemots of all ages from breeding colonies in the UK and beyond. The relevant non-breeding season reference population is the UK Western Waters BDMPS, consisting of 1,139,220 individuals (August to February) (Furness, 2015).

2188. Furness (2015) estimated that 95% of the Sule Skerry and Sule Stack SPA breeding adults are present within the UK Western Waters BDMPS during the non-breeding season, which is 14,503 birds. This represents 1.3% of the BDMPS population for this period (1,139,220). It is therefore assumed that 1.3% of guillemots present at the Project site are breeding adults from Sule Skerry and Sule Stack SPA.

## Potential effects on the qualifying feature

### Operation and maintenance phase disturbance/displacement/barrier effects

*Project-alone*

2189. The mean peak abundance of guillemots present within the windfarm site and 2km buffer during the non-breeding season is 8,315 (6,085-12,047) individuals (refer to **Appendix 12.1** of the ES). Of these, 108 birds (79-157) are likely to be breeding adults from the Sule Skerry and Sule Stack SPA.

2190. **Table 8.140** sets out the predicted impacts on guillemots from Sule Skerry and Sule Stack SPA during the non-breeding season. Displacement rates of 30% to 70% are considered for this species, along with a range of mortality rates of 1% to 10% of displaced birds (UK SNCBs, 2017).

2191. The available evidence suggests that the upper ranges of these displacement and mortality rates may be excessively precautionary. Whilst it is true that guillemots and razorbills tend to be displaced by OWFs they do not avoid them completely, and displacement rates vary between sites. MacArthur Green (2019b) concluded that, on average, densities within OWFs were around half of the density found in the habitats around the OWF. At some OWFs there was also displacement of birds from a buffer zone surrounding it. The size of the buffer zone varied between OWFs and was generally less than 2km, with auk density increasing across the buffer zone as distance from turbines increased, up to the density in the wider area. Another recent review (APEM, 2022) found that the current displacement rates suggested by the SNCBs for guillemot and razorbill (30% to 70%) did not account for the quality of or confidence in the studies which were used to inform this position, and that studies where no significant effects were recorded were not accounted for during the provision of the advice. APEM (2022) suggested that in the case of Hornsea Project Four, an evidence-based displacement rate of 50% was appropriate. However, the study also recognised that larger displacement effects were possible.

2192. Mortality due to displacement could arise due to increased energy costs and/or decreased energy intake, if displacement results in increased flying time to avoid OWFs, and/or increased bird densities and competition for prey in areas of unimpacted habitat outside the OWF. Given that UK OWFs in the Irish and Celtic Seas represent a very small proportion of the available foraging habitat for guillemots and razorbills within UK Western Waters, the increase in density of auks outside OWFs due to displacement is likely to be negligible (MacArthur Green, 2019b). It is considered unlikely that the mortality rate due to displacement would be as high as the empirically estimated 6.1% annual mortality for adult guillemots that occurs due to the combination of 'natural' factors and anthropogenic activities (Horswill and Robinson, 2015). Indeed, it may be much lower; MacArthur Green (2019b) recommended precautionary rates of 50% displacement for guillemot and 1% mortality of displaced birds based on a review of available evidence. Modelling undertaken by APEM (2022) found that in the case of Hornsea Project Four, a mortality rate of 1% for displaced auks was considered to be precautionary, and that this may be an overestimate given that Hornsea Project Four is located towards the upper end of the mean maximum foraging range for guillemot from the closest designated sites. It is also noted that in the recent decision on Hornsea Project Four, the SoS determined that a rate of 70% displacement/2% mortality was appropriate to inform the assessment of effects on guillemots from Flamborough and Filey Coast SPA (DESNZ, 2023b).

2193. The full ranges of recommended displacement and mortality effects are considered by the assessment, along with evidence-based displacement and mortality rates of 50% and 1%, respectively (APEM, 2022; MacArthur Green, 2019b).

*Table 8.140 Guillemot – predicted operation and maintenance phase displacement and mortality from Sule Skerry and Sule Stack SPA*

| Mean peak abundance estimate type | Mean peak abundance estimate | Number of Sule Skerry and Sule Stack SPA breeding adults present (non-breeding season)[1] | Annual mortality range[2] | Annual baseline mortality increase range[3] |
|---|---|---|---|---|
| Upper 95% CI | 12,047 | 157 | 0-11 | 0.08-1.79% |
| Mean | 8,315 | 108 | 0-8 | 0.05-1.23% |
| Lower 95% CI | 6,085 | 79 | 0-6 | 0.04-0.90% |

[1] Assumes 1.3% of birds present during the non-breeding season are Sule Skerry and Sule Stack SPA breeding adults

[2] Assumes displacement rates of 30-70% and mortality rates of 1-10%

[3] Background mortality rate of 6.1% (Horswill and Robinson, 2015)

2194. Based on the mean peak abundances, the annual total of guillemots from the Sule Skerry and Sule Stack SPA at risk of displacement is 108 birds (**Table 8.140**). At displacement rates of 30% to 70%, and mortality rates of 1% to 10% for displaced birds, 0 to 8 SPA breeding adults would be predicted to die each year due to displacement from the Project.

2195. Assuming a displacement rate of 70% and a mortality rate of 10% of displaced birds, annual mortality within the SPA breeding adult population would increase by 1.23%. Using an evidence-based displacement rate of 50%, and a mortality rate for displaced birds of 1%, annual mortality in the population would instead increase by 0.09% (1 bird).

2196. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that no detectable changes in mortality rates would occur under almost any combination of displacement and mortality rates when the mean peak abundance estimate assessments are considered. Mortality rate increases of over 1% are predicted for mean peak abundance estimate assessments only when a displacement rate of 70%/60% and a mortality rate of 10% is considered. These displacement and mortality rates are much higher than evidence suggested will actually be the case. Use of the evidence-based displacement (50%) and mortality rate (1%) would result in a mortality increase of significantly less than 1%, as would a rate of 70%/2% agreed by the SoS in respect of Hornsea Project Four (DESNZ, 2023b).

2197. Increases of over 1% are predicted if the upper 95% CIs for mean peak abundances are used as inputs to the assessment alongside a 10% mortality rate for displaced birds. The probability of this occurring is extremely small for two reasons. Firstly, the upper 95% CI for the mean peak abundances are highly unlikely to occur regularly at the windfarm site. Secondly, mortality rates for displaced birds of 10% are much higher than evidence suggests will actually be the case, and use of the evidence-based displacement (50%) and mortality rate (1%) (and also 70%/2%) would again result in a mortality increase of significantly less than 1%.

2198. **It is concluded that predicted guillemot mortality due to operational phase displacement for the project alone would not adversely affect the integrity of the Sule Skerry and Sule Stack SPA.**

2199. The confidence in the assessment is high for several reasons. Firstly, the evidence used to inform the evidence-based displacement rates is of high applicability and quality (based on the criteria discussed in **Chapter 12 Offshore Ornithology** of the ES). Whilst there was limited available evidence to inform mortality rates, 1% is considered to be sufficiently precautionary based on expert opinion. This species is not regarded as being highly specialised in its habitat requirements (Bradbury *et al*., 2014; Furness and Wade, 2012; Garthe and Hüppop, 2004), and it is therefore anticipated that displaced birds will find alternative habitat in the vast majority of cases. Finally, the conclusion of the assessment is the same irrespective of whether the mean or 95% upper CI mean peak abundances are used to calculate potential mortality and increases in the baseline mortality rate of the background population, provided the evidence-based displacement and mortality rates are used.

*In-combination*

2200. The in-combination assessment for guillemots from Sule Skerry and Sule Stack SPA has been undertaken in accordance with the approach presented in **Section 8.1**. The total population apportioned to Sule Skerry and Sule Stack SPA at risk of displacement is estimated to be 777 breeding adults (**Appendix 12.1** of the ES). Annual in-combination displacement and mortality rates for birds from Sule Skerry and Sule Stack SPA are presented in **Table 8.141**.

2201. Assuming a displacement rate of 70% and a mortality rate of 10% of displaced birds, 54 breeding adult SPA birds would be lost to displacement annually. Including additional apportioned mortality from the Morlais and Holyhead tidal projects (total 0.41 birds), this would increase the existing mortality within the SPA population (614 breeding adult birds per year) by 8.92%. Using an evidence-based displacement rate of 50%, and a mortality rate for displaced birds of 1%, the annual in-combination displacement mortality would be 4 birds. This would increase the existing mortality within this population by 0.70%. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that detectable changes in mortality rates will not occur due to the level or mortality predicted if the more realistic rates for mortality are used.

2202. **It is concluded that predicted guillemot mortality due to of operational phase displacement impacts of the Project in-combination with other projects, would not adversely affect the integrity of Sule Skerry and Sule Stack SPA.**

*Table 8.141 In-combination year-round displacement matrix for guillemot from Sule Skerry and Sule Stack SPA*

| Annual | Mortality | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| **Displacement** | **1%** | **2%** | **3%** | **4%** | **5%** | **10%** | **20%** | **30%** | **50%** | **80%** | **100%** |
| **10%** | 1 | 2 | 2 | 3 | 4 | 8 | 16 | 23 | 39 | 62 | 78 |
| **20%** | 2 | 3 | 5 | 6 | 8 | 16 | 31 | 47 | 78 | 124 | 155 |
| **30%** | 2 | 5 | 7 | 9 | 12 | 23 | 47 | 70 | 117 | 186 | 233 |
| **40%** | 3 | 6 | 9 | 12 | 16 | 31 | 62 | 93 | 155 | 249 | 311 |
| **50%** | 4 | 8 | 12 | 16 | 19 | 39 | 78 | 117 | 194 | 311 | 388 |
| **60%** | 5 | 9 | 14 | 19 | 23 | 47 | 93 | 140 | 233 | 373 | 466 |
| **70%** | 5 | 11 | 16 | 22 | 27 | 54 | 109 | 163 | 272 | 435 | 544 |
| **80%** | 6 | 12 | 19 | 25 | 31 | 62 | 124 | 186 | 311 | 497 | 621 |
| **90%** | 7 | 14 | 21 | 28 | 35 | 70 | 140 | 210 | 350 | 559 | 699 |
| **100%** | 8 | 16 | 23 | 31 | 39 | 78 | 155 | 233 | 388 | 621 | 777 |

Note: The cells show the number of birds predicted to die (rounded to the nearest integer) at a given rate of displacement and mortality. Highlighted cells are considered to be the most realistic scenario, in accordance with SNCB advice (SNCBs, 2022).

#### 8.52.3.4 Puffin

##### Status

2203. The Sule Skerry and Sule Stack SPA breeding puffin population is cited as 43,380 pairs, or 86,760 breeding adults, in 1993 (Furness, 2015, Stroud *et al*., 2016). Furness (2015) gave a breeding population of 59,471 breeding pairs, or 118,942 breeding adults, in 1998. The most recent count is 47,742 pairs (apparently occupied burrows), or 95,484 breeding adults, in 2018 (JNCC, 2023a); this is used as the reference population for the assessment.

2204. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.094 (1 – 0.906, Horswill and Robinson (2015), the expected annual mortality from the SPA population would be 8,975 breeding adults.

##### Functional linkage and seasonal apportionment of potential effects

2205. The mean maximum foraging range of puffin is 137.1km (±128.3km) and the maximum foraging range is 383km (Woodward *et al*., 2019). The Project is located approximately 575km from Sule Skerry and Sule Stack SPA, which means the Project is beyond the maximum foraging range of puffins from the SPA. No impacts during the breeding season from the Project are therefore apportioned to puffins breeding at this SPA.

2206. Outside of the breeding season, breeding puffins, including those from the Sule Skerry and Sule Stack SPA, are not constrained by requirements to visit nests to incubate eggs or provision chicks. At this time, they are assumed to range more widely and to mix with puffins of all age classes from breeding colonies in the UK and further afield. The background population during these seasons is the UK Western Waters BDMPS. This consists of 304,557 individuals during the non-breeding season (August to March).

2207. Furness (2015) estimates that 18% of the Sule Skerry and Sule Stack SPA breeding adults (118,942) are present within the UK Western Waters BDMPS during the non-breeding season, which is 21,410 birds. This represents 7.0% of the BDMPS population for this period (304,557). It is therefore assumed that 7.0% of puffins present at the Project site are breeding adults from Sule Skerry and Sule Stack SPA.

##### Potential effects on the qualifying feature

**Operation and maintenance phase disturbance/displacement/barrier effects**

*Project-alone*

2208. The mean peak abundance of puffins present within the windfarm site and 2km buffer during the non-breeding season is 19.7 (1.9-50.8) individuals (refer to **Appendix 12.1** of the ES). Of these, approximately one bird (1.4 (0.1-3.6)) is likely to be a breeding adult from Sule Skerry and Sule Stack SPA.

2209. **Table 8.142** sets out the predicted impacts on puffins from Sule Skerry and Sule Stack SPA during the non-breeding season. This estimates that there would be no measurable increase in mortality from the SPA population, assuming a displacement rate of 30-70% and a mortality of 1-10% for displaced birds.

*Table 8.142 Puffin – predicted operation and maintenance phase displacement and mortality from Sule Skerry and Sule Stack SPA*

| Mean peak abundance estimate type | Mean peak abundance estimate | Number of Sule Skerry and Sule Stack SPA breeding adults present (non-breeding season)[1] | Annual mortality range[2] | Annual baseline mortality increase range[3] |
|---|---|---|---|---|
| Upper 95% CI | 50.8 | 3.6 | 0-1 | 0.00-0.00% |
| Mean | 19.7 | 1.4 | 0-0 | 0.00-0.00% |
| Lower 95% CI | 1.9 | 0.1 | 0-0 | 0.00-0.00% |

[1] Assumes 7.0% of birds present during the non-breeding season are Sule Skerry and Sule Stack SPA breeding adults
[2] Assumes displacement rates of 30-70% and mortality rates of 1-10%
[3] Background mortality rate of 9.4% (Horswill and Robinson, 2015)

2210. **It is concluded that predicted puffin mortality due to operational phase displacement would not adversely affect the integrity of the Sule Skerry and Sule Stack SPA.**

2211. The confidence in the assessment is high for several reasons. Firstly, the evidence used to inform the evidence-based displacement rates is of high applicability and quality (based on the criteria discussed in **Chapter 12 Offshore Ornithology** of the ES). Whilst there is limited available evidence to inform mortality rates, 1% is considered to be sufficiently precautionary based on expert opinion. This species is not regarded as being highly specialised in its habitat requirements (Bradbury *et al*., 2014; Furness and Wade, 2012; Garthe and Hüppop, 2004), and it is therefore anticipated that displaced birds will find alternative habitat in the vast majority of cases. Finally, the conclusion of the assessment is the same irrespective of whether the mean or 95% upper CI mean peak abundances are used to calculate potential mortality and increases in the baseline mortality rate of the background population.

*In-combination*

2212. As the Project would have no measurable effect on puffin populations from the Sule Skerry and Sule Stack SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that predicted puffin mortality due to operational phase displacement due to the Project in-combination with other plans or projects would not adversely affect the integrity of the Sule Skerry and Sule Stack SPA.**

# 8.53 Rousay SPA

2213. Rousay SPA is located approximately 589km from the windfarm site.

## 8.53.1 Description of designation

2214. The island of Rousay lies to the north-west of the Orkney mainland. The site is composite and consists of two parts located at the north-west and north-east ends of the island. Here, sea-cliffs grade inland to areas of maritime heath and grassland. The maritime heath contains numerous base-rich flushes characterised by black bog-rush, various sedges and grasses. The maritime heath also supports colonies of the nationally scarce Scottish primrose. The site holds a diverse assemblage of breeding seabirds, which feed in the waters around Rousay outside the SPA, as well as further away.

## 8.53.2 Conservation objectives

2215. The overarching conservation objectives for the site are:

- To avoid deterioration of the habitats of the qualifying species or significant disturbance to the qualifying species, thus ensuring that the integrity of the site is maintained
- To ensure for the qualifying species that the following are maintained in the long term:
  - Population of the species as a viable component of the site
  - Distribution of the species within site
  - Distribution and extent of habitats supporting the species
  - Structure, function and supporting processes of habitats supporting the species
  - No significant disturbance of the species

## 8.53.3 Assessment

2216. One qualifying features of Rousay SPA has been screened into the Appropriate Assessment (**Table 5.2**): breeding fulmar. This species has also been screened in as a component of the seabird assemblage.

### 8.53.3.1 Fulmar

#### Status

2217. The Rousay SPA breeding fulmar population was cited as 1,240 pairs, or 2,480 breeding adults, in 2000 (Furness, 2015, Stroud *et al*., 2016). Furness

(2015) gave a breeding population of 1,030 pairs, or 2,060 breeding adults, in 2009. The most recent complete count is 2,129 pairs, or 4,258 breeding adults, in 2016 (JNCC 2023); this is used as the reference population for the assessment.

2218. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.064 (1 – 0.936; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 273 breeding adults.

### Functional linkage and seasonal apportionment of potential effects

2219. The mean maximum foraging range of fulmar is 542.3km (±657.9km) and the maximum foraging range is 2,736km (Woodward *et al*., 2019). The Project is located approximately 589km from Rousay SPA, which means that the Project is beyond the mean maximum foraging range of fulmars breeding at this SPA, but within the mean maximum foraging range +1SD, and the maximum foraging range.

### Potential effects on the qualifying feature from the Project-alone

2220. Fulmar are considered to have low vulnerability to both collision and displacement impacts from offshore windfarms (Bradbury *et al*., 2014). This species was recorded infrequently within the windfarm site (in only five of 24 surveys, all during the breeding season (January to August; Furness, 2015) and in low numbers, with densities in four of the five months less than 0.1 birds/km$^2$. A peak density of 0.23 birds/km$^2$ was recorded in May 2022.

2221. On the basis of the low vulnerability of fulmars to potential impacts, and the low frequency and abundance of birds present at the windfarm site, it is concluded that any effects on breeding populations at Rousay SPA are very unlikely, both during and outside of the breeding season.

2222. **It is therefore concluded that there would be no measurable effects on fulmar due to the project alone, and no adverse effect on the integrity of the Rousay SPA is predicted.**

### Potential effects on the qualifying feature in-combination with other projects

2223. As the Project would have no measurable effect on fulmar populations from the Rousay SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of Rousay SPA, when assessed in-combination with other plans or projects.**

# 8.54 North Rona and Sula Sgeir SPA

2224. North Rona and Sula Sgeir SPA is located approximately 597km from the windfarm site.

## 8.54.1 Description of designation

2225. The uninhabited islands of North Rona and Sula Sgeir, together with several outlying rocky islets and adjacent waters, lie 65km north of Lewis. The coastlines of both islands consist mainly of cliffs except for two low-lying peninsulas on North Rona. North Rona is well covered by peat or soil and vegetated by sub-maritime grassland. Sula Sgeir lies about 15km west of North Rona. It is much the smaller of the two islands and has little soil or vegetation. The boundary of the Special Protection Area overlaps with the boundary of North Rona & Sula Sgeir SSSI, and the seaward elements extend approximately 2km into the marine environment to include the seabed, water column and surface. Qualifying seabird species of the SPA comprise fulmar, Leach's petrel, storm petrel, great black-backed gull, kittiwake, gannet, guillemot, razorbill and puffin.

## 8.54.2 Conservation objectives

2226. The overarching conservation objectives for the site are:

- To avoid deterioration of the habitats of the qualifying species or significant disturbance to the qualifying species, thus ensuring that the integrity of the site is maintained
- To ensure for the qualifying species that the following are maintained in the long term:
  - Population of the species as a viable component of the site
  - Distribution of the species within site
  - Distribution and extent of habitats supporting the species
  - Structure, function and supporting processes of habitats supporting the species
  - No significant disturbance of the species

## 8.54.3 Assessment

2227. The qualifying features of North Rona and Sula Sgeir SPA screened into the Appropriate Assessment are listed in **Table 5.2**. These are breeding fulmar, breeding Leach's storm-petrel, breeding gannet and breeding guillemot. The seabird assemblage has also been screened in for these species.

#### 8.54.3.1 Fulmar

**Status**

2228. The North Rona and Sula Sgeir SPA breeding fulmar population was cited as 11,500 pairs, or 23,000 breeding adults, in 1985-86 (Furness, 2015, Stroud *et al.*, 2016). Furness (2015) gave the breeding population for North Rona only as 1,438 pairs, or 2,876 breeding adults in 2012. The most recent count is 2,210 nests (AON), or 4,420 breeding adults, in 2021 (JNCC, 2023a); this is used as the reference population for the assessment.

2229. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.064 (1 – 0.936; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 283 breeding adults.

**Functional linkage and seasonal apportionment of potential effects**

2230. The mean maximum foraging range of fulmar is 542.3km (±657.9km) and the maximum foraging range is 2,736km (Woodward *et al.*, 2019). The Project is located approximately 597km from North Rona and Sula Sgeir SPA, which means that the Project is beyond the mean maximum foraging range of fulmars breeding at this SPA, but within the mean maximum foraging range +1SD, and the maximum foraging range.

**Potential effects on the qualifying feature from the Project-alone**

2231. Fulmar were considered to have low vulnerability to both collision and displacement impacts from offshore windfarms (Bradbury *et al.*, 2014). This species was recorded infrequently within the windfarm site (in only five of 24 surveys, all during the breeding season (January to August; Furness, 2015)) and in low numbers, with densities in four of the five months less than 0.1 birds/km$^2$. A peak density of 0.23 birds/km$^2$ was recorded in May 2022.

2232. On the basis of the low vulnerability of fulmars to potential impacts, and the low frequency and abundance of birds present at the windfarm site, it is concluded that any effects on breeding populations at North Rona and Sula Sgeir SPA are very unlikely, both during and outside of the breeding season.

2233. **It is therefore concluded that there would be no measurable effects on fulmar due to the project alone, and no adverse effect on the integrity of the North Rona and Sula Sgeir SPA is predicted.**

**Potential effects on the qualifying feature in-combination with other projects**

2234. As the Project would have no measurable effect on fulmar populations from the North Rona and Sula Sgeir SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there**

**would be no adverse effect on the integrity of North Rona and Sula Sgeir SPA, when assessed in-combination with other plans or projects.**

### 8.54.3.2 Leach's storm-petrel

#### Status

2235. The North Rona and Sula Sgeir SPA breeding Leach's storm-petrel population at classification (2001) was 2,750 pairs, or 5,500 breeding adults. Stroud *et al.,* (2016) gave a population of 713 pairs, or 1,426 breeding adults, in 2009; in the absence of more recent SMP data, this is used as the reference population for the assessment.

#### Functional linkage and seasonal apportionment of potential effects

2236. The mean foraging range of Leach's storm-petrel is 657km (Woodward *et al*., 2019); estimates for maximum and mean maximum foraging ranges are not available. The Project is located approximately 597km from North Rona and Sula Sgeir SPA, which means that the Project is within the mean foraging range of Leach's storm-petrels breeding at this SPA.

#### Potential effects on the qualifying feature from the Project-alone

2237. Leach's storm-petrel was not recorded during baseline surveys of the windfarm site (including buffer areas). It is therefore concluded that this species does not occur regularly in this area. It is noted that Leach's storm-petrel is considered to have low vulnerability to collision risk and very low vulnerability to displacement impacts (Bradbury *et al*., 2014), and therefore the risk of significant effects would be low, even if this species occurred at the windfarm site.

2238. **It is therefore concluded that there would be no measurable effects on Leach's storm-petrel due to the project alone, and no adverse effect on the integrity of the North Rona and Sula Sgeir SPA is predicted**.

#### Potential effects on the qualifying feature in-combination with other projects

2239. As the Project would have no measurable effect on Leach's storm-petrel populations from the North Rona and Sula Sgeir SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of North Rona and Sula Sgeir SPA, when assessed in-combination with other plans or projects.**

#### 8.54.3.3 Gannet

##### Status

2240. The North Rona and Sula Sgeir SPA breeding gannet population was cited as 9,000 pairs, or 18,000 breeding adults, in 2001 (Furness, 2015, Stroud *et al*., 2016). Furness (2015) gave a breeding population of 9,225 pairs, or 18,450 breeding adults in 2004. The most recent count is 11,230 pairs (AOS), or 22,460 breeding adults, in 2013 (JNCC, 2023a); this is used as the reference population for the assessment.

2241. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.081 (1 – 0.919; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 1,819 breeding adults.

##### Functional linkage and seasonal apportionment of potential effects

2242. The mean maximum foraging range of gannet is 315.2km (±194.2km), and the maximum foraging range is 709km (Woodward *et al*., 2019). The Project is located approximately 597km from North Rona and Sula Sgeir SPA, which means that the Project is beyond the mean maximum foraging range +1SD for gannets from the SPA, but within the maximum foraging range. The maximum foraging range is a poor indicator of typical foraging behaviour. It would be expected that few birds or foraging trips will occur at this distance from the colony, and even fewer with any regularity. No impacts during the breeding season are therefore apportioned to birds breeding at this colony.

2243. Outside the breeding season breeding gannets, including those from the North Rona and Sula Sgeir SPA, are not constrained by requirements to visit nests to incubate eggs or provision chicks. At this time, they are assumed to range more widely and to mix with gannets of all age classes from breeding colonies in the UK and further afield. The background population during these seasons is the UK Western Waters BDMPS. This consists of 545,954 individuals during autumn migration (September to November), and 661,888 individuals during spring migration (December to March) (Furness, 2015).

2244. Furness (2015) estimates that 90% of the North Rona and Sula Sgeir SPA breeding adults (9,350) are present within the UK Western Waters BDMPS during the autumn migration period, which is 16,605 birds, but that all of the SPA population (i.e. 18,450 birds) is present during spring migration. Estimates of the proportion of gannets present at the windfarm site which originate from the North Rona and Sula Sgeir SPA during the non-breeding season (and therefore the proportion of predicted mortalities from the SPA population) are based on these population estimates as a proportion of the UK Western Waters BDMPS during the relevant season. During autumn migration and spring migration, 3.0%, and 2.8% of impacts are considered to affect birds from the SPA respectively (Furness, 2015).

## Potential effects on the qualifying feature

2245. The gannet qualifying feature of the North Rona and Sula Sgeir SPA has been screened into the assessment due to the potential risk of collision and operational phase displacement/barrier effects during the operation and maintenance phase of the Project.

### Operation and maintenance phase displacement/barrier effects

2246. Displacement effects for gannet for the Project were assessed during the autumn and spring migration periods, based on an unapportioned peak mean population of 124 and eight birds respectively, calculated for the windfarm site and a 2km buffer, in line with recommendations within the SNCB guidance (SNCB 2017). As set out above, no gannets present at the windfarm site have been apportioned to North Rona and Sula Sgeir SPA during the breeding season. The displacement matrices used to calculate potential impacts are presented in **Appendix 12.1** of the ES, and summarised in **Table 8.143**. The inclusion of all birds within the 2km buffer, to determine the total number of birds subject to displacement, is precautionary, as in reality the avoidance rate is likely to fall with distance from the windfarm site.

2247. A displacement rate of 60-80% and mortality rate of 1% has been presented. A maximum 1% mortality value has been selected firstly because gannet are known to possess high habitat flexibility (Furness and Wade, 2012). This suggests that displaced birds will readily find alternative habitats including foraging areas. Secondly, no evidence of displacement-induced mortality has been identified, which means there is limited justification for setting predicted mortality rates at a higher level. Given the extensive foraging range of this species (Woodward *et al.*, 2019), there may be no mortality costs to displacement from the relatively very small footprints of OWFs.

*Table 8.143 Gannet – predicted operation and maintenance phase displacement and mortality from North Rona and Sula Sgeir SPA*

| Mean peak abundance estimate type | Mean peak abundance estimate | Number of SPA breeding adults present by season[1] | Annual mortality range[2] | Annual baseline mortality increase range[3] |
|---|---|---|---|---|
| Upper 95% CI | 809 (breeding)<br>189 (autumn)<br>16 (spring)<br>1,014 (year round) | 0 (breeding)<br>6 (autumn)<br>0 (spring)<br>6 (year round) | 0-0 | 0.00-0.00% |
| Mean | 541 (breeding)<br>124 (autumn)<br>8 (spring)<br>673 (year round) | 0 (breeding)<br>4 (autumn)<br>0 (spring)<br>4 (year round) | 0-0 | 0.00-0.00% |
| Lower 95% CI | 160 (breeding)<br>0 (autumn)<br>0 (spring)<br>160 (year round) | 0 (breeding)<br>0 (autumn)<br>0 (spring)<br>0 (year round) | 0-0 | 0.00-0.00% |

[1]3.0% and 2.8% of birds are assumed to be breeding adults from the SPA population during the autumn and spring migration periods respectively.

[2] Assumes displacement rates of 60-80% and mortality rate of 1%

[3] Background population is North Rona and Sula Sgeir SPA breeding adults (9,030 individuals), adult age class annual mortality rate of 8.1% (Horswill and Robinson, 2015)

2248. Using the maximum potential mortality value, there would be no measurable increase in gannet mortality. Accordingly, no significant effects on gannet are predicted during the operation and maintenance phase, and **it is concluded that there would be no potential for the Project to have an adverse effect on the integrity of North Rona and Sula Sgeir SPA**.

2249. The confidence in the assessment is high for several reasons. Firstly, the evidence used to set the displacement rates presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES is of high applicability and quality. Whilst there is limited available evidence to inform mortality rates, 1% is considered to be sufficiently precautionary based on expert opinion. Finally, the conclusion of the assessment is the same irrespective of whether the mean or 95% upper CI mean peak abundances are used to calculate potential mortality and increases in the baseline mortality rate of the background population.

**Operation and maintenance phase collision risk**

2250. Information to support the Appropriate Assessment for collision risk on breeding adult gannets belonging to the North Rona and Sula Sgeir SPA population is presented in **Table 8.144**. Collision estimates, calculated using the sCRM, are presented by biological season. A summary of the annual outputs and the corresponding increase in the annual baseline mortality rate is also presented. Parameters used in the sCRM were agreed with Natural England during the ETG process and are described in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES. In accordance with Natural England advice, a 70% macro-avoidance correction was applied to gannet abundance data used in the sCRM.

2251. Based on the mean collision rates, no breeding adult gannets from North Rona and Sula Sgeir SPA are considered at risk of collision as a result of the Project. Therefore, there would be no measurable increase the existing mortality of the SPA breeding population.

*Table 8.144 Predicted seasonal and annual collision mortality (Stochastic model Option 2, avoidance rate 0.993 (±0.0003), plus 70% macro-avoidance) for breeding adult gannets at the windfarm site, apportioned to North Rona and Sula Sgeir SPA, with corresponding increases to baseline mortality of the population*

| Period | Breeding Season Apr-Aug | Autumn Migration Sep-Nov | Non-breeding/winter - | Spring Migration Dec-Mar | Annual Jan-Dec |
|---|---|---|---|---|---|
| Total collisions[1] (mean and 95% CIs) | 0.83 (0.00-3.35) | 0.14 (0.00-0.74) | - | 0.00 | 0.97 (0.00-4.10) |
| % apportioned to the SPA | 0.0% | 3.0% | - | 2.8% | - |
| Total SPA collisions (mean and 95% CIs) | 0.00 (0.00-0.00) | 0.00 (0.00-0.02) | - | 0.00 (0.00-0.00) | 0.00 (0.00-0.02) |
| Mortality increase[2] (mean and 95% CIs) | 0.00% (0.00-0.00%) | 0.00% | - | 0.00% | 0.00% (0.00-0.00%) |

[1] Breeding season collision values reduced to 73.8% of modelled value to reflect proportion of adult birds recorded during site surveys

[2] Assuming predicted annual SPA mortality of 1,819 birds (22,460 x 0.081)

2252. Accordingly, no significant effects on gannet are predicted during the operation and maintenance phase, and **it is concluded that there would be no potential for the Project to have an adverse effect on the integrity of North Rona and Sula Sgeir SPA.** Comments received from RSPB during the ETG process, indicating that they do not accept the 70% macro-avoidance rate for collision risk recommended by Natural England, are noted. However, even in the absence of this correction factor, the net increase in mortality would be unchanged (i.e. zero).

2253. The confidence in the assessment is high. The evidence used to define the CRM input parameters presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES submitted alongside this RIAA is of high applicability and quality. Whilst there is uncertainty around some of the input parameters (e.g. avoidance rate), the rates selected are considered to be sufficiently precautionary based on expert opinion to provide confidence that collision rates are not underestimated.

**Combined displacement/barrier effects and collision risk**

2254. As no measurable increase in mortality is predicted for both displacement and collision risk, the mean combined displacement and collision rates for breeding adult gannet from the North Rona and Sula Sgeir SPA would be zero. Therefore, there would be no net increase in existing mortality rates.

2255. **It is concluded that based on predicted gannet mortality due to the combined effects of operational phase displacement and collision there is no potential for the Project-alone or in-combination to have an adverse effect on the integrity of the North Rona and Sula Sgeir SPA**.

2256. The confidence in the assessment is high, for the reasons provided in the individual displacement and collision assessments.

### Potential effects in-combination with other projects

2257. As no measurable effects of displacement /barrier and collision on gannet are predicted as a result of the Project-alone, there would be no contribution to other plans or projects in-combination. **It is therefore concluded that there would be no potential for the Project-alone or in-combination to have an adverse effect on the integrity of North Rona and Sula Sgeir SPA**.

### 8.54.3.4 Guillemot

### Status

2258. The North Rona and Sula Sgeir SPA breeding guillemot population was cited as 28,944 pairs, or 57,888 breeding adults, in 1986 (Furness, 2015, Stroud *et al*., 2016). Furness (2015) gave a breeding population of 5,000 pairs, or 10,000 breeding adults in 2012. The most recent count is 7,727 individuals in

2021 (JNCC, 2023a); this is used as the reference population for the assessment.

2259. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.061 (1 – 0.939; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 471 breeding adults.

## Functional linkage and seasonal apportionment of potential effects

2260. The mean maximum foraging range of guillemot is 73.2km (±80.5km) and the maximum foraging range is 338km (Woodward *et al*., 2019). The Project is located approximately 597km from North Rona and Sula Sgeir SPA, which means that the Project is beyond the maximum foraging range for guillemots from the SPA. No impacts during the breeding season from the Project are therefore apportioned to guillemots breeding at this SPA.

2261. Outside the breeding season, breeding guillemots from the SPA are assumed to range widely and to mix with guillemots of all ages from breeding colonies in the UK and beyond. The relevant non-breeding season reference population is the UK Western Waters BDMPS, consisting of 1,139,220 individuals (August to February) (Furness, 2015).

2262. Furness (2015) estimates that 95% of the North Rona and Sula Sgeir SPA breeding adults are present within the UK Western Waters BDMPS during the non-breeding season, which is 9,500 birds. This represents 0.8% of the BDMPS population for this period (1,139,220). It is therefore assumed that 0.8% of guillemots present at the Project site are breeding adults from North Rona and Sula Sgeir SPA.

## Potential effects on the qualifying feature

### Operation and maintenance phase disturbance /displacement/barrier effects

*Project-alone*

2263. The mean peak abundance of guillemots present within the windfarm site and 2km buffer during the non-breeding season was 8,315 (6,085-12,047) individuals (refer to **Appendix 12.1** of the ES). Of these, 67 birds (49-96) were likely to be breeding adults from the North Rona and Sula Sgeir SPA.

2264. **Table 8.145** sets out the predicted impacts on guillemots from North Rona and Sula Sgeir SPA during the non-breeding season. Displacement rates of 30% to 70% are considered for this species, along with a range of mortality rates of 1% to 10% of displaced birds (UK SNCBs, 2017).

2265. The available evidence suggests that the upper ranges of these displacement and mortality rates may be excessively precautionary. Whilst it is true that guillemots and razorbills tend to be displaced by OWFs they do not avoid them completely, and displacement rates vary between sites. MacArthur Green (2019b) concluded that, on average, densities within OWFs were around half of the density found in the habitats around the OWF. At some OWFs there was also displacement of birds from a buffer zone surrounding it. The size of the buffer zone varied between OWFs and was generally less than 2km, with auk density increasing across the buffer zone as distance from turbines increased, up to the density in the wider area. Another recent review (APEM, 2022) found that the current displacement rates suggested by the SNCBs for guillemot and razorbill (30% to 70%) did not account for the quality of or confidence in the studies which were used to inform this position, and that studies where no significant effects were recorded were not accounted for during the provision of the advice. APEM (2022) suggested that in the case of Hornsea Project Four, an evidence-based displacement rate of 50% was appropriate. However, the study also recognised that larger displacement effects were possible.

2266. Mortality due to displacement could arise due to increased energy costs and/or decreased energy intake, if displacement results in increased flying time to avoid OWFs, and/or increased bird densities and competition for prey in areas of unimpacted habitat outside the OWF. Given that UK OWFs in the Irish and Celtic Seas represent a very small proportion of the available foraging habitat for guillemots and razorbills within UK Western Waters, the increase in density of auks outside OWFs due to displacement is likely to be negligible (MacArthur Green, 2019b). It is considered unlikely that the mortality rate due to displacement would be as high as the empirically estimated 6.1% annual mortality for adult guillemots that occurs due to the combination of ‘natural’ factors and anthropogenic activities (Horswill and Robinson, 2015). Indeed, it may be much lower; MacArthur Green (2019b) recommended precautionary rates of 50% displacement for guillemot and 1% mortality of displaced birds based on a review of available evidence. Modelling undertaken by APEM (2022) found that in the case of Hornsea Project Four, a mortality rate of 1% for displaced auks was considered to be precautionary, and that this may be an overestimate given that Hornsea Project Four is located towards the upper end of the mean maximum foraging range for guillemot from the closest designated sites. It is also noted that in the recent decision on Hornsea Project Four, the SoS determined that a rate of 70% displacement/2% mortality was appropriate to inform the assessment of effects on guillemots from Flamborough and Filey Coast SPA (DESNZ, 2023b).

2267. The full ranges of recommended displacement and mortality effects are considered by the assessment, along with evidence-based displacement and mortality rates of 50% and 1%, respectively (APEM, 2022; MacArthur Green, 2019b).

*Table 8.145 Guillemot – predicted operation and maintenance phase displacement and mortality from North Rona and Sula Sgeir SPA*

| Mean peak abundance estimate type | Mean peak abundance estimate | Number of North Rona and Sula Sgeir SPA breeding adults present (non-breeding season)[1] | Annual mortality range[2] | Annual baseline mortality increase range[3] |
|---|---|---|---|---|
| Upper 95% CI | 12,047 | 96 | 0-7 | 0.05-1.11% |
| Mean | 8,315 | 67 | 0-5 | 0.03-0.76% |
| Lower 95% CI | 6,085 | 49 | 0-3 | 0.02-0.56% |

[1] Assumes 0.8% of birds present during the non-breeding season are North Rona and Sula Sgeir SPA breeding adults

[2] Assumes displacement rates of 30-70% and mortality rates of 1-10%

[3] Background mortality rate of 6.1% (Horswill and Robinson, 2015)

2268. Based on the mean peak abundances, the annual total of guillemots from the North Rona and Sula Sgeir SPA at risk of displacement is 67 birds (**Table 8.145**). At displacement rates of 30% to 70%, and mortality rates of 1% to 10% for displaced birds, 0-5 SPA breeding adults would be predicted to die each year due to displacement from the Project.

2269. Assuming a displacement rate of 70% and a mortality rate of 10% of displaced birds, annual mortality within the SPA breeding adult population would increase by 0.76%. Using an evidence-based displacement rate of 50%, and a mortality rate for displaced birds of 1%, annual mortality in the population would instead increase by 0.05% (<1 bird).

2270. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that no detectable changes in mortality rates would occur under any combination of displacement and mortality rates when the mean peak abundance estimate assessments are considered.

2271. Increases of over 1% are predicted if the upper 95% CIs for mean peak abundances are used as inputs to the assessment alongside a 10% mortality rate for displaced birds. The probability of this occurring is extremely small for two reasons. Firstly, the upper 95% CI for the mean peak abundances are highly unlikely to occur regularly at the windfarm site. Secondly, mortality rates for displaced birds of 10% are much higher than evidence suggests will actually be the case, and use of the evidence-based displacement (50%) and mortality rate (1%) (and also 70%/2%) would again result in a mortality increase of significantly less than 1%.

2272. **It is concluded that predicted guillemot mortality due to operational phase displacement for the project alone would not adversely affect the integrity of the North Rona and Sula Sgeir SPA.**

2273. The confidence in the assessment is high for several reasons. Firstly, the evidence used to inform the evidence-based displacement rates is of high applicability and quality (based on the criteria discussed in **Chapter 12 Offshore Ornithology** of the ES). Whilst there is limited available evidence to inform mortality rates, 1% is considered to be sufficiently precautionary based on expert opinion. This species is not regarded as being highly specialised in its habitat requirements (Bradbury *et al*., 2014; Furness and Wade, 2012; Garthe and Hüppop, 2004), and it is therefore anticipated that displaced birds will find alternative habitat in the vast majority of cases. Finally, the conclusion of the assessment is the same irrespective of whether the mean or 95% upper CI mean peak abundances are used to calculate potential mortality and increases in the baseline mortality rate of the background population, provided the evidence-based displacement and mortality rates are used.

*In-combination*

2274. The in-combination assessment for guillemots from North Rona and Sula Sgeir SPA has been undertaken in accordance with the approach presented in **Section 8.1**. The total population apportioned to North Rona and Sula Sgeir SPA at risk of displacement is estimated to be 397 breeding adults (**Appendix 12.1** of the ES). Annual in-combination displacement and mortality rates for birds from North Rona and Sula Sgeir SPA are presented in **Table 8.146**.

2275. Assuming a displacement rate of 70% and a mortality rate of 10% of displaced birds, 28 breeding adult SPA birds would be lost to displacement annually. Including additional apportioned mortality from the Morlais and Holyhead tidal projects (total 0.25 birds), this would increase the existing mortality within the SPA population (471 breeding adult birds per year) by 5.95%. Using an evidence-based displacement rate of 50%, and a mortality rate for displaced birds of 1%, the annual in-combination displacement mortality would be 2 birds. This would increase the existing mortality within this population by 0.47%. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that detectable changes in mortality rates will not occur due to the level or mortality predicted if the more realistic rates for mortality are used.

2276. **It is concluded that predicted guillemot mortality due to of operational phase displacement impacts of the Project in-combination with other projects, would not adversely affect the integrity of North Rona and Sula Sgeir SPA.**

*Table 8.146 In-combination year-round displacement matrix for guillemot from North Rona and Sula Sgeir SPA*

| Annual | Mortality | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| **Displacement** | **1%** | **2%** | **3%** | **4%** | **5%** | **10%** | **20%** | **30%** | **50%** | **80%** | **100%** |
| **10%** | 0 | 1 | 1 | 2 | 2 | 4 | 8 | 12 | 20 | 32 | 40 |
| **20%** | 1 | 2 | 2 | 3 | 4 | 8 | 16 | 24 | 40 | 64 | 79 |
| **30%** | 1 | 2 | 4 | 5 | 6 | 12 | 24 | 36 | 60 | 95 | 119 |
| **40%** | 2 | 3 | 5 | 6 | 8 | 16 | 32 | 48 | 79 | 127 | 159 |
| **50%** | 2 | 4 | 6 | 8 | 10 | 20 | 40 | 60 | 99 | 159 | 199 |
| **60%** | 2 | 5 | 7 | 10 | 12 | 24 | 48 | 72 | 119 | 191 | 238 |
| **70%** | 3 | 6 | 8 | 11 | 14 | 28 | 56 | 83 | 139 | 223 | 278 |
| **80%** | 3 | 6 | 10 | 13 | 16 | 32 | 64 | 95 | 159 | 254 | 318 |
| **90%** | 4 | 7 | 11 | 14 | 18 | 36 | 72 | 107 | 179 | 286 | 358 |
| **100%** | 4 | 8 | 12 | 16 | 20 | 40 | 79 | 119 | 199 | 318 | 397 |

Note: The cells show the number of birds predicted to die (rounded to the nearest integer) at a given rate of displacement and mortality. Highlighted cells are considered to be the most realistic scenario, in accordance with SNCB advice (SNCBs, 2022).

## 8.55 Calf of Eday SPA

2277. Calf of Eday SPA is located approximately 599km from the windfarm site.

### 8.55.1 Description of designation

2278. The Calf of Eday is a small, uninhabited island located to the north of the island of Eday, Orkney. The island has a rocky coastline with cliffs on the north and east coasts. The dominant vegetation on the island is dry dwarf-shrub heath dominated by heather, with smaller areas of wet heath, semi-improved grassland and coastal grassland. The site is of importance as a nesting area for breeding seabirds, which feed in surrounding waters outside the SPA and use most of the island for loafing.

### 8.55.2 Conservation objectives

2279. The overarching conservation objectives for the site are:

- To avoid deterioration of the habitats of the qualifying species or significant disturbance to the qualifying species, thus ensuring that the integrity of the site is maintained
- To ensure for the qualifying species that the following are maintained in the long term:
  - Population of the species as a viable component of the site
  - Distribution of the species within site
  - Distribution and extent of habitats supporting the species
  - Structure, function and supporting processes of habitats supporting the species
  - No significant disturbance of the species

### 8.55.3 Assessment

2280. One qualifying feature of the Calf of Eday SPA has been screened into the Appropriate Assessment (**Table 5.2**): breeding fulmar. This species has also been screened in as a component of the seabird assemblage.

#### 8.55.3.1 Fulmar

#### Status

2281. The Calf of Eday SPA breeding fulmar population was cited as 1,955 pairs, or 3,910 breeding adults, in 1998 (Furness, 2015, Stroud *et al*., 2016). Furness (2015) gave the breeding population of 1,842 pairs, or 3,684 breeding adults, in 2002. The most recent count was 1,758 pairs (AOS), or 3,516 breeding adults, in 2016 (JNCC 2023); this is used as the reference population for the assessment.

2282. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.064 (1 – 0.936; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 225 breeding adults.

### Functional linkage and seasonal apportionment of potential effects

2283. The mean maximum foraging range of fulmar is 542.3km (±657.9km) and the maximum foraging range is 2,736km (Woodward *et al*., 2019). The Project is located approximately 599km from Calf of Eday SPA, which means that the Project is beyond the mean maximum foraging range of fulmars breeding at this SPA, but within the mean maximum foraging range +1SD, and the maximum foraging range.

### Potential effects on the qualifying feature from the Project-alone

2284. Fulmar are considered to have low vulnerability to both collision and displacement impacts from offshore windfarms (Bradbury *et al*., 2014). This species was recorded infrequently within the windfarm site (in only five of 24 surveys, all during the breeding season (January to August; Furness, 2015)) and in low numbers, with densities in four of the five months less than 0.1 birds/km$^2$. A peak density of 0.23 birds/km$^2$ was recorded in May 2022.

2285. On the basis of the low vulnerability of fulmars to potential impacts, and the low frequency and abundance of birds present at the windfarm site, it is concluded that any effects on breeding populations at Calf of Eday SPA are very unlikely, both during and outside of the breeding season.

2286. **It is therefore concluded that there would be no measurable effects on fulmar due to the project alone, and no adverse effect on the integrity of the Calf of Eday SPA is predicted.**

### Potential effects on the qualifying feature in-combination with other projects

2287. As the Project would have no measurable effect on fulmar populations from the Calf of Eday SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of Calf of Eday SPA, when assessed in-combination with other plans or projects.**

## 8.56 West Westray SPA

2288. West Westray SPA is located approximately 603km from the windfarm site.

### 8.56.1 Description of designation

2289. West Westray SPA is an 8km stretch of sea cliffs, together with adjacent grassland and heathland, along the west coast of the island of Westray in Orkney. The cliffs support large colonies of breeding auks and kittiwakes while the grassland and heathland areas support breeding colonies of skuas and terns. The seaward elements of the SPA extend approximately 2km into the marine environment and include the seabed, water column and surface. Seabirds included within the designation feed both inside and outside the SPA in nearby waters, as well as more distantly in the wider North Sea.

### 8.56.2 Conservation objectives

2290. The overarching conservation objectives for the site are:

- To avoid deterioration of the habitats of the qualifying species or significant disturbance to the qualifying species, thus ensuring that the integrity of the site is maintained
- To ensure for the qualifying species that the following are maintained in the long term:
  - Population of the species as a viable component of the site
  - Distribution of the species within site
  - Distribution and extent of habitats supporting the species
  - Structure, function and supporting processes of habitats supporting the species
  - No significant disturbance of the species

### 8.56.3 Assessment

2291. The qualifying features of West Westray SPA screened into the Appropriate Assessment are listed in Table 5.2. These are breeding fulmar and breeding kittiwake. The seabird assemblage has also been screened in for these species.

#### 8.56.3.1 Fulmar

##### Status

2292. The West Westray SPA breeding fulmar population was cited as 1,400 pairs, or 2,800 breeding adults, in 1996 (Furness, 2015, Stroud *et al*., 2016). Furness (2015) gave a breeding population of 677 pairs, or 1,354 breeding adults, in 2007. The most recent complete count is 1,995 pairs (AOS), or 2,390 breeding adults, in 2017 (JNCC, 2023a); this is used as the reference population for the assessment.

2293. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.064 (1 – 0.936; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 153 breeding adults.

##### Functional linkage and seasonal apportionment of potential effects

2294. The mean maximum foraging range of fulmar is 542.3km (±657.9km) and the maximum foraging range is 2,736km (Woodward *et al*., 2019). The Project is located approximately 603km from West Westray SPA, which means that the Project is beyond the mean maximum foraging range of fulmars breeding at this SPA, but within the mean maximum foraging range +1SD, and the maximum foraging range.

##### Potential effects on the qualifying feature from the Project-alone

2295. Fulmar are considered to have low vulnerability to both collision and displacement impacts from offshore windfarms (Bradbury *et al*., 2014). This species was recorded infrequently within the windfarm site (in only five of 24 surveys, all during the breeding season (January to August; Furness, 2015)) and in low numbers, with densities in four of the five months less than 0.1 birds/km$^2$. A peak density of 0.23 birds/km$^2$ was recorded in May 2022.

2296. On the basis of the low vulnerability of fulmars to potential impacts, and the low frequency and abundance of birds present at the windfarm site, it is concluded that any effects on breeding populations at West Westray SPA are very unlikely, both during and outside of the breeding season.

2297. **It is therefore concluded that there would be no measurable effects on fulmar due to the project alone, and no adverse effect on the integrity of the West Westray SPA is predicted.**

##### Potential effects on the qualifying feature in-combination with other projects

2298. As the Project would have no measurable effect on fulmar populations from the West Westray SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no**

**adverse effect on the integrity of West Westray SPA, when assessed in-combination with other plans or projects.**

#### 8.56.3.2 Kittiwake

##### Status

2299. The West Westray SPA breeding kittiwake population was cited as 24,000 pairs, or 48,000 breeding adults, in 1996 (Furness, 2015, Stroud *et al*., 2016). Furness (2015) gave a breeding population of 12,055 pairs, or 24,110 breeding adults, in 2007. The most recent count is 2,755 pairs (AON), or 5,510 breeding adults, in 2017 (JNCC, 2023a); this is used as the reference population for the assessment.

2300. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.146 (1 – 0.854; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 804 breeding adults.

##### Functional linkage and seasonal apportionment of potential effects

2301. The mean maximum foraging range of kittiwake is 156.1km (±144.5km) and the maximum foraging range is 770km (Woodward *et al*., 2019). The Project is located approximately 603km from West Westray SPA, which means that the Project is beyond the mean maximum foraging range +1SD of kittiwakes breeding at this SPA, but within the maximum foraging range.

2302. The maximum foraging range is a poor indicator of typical foraging behaviour. It would be expected that few birds or foraging trips will occur at this distance from the colony, and even fewer with any regularity. No impacts during the breeding season are therefore apportioned to birds breeding at this colony.

2303. Outside the breeding season, breeding kittiwakes from the SPA are assumed to range widely and to mix with kittiwakes of all ages from breeding colonies in the UK and beyond. The relevant non-breeding season reference population is the UK Western Waters and Channel BDMPS, consisting of 911,586 individuals during autumn migration (September to December) and 627,816 during spring migration (January to February) (Furness, 2015).

2304. Furness (2015) estimated that 20% of the West Westray SPA breeding adults are present within the UK Western Waters and Channel BDMPS during the autumn migration period, which is 4,822 birds. During the spring migration period 30% of the population is estimated to be present, which is 7,233 birds. This represents 0.53% and 1.15% of the BDMPS population for the autumn and spring periods respectively. During autumn migration and spring migration, 0.53%, and 1.15% of impacts are therefore considered to affect birds from the SPA.

## Potential effects on the qualifying feature from the Project-alone

2305. The kittiwake qualifying feature of the West Westray SPA has been screened into the Appropriate Assessment due to the potential risk of collision.

### Operation and maintenance phase collision risk

2306. Information for collision risk on breeding adult kittiwakes belonging to the West Westray SPA population is presented in **Table 8.147**. Collision estimates, calculated using the sCRM, are presented by biological season. A summary of the annual outputs and the corresponding increase in the annual baseline mortality rate is also presented. Parameters used in the sCRM were agreed with Natural England during the ETG process and are described in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES.

2307. Based on the mean collision rates, the annual total of breeding adult kittiwakes from West Westray SPA at risk of collision as a result of the Project is less than one bird (0.05). This would increase the existing mortality of the SPA breeding population by 0.01%.

*Table 8.147 Predicted seasonal and annual collision mortality (Stochastic model Option 2, avoidance rate 0.993 (±0.0003)), for breeding adult kittiwakes at the windfarm site, apportioned to West Westray SPA, with corresponding increases to baseline mortality of the population*

| Period | Breeding Season<br>Mar-Aug | Autumn Migration<br>Sep-Dec | Non-breeding/winter<br>- | Spring Migration<br>Jan-Feb | Annual<br>Jan-Dec |
|---|---|---|---|---|---|
| Total collisions[1] (mean and 95% CIs) | 15.36<br>(4.17-33.87) | 8.49<br>(2.20-18.85) | - | 0.63<br>(0.00-1.46) | 24.48<br>(6.37-54.17) |
| % apportioned to the SPA | 0.00% | 0.53% | - | 1.15% | - |
| Total SPA collisions (mean and 95% CIs) | 0.00<br>(0.00-0.00) | 0.04<br>(0.01-0.10) | - | 0.01<br>(0.00-0.02) | 0.05<br>(0.01-0.12) |
| Mortality increase[2] (mean and 95% CIs) | 0.00%<br>(0.00-0.00%) | 0.01%<br>(0.00-0.01%) | - | 0.00%<br>(0.00-0.00%) | 0.01%<br>(0.00-0.01%) |

[1] Breeding season collision values reduced to 94.1% of modelled value to reflect proportion of adult birds recorded during site surveys

[2] Assuming predicted annual SPA mortality of 804 birds (5,510 x 0.146)

2308. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that no detectable changes in mortality rates would occur on this population for the mean monthly collision estimates for the Project, or for the upper 95% CI collision estimate.

2309. **It is concluded that based on predicted kittiwake mortality due to collision at the windfarm site there is no potential for the Project to have an adverse effect on the integrity of the West Westray SPA**.

2310. The confidence in the assessment is high. The evidence used to define the CRM input parameters presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** as part of the ES submitted alongside this RIAA is of high applicability and quality. Whilst there is uncertainty around some of the input parameters (e.g. avoidance rate), the rates selected are considered to be sufficiently precautionary based on expert opinion to provide confidence that collision rates are not underestimated.

### Potential effects on the qualifying feature in-combination with other projects

2311. As the Project would have no measurable effect on kittiwake populations from the West Westray SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of West Westray SPA, when assessed in-combination with other plans or projects**.

# 8.57 Fair Isle SPA

2312. Fair Isle SPA is located approximately 639km from the windfarm site.

## 8.57.1 Description of designation

2313. Fair Isle SPA is situated on the most southerly island of the Shetland group, lying halfway between the Shetland Mainland and Orkney. It has a rocky, cliff coastline and supports a wide range of breeding seabird populations of international importance. The seaward elements of the SPA extend approximately 2km into the marine environment and includes the seabed, water column and surface. Seabirds included within the designation feed both inside and outside the SPA in nearby waters, as well as more distantly in the wider North Sea.

## 8.57.2 Conservation objectives

2314. The overarching conservation objectives for the site are:

- To avoid deterioration of the habitats of the qualifying species or significant disturbance to the qualifying species, thus ensuring that the integrity of the site is maintained
- To ensure for the qualifying species that the following are maintained in the long term:
    - Population of the species as a viable component of the site
    - Distribution of the species within site
    - Distribution and extent of habitats supporting the species
    - Structure, function and supporting processes of habitats supporting the species
    - No significant disturbance of the species

## 8.57.3 Assessment

2315. The qualifying features of Fair Isle SPA screened into the Appropriate Assessment are listed in **Table 5.2**. These are breeding fulmar and breeding great skua. The seabird assemblage has also been screened in for these species.

#### 8.57.3.1 Fulmar

##### Status

2316. The Fair Isle SPA breeding fulmar population was cited as 43,320 pairs, or 86,640 breeding adults, in 1994 (Furness, 2015, Stroud *et al*., 2016). Furness (2015) gave a breeding population of 29,649 pairs, or 59,298 breeding adults, in 2011. The most recent count is 32,491 pairs (AOS), or 64,982 breeding adults, in 2021 (JNCC, 2023a); this is used as the reference population for the assessment.

2317. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.064 (1 – 0.936; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 4,159 breeding adults.

##### Functional linkage and seasonal apportionment of potential effects

2318. The mean maximum foraging range of fulmar is 542.3km (±657.9km) and the maximum foraging range is 2,736km (Woodward *et al*., 2019). The Project is located approximately 639km from Fair Isle SPA, which means that the Project is beyond the mean maximum foraging range of fulmars breeding at this SPA, but within the mean maximum foraging range +1SD, and the maximum foraging range.

##### Potential effects on the qualifying feature from the Project-alone

2319. Fulmar are considered to have low vulnerability to both collision and displacement impacts from offshore windfarms (Bradbury *et al*., 2014). This species was recorded infrequently within the windfarm site (in only five of 24 surveys, all during the breeding season (January to August; Furness, 2015)) and in low numbers, with densities in four of the five months less than 0.1 birds/km$^2$. A peak density of 0.23 birds/km$^2$ was recorded in May 2022.

2320. On the basis of the low vulnerability of fulmars to potential impacts, and the low frequency and abundance of birds present at the windfarm site, it is concluded that any effects on breeding populations at Fair Isle SPA are very unlikely, both during and outside of the breeding season.

2321. **It is therefore concluded that there would be no measurable effects on fulmar due to the project alone, and no adverse effect on the integrity of the Fair Isle is predicted.**

##### Potential effects on the qualifying feature in-combination with other projects

2322. As the Project would have no measurable effect on fulmar populations from the Fair Isle SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse**

**effect on the integrity of Fair Isle SPA, when assessed in-combination with other plans or projects.**

### 8.57.3.2 Great skua

#### Status

2323. The Fair Isle SPA breeding great skua population was cited as 130 pairs, or 260, breeding adults, in 1994 (Furness, 2015, Stroud *et al*., 2016). Furness (2015) gave a breeding population of 266 pairs, or 532 breeding adults, in 2013. The most recent count is 430 pairs (AOT), or 860 breeding adults, in 2020 (JNCC, 2023a); this is used as the reference population for the assessment.

2324. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.118 (1 – 0.882; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 101 breeding adults.

#### Functional linkage and seasonal apportionment of potential effects

2325. The mean maximum foraging range of great skua is 443.3km (±487.9km) and the maximum foraging range is 1,003km (Woodward *et al*., 2019). The Project is located approximately 639km from Fair Isle SPA, which means that the Project is beyond the mean maximum foraging range of great skuas breeding at this SPA, but within the mean maximum foraging range +1SD, and the maximum foraging range.

#### Potential effects on the qualifying feature from the Project-alone

**Operation and maintenance phase collision risk**

2326. The great skua qualifying feature of the Fair Isle SPA has been screened into the Appropriate Assessment due to the potential risk of collision. However, this species was not recorded within the windfarm site during site surveys, and therefore there would be no risk that collision mortality would affect great skua populations from Fair Isle SPA. It is noted that a separate assessment of collision risk to migrant great skuas has also been undertaken, as set out in Chapter 12 of the ES. This also predicted negligible annual mortality for this species (0.03 birds), which would equate to no measurable increase in mortality apportioned to populations from Fair Isle SPA. **It is concluded that there would be no adverse effect on the integrity of Fair Isle SPA**.

2327. The confidence in the assessment is high. As both the surveys of the Project site and separate migrant collision risk assessment indicate that there would be negligible collision impacts on this species, it is considered extremely unlikely that there would be any effects on populations from the SPA.

### Potential effects on the qualifying feature in-combination with other projects

2328. As the Project would have no measurable effect on great skua populations from the Fair Isle SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of Fair Isle SPA, when assessed in-combination with other plans or projects.**

# 8.58 Sumburgh Head SPA

2329. Sumburgh Head SPA is located approximately 681km from the windfarm site.

## 8.58.1 Description of designation

2330. Sumburgh Head SPA covers an area of cliffs and boulder beaches at the southern tip of Mainland, Shetland. The boundary of the SPA is coincident with that of Sumburgh Head SSSI and the seaward elements extend approximately 2km into the marine environment to include the seabed, water column and surface. Qualifying seabird species for which the SPA is designated comprise fulmar, Arctic tern, kittiwake and guillemot.

## 8.58.2 Conservation objectives

2331. The overarching conservation objectives for the site are:

- To avoid deterioration of the habitats of the qualifying species or significant disturbance to the qualifying species, thus ensuring that the integrity of the site is maintained
- To ensure for the qualifying species that the following are maintained in the long term:
    - Population of the species as a viable component of the site
    - Distribution of the species within site
    - Distribution and extent of habitats supporting the species
    - Structure, function and supporting processes of habitats supporting the species
    - No significant disturbance of the species

## 8.58.3 Assessment

2332. One qualifying feature of Sumburgh Head SPA has been screened into the Appropriate Assessment (**Table 5.2**): breeding fulmar. This species has also been screened in as a component of the seabird assemblage.

### 8.58.3.1 Fulmar

#### Status

2333. The Sumburgh Head SPA breeding fulmar population was cited as 2,542 pairs, or 5,084 breeding adults, in 1996 (Furness, 2015, Stroud *et al.*, 2016). Furness (2015) gave a breeding population of 233 pairs, or 466 breeding adults, in 2009. The most recent complete count is 4,431 pairs (AON) or 8,862 breeding adults, in 2017 (JNCC 2023); this is used as the reference population for the assessment.

2334. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.064 (1 – 0.936; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 567 breeding adults.

### Functional linkage and seasonal apportionment of potential effects

2335. The mean maximum foraging range of fulmar is 542.3km (±657.9km) and the maximum foraging range is 2,736km (Woodward *et al*., 2019). The Project is located approximately 681km from Sumburgh Head SPA, which means that the Project is beyond the mean maximum foraging range of fulmars breeding at this SPA, but within the mean maximum foraging range +1SD, and the maximum foraging range.

### Potential effects on the qualifying feature from the Project-alone

2336. Fulmar are considered to have low vulnerability to both collision and displacement impacts from offshore windfarms (Bradbury *et al*., 2014). This species was recorded infrequently within the windfarm site (in only five of 24 surveys, all during the breeding season (January to August; Furness, 2015)) and in low numbers, with densities in four of the five months less than 0.1 birds/km$^2$. A peak density of 0.23 birds/km$^2$ was recorded in May 2022.

2337. On the basis of the low vulnerability of fulmars to potential impacts, and the low frequency and abundance of birds present at the windfarm site, it is concluded that any effects on breeding populations at Sumburgh Head SPA are very unlikely, both during and outside of the breeding season.

2338. **It is therefore concluded that there would be no measurable effects on fulmar due to the project alone, and no adverse effect on the integrity of the Sumburgh Head SPA is predicted.**

### Potential effects on the qualifying feature in-combination with other projects

2339. As the Project would have no measurable effect on fulmar populations from the Sumburgh Head SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would** be **no adverse effect on the integrity of Sumburgh Head SPA, when assessed in-combination with other plans or projects**.

## 8.59 Foula SPA

2340. Foula SPA is located approximately 701km from the windfarm site.

### 8.59.1 Description of designation

2341. Foula is the most westerly of the Shetland Islands, lying 20km west of Shetland Mainland. It consists of a rocky coastline, large areas of mire, and adjacent coastal waters which support internationally important breeding populations of seabirds. The boundary of the SPA extends approximately 2km into the marine environment to include the seabed, water column and surface.

### 8.59.2 Conservation objectives

2342. The overarching conservation objectives for the site are:

- To avoid deterioration of the habitats of the qualifying species or significant disturbance to the qualifying species, thus ensuring that the integrity of the site is maintained
- To ensure for the qualifying species that the following are maintained in the long term:
  - Population of the species as a viable component of the site
  - Distribution of the species within site
  - Distribution and extent of habitats supporting the species
  - Structure, function and supporting processes of habitats supporting the species
  - No significant disturbance of the species

### 8.59.3 Assessment

2343. The qualifying features of Foula SPA screened into the Appropriate Assessment are listed in **Table 5.2**. These are breeding fulmar, breeding great skua, breeding red-throated diver and breeding puffin. The seabird assemblage has also been screened in for these species.

#### 8.59.3.1 Fulmar

#### Status

2344. The Foula SPA breeding fulmar population was cited as 46,800 pairs, or 93,600 breeding adults, in 1995 (Furness, 2015, Stroud *et al*., 2016). Furness (2015) gave a breeding population of 19,758 pairs, or 39,516 breeding adults, in 2007. The most recent count is 10,253 pairs (AOS), or 20,506 breeding adults, in 2021 (JNCC, 2023a); this is used as the reference population for the assessment.

2345. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.064 (1 – 0.936; Horswill and Robinson, 2015), the expected annual mortality from the SPA population would be 1,312 breeding adults.

### Functional linkage and seasonal apportionment of potential effects

2346. The mean maximum foraging range of fulmar is 542.3km (±657.9km) and the maximum foraging range is 2,736km (Woodward *et al*., 2019). The Project is located approximately 701km from Foula SPA, which means that the Project is beyond the mean maximum foraging range of fulmars breeding at this SPA, but within the mean maximum foraging range +1SD, and the maximum foraging range.

### Potential effects on the qualifying feature from the Project-alone

2347. Fulmar are considered to have low vulnerability to both collision and displacement impacts from offshore windfarms (Bradbury *et al*., 2014). This species was recorded infrequently within the windfarm site (in only five of 24 surveys, all during the breeding season (January to August; Furness, 2015) and in low numbers, with densities in four of the five months less than 0.1 birds/km$^2$. A peak density of 0.23 birds/km$^2$ was recorded in May 2022.

2348. On the basis of the low vulnerability of fulmars to potential impacts, and the low frequency and abundance of birds present at the windfarm site, it is concluded that any effects on breeding populations at Foula SPA are very unlikely, both during and outside of the breeding season.

2349. **It is therefore concluded that there would be no measurable effects on fulmar due to the project alone, and no adverse effect on the integrity of the Foula SPA is predicted.**

### Potential effects on the qualifying feature in-combination with other projects

2350. As the Project would have no measurable effect on fulmar populations from the Foula SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of Foula SPA, when assessed in-combination with other plans or projects.**

#### 8.59.3.2 Great skua

##### Status

2351. The Foula SPA breeding great skua population was cited as 2,170 pairs, or 4,340 breeding adults, in 1992 (Furness, 2015, Stroud *et al*., 2016). Furness (2015) gave a breeding population of 1,657 pairs, or 3,314 breeding adults, 2007. The most recent count is 1,846 pairs (AOT), or 3,692 breeding adults, in 2015 (JNCC, 2023a).

2352. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.118 (1 – 0.882; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 436 breeding adults.

##### Functional linkage and seasonal apportionment of potential effects

2353. The mean maximum foraging range of great skua is 443.3km (±487.9km) and the maximum foraging range is 1003km (Woodward *et al*., 2019). The Project is located approximately 701km from Foula SPA, which means that the Project is beyond the mean maximum foraging range of great skuas breeding at this SPA, but within the mean maximum foraging range +1SD, and the maximum foraging range.

##### Potential effects on the qualifying feature from the Project-alone

**Operation and maintenance phase collision risk**

2354. The great skua qualifying feature of the Foula SPA has been screened into the Appropriate Assessment due to the potential risk of collision. However, this species was not recorded within the windfarm site during site surveys, and therefore there would be no risk that collision mortality would affect great skua populations from Foula SPA. It is noted that a separate assessment of collision risk to migrant great skuas has also been undertaken, as set out in Chapter 12 of the ES. This also predicted negligible annual mortality for this species (0.03 birds), which would equate to no measurable increase in mortality apportioned to populations from Foula SPA. **It is concluded that there would be no adverse effect on the integrity of Foula SPA**.

2355. The confidence in the assessment is high. As both the surveys of the Project site and separate migrant collision risk assessment indicate that there would be negligible collision impacts on this species, it is considered extremely unlikely that there would be any effects on populations from the SPA.

##### Potential effects on the qualifying feature in-combination with other projects

2356. As the Project would have no measurable effect on great skua populations from the Foula SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of Foula SPA, when assessed in-combination with other plans or projects**.

#### 8.59.3.3 Red-throated diver

#### Status

2357. The Foula SPA breeding red-throated diver population was cited as 11 pairs, or 22 breeding adults, in 1994 (Furness, 2015, Stroud *et al*., 2016). Furness (2015) gave the breeding population of 12 pairs, or 24 breeding adults in 2013; this is used as the reference population for the assessment.

2358. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.16 (1 – 0.840; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be four breeding adults.

#### Functional linkage and seasonal apportionment of potential effects

2359. The mean maximum foraging range of red-throated diver is 9km (±0km), as is the maximum foraging range (Woodward *et al*., 2019). Foula SPA is located approximately 701km from the Project, which means that the Project is beyond the maximum foraging range for red-throated divers from the SPA. No impacts during the breeding season from the Project are therefore apportioned to red-throated divers breeding at this SPA.

2360. Outside the breeding season, breeding red-throated divers from the SPA are assumed to range widely and to mix with red-throated divers of all ages from breeding colonies in the UK and further afield. The relevant background population during the autumn and spring migration seasons is the UK Western waters plus Channel BDMPS, consisting of 4,373 individuals during autumn and spring passage periods (September to November and February to April) (Furness, 2015). The relevant background population during the winter season is the NW England and Wales BDMPS, consisting of 1,657 individuals (Furness, 2015).

2361. During the spring and autumn migration seasons, Furness (2015) estimated that 5% of breeding adults from Foula SPA (24 birds) are present within the UK Western waters plus Channel BDMPS, which is one bird. This represents 0.02% of the BDMPS population for that period (4,373). During the winter period it is estimated that 2% of breeding adults from Foula SPA (24 birds) are present within the NW England and Wales BDMPS, which is zero birds. This represents 0.0% of the BDMPS population for that period (1,657). These percentages (i.e. 0.02% and 0.0%) are the proportions of birds present at the windfarm site that are presumed to originate from the Foula SPA during the relevant seasons.

## Potential effects on the qualifying feature

### Operation and maintenance phase disturbance/displacement/barrier effects

*Project-alone*

2362. The year-round mean peak abundance of red-throated divers present within the windfarm site and hybrid 10km buffer was 74 (0-240) individuals (refer to **Appendix 12.1** of the ES). Of these, less than one bird (0.01 birds (0-0.03)) were likely to be breeding adults from the Foula SPA.

2363. Red-throated divers have a very high sensitivity to disturbance and displacement from operational OWFs. The majority of birds present before OWFs are constructed are displaced by the operation of OWFs. It is expected (based on expert opinion), that this is due to a combination of anthropogenic activities (mainly vessel movements), as well as the presence of OWF infrastructure. A large body of work investigating the effects of displacement of red-throated divers due to operational OWFs exists (Dorsch *et al*., 2020; Elston *et al*., 2016; Gill *et al*., 2018; Hi Def Aerial Surveying, 2017; Irwin *et al*., 2019; MacArthur Green and Royal HaskoningDHV, 2021; McGovern *et al*., 2016; Mendel *et al*., 2019; Percival, 2014; Percival and Ford, 2017; Petersen *et al*., 2014, 2006; Vilela *et al*., 2020; Welcker and Nehls, 2016).

2364. There was a high degree of concordance of the available literature with respect to effects of operation of OWFs on red-throated diver distribution and abundance within OWFs. There was also a high degree of concordance that displacement effects extended beyond OWF boundaries. However, there was considerable variation with respect to the distance at which this effect remained detectable. Studies within the UK have ranged from no significant displacement effects being reported (McGovern *et al*., 2016), displacement effects being restricted to 1km to 2km of an OWF (Percival, 2014; Percival and Ford, 2017), to clear displacement effects across many years. These effects have been reported extending to 7km from OWFs (MacArthur Green and Royal HaskoningDHV, 2021), 9km from OWFs (Elston *et al*., 2016; HiDef Aerial Surveying, 2017), and beyond, though not all evidence was available to be referenced by this assessment. Studies from other countries have also recorded variable displacement distances, ranging from 1.5km to 2km (Welcker and Nehls, 2016) to 10km and beyond (Dorsch *et al*., 2020; Vilela *et al*., 2020). Displacement effects were detectable up to 20km from OWFs in one case.

2365. There was also concordance in the studies reviewed that displacement effects on red-throated diver due to operational OWFs occurred on a gradient, with the strongest effects observed either within, or close to OWFs. As the distance from the OWF increased, the magnitude of the effect reduced, until a distance was reached at which the effect is no longer detectable.

2366. No study to date has managed to provide insight into whether changes in red-throated diver distribution at any spatial scale have the potential to result in population level effects, either at local, regional, national or international levels. Red-throated divers are capable of utilising a range of marine habitats and prey species (Dierschke *et al*., 2017; Guse *et al*., 2009; Kleinschmidt *et al*., 2016). Recent data from the Outer Thames Estuary SPA indicated that birds were much more commonly recorded in water depths of less than 20m (Irwin *et al*., 2019). During the non-breeding season, red-throated divers were mostly widely dispersed, at densities often less than four birds per km$^2$ (Dierschke *et al*., 2017), and were highly mobile (Dorsch *et al*., 2020; Duckworth *et al*., 2020). In some instances, home ranges of many thousands of square kilometres have been demonstrated (Nehls *et al*., 2018). This implies that following displacement, red-throated divers will be able to find alternative foraging sites, in some cases distant from the original area of displacement, which may already have been part of their existing non-breeding season range. It seems likely that in the vast majority of cases, mortality is not a consequence of displacement.

2367. Displacement rates of 1.000, along with a range of mortality rates of 1% to 10% of displaced birds is considered for this species at this SPA (UK SNCBs, 2017). However, it is considered that there is a high possibility that displacement and mortality rates are substantially lower than this.

2368. **Table 8.148** sets out the predicted impacts on red-throated divers from Foula SPA during the non-breeding season. A displacement rate of 1.000 is presented for this species, along with a range of mortality rates of 1% to 10% of displaced birds (UK SNCBs, 2017). An estimated annual mortality for the population is provided, along with the increase of existing mortality that would occur through such an impact. The displacement matrices used to calculate potential impacts are presented in **Appendix 12.1** of the ES.

*Table 8.148 Red-throated diver – predicted operation and maintenance phase displacement and mortality from Foula SPA*

| Mean peak abundance estimate type | Mean peak abundance estimate by season[1] | Number of Foula SPA breeding adults present by season)[2] | Annual mortality range[3] | Annual baseline mortality increase range[4] |
|---|---|---|---|---|
| **Upper 95% CI** | 35 (b)<br>23 (aut)<br>55 (win)<br>127 (spr)<br>240 (year round) | 0 (b)<br>0 (aut)<br>0 (win)<br>0 (spr)<br>0 (year round) | 0-0 | 0.01-0.08% |

| Mean peak abundance estimate type | Mean peak abundance estimate by season[1] | Number of Foula SPA breeding adults present by season)[2] | Annual mortality range[3] | Annual baseline mortality increase range[4] |
|---|---|---|---|---|
| **Mean** | 8 (b)<br>10 (aut)<br>12 (win)<br>43 (spr)<br>74 (year round) | 0 (b)<br>0 (aut)<br>0 (win)<br>0 (spr)<br>0 (year round) | 0-0 | 0.00-0.03% |
| **Lower 95% CI** | 0 (b)<br>0 (aut)<br>0 (win)<br>0 (spr)<br>0 (year round) | 0 (b)<br>0 (aut)<br>0 (win)<br>0 (spr)<br>0 (year round) | 0-0 | 0.00-0.00% |

[1] Breeding season = b, autumn migration season = aut, winter season = win, spring migration season = spr

[2] Assumes breeding adult apportioning of 0.0% (breeding season), 0.02% (spring and autumn migration) and 0.0% (winter) to Foula SPA.

[3] Assumes displacement rate of 100% and mortality rates of 1-10%.

[4] Background mortality rate of 16.0% (Horswill and Robinson, 2015)

2369. Based on the mean peak abundances, less than one red-throated diver from the Foula SPA would be at risk of displacement (**Table 8.148**). At displacement rates of 1.000, and mortality rates of 1% to 10% for displaced birds, <0.01 SPA breeding adults would be predicted to die each year due to displacement from the Project.

2370. Assuming a displacement rate of 1.00 and a mortality rate of 10% of displaced birds, annual mortality within the SPA breeding adult population would increase by 0.03%. Using a more realistic mortality rate for displaced birds of 1%, annual mortality in the population would be less than 0.01%.

2371. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that no detectable changes in mortality rates would occur under any combination of displacement and mortality rates when the mean peak abundance estimate assessments are considered. This would be the case even when upper 95% CIs are considered.

2372. **It is concluded that predicted red-throated diver mortality due to operational phase displacement for the project alone would not adversely affect the integrity of the Foula SPA.**

2373. The confidence in the assessment is high for several reasons. Firstly, the evidence used to inform the evidence-based displacement rates is of high

applicability and quality (based on the criteria discussed in **Chapter 12 Offshore Ornithology** of the ES). Whilst there was limited available evidence to inform mortality rates, 1% is considered to be sufficiently precautionary based on expert opinion. This species has been demonstrated to be highly mobile during the non-breeding season, and individuals frequently possess very large home ranges during this time (Dorsch *et al*., 2020; Nehls *et al*., 2018). It is therefore anticipated that displaced birds will find alternative habitat in the vast majority of cases. Finally, the conclusion of the assessment is the same irrespective of whether 1% or 10% mortality, or the mean or 95% upper CI mean peak abundances, are used to calculate potential mortality and increases in the baseline mortality rate of the background population.

*In-combination*

2374. As the Project would have no measurable effect on red-throated diver populations from the Foula SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of Foula SPA, when assessed in-combination with other plans or projects.**

#### 8.59.3.4 Puffin

#### Status

2375. The Foula SPA breeding puffin population was cited as 48,000 pairs, or 96,000 breeding adults, in 1987 (Furness, 2015, Stroud *et al*., 2016). Furness (2015) gave a breeding population of 22,500 pairs, or 45,000 breeding adults, in 2000. The most recent count is 6,351 individuals in 2016 (JNCC, 2023a); this is used as the reference population for the assessment.

2376. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.094 (1 – 0.906; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 597 breeding adults.

#### Functional linkage and seasonal apportionment of potential effects

2377. The mean maximum foraging range of puffin is 137.1km (±128.3km) and the maximum foraging range is 383km (Woodward *et al*., 2019). Foula SPA is located approximately 701km from the Project, which means that breeding puffins from this SPA are beyond the maximum foraging range for this species from the Project. No impacts during the breeding season from the Project are therefore apportioned to puffins breeding at this SPA.

2378. Outside of the breeding season, puffins, including those from the Foula SPA, are not constrained by requirements to visit nests to incubate eggs or provision chicks. At this time, they are assumed to range more widely and to mix with puffins of all age classes from breeding colonies in the UK and further afield. The background population during these seasons is the UK Western Waters BDMPS. This consists of 304,557 individuals during the non-breeding season (August to March).

2379. Furness (2015) estimated that 8% of the Foula SPA breeding adults (45,000) are present within the UK Western Waters BDMPS during the non-breeding season, which is 3,600 birds. This represents 1.2% of the BDMPS population for this period (304,557). It is therefore assumed that 1.2% of puffins present at the Project site are breeding adults from Foula SPA.

## Potential effects on the qualifying feature

### Operation and maintenance phase disturbance/displacement/barrier effects

*Project-alone*

2380. The mean peak abundance of puffins present within the windfarm site and 2km buffer during the non-breeding season was 19.7 (1.9-50.8) individuals (refer to **Appendix 12.1** of the ES). Of these, less than one bird (0.2 (0.0-0.6)) was likely to be a breeding adult from Foula SPA.

2381. **Table 8.149** sets out the predicted impacts on puffins from Foula SPA during the non-breeding season. This estimates that there would be no measurable increase in mortality from the SPA population, assuming a displacement rate of 30-70% and a mortality of 1-10% for displaced birds.

*Table 8.149 Puffin – predicted operation and maintenance phase displacement and mortality from Foula SPA*

| Mean peak abundance estimate type | Mean peak abundance estimate | Number of Foula SPA breeding adults present (non-breeding season)[1] | Annual mortality range[2] | Annual baseline mortality increase range[3] |
|---|---|---|---|---|
| Upper 95% CI | 50.8 | 0.6 | 0-0 | 0.00-0.00% |
| Mean | 19.7 | 0.2 | 0-0 | 0.00-0.00% |
| Lower 95% CI | 1.9 | 0.0 | 0-0 | 0.00-0.00% |

[1] Assumes 1.2% of birds present during the non-breeding season are Foula SPA breeding adults
[2] Assumes displacement rates of 30-70% and mortality rates of 1-10%
[3] Background mortality rate of 9.4% (Horswill and Robinson, 2015)

2382. **It is concluded that predicted puffin mortality due to operational phase displacement would not adversely affect the integrity of the Foula SPA.**

2383. The confidence in the assessment is high for several reasons. Firstly, the evidence used to inform the evidence-based displacement rates is of high

applicability and quality (based on the criteria discussed in **Chapter 12 Offshore Ornithology** of the ES). Whilst there is limited available evidence to inform mortality rates, 1% is considered to be sufficiently precautionary based on expert opinion. This species is not regarded as being highly specialised in its habitat requirements (Bradbury *et al*., 2014; Furness and Wade, 2012; Garthe and Hüppop, 2004), and it is therefore anticipated that displaced birds will find alternative habitat in the vast majority of cases. Finally, the conclusion of the assessment is the same irrespective of whether the mean or 95% upper CI mean peak abundances are used to calculate potential mortality and increases in the baseline mortality rate of the background population.

*In-combination*

2384. As the Project would have no measurable effect on puffin populations from the Foula SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that predicted puffin mortality due to operational phase displacement due to the Project in-combination with other plans or projects would not adversely affect the integrity of the Foula SPA.**

# 8.60 Noss SPA

2385. Noss SPA is located approximately 715km from the windfarm site.

## 8.60.1 Description of designation

2386. Noss is an offshore island lying 5km east of Lerwick, Shetland. The SPA supports breeding seabirds on cliffs, inland heathlands and grasslands. The seaward extension of the SPA extends approximately 2km into the marine environment and includes the seabed, water column and surface. Seabirds included within the designation feed both inside and outside the SPA in nearby waters, as well as more distantly in the wider North Sea.

## 8.60.2 Conservation objectives

2387. The overarching conservation objectives for the site are:

- To avoid deterioration of the habitats of the qualifying species or significant disturbance to the qualifying species, thus ensuring that the integrity of the site is maintained
- To ensure for the qualifying species that the following are maintained in the long term:
  - Population of the species as a viable component of the site
  - Distribution of the species within site
  - Distribution and extent of habitats supporting the species
  - Structure, function and supporting processes of habitats supporting the species
  - No significant disturbance of the species

## 8.60.3 Assessment

2388. The qualifying features of Noss SPA screened into the Appropriate Assessment are listed in **Table 5.2**. These are breeding fulmar, breeding great skua and breeding gannet. The seabird assemblage has also been screened in for these species.

### 8.60.3.1 Fulmar

#### Status

2389. The Noss SPA breeding fulmar population at classification (1996) was 6,350 pairs, or 12,700 breeding adults (SNH, 2009f). Furness (2015) gave a population of 5,248 pairs, or 10,496 breeding adults, in 2011. The most recent count is 4,347 pairs (AOS), or 8,694 breeding adults, in 2022 (JNCC, 2023a); this is used as the reference population for the assessment.

2390. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.064 (1 – 0.936; Horswill and Robinson, 2015), the expected annual mortality from the SPA population would be 556 breeding adults.

### Functional linkage and seasonal apportionment of potential effects

2391. The mean maximum foraging range of fulmar is 542.3km (±657.9km) and the maximum foraging range is 2,736km (Woodward *et al*., 2019). The Project is located approximately 715km from Noss SPA, which means that the Project is beyond the mean maximum foraging range of fulmars breeding at this SPA, but within the mean maximum foraging range +1SD, and the maximum foraging range.

### Potential effects on the qualifying feature from the Project-alone

2392. Fulmar are considered to have low vulnerability to both collision and displacement impacts from offshore windfarms (Bradbury *et al*., 2014). This species was recorded infrequently within the windfarm site (in only five of 24 surveys, all during the breeding season (January to August; Furness, 2015)) and in low numbers, with densities in four of the five months less than 0.1 birds/km$^2$. A peak density of 0.23 birds/km$^2$ was recorded in May 2022.

2393. On the basis of the low vulnerability of fulmars to potential impacts, and the low frequency and abundance of birds present at the windfarm site, it is concluded that any effects on breeding populations at Noss SPA are very unlikely, both during and outside of the breeding season.

2394. **It is therefore concluded that there would be no measurable effects on fulmar due to the project alone, and no adverse effect on the integrity of the Noss SPA is predicted.**

### Potential effects on the qualifying feature in-combination with other projects

2395. As the Project would have no measurable effect on fulmar populations from the Noss SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of Noss SPA, when assessed in-combination with other plans or projects.**

#### 8.60.3.2 Great skua

##### Status

2396. The Noss SPA breeding great skua population at classification (1996) was 420 pairs, or 840 breeding adults (SNH, 2009f). Furness (2015) gave a population of 465 pairs, or 930 breeding adults, in 2013. The most recent count is 103 pairs (AOT), or 206 breeding adults, in 2022 (JNCC, 2023a); this is used as the reference population for the assessment.

2397. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.118 (1 – 0.882; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 24 breeding adults.

##### Functional linkage and seasonal apportionment of potential effects

2398. The mean maximum foraging range of great skua is 443.3km (±487.9km) and the maximum foraging range is 1,003km (Woodward *et al*., 2019). The Project is located approximately 715km from Noss SPA, which means that the project is beyond the mean maximum foraging range of great skuas breeding at this SPA, but within the maximum foraging range +1SD, and the maximum foraging range.

##### Potential effects on the qualifying feature from the Project-alone

**Operation and maintenance phase collision risk**

2399. The great skua qualifying feature of the Noss SPA has been screened into the Appropriate Assessment due to the potential risk of collision. However, this species was not recorded within the windfarm site during site surveys, and therefore there would be no risk that collision mortality would affect great skua populations from Noss SPA. It is noted that a separate assessment of collision risk to migrant great skuas has also been undertaken, as set out in Chapter 12 of the ES. This also predicted negligible annual mortality for this species (0.03 birds), which would equate to no measurable increase in mortality apportioned to populations from Noss SPA. **It is concluded that there would be no adverse effect on the integrity of Noss SPA**.

2400. The confidence in the assessment is high. As both the surveys of the Project site and separate migrant collision risk assessment indicate that there would be negligible collision impacts on this species, it is considered extremely unlikely that there would be any effects on populations from the SPA.

##### Potential effects on the qualifying feature in-combination with other projects

2401. As the Project would have no measurable effect on great skua populations from the Noss SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of Noss SPA, when assessed in-combination with other plans or projects**.

### 8.60.3.3 Gannet

#### Status

2402. The Noss SPA breeding gannet population at classification (1996) was 6,860 pairs, or 13,720 breeding adults (SNH, 2009f). Furness (2015) gave a population of 9,767 pairs, or 19,534 breeding adults, in 2008. The most recent count is 11,472 pairs (AON), or 22,944 breeding adults, in 2022 (JNCC, 2023a); this is used as the reference population in the assessment.

2403. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.081 (1 – 0.919; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 1,858 breeding adults.

#### Functional linkage and seasonal apportionment of potential effects

2404. The mean maximum foraging range of gannet is 315.2km (±194.2km), and the maximum foraging range is 709km (Woodward *et al*., 2019). Noss SPA is approximately 715km from the Project, which means that breeding gannets from this SPA are beyond the maximum foraging range for this species from the Project. No impacts during the breeding season from the Project are therefore apportioned to gannets breeding at this SPA.

2405. Outside the breeding season breeding gannets, including those from the Noss SPA, are not constrained by requirements to visit nests to incubate eggs or provision chicks. At this time, they are assumed to range more widely and to mix with gannets of all age classes from breeding colonies in the UK and further afield. The background population during these seasons is the UK Western Waters BDMPS. This consists of 545,954 individuals during autumn migration (September to November), and 661,888 individuals during spring migration (December to March) (Furness, 2015).

2406. Furness (2015) estimated that 20% of the Noss SPA breeding adults (19,534) are present within the UK Western Waters BDMPS during the autumn migration period, which is 3,907 birds, and that 30% the SPA population (i.e. 5,860) birds) is present during spring migration. Estimates of the proportion of gannets present at the windfarm site which originate from the Noss SPA during the non-breeding season (and therefore the proportion of predicted mortalities from the SPA population) are based on these population estimates as a proportion of the UK Western Waters BDMPS during the relevant season. During autumn migration and spring migration, 0.7%, and 0.9% of impacts are considered to affect birds from the SPA respectively (Furness, 2015).

### Potential effects on the qualifying feature from the Project-alone

2407. The gannet qualifying feature of the Noss SPA has been screened into the assessment due to the potential risk of collision and operational phase displacement/barrier effects during the operation and maintenance phase of the Project.

**Operation and maintenance phase displacement/barrier effects**

2408. Displacement effects for gannet for the Project were assessed during the autumn and spring migration periods, based on an unapportioned peak mean population of 124 and eight birds respectively, calculated for the windfarm site and a 2km buffer, in line with recommendations within the SNCB guidance (SNCB 2017). As set out above, no gannets present at the windfarm site have been apportioned to Noss SPA during the breeding season. The displacement matrices used to calculate potential impacts are presented in **Appendix 12.1** of the ES, and summarised in **Table 8.150**. The inclusion of all birds within the 2km buffer, to determine the total number of birds subject to displacement, is precautionary, as in reality the avoidance rate is likely to fall with distance from the windfarm site.

2409. A displacement rate of 60-80% and mortality rate of 1% has been presented. A maximum 1% mortality value has been selected firstly because gannet are known to possess high habitat flexibility (Furness and Wade, 2012). This suggests that displaced birds will readily find alternative habitats including foraging areas. Secondly, no evidence of displacement-induced mortality has been identified, which means there is limited justification for setting predicted mortality rates at a higher level. Given the extensive foraging range of this species (Woodward *et al.*, 2019), there may be no mortality costs to displacement from the relatively very small footprints of OWFs.

*Table 8.150 Gannet – predicted operation and maintenance phase displacement and mortality from Noss SPA*

| Mean peak abundance estimate type | Mean peak abundance estimate | Number of SPA breeding adults present by season[1] | Annual mortality range[2] | Annual baseline mortality increase range[3] |
|---|---|---|---|---|
| Upper 95% CI | 809 (breeding)<br>189 (autumn)<br>16 (spring)<br>1,014 (year round) | 0 (breeding)<br>1 (autumn)<br>0 (spring)<br>1 (year round) | 0-0 | 0.00-0.00% |
| Mean | 541 (breeding)<br>124 (autumn)<br>8 (spring)<br>673 (year round) | 0 (breeding)<br>1 (autumn)<br>0 (spring)<br>1 (year round) | 0-0 | 0.00-0.00% |
| Lower 95% CI | 160 (breeding)<br>0 (autumn)<br>0 (spring)<br>160 (year round) | 0 (breeding)<br>0 (autumn)<br>0 (spring)<br>0 (year round) | 0-0 | 0.00-0.00% |

[1]0.7% and 0.9% of birds are assumed to be breeding adults from the SPA population during the autumn and spring migration periods respectively.
[2] Assumes displacement rates of 60-80% and mortality rate of 1%
[3] Background population is Noss SPA breeding adults (22,944 individuals), adult age class annual mortality rate of 8.1% (Horswill and Robinson, 2015)

2410. Using the maximum potential mortality value, there would be no measurable increase in gannet mortality. Accordingly, no significant effects on gannet are predicted during the operation and maintenance phase, and **it is concluded that there would be no potential for the Project to have an adverse effect on the integrity of Noss SPA**.

2411. The confidence in the assessment is high for several reasons. Firstly, the evidence used to set the displacement rates presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES is of high applicability and quality. Whilst there is limited available evidence to inform mortality rates, 1% is considered to be sufficiently precautionary based on expert opinion. Finally, the conclusion of the assessment is the same irrespective of whether the mean or 95% upper CI mean peak abundances are used to calculate potential mortality and increases in the baseline mortality rate of the background population.

**Operation and maintenance phase collision risk**

2412. Information to support the Appropriate Assessment for collision risk on breeding adult gannets belonging to the Noss SPA population is presented in **Table 8.151**. Collision estimates, calculated using the sCRM, are presented by biological season. A summary of the annual outputs and the corresponding increase in the annual baseline mortality rate is also presented. Parameters used in the sCRM were agreed with Natural England during the ETG process and are described in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES. In accordance with Natural England advice, a 70% macro-avoidance correction was applied to gannet abundance data used in the sCRM.

2413. Based on the mean collision rates, no breeding adult gannets from Noss SPA are considered at risk of collision as a result of the Project. Therefore, there would be no measurable increase the existing mortality of the SPA breeding population.

*Table 8.151 Predicted seasonal and annual collision mortality (Stochastic model Option 2, avoidance rate 0.993 (±0.0003), plus 70% macro-avoidance) for breeding adult gannets at the windfarm site, apportioned to Noss SPA, with corresponding increases to baseline mortality of the population*

| Period | Breeding Season Apr-Aug | Autumn Migration Sep-Nov | Non-breeding/winter - | Spring Migration Dec-Mar | Annual Jan-Dec |
|---|---|---|---|---|---|
| Total collisions[1] (mean and 95% CIs) | 0.83 (0.00-3.35) | 0.14 (0.00-0.74) | - | 0.00 | 0.97 (0.00-4.10) |
| % apportioned to the SPA | 0.0% | 0.7% | - | 0.9% | - |
| Total SPA collisions (mean and 95% CIs) | 0.00 (0.00-0.00) | 0.00 (0.00-0.01) | - | 0.00 (0.00-0.00) | 0.00 (0.00-0.00) |
| Mortality increase[2] (mean and 95% CIs) | 0.00% (0.00-0.00%) | 0.00% (0.00-0.00%) | - | 0.00% (0.00-0.00%) | 0.00% (0.00-0.00%) |

[1] Breeding season collision values reduced to 73.8% of modelled value to reflect proportion of adult birds recorded during site surveys

[2] Assuming predicted annual SPA mortality of 1,859 birds (22,944 x 0.081)

2414. Accordingly, no significant effects on gannet are predicted during the operation and maintenance phase, and **it is concluded that there would be no potential for the Project to have an adverse effect on the integrity of Noss SPA**. Comments received from RSPB during the ETG process, indicating that they do not accept the 70% macro-avoidance rate for collision risk recommended by Natural England, are noted. However, even in the absence of this correction factor, the net increase in mortality would be unchanged (i.e. zero).

2415. The confidence in the assessment is high. The evidence used to define the CRM input parameters presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES submitted alongside this RIAA is of high applicability and quality. Whilst there is uncertainty around some of the input parameters (e.g. avoidance rate), the rates selected are considered to be sufficiently precautionary based on expert opinion to provide confidence that collision rates are not underestimated.

**Combined displacement/barrier effects and collision risk**

2416. As no measurable increase in mortality is predicted for both displacement and collision risk, the mean combined displacement and collision rates for breeding adult gannet from the Noss SPA would be zero. Therefore, there would be no net increase in existing mortality rates.

2417. **It is concluded that based on predicted gannet mortality due to the combined effects of operational phase displacement and collision there is no potential for the Project to have an adverse effect on the integrity of the Noss SPA**.

2418. The confidence in the assessment is high, for the reasons provided in the individual displacement and collision assessments.

### Potential effects in-combination with other projects

2419. As no measurable effects of displacement/barrier and collision on gannet are predicted as a result of the Project-alone, there would be no contribution to other plans or projects in-combination. **It is therefore concluded that there would be no potential for the Project-alone or in-combination to have an adverse effect on the integrity of Noss SPA**.

# 8.61 Ronas Hill – North Roe and Tingon SPA and Ramsar

2420. Ronas Hill – North Roe and Tingon SPA and Ramsar site is located approximately 753km from the windfarm site.

## 8.61.1 Description of designation

2421. Ronas Hill – North Roe and Tingon SPA and Ramsar site comprises two adjacent headlands separated by Ronas Voe in the North Mainland of Shetland. Most of the site is composed of active blanket bog with numerous lochans and pools that support a typical peatland avifauna. Qualifying bird species of the SPA comprise great skua and red-throated diver.

## 8.61.2 Conservation objectives

2422. The overarching conservation objectives for the SPA are:

- To avoid deterioration of the habitats of the qualifying species or significant disturbance to the qualifying species, thus ensuring that the integrity of the site is maintained
- To ensure for the qualifying species that the following are maintained in the long term:
  - Population of the species as a viable component of the site
  - Distribution of the species within site
  - Distribution and extent of habitats supporting the species
  - Structure, function and supporting processes of habitats supporting the species
  - No significant disturbance of the species

## 8.61.3 Assessment

2423. Two qualifying features of Ronas Hill – North Roe and Tingon SPA and Ramsar site have been screened into the Appropriate Assessment (**Table 5.2**): red-throated diver and great skua.

### 8.61.3.1 Red-throated diver

#### Status

2424. The Ronas Hill – North Roe and Tingon SPA breeding red-throated diver population stood at 50 pairs, or 100 breeding adults, in 1997 and 2006 (Furness, 2015); this is the most recent available count and is used as the reference population for the assessment.

2425. Based on the most recent available SPA population of breeding adults, and an annual adult baseline mortality rate of 0.160 (1 – 0.840; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 16 breeding adults.

## Functional linkage and seasonal apportionment of potential effects

2426. The mean maximum foraging range of red-throated diver is 9km (±0km), as is the maximum foraging range (Woodward *et al*., 2019). The Project is located approximately 753km from Ronas Hill – North Roe and Tingon SPA, which means that the Project is beyond the maximum foraging range for red-throated divers from the SPA. No impacts during the breeding season from the Project are therefore apportioned to red-throated divers breeding at this SPA.

2427. Outside the breeding season, breeding red-throated divers from the SPA are assumed to range widely and to mix with red-throated divers of all ages from breeding colonies in the UK and further afield. The relevant background population during the autumn and spring migration seasons is the UK Western waters plus Channel BDMPS, consisting of 4,373 individuals during autumn and spring passage periods (September to November and February to April) (Furness, 2015). The relevant background population during the winter season is the NW England and Wales BDMPS, consisting of 1,657 individuals (Furness, 2015).

2428. During the spring and autumn migration seasons, Furness (2015) estimated that 5% of breeding adults from Ronas Hill – North Roe and Tingon SPA (100 birds) are present within the UK Western waters plus Channel BDMPS, which is five birds. This represents 0.1% of the BDMPS population for that period (4,373). During the winter period it is estimated that 2% of breeding adults from Ronas Hill – North Roe and Tingon SPA (100 birds) are present within the NW England and Wales BDMPS, which is two birds. This represents 0.1% of the BDMPS population for that period (1,657). These percentages (i.e. 0.1% and 0.1%) are the proportions of birds present at the windfarm site that are presumed to originate from the Ronas Hill – North Roe and Tingon SPA during the relevant seasons.

## Potential effects on the qualifying feature

### Operation and maintenance phase disturbance/displacement/barrier effects

*Project-alone*

2429. The year-round mean peak abundance of red-throated divers present within the windfarm site and hybrid 10km buffer is 74 (0-240) individuals (refer to **Appendix 12.1** of the ES). Of these, less than one bird (0.07 birds (0-0.21)) is likely to be a breeding adult from the Ronas Hill – North Roe and Tingon SPA.

2430. Red-throated divers have a very high sensitivity to disturbance and displacement from operational OWFs. The majority of birds present before OWFs are constructed are displaced by the operation of OWFs. It is expected (based on expert opinion), that this is due to a combination of anthropogenic activities (mainly vessel movements), as well as the presence of OWF infrastructure. A large body of work investigating the effects of displacement of red-throated divers due to operational OWFs exists (Dorsch *et al*., 2020; Elston *et al*., 2016; Gill *et al*., 2018; Hi Def Aerial Surveying, 2017; Irwin *et al*., 2019; MacArthur Green and Royal HaskoningDHV, 2021; McGovern *et al*., 2016; Mendel *et al*., 2019; Percival, 2014; Percival and Ford, 2017; Petersen *et al*., 2014, 2006; Vilela *et al*., 2020; Welcker and Nehls, 2016).

2431. There was a high degree of concordance of the available literature with respect to effects of operation of OWFs on red-throated diver distribution and abundance within OWFs. There was also a high degree of concordance that displacement effects extended beyond OWF boundaries. However, there was considerable variation with respect to the distance at which this effect remained detectable. Studies within the UK have ranged from no significant displacement effects being reported (McGovern *et al*., 2016), displacement effects being restricted to 1km to 2km of an OWF (Percival, 2014; Percival and Ford, 2017), to clear displacement effects across many years. These effects have been reported extending to 7km from OWFs (MacArthur Green and Royal HaskoningDHV, 2021), 9km from OWFs (Elston *et al*., 2016; Hi Def Aerial Surveying, 2017), and beyond, though not all evidence was available to be referenced by this assessment. Studies from other countries have also recorded variable displacement distances, ranging from 1.5km to 2km (Welcker and Nehls, 2016) to 10km and beyond (Dorsch *et al*., 2020; Vilela *et al*., 2020). Displacement effects were detectable up to 20km from OWFs in one case.

2432. There was also concordance in the studies reviewed that displacement effects on red-throated diver due to operational OWFs occurred on a gradient, with the strongest effects observed either within, or close to OWFs. As the distance from the OWF increased, the magnitude of the effect reduced, until a distance was reached at which the effect was no longer detectable.

2433. No study to date has managed to provide insight into whether changes in red-throated diver distribution at any spatial scale had the potential to result in population level effects, either at local, regional, national or international levels. Red-throated divers have been noted to be capable of utilising a range of marine habitats and prey species (Dierschke *et al*., 2017; Guse *et al*., 2009; Kleinschmidt *et al*., 2016). Recent data from the Outer Thames Estuary SPA indicated that birds were much more commonly recorded in water depths of less than 20m (Irwin *et al*., 2019). During the non-breeding season, red-throated divers were mostly widely dispersed, at densities often less than four birds per km$^2$ (Dierschke *et al*., 2017), and were highly mobile (Dorsch *et al*., 2020; Duckworth *et al*., 2020). In some instances, home ranges of many thousands of square kilometres have been demonstrated (Nehls *et al*., 2018). This implies that following displacement, red-throated divers will be able to find alternative foraging sites, in some cases distant from the original area of displacement, which may already have been part of their existing non-breeding season range. It seems likely that in the vast majority of cases, mortality is not a consequence of displacement.

2434. Displacement rates of 1.000, along with a range of mortality rates of 1% to 10% of displaced birds is considered for this species at this SPA (UK SNCBs, 2017). However, it is considered that there is a high possibility that displacement and mortality rates are substantially lower than this.

2435. **Table 8.152** sets out the predicted impacts on red-throated divers from Ronas Hill – North Roe and Tingon SPA during the non-breeding season. A displacement rate of 1.000 is presented for this species, along with a range of mortality rates of 1% to 10% of displaced birds (UK SNCBs, 2017). An estimated annual mortality for the population is provided, along with the increase of existing mortality that would occur through such an impact. The displacement matrices used to calculate potential impacts are presented in **Appendix 12.1** of the ES.

*Table 8.152 Red-throated diver – predicted operation and maintenance phase displacement and mortality from Ronas Hill – North Roe and Tingon SPA*

| Mean peak abundance estimate type | Mean peak abundance estimate by season[1] | Number of Ronas Hill – North Roe and Tingon SPA breeding adults present by season)[2] | Annual mortality range[3] | Annual baseline mortality increase range[4] |
|---|---|---|---|---|
| Upper 95% CI | 35 (b)<br>23 (aut)<br>55 (win)<br>127 (spr)<br>240 (year round) | 0 (b)<br>0 (aut)<br>0 (win)<br>0 (spr)<br>0 (year round) | 0-0 | 0.01-0.13% |
| Mean | 8 (b)<br>10 (aut)<br>12 (win)<br>43 (spr)<br>74 (year round) | 0 (b)<br>0 (aut)<br>0 (win)<br>0 (spr)<br>0 (year round) | 0-0 | 0.00-0.04% |

| Mean peak abundance estimate type | Mean peak abundance estimate by season[1] | Number of Ronas Hill – North Roe and Tingon SPA breeding adults present by season)[2] | Annual mortality range[3] | Annual baseline mortality increase range[4] |
|---|---|---|---|---|
| Lower 95% CI | 0 (b)<br>0 (aut)<br>0 (win)<br>0 (spr)<br>0 (year round) | 0 (b)<br>0 (aut)<br>0 (win)<br>0 (spr)<br>0 (year round) | 0-0 | 0.00-0.00% |

[1] Breeding season = b, autumn migration season = aut, winter season = win, spring migration season = spr

[2] Assumes breeding adult apportioning of 0.0% (breeding season), 0.02% (spring and autumn migration) and 0.0% (winter) to Ronas Hill – North Roe and Tingon SPA.

[3] Assumes displacement rate of 100% and mortality rates of 1-10%.

[4] Background mortality rate of 16.0% (Horswill and Robinson, 2015)

2436. Based on the mean peak abundances, less than one red-throated diver from the Ronas Hill – North Roe and Tingon SPA would be at risk of displacement (**Table 8.152**). At displacement rates of 1.000, and mortality rates of 1% to 10% for displaced birds, <0.01 SPA breeding adults would be predicted to die each year due to displacement from the Project.

2437. Assuming a displacement rate of 1.00 and a mortality rate of 10% of displaced birds, annual mortality within the SPA breeding adult population would increase by 0.04%. Using a more realistic mortality rate for displaced birds of 1%, annual mortality in the population would be less than 0.01%.

2438. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that no detectable changes in mortality rates would occur under any combination of displacement and mortality rates when the mean peak abundance estimate assessments are considered. This would be the case even when upper 95% CIs are considered.

2439. **It is concluded that predicted red-throated diver mortality due to operational phase displacement for the project alone would not adversely affect the integrity of the Ronas Hill – North Roe and Tingon SPA.**

2440. The confidence in the assessment is high for several reasons. Firstly, the evidence used to inform the evidence-based displacement rates is of high applicability and quality (based on the criteria discussed in **Chapter 12 Offshore Ornithology** of the ES). Whilst there was limited available evidence to inform mortality rates, 1% is considered to be sufficiently precautionary based on expert opinion. This species has been demonstrated to be highly mobile during the non-breeding season, and individuals frequently possessed very large home ranges during this time (Dorsch *et al*., 2020; Nehls *et al*., 2018). It is therefore anticipated that displaced birds will find alternative habitat in the vast majority of cases. Finally, the conclusion of the assessment is the same irrespective of whether 1% or 10% mortality, or the mean or 95% upper CI mean peak abundances, are used to calculate potential mortality and increases in the baseline mortality rate of the background population.

*In-combination*

2441. As the Project would have no measurable effect on red-throated diver populations from the Ronas Hill – North Roe and Tingon SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of Ronas Hill – North Roe and Tingon SPA, when assessed in-combination with other plans or projects.**

#### 8.61.3.2 Great skua

### Status

2442. The Ronas Hill – North Roe and Tingon SPA and Ramsar site breeding great skua population was cited as 130 pairs, or 260 breeding adults, in 1997 (Furness, 2015). Stroud *et al.,* (2014) gave a breeding population of 189 pairs, or 378 breeding adults, in 2002. The most recent count is 43 pairs, or 86 breeding adults, in 2019 (JNCC, 2023a) however it is unclear if this count covered the full extent of the SPA. The 2002 estimate is therefore used as the reference population for the assessment.

2443. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.118 (1 – 0.882; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 45 breeding adults.

### Functional linkage and seasonal apportionment of potential effects

2444. The mean maximum foraging range of great skua is 443.3km (±487.9km) and the maximum foraging range is 1,003km (Woodward *et al*., 2019). The Project is located approximately 753km from Ronas Hill – North Roe and Tingon SPA, which means that the Project is beyond the mean maximum foraging range of great skuas breeding at this SPA, but within the mean maximum foraging range +1SD, and the maximum foraging range.

## Potential effects on the qualifying feature from the Project-alone

### Operation and maintenance phase collision risk

2445. The great skua qualifying feature of the Ronas Hill – North Roe and Tingon SPA has been screened into the Appropriate Assessment due to the potential risk of collision. However, this species was not recorded within the windfarm site during site surveys, and therefore there would be no risk that collision mortality would affect great skua populations from Ronas Hill – North Roe and Tingon SPA. It is noted that a separate assessment of collision risk to migrant great skuas has also been undertaken, as set out in Chapter 12 of the ES. This also predicted negligible annual mortality for this species (0.03 birds), which would equate to no measurable increase in mortality apportioned to populations from Ronas Hill – North Roe and Tingon SPA. **It is concluded that there would be no adverse effect on the integrity of Ronas Hill – North Roe and Tingon SPA**.

2446. The confidence in the assessment is high. As both the surveys of the Project site and separate migrant collision risk assessment indicate that there would be negligible collision impacts on this species, it is considered extremely unlikely that there would be any effects on populations from the SPA.

## Potential effects on the qualifying feature in-combination with other projects

2447. As the Project would have no measurable effect on great skua populations from the Ronas Hill – North Roe and Tingon SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of Ronas Hill – North Roe and Tingon SPA, when assessed in-combination with other plans or projects**.

# 8.62 Fetlar SPA

2448. Fetlar SPA is located approximately 763km from the windfarm site.

## 8.62.1 Description of designation

2449. Fetlar is an island in the Shetland group, lying to the east and south respectively of the larger islands of Yell and Unst. The species-rich heath, bog and mire communities on the island support an important and characteristic breeding bird community, with the cliffs, rocky shores, and adjacent coastal waters important for breeding seabirds. The seaward elements extend approximately 2km into the marine environment to include the seabed, water column and surface.

## 8.62.2 Conservation objectives

2450. The overarching conservation objectives for the SPA are:

- To avoid deterioration of the habitats of the qualifying species or significant disturbance to the qualifying species, thus ensuring that the integrity of the site is maintained
- To ensure for the qualifying species that the following are maintained in the long term:
    - Population of the species as a viable component of the site
    - Distribution of the species within site
    - Distribution and extent of habitats supporting the species
    - Structure, function and supporting processes of habitats supporting the species
    - No significant disturbance of the species

## 8.62.3 Assessment

2451. Two qualifying features of Fetlar SPA have been screened into the Appropriate Assessment (**Table 5.2**): breeding fulmar and breeding great skua. These species have also been screened in as named components of the seabird assemblage.

### 8.62.3.1 Fulmar

#### Status

2452. The Fetlar SPA breeding fulmar population was cited as 9,800 pairs, or 19,600 breeding adults, in 1994 (Furness, 2015). Stroud *et al.,* (2014) gave a population of 8,912 pairs, or 17,824 breeding adults, for the period 1999 – 2002. The most recent count was 9,165 pairs (AOS), or 18,330 breeding adults, in 2016 (JNCC, 2023a); this is used as the reference population for the assessment.

2453. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.064 (1 – 0.936, Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 1,173 breeding adults.

### Functional linkage and seasonal apportionment of potential effects

2454. The mean maximum foraging range of fulmar is 542.3km (±657.9km) and the maximum foraging range is 2,736km (Woodward *et al*., 2019). The Project is located approximately 763km from Fetlar SPA, which means that the Project is beyond the mean maximum foraging range of fulmars breeding at this SPA, but within the mean maximum foraging range +1SD, and the maximum foraging range.

### Potential effects on the qualifying feature from the Project-alone

2455. Fulmar are considered to have low vulnerability to both collision and displacement impacts from offshore windfarms (Bradbury *et al*., 2014). This species was recorded infrequently within the windfarm site (in only five of 24 surveys, all during the breeding season (January to August; Furness, 2015)) and in low numbers, with densities in four of the five months less than 0.1 birds/km$^2$. A peak density of 0.23 birds/km$^2$ was recorded in May 2022.

2456. On the basis of the low vulnerability of fulmars to potential impacts, and the low frequency and abundance of birds present at the windfarm site, it is concluded that any effects on breeding populations at Fetlar SPA are very unlikely, both during and outside of the breeding season.

2457. **It is therefore concluded that there would be no measurable effects on fulmar due to the project alone, and no adverse effect on the integrity of the Fetlar SPA is predicted.**

### Potential effects on the qualifying feature in-combination with other projects

2458. As the Project would have no measurable effect on fulmar populations from the Fetlar SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of Fetlar SPA, when assessed in-combination with other plans or projects.**

#### 8.62.3.2 Great skua

##### Status

2459. The Fetlar SPA breeding great skua population was cited as 512 pairs, or 1,024 breeding adults, in 1994 (Furness, 2015). Stroud *et al.,* (2014) gave a breeding population of 585 pairs, or 1,170 breeding adults, in 2002. The most recent count is 852 pairs (AOT), or 1,704 breeding adults, in 2017 (JNCC, 2023a); this is used as the reference population for the assessment.

2460. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.118 (1 – 0.882; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 201 breeding adults.

##### Functional linkage and seasonal apportionment of potential effects

2461. The mean maximum foraging range of great skua is 443.3km (±487.9km) and the maximum foraging range is 1,003km (Woodward *et al*., 2019). The Project is located approximately 763km from Fetlar SPA, which means that the Project is beyond the mean maximum foraging range of great skuas breeding at this SPA, but within the mean maximum foraging range +1SD, and the maximum foraging range.

##### Potential effects on the qualifying feature from the Project-alone

**Operation and maintenance phase collision risk**

2462. The great skua qualifying feature of the Fetlar SPA has been screened into the Appropriate Assessment due to the potential risk of collision. However, this species was not recorded within the windfarm site during site surveys, and therefore there would be no risk that collision mortality would affect great skua populations from Fetlar SPA. It is noted that a separate assessment of collision risk to migrant great skuas has also been undertaken, as set out in Chapter 12 of the ES. This also predicted negligible annual mortality for this species (0.03 birds), which would equate to no measurable increase in mortality apportioned to populations from Fetlar SPA. **It is concluded that there would be no adverse effect on the integrity of Fetlar SPA**.

2463. The confidence in the assessment is high. As both the surveys of the Project site and separate migrant collision risk assessment indicate that there would be negligible collision impacts on this species, it is considered extremely unlikely that there would be any effects on populations from the SPA.

##### Potential effects on the qualifying feature in-combination with other projects

2464. As the Project would have no measurable effect on great skua populations from the Fetlar SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of Fetlar SPA, when assessed in-combination with other plans or projects**.

## 8.63 Hermaness, Saxa Vord and Valla Field SPA

2465. Hermaness, Saxa Vord and Valla Field SPA is located approximately 782km from the windfarm site.

### 8.63.1 Description of designation

2466. Hermaness, Saxa Vord and Valla Field SPA lies in the north-west corner of the island of Unst, Shetland. It consists of 100m to 200m high sea cliffs and adjoining areas of grassland, heath and blanket bog. The seaward elements extend approximately 2km into the marine environment to include the seabed, water column and surface. Qualifying seabird species of the SPA comprise fulmar, great skua, gannet, kittiwake, red-throated diver, shag, guillemot and puffin.

### 8.63.2 Conservation objectives

2467. The overarching conservation objectives for the SPA are:

- To avoid deterioration of the habitats of the qualifying species or significant disturbance to the qualifying species, thus ensuring that the integrity of the site is maintained
- To ensure for the qualifying species that the following are maintained in the long term:
  - Population of the species as a viable component of the site
  - Distribution of the species within site
  - Distribution and extent of habitats supporting the species
  - Structure, function and supporting processes of habitats supporting the species
  - No significant disturbance of the species

### 8.63.3 Assessment

2468. The qualifying features of Hermaness, Saxa Vord and Valla Field SPA screened into the Appropriate Assessment are listed in **Table 5.2**. These are breeding fulmar, breeding great skua, breeding gannet, breeding red-throated diver and breeding puffin. The seabird assemblage has also been screened in for these species.

#### 8.63.3.1 Fulmar

##### Status

2469. The Hermaness, Saxa Vord and Valla Field SPA breeding fulmar population was cited at 19,539 pairs, or 39,078 breeding adults, in 1999 (SNH 2009g). The most recent count is 13,208 pairs (AOS), or 26,416 breeding adults, in 2016 (JNCC, 2023a); this is used as the reference population for the assessment.

2470. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.064 (1 - 0.936, Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 1,691 breeding adults.

##### Functional linkage and seasonal apportionment of potential effects

2471. The mean maximum foraging range of fulmar is 542.3km (±657.9km) and the maximum foraging range is 2,736km (Woodward *et al*., 2019). The Project is located approximately 782km from Hermaness, Saxa Vord and Valla Field SPA, which means that the Project is beyond the mean maximum foraging range of fulmars breeding at this SPA, but within the mean maximum foraging range +1SD, and the maximum foraging range.

##### Potential effects on the qualifying feature from the Project-alone

2472. Fulmar are considered to have low vulnerability to both collision and displacement impacts from offshore windfarms (Bradbury *et al*., 2014). This species was recorded infrequently within the windfarm site (in only five of 24 surveys, all during the breeding season (January to August; Furness, 2015)) and in low numbers, with densities in four of the five months less than 0.1 birds/km$^2$. A peak density of 0.23 birds/km$^2$ was recorded in May 2022.

2473. On the basis of the low vulnerability of fulmars to potential impacts, and the low frequency and abundance of birds present at the windfarm site, it is concluded that any effects on breeding populations at Hermaness, Saxa Vord and Valla Field SPA are very unlikely, both during and outside of the breeding season.

2474. **It is therefore concluded that there would be no measurable effects on fulmar due to the project alone, and no adverse effect on the integrity of the Hermaness, Saxa Vord and Valla Field SPA is predicted.**

##### Potential effects on the qualifying feature in-combination with other projects

2475. As the Project would have no measurable effect on fulmar populations from the Hermaness, Saxa Vord and Valla Field SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is**

**concluded that there would be no adverse effect on the integrity of Hermaness, Saxa Vord and Valla Field SPA, when assessed in-combination with other plans or projects.**

#### 8.63.3.2 Great skua

##### Status

2476. The Hermaness, Saxa Vord and Valla Field SPA breeding great skua population was cited as 788 pairs, or 1,576 breeding adults, in 1997 (SNH 2009g). Furness (2015) gave a breeding population of 979 pairs, or 1,958 breeding adults, in 2013. The most recent complete count is 955 pairs (AOT), or 1,910 breeding adults, in 2018 (JNCC, 2023a); this is used as the reference population for the assessment.

2477. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.118 (1 – 0.882; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 225 breeding adults.

##### Functional linkage and seasonal apportionment of potential effects

2478. The mean maximum foraging range of great skua is 443.3km (±487.9km) and the maximum foraging range is 1,003km (Woodward *et al*., 2019). The Project is located approximately 782km from Hermaness, Saxa Vord and Valla Field SPA, which means that the Project is beyond the mean maximum foraging range of great skuas breeding at this SPA, but within the mean maximum foraging range +1SD, and the maximum foraging range.

##### Potential effects on the qualifying feature from the Project-alone

**Operation and maintenance phase collision risk**

2479. The great skua qualifying feature of the Hermaness, Saxa Vord and Valla Field SPA has been screened into the Appropriate Assessment due to the potential risk of collision. However, this species was not recorded within the windfarm site during site surveys, and therefore there would be no risk that collision mortality would affect great skua populations from Hermaness, Saxa Vord and Valla Field SPA. It is noted that a separate assessment of collision risk to migrant great skuas has also been undertaken, as set out in Chapter 12 of the ES. This also predicted negligible annual mortality for this species (0.03 birds), which would equate to no measurable increase in mortality apportioned to populations from Hermaness, Saxa Vord and Valla Field SPA. **It is concluded that there would be no adverse effect on the integrity of Hermaness, Saxa Vord and Valla Field SPA**.

2480. The confidence in the assessment is high. As both the surveys of the Project site and separate migrant collision risk assessment indicate that there would be negligible collision impacts on this species, it is considered extremely unlikely that there would be any effects on populations from the SPA.

### Potential effects on the qualifying feature in-combination with other projects

2481. As the Project would have no measurable effect on great skua populations from the Hermaness, Saxa Vord and Valla Field SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of Hermaness, Saxa Vord and Valla Field SPA, when assessed in-combination with other plans or projects**.

### 8.63.3.3 Gannet

### Status

2482. The Hermaness, Saxa Vord and Valla Field SPA breeding gannet population at classification (2009) was cited as 16,400 pairs, or 32,800 breeding adults (SNH 2009g). Furness (2015) gave a breeding population of 24,353 pairs, or 48,706 breeding adults, in 2008. The most recent count is 25,580 pairs (AOS), or 51,160 breeding adults, in 2014 (JNCC, 2023a); this is used as the reference population for the assessment.

2483. Based on the most recent SPA population of breeding adults, and an annual breeding adult baseline mortality rate of 0.081 (1 – 0.919; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 4,144 breeding adults.

### Functional linkage and seasonal apportionment of potential effects

2484. The mean maximum foraging range of gannet is 315.2km (±194.2km), and the maximum foraging range is 709km (Woodward *et al*., 2019). The Project is located approximately 782km from Hermaness, Saxa Vord and Valla Field SPA, which means that the Project is beyond the maximum foraging range for gannets from the SPA. No impacts during the breeding season from the Project are therefore apportioned to gannets breeding at this SPA.

2485. Outside the breeding season breeding gannets, including those from the Hermaness, Saxa Vord and Valla Field SPA, are not constrained by requirements to visit nests to incubate eggs or provision chicks. At this time, they are assumed to range more widely and to mix with gannets of all age classes from breeding colonies in the UK and further afield. The background population during these seasons is the UK Western Waters BDMPS. This consists of 545,954 individuals during autumn migration (September to

November), and 661,888 individuals during spring migration (December to March) (Furness, 2015).

2486. Furness (2015) estimated that 20% of the Hermaness, Saxa Vord and Valla Field SPA breeding adults (48,706) are present within the UK Western Waters BDMPS during the autumn migration period, which is 9,741 birds, and that 30% the SPA population (i.e. 14,612 birds) is present during spring migration. Estimates of the proportion of gannets present at the windfarm site which originate from the Hermaness, Saxa Vord and Valla Field SPA during the non-breeding season (and therefore the proportion of predicted mortalities from the SPA population) are based on these population estimates as a proportion of the UK Western Waters BDMPS during the relevant season. During autumn migration and spring migration, 1.8%, and 2.2% of impacts are considered to affect birds from the SPA respectively (Furness, 2015).

## Potential effects on the qualifying feature from the Project-alone

2487. The gannet qualifying feature of the Hermaness, Saxa Vord and Valla Field SPA has been screened into the assessment due to the potential risk of collision and operational phase displacement/barrier effects during the operation and maintenance phase of the Project.

### Operation and maintenance phase displacement/barrier effects

2488. Displacement effects for gannet for the Project were assessed during the autumn and spring migration periods, based on an unapportioned peak mean population of 124 and eight birds respectively, calculated for the windfarm site and a 2km buffer, in line with recommendations within the SNCB guidance (SNCB 2017). As set out above, no gannets present at the windfarm site have been apportioned to Hermaness, Saxa Vord and Valla Field SPA during the breeding season. The displacement matrices used to calculate potential impacts are presented in **Appendix 12.1** of the ES, and summarised in **Table 8.153**. The inclusion of all birds within the 2km buffer, to determine the total number of birds subject to displacement, is precautionary, as in reality the avoidance rate is likely to fall with distance from the windfarm site.

2489. A displacement rate of 60-80% and mortality rate of 1% has been presented. A maximum 1% mortality value has been selected firstly because gannet are known to possess high habitat flexibility (Furness and Wade, 2012). This suggests that displaced birds will readily find alternative habitats including foraging areas. Secondly, no evidence of displacement-induced mortality has been identified, which means there is limited justification for setting predicted mortality rates at a higher level. Given the extensive foraging range of this species (Woodward *et al.*, 2019), there may be no mortality costs to displacement from the relatively very small footprints of OWFs.

*Table 8.153 Gannet – predicted operation and maintenance phase displacement and mortality from Hermaness, Saxa Vord and Valla Field SPA*

| Mean peak abundance estimate type | Mean peak abundance estimate | Number of SPA breeding adults present by season[1] | Annual mortality range[2] | Annual baseline mortality increase range[3] |
|---|---|---|---|---|
| Upper 95% CI | 809 (breeding)<br>189 (autumn)<br>16 (spring)<br>1,014 (year round) | 0 (breeding)<br>3 (autumn)<br>0 (spring)<br>4 (year round) | 0-0 | 0.00-0.00% |
| Mean | 541 (breeding)<br>124 (autumn)<br>8 (spring)<br>673 (year round) | 0 (breeding)<br>2 (autumn)<br>0 (spring)<br>2 (year round) | 0-0 | 0.00-0.00% |
| Lower 95% CI | 160 (breeding)<br>0 (autumn)<br>0 (spring)<br>160 (year round) | 0 (breeding)<br>0 (autumn)<br>0 (spring)<br>0 (year round) | 0-0 | 0.00-0.00% |

[1] 1.8% and 2.2% of birds are assumed to be breeding adults from the SPA population during the autumn and spring migration periods respectively.

[2] Assumes displacement rates of 60-80% and mortality rate of 1%

[3] Background population is Hermaness, Saxa Vord and Valla Field SPA breeding adults (51,560 individuals), adult age class annual mortality rate of 8.1% (Horswill and Robinson, 2015)

2490. Using the maximum potential mortality value, there would be no measurable increase in gannet mortality. Accordingly, no significant effects on gannet are predicted during the operation and maintenance phase, and **it is concluded that there would be no potential for the Project to have an adverse effect on the integrity of Hermaness, Saxa Vord and Valla Field SPA**.

2491. The confidence in the assessment is high for several reasons. Firstly, the evidence used to set the displacement rates presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES is of high applicability and quality. Whilst there is limited available evidence to inform mortality rates, 1% is considered to be sufficiently precautionary based on expert opinion. Finally, the conclusion of the assessment is the same irrespective of whether the mean or 95% upper CI mean peak abundances are used to calculate potential mortality and increases in the baseline mortality rate of the background population.

**Operation and maintenance phase collision risk**

2492. Information to support the Appropriate Assessment for collision risk on breeding adult gannets belonging to the Hermaness, Saxa Vord and Valla Field SPA population is presented in **Table 8.154**. Collision estimates, calculated using the sCRM, are presented by biological season. A summary of the annual outputs and the corresponding increase in the annual baseline mortality rate is also presented. Parameters used in the sCRM were agreed with Natural England during the ETG process and are described in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES. In accordance with Natural England advice, a 70% macro-avoidance correction was applied to gannet abundance data used in the sCRM.

2493. Based on the mean collision rates, no breeding adult gannets from Hermaness, Saxa Vord and Valla Field SPA are considered at risk of collision as a result of the Project. Therefore, there would be no measurable increase the existing mortality of the SPA breeding population.

*Table 8.154 Predicted seasonal and annual collision mortality (Stochastic model Option 2, avoidance rate 0.993 (±0.0003), plus 70% macro-avoidance) for breeding adult gannets at the windfarm site, apportioned to Hermaness, Saxa Vord and Valla Field SPA, with corresponding increases to baseline mortality of the population*

| Period | Breeding Season Apr-Aug | Autumn Migration Sep-Nov | Non-breeding/winter - | Spring Migration Dec-Mar | Annual Jan-Dec |
|---|---|---|---|---|---|
| Total collisions[1] (mean and 95% CIs) | 0.83 (0.00-3.35) | 0.14 (0.00-0.74) | - | 0.00 | 0.97 (0.00-4.10) |
| % apportioned to the SPA | 0.0% | - | - | - | - |
| Total SPA collisions (mean and 95% CIs) | 0.00 (0.00-0.00) | 0.00 (0.00-0.01) | - | 0.00 (0.00-0.00) | 0.00 (0.00-0.01) |
| Mortality increase[2] (mean and 95% CIs) | 0.00% (0.00-0.00%) | 0.00% (0.00-0.00%) | - | 0.00% (0.00-0.00%) | 0.00% (0.00-0.00%) |

[1] Breeding season collision values reduced to 73.8% of modelled value to reflect proportion of adult birds recorded during site surveys
[2] Assuming predicted annual SPA mortality of 4,144 birds (51,160 x 0.081)

2494. Accordingly, no significant effects on gannet are predicted during the operation and maintenance phase, and **it is concluded that there would be no potential for the Project to have an adverse effect on the integrity of Hermaness, Saxa Vord and Valla Field SPA**. Comments received from RSPB during the ETG process, indicating that they do not accept the 70% macro-avoidance rate for collision risk recommended by Natural England, are noted. However, even in the absence of this correction factor, the net increase in mortality would be unchanged (i.e. zero).

2495. The confidence in the assessment is high. The evidence used to define the CRM input parameters presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES submitted alongside this RIAA is of high applicability and quality. Whilst there is uncertainty around some of the input parameters (e.g. avoidance rate), the rates selected are considered to be sufficiently precautionary based on expert opinion to provide confidence that collision rates are not underestimated.

**Combined displacement/barrier effects and collision risk**

2496. As no measurable increase in mortality is predicted for both displacement and collision risk, the mean combined displacement and collision rates for breeding adult gannet from the Hermaness, Saxa Vord and Valla Field SPA would be zero. Therefore, there would be no net increase in existing mortality rates.

2497. **It is concluded that based on predicted gannet mortality due to the combined effects of operational phase displacement and collision there is no potential for the Project-alone or in-combination to have an adverse effect on the integrity of the Hermaness, Saxa Vord and Valla Field SPA**.

2498. The confidence in the assessment is high, for the reasons provided in the individual displacement and collision assessments.

### Potential effects in-combination with other projects

2499. As no measurable effects of displacement/barrier and collision on gannet are predicted as a result of the Project-alone, there would be no contribution to other plans or projects in-combination. **It is therefore concluded that there would be no potential for the Project-alone or in-combination to have an adverse effect on the integrity of Hermaness, Saxa Vord and Valla Field SPA**.

### 8.63.3.4 Red-throated diver

### Status

2500. The Hermaness, Saxa Vord and Valla Field SPA breeding red-throated diver population at classification (2009) was cited as 26 pairs, or 52 breeding adults (average 1994 – 1999; SNH 2009g). Furness (2015) gave a breeding population of 16 pairs, or 32 breeding adults, in 2013; this is used as the reference population for the assessment.

2501. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.16 (1 – 0.840; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be five breeding adults.

### Functional linkage and seasonal apportionment of potential effects

2502. The mean maximum foraging range of red-throated diver is 9km (±0km), as is the maximum foraging range (Woodward *et al*., 2019). The Project is located approximately 782km from Hermaness, Saxa Vord and Valla Field SPA, which means that the Project is beyond the maximum foraging range for red-throated divers from the SPA. No impacts during the breeding season from the Project are therefore apportioned to red-throated divers breeding at this SPA.

2503. Outside the breeding season, breeding red-throated divers from the SPA are assumed to range widely and to mix with red-throated divers of all ages from breeding colonies in the UK and further afield. The relevant background population during the autumn and spring migration seasons is the UK Western waters plus Channel BDMPS, consisting of 4,373 individuals during autumn and spring passage periods (September to November and February to April) (Furness, 2015). The relevant background population during the winter season is the NW England and Wales BDMPS, consisting of 1,657 individuals (Furness, 2015).

2504. During the spring and autumn migration seasons, Furness (2015) estimated that 5% of breeding adults from Hermaness, Saxa Vord and Valla Field SPA (32 birds) are present within the UK Western waters plus Channel BDMPS, which is two birds. This represents 0.04% of the BDMPS population for that period (4,373). During the winter period it is estimated that 2% of breeding adults from Hoy SPA (32 birds) are present within the NW England and Wales BDMPS, which is less than one bird. This represents 0.03% of the BDMPS population for that period (1,657). These percentages (i.e. 0.04% and 0.03%) are the proportions of birds present at the windfarm site that are presumed to originate from the Hermaness, Saxa Vord and Valla Field SPA during the relevant seasons.

### Potential effects on the qualifying feature

#### Operation and maintenance phase disturbance/displacement/barrier effects

*Project-alone*

2505. The year-round mean peak abundance of red-throated divers present within the windfarm site and hybrid 10km buffer was 74 (0-240) individuals (refer to

**Appendix 12.1** of the ES). Of these, less than one bird (0.03 birds (0-0.08)) was likely to be a breeding adult from the Hermaness, Saxa Vord and Valla Field SPA.

2506. Red-throated divers have a very high sensitivity to disturbance and displacement from operational OWFs. The majority of birds present before OWFs are constructed are displaced by the operation of OWFs. It is expected (based on expert opinion), that this is due to a combination of anthropogenic activities (mainly vessel movements), as well as the presence of OWF infrastructure. A large body of work investigating the effects of displacement of red-throated divers due to operational OWFs exists (Dorsch *et al*., 2020; Elston *et al*., 2016; Gill *et al*., 2018; Hi Def Aerial Surveying, 2017; Irwin *et al*., 2019; MacArthur Green and Royal HaskoningDHV, 2021; McGovern *et al*., 2016; Mendel *et al*., 2019; Percival, 2014; Percival and Ford, 2017; Petersen *et al*., 2014, 2006; Vilela *et al*., 2020; Welcker and Nehls, 2016).

2507. There was a high degree of concordance of the available literature with respect to effects of operation of OWFs on red-throated diver distribution and abundance within OWFs. There was also a high degree of concordance that displacement effects extended beyond OWF boundaries. However, there was considerable variation with respect to the distance at which this effect remained detectable. Studies within the UK have ranged from no significant displacement effects being reported (McGovern *et al*., 2016), displacement effects being restricted to 1km to 2km of an OWF (Percival, 2014; Percival and Ford, 2017), to clear displacement effects across many years. These effects have been reported extending to 7km from OWFs (MacArthur Green and Royal HaskoningDHV, 2021), 9km from OWFs (Elston *et al*., 2016; Hi Def Aerial Surveying, 2017), and beyond, though not all evidence was available to be referenced by this assessment. Studies from other countries have also recorded variable displacement distances, ranging from 1.5km to 2km (Welcker and Nehls, 2016) to 10km and beyond (Dorsch *et al*., 2020; Vilela *et al*., 2020). Displacement effects were detectable up to 20km from OWFs in one case.

2508. There was also concordance in the studies reviewed that displacement effects on red-throated diver due to operational OWFs occurred on a gradient, with the strongest effects observed either within, or close to OWFs. As the distance from the OWF increased, the magnitude of the effect reduced, until a distance was reached at which the effect was no longer detectable.

2509. No study to date has managed to provide insight into whether changes in red-throated diver distribution at any spatial scale had the potential to result in population level effects, either at local, regional, national or international levels. Red-throated divers have been noted to be capable of utilising a range of marine habitats and prey species (Dierschke *et al*., 2017; Guse *et al*., 2009; Kleinschmidt *et al*., 2016). Recent data from the Outer Thames Estuary SPA indicated that birds were much more commonly recorded in water depths of less than 20m (Irwin *et al*., 2019). During the non-breeding season, red-throated divers were mostly widely dispersed, at densities often less than four birds per km$^2$ (Dierschke *et al*., 2017), and were highly mobile (Dorsch *et al*., 2020; Duckworth *et al*., 2020). In some instances, home ranges of many thousands of square kilometres have been demonstrated (Nehls *et al*., 2018). This implies that following displacement, red-throated divers will be able to find alternative foraging sites, in some cases distant from the original area of displacement, which may already have been part of their existing non-breeding season range. It seems likely that in the vast majority of cases, mortality is not a consequence of displacement.

2510. Displacement rates of 1.000, along with a range of mortality rates of 1% to 10% of displaced birds is considered for this species at this SPA (UK SNCBs, 2017). However, it is considered that there is a high possibility that displacement and mortality rates are substantially lower than this.

2511. **Table 8.155** sets out the predicted impacts on red-throated divers from Hermaness, Saxa Vord and Valla Field SPA during the non-breeding season. A displacement rate of 1.000 is presented for this species, along with a range of mortality rates of 1% to 10% of displaced birds (UK SNCBs, 2017). An estimated annual mortality for the population is provided, along with the increase of existing mortality that would occur through such an impact. The displacement matrices used to calculate potential impacts are presented in **Appendix 12.1** of the ES.

*Table 8.155 Red-throated diver – predicted operation and maintenance phase displacement and mortality from Hermaness, Saxa Vord and Valla Field SPA*

| Mean peak abundance estimate type | Mean peak abundance estimate by season[1] | Number of SPA breeding adults present by season)[2] | Annual mortality range[3] | Annual baseline mortality increase range[4] |
|---|---|---|---|---|
| Upper 95% CI | 35 (b)<br>23 (aut)<br>55 (win)<br>127 (spr)<br>240 (year round) | 0 (b)<br>0 (aut)<br>0 (win)<br>0 (spr)<br>0 (year round) | 0-0 | 0.00-0.05% |
| Mean | 8 (b)<br>10 (aut)<br>12 (win)<br>43 (spr)<br>74 (year round) | 0 (b)<br>0 (aut)<br>0 (win)<br>0 (spr)<br>0 (year round) | 0-0 | 0.00-0.02% |

| Mean peak abundance estimate type | Mean peak abundance estimate by season[1] | Number of SPA breeding adults present by season)[2] | Annual mortality range[3] | Annual baseline mortality increase range[4] |
|---|---|---|---|---|
| Lower 95% CI | 0 (b)<br>0 (aut)<br>0 (win)<br>0 (spr)<br>0 (year round) | 0 (b)<br>0 (aut)<br>0 (win)<br>0 (spr)<br>0 (year round) | 0-0 | 0.00-0.00% |

[1] Breeding season = b, autumn migration season = aut, winter season = win, spring migration season = spr

[2] Assumes breeding adult apportioning of 0.0% (breeding season), 0.04% (spring and autumn migration) and 0.03% (winter) to Hermaness, Saxa Vord and Valla Field SPA.

[3] Assumes displacement rate of 100% and mortality rates of 1-10%.

[4] Background mortality rate of 16.0% (Horswill and Robinson, 2015)

2512. Based on the mean peak abundances, less than one red-throated diver from the Hermaness, Saxa Vord and Valla Field SPA would be at risk of displacement (**Table 8.155**). At displacement rates of 1.000, and mortality rates of 1% to 10% for displaced birds, <0.02 SPA breeding adults would be predicted to die each year due to displacement from the Project.

2513. Assuming a displacement rate of 1.00 and a mortality rate of 10% of displaced birds, annual mortality within the SPA breeding adult population would increase by 0.02%. Using a more realistic mortality rate for displaced birds of 1%, annual mortality in the population would be less than 0.01%.

2514. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that no detectable changes in mortality rates would occur under any combination of displacement and mortality rates when the mean peak abundance estimate assessments are considered. This would be the case even when upper 95% CIs are considered.

2515. **It is concluded that predicted red-throated diver mortality due to operational phase displacement for the project alone would not adversely affect the integrity of the Hermaness, Saxa Vord and Valla Field SPA.**

2516. The confidence in the assessment is high for several reasons. Firstly, the evidence used to inform the evidence-based displacement rates is of high applicability and quality (based on the criteria discussed in **Chapter 12 Offshore Ornithology** of the ES). Whilst there was limited available evidence to inform mortality rates, 1% is considered to be sufficiently precautionary based on expert opinion. This species has been demonstrated to be highly mobile during the non-breeding season, and individuals frequently possessed very large home ranges during this time (Dorsch *et al*., 2020; Nehls *et al*., 2018). It is therefore anticipated that displaced birds will find alternative habitat in the vast majority of cases. Finally, the conclusion of the assessment is the same irrespective of whether 1% or 10% mortality, or the mean or 95% upper CI mean peak abundances, are used to calculate potential mortality and increases in the baseline mortality rate of the background population.

*In-combination*

2517. As the Project would have no measurable effect on red-throated diver populations from the Hermaness, Saxa Vord and Valla Field SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of Hermaness, Saxa Vord and Valla Field SPA, when assessed in-combination with other plans or projects.**

#### 8.63.3.5 Puffin

#### Status

2518. The Hermaness, Saxa Vord and Valla Field SPA breeding puffin population at classification (2009) was cited as 55,000 individuals (SNH, 2009g) Furness (2015) gave a breeding population of 23,661 pairs, or 47,322 pairs, in 2002. Owen *et al.,* (2018) estimated 13,773 apparently occupied burrows at Hermaness in 2017, or 27,546 breeding adults; this is used as the reference population for the assessment.

2519. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.094 (1 – 0.906, Horswill and Robinson (2015), the expected annual mortality from the SPA population would be 2,589 breeding adults.

#### Functional linkage and seasonal apportionment of potential effects

2520. The mean maximum foraging range of puffin is 137.1km (±128.3km) and the maximum foraging range is 383km (Woodward *et al*., 2019). The Project is located approximately 782km from Hermaness, Saxa Vord and Valla Field SPA, which means the Project is beyond the maximum foraging range for puffins from the SPA. No impacts during the breeding season from the Project are therefore apportioned to puffins breeding at this SPA.

2521. Outside of the breeding season, breeding puffins, including those from the Hermaness, Saxa Vord and Valla Field SPA, are not constrained by requirements to visit nests to incubate eggs or provision chicks. At this time, they are assumed to range more widely and to mix with puffins of all age classes from breeding colonies in the UK and further afield. The background population during these seasons is the UK Western Waters BDMPS. This consists of 304,557 individuals during the non-breeding season (August to March).

2522. Furness (2015) estimated that 8% of the Hermaness, Saxa Vord and Valla Field SPA breeding adults (47,322) are present within the UK Western Waters BDMPS during the non-breeding season, which is 3,786 birds. This represents 1.2% of the BDMPS population for this period (304,557). It is therefore assumed that 1.2% of puffins present at the Project site are breeding adults from Hermaness, Saxa Vord and Valla Field SPA.

## Potential effects on the qualifying feature

### Operation and maintenance phase disturbance / displacement / barrier effects

*Project-alone*

2523. The mean peak abundance of puffins present within the windfarm site and 2km buffer during the non-breeding season was 19.7 (1.9-50.8) individuals (refer to **Appendix 12.1** of the ES). Of these, less than one bird (0.2 (0.0-0.6)) was likely to be a breeding adult from Hermaness, Saxa Vord and Valla Field SPA.

2524. **Table 8.156** sets out the predicted impacts on puffins from Hermaness, Saxa Vord and Valla Field SPA during the non-breeding season. This estimates that there would be no measurable increase in mortality from the SPA population, assuming a displacement rate of 30-70% and a mortality of 1-10% for displaced birds.

*Table 8.156 Puffin – predicted operation and maintenance phase displacement and mortality from Hermaness, Saxa Vord and Valla Field SPA*

| Mean peak abundance estimate type | Mean peak abundance estimate | Number SPA breeding adults present (non-breeding season)[1] | Annual mortality range[2] | Annual baseline mortality increase range[3] |
|---|---|---|---|---|
| Upper 95% CI | 50.8 | 0.6 | 0-0 | 0.00-0.00% |
| Mean | 19.7 | 0.2 | 0-0 | 0.00-0.00% |
| Lower 95% CI | 1.9 | 0.0 | 0-0 | 0.00-0.00% |

[1] Assumes 1.2% of birds present during the non-breeding season are Hermaness, Saxa Vord and Valla Field SPA breeding adults
[2] Assumes displacement rates of 30-70% and mortality rates of 1-10%
[3] Background mortality rate of 9.4% (Horswill and Robinson, 2015)

2525. **It is concluded that predicted puffin mortality due to operational phase displacement would not adversely affect the integrity of the Hermaness, Saxa Vord and Valla Field SPA.**

2526. The confidence in the assessment is high for several reasons. Firstly, the evidence used to inform the evidence-based displacement rates is of high applicability and quality (based on the criteria discussed in **Chapter 12 Offshore Ornithology** of the ES). Whilst there was limited available evidence to inform mortality rates, 1% is considered to be sufficiently precautionary based on expert opinion. This species is not regarded as being highly specialised in its habitat requirements (Bradbury *et al*., 2014; Furness and Wade, 2012; Garthe and Hüppop, 2004), and it is therefore anticipated that displaced birds will find alternative habitat in the vast majority of cases. Finally, the conclusion of the assessment is the same irrespective of whether the mean or 95% upper CI mean peak abundances are used to calculate potential mortality and increases in the baseline mortality rate of the background population.

*In-combination*

2527. As the Project would have no measurable effect on puffin populations from the Hermaness, Saxa Vord and Valla Field SPA , there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that predicted puffin mortality due to operational phase displacement due to the Project in-combination with other plans or projects would not adversely affect the integrity of the Hermaness, Saxa Vord and Valla Field SPA.**

# 8.64 Ballaugh Curragh Ramsar (transboundary site)

2528. Ballaugh Curragh Ramsar site is located on the Isle of Man, approximately 85km from the windfarm site.

## 8.64.1 Description of designation

2529. The Ballaugh Curragh consists of a complex mosaic of interrelated peatland habitats dominated by willow and birch scrub (a habitat known locally as 'curragh'). Associated wetland habitats include bog pools, wet woodland, man-made ditch systems and fen grassland. The area supports a large winter roost of hen harriers - at times the largest recorded in Europe. It has a very high diversity of breeding birds and a range of mire and aquatic plants including local rarities and species protected under Manx law.

## 8.64.2 Assessment

### 8.64.2.1 Migratory bird qualifying features

#### Status

2530. Ballaugh Curragh Ramsar site supports the largest hen harrier winter roost on the Island, with a 5-year mean peak count of 82 (1996/7-2000/01); well over 100 may be seen. This represents a high proportion of all winter roosting hen harriers in the region; in some years it has been recorded as having the highest number in Western Europe. It was also one of the last strongholds for corncrake, but it is not thought that this species currently breeds within the site.

#### Functional linkage and seasonal apportionment of potential effects

2531. Hen harrier has been screened into the Appropriate Assessment due to the risk of potential impacts occurring during the spring and autumn migration seasons. This species was not recorded in the aerial survey study area during the baseline surveys undertaken at the windfarm site. However, it is recognised that it may pass through the habitat in the windfarm site during migration periods, and may have been missed by the surveys.

2532. The apportioning of impacts to this designated site was calculated by dividing the number of collisions calculated at the national level by the proportion of the national population that were members of the designated site population at citation. The numbers used to define the national populations were the Great Britain populations presented in Wright *et al.,* (2012). The designated site population was obtained from the Ramsar site population.

## Potential effects on the qualifying feature from the Project-alone

2533. The qualifying feature of this designated site (hen harrier) has been screened into the Appropriate Assessment due to the potential risk of collision.

2534. The magnitude of potential collision impacts has been investigated using the SOSSMAT tool (Wright *et al*., 2012).

### Collision risk

2535. The estimated annual collision risk for hen harrier, along with the conclusion of the assessment based on this annual collision rate, is presented in **Table 8.157**. An avoidance rate of 0.980 has been assumed.

2536. The number of annual collisions predicted for hen harrier is effectively zero. Therefore, it is expected that the increases to existing mortality due to this impact would be undetectable within the site population. Such impacts would consequently not result in any measurable effect.

2537. It is concluded that the predicted hen harrier mortality due to collision at the windfarm site would not adversely affect the integrity of the Ballaugh Curragh Ramsar site.

2538. Whilst extensive information exists on the responses of waterbirds to onshore OWFs, there is substantial uncertainty regarding waterbird movements at sea. The confidence level assigned to this section of the assessment is therefore medium. However, since such low levels of collision are predicted, an adverse effect on the integrity of the site is considered highly unlikely, even in the unlikely event that impacts have been underestimated.

## Potential effects on the qualifying feature in-combination with other projects

### Collision risk

2539. The migration corridors identified by Wright *et al.,* (2012) indicated that hen harrier migration activity is widespread across UK waters. Similarly low numbers of birds, and hence collisions, are therefore expected at other OWFs in UK waters. The total collision mortality of hen harriers at all UK OWFs is still likely to be small in the context of their respective national populations, and the number of collisions associated with this designated site will be smaller still. It is expected that the increases to existing mortality rates due to this impact would be undetectable within the site population. Such impacts would consequently not result in any measurable effect.

2540. **It is concluded that predicted hen harrier mortality due to collision at the windfarm site, in-combination with other projects, would not adversely affect the integrity of the Ballaugh Curragh Ramsar site**.

*Table 8.157 Information to support the Appropriate Assessment for Ballaugh Curragh Ramsar site (migratory bird qualifying features)*

| Qualifying feature | GB population (Wright *et al*., 2012) | Ramsar site population (citation) | Apportioning rate | Unapportioned predicted mean annual collisions (avoidance rate 0.980) | Annual collisions apportioned to SPA | Conclusion of adverse effect on site integrity |
|---|---|---|---|---|---|---|
| Hen harrier | 1,140 | 82 | 7.2% | 0.00 | 0.00 | No adverse effect on site integrity. Numbers of collisions so small that effects on population would be negligible. It would not be possible for impacts of this magnitude to have an effect at the site level given the background populations |

# 8.65 Lambay Island SPA (transboundary site)

2541. Lambay Island SPA is located on the east coast of Ireland approximately 156km from the windfarm site.

## 8.65.1 Description of designation

2542. Lambay Island SPA lies approximately 4km off the north County Dublin coastline, and is separated from it by a channel of 10-13m in depth. East of Lambay Island, the water deepens rapidly into the Irish Sea basin. The island, which rises to 127m, has an area of 250ha above high tide mark. Lambay Island SPA is internationally important for its breeding seabirds and is of particular note for the diversity of these, with 12 species breeding regularly.

## 8.65.2 Conservation objectives

2543. The conservation objective of the SPA is 'to maintain or restore the favourable conservation condition of the bird species listed as Special Conservation Interests for this SPA.'

## 8.65.3 Assessment

2544. Nine qualifying features of Lambay Island SPA have been screened into the Appropriate Assessment (**Table 5.2**): guillemot, razorbill, puffin, fulmar, lesser black-backed gull, herring gull, kittiwake, shag and cormorant.

### 8.65.3.1 Guillemot

#### Status

2545. The Lambay Island SPA breeding guillemot population stood at 40,705 pairs (or 81,410 breeding adults) in 1999, and 38,999 pairs (or 77,998 breeding adults) in 2004 (NPWS, 2011a). The most recent count is 59,983 individuals in 2015 (JNCC, 2023a); this is used as the reference population for the assessment.

2546. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.061 (1 – 0.939; Horswill and Robinson, 2015), the expected annual mortality rate from the SPA population would be 3,659 breeding adults.

#### Functional linkage and seasonal apportionment of potential effects

2547. The mean maximum foraging range of guillemot is 73.2km (±80.5km) and the maximum foraging range is 338km (Woodward *et al.*, 2019). The Project is located approximately 156km from Lambay Island SPA, which means that the Project is beyond the mean maximum foraging range and mean maximum foraging range +1SD of guillemots breeding at this SPA, but within the maximum foraging range. The maximum foraging range is a poor indicator of typical foraging behaviour. It would be expected that few birds or foraging trips will occur at this distance from the colony, and even fewer with any regularity. No impacts during the breeding season are therefore apportioned to birds breeding at this colony.

2548. Outside the breeding season, breeding guillemots from the SPA are assumed to range widely and to mix with guillemots of all ages from breeding colonies in Ireland, the UK and beyond. The relevant non-breeding season reference population is the UK Western Waters BDMPS, consisting of 1,139,220 individuals (August to February) (Furness, 2015).

2549. As no published estimate was available, it is assumed that 95% of Lambay Island SPA breeding adults are present within the UK Western Waters BDMPS during the non-breeding season; the same proportion as 'West Coast UK non-SPA populations' (and other comparable SPAs around the Irish Sea) identified by Furness (2015). It is assumed that the most recent count prior to the publication of Furness (2015) was used to inform the BDMPS total which is 77,998 breeding adults; 95% of this population is 74,098 birds. This represents 6.5% of the BDMPS population for this period (1,139,220). It is therefore assumed that 6.5% of guillemots present at the Project site are breeding adults from Lambay Island SPA.

## Potential effects on the qualifying feature

### Operation and maintenance phase disturbance/displacement/barrier effects

*Project-alone*

2550. The mean peak abundance of guillemots present within the windfarm site and 2km buffer during the non-breeding season was 8,315 (6,085-12,047) individuals (refer to **Appendix 12.1** of the ES). Of these, 540 birds (396-783) were likely to be breeding adults from the Lambay Island SPA.

2551. **Table 8.158** sets out the predicted impacts on guillemots from Lambay Island SPA during the non-breeding season. Displacement rates of 30% to 70% are considered for this species, along with a range of mortality rates of 1% to 10% of displaced birds (UK SNCBs, 2017).

2552. The available evidence suggests that the upper ranges of these displacement and mortality rates may be excessively precautionary. Whilst it is true that guillemots and razorbills tend to be displaced by OWFs they do not avoid them completely, and displacement rates vary between sites. MacArthur Green (2019b) concluded that, on average, densities within OWFs were around half of the density found in the habitats around the OWF. At some OWFs there was also displacement of birds from a buffer zone surrounding it. The size of the buffer zone varied between OWFs and was generally less than 2km, with auk density increasing across the buffer zone as distance from turbines increased, up to the density in the wider area. Another recent review (APEM, 2022) found that the current displacement rates suggested by the SNCBs for guillemot and razorbill (30% to 70%) did not account for the quality of or confidence in the studies which were used to inform this position, and that studies where no significant effects were recorded were not accounted for during the provision of the advice. APEM (2022) suggested that in the case of Hornsea Project Four, an evidence-based displacement rate of 50% was appropriate. However, the study also recognised that larger displacement effects were possible.

2553. Mortality due to displacement could arise due to increased energy costs and/or decreased energy intake, if displacement results in increased flying time to avoid OWFs, and/or increased bird densities and competition for prey in areas of unimpacted habitat outside the OWF. Given that UK OWFs in the Irish and Celtic Seas represent a very small proportion of the available foraging habitat for guillemots and razorbills within UK Western Waters, the increase in density of auks outside OWFs due to displacement is likely to be negligible (MacArthur Green, 2019b). It is considered unlikely that the mortality rate due to displacement would be as high as the empirically estimated 6.1% annual mortality for adult guillemots that occurs due to the combination of 'natural' factors and anthropogenic activities (Horswill and Robinson, 2015). Indeed, it may be much lower; MacArthur Green (2019b) recommended precautionary rates of 50% displacement for guillemot and 1% mortality of displaced birds based on a review of available evidence. Modelling undertaken by APEM (2022) found that in the case of Hornsea Project Four, a mortality rate of 1% for displaced auks was considered to be precautionary, and that this may be an overestimate given that Hornsea Project Four is located towards the upper end of the mean maximum foraging range for guillemot from the closest designated sites. It is also noted that in the recent decision on Hornsea Project Four, the SoS determined that a rate of 70% displacement/2% mortality was appropriate to inform the assessment of effects on guillemots from Flamborough and Filey Coast SPA (DESNZ, 2023b).

2554. The full ranges of recommended displacement and mortality effects are considered by the assessment, along with evidence-based displacement and mortality rates of 50% and 1%, respectively (APEM, 2022; MacArthur Green, 2019b).

*Table 8.158 Guillemot – predicted operation and maintenance phase displacement and mortality from Lambay Island SPA*

| Mean peak abundance estimate type | Mean peak abundance estimate | Number of SPA breeding adults present (non-breeding season)[1] | Annual mortality range[2] | Annual baseline mortality increase range[3] |
|---|---|---|---|---|
| Upper 95% CI | 12,047 | 780 | 2-55 | 0.06-1.50% |
| Mean | 8,315 | 540 | 2-38 | 0.04-1.03% |
| Lower 95% CI | 6,085 | 396 | 1-28 | 0.03-0.76% |

[1] Assumes 6.5% of birds present during the non-breeding season are Lambay Island SPA breeding adults

[2] Assumes displacement rates of 30-70% and mortality rates of 1-10%

[3] Background mortality rate of 6.1% (Horswill and Robinson, 2015)

2555. Based on the mean peak abundances, the annual total of guillemots from the Lambay Island SPA at risk of displacement is 540 birds (**Table 8.158**). At displacement rates of 30% to 70%, and mortality rates of 1% to 10% for displaced birds, 2 to 55 SPA breeding adults would be predicted to die each year due to displacement from the Project.

2556. Assuming a displacement rate of 70% and a mortality rate of 10% of displaced birds, annual mortality within the SPA breeding adult population would increase by 1.03%. Using an evidence-based displacement rate of 50%, and a mortality rate for displaced birds of 1%, annual mortality in the population would instead increase by 0.07% (3 birds).

2557. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that no detectable changes in mortality rates would occur under almost any combination of displacement and mortality rates when the mean peak abundance estimate assessments are considered. Mortality rate increases of over 1% are predicted for mean peak abundance estimate assessments only when a displacement rate of 70% and a mortality rate of 10% is considered. These displacement and mortality rates are much higher than evidence suggested will actually be the case. Use of the evidence-based displacement (50%) and mortality rate (1%) would result in a mortality increase of significantly less than 1%, as would a rate of 70%/2% agreed by the SoS in respect of Hornsea Project Four (DESNZ, 2023b).

2558. Increases of over 1% are also predicted if the upper 95% CIs for mean peak abundances are used as inputs to the assessment alongside a 10% mortality rate for displaced birds. The probability of this occurring is extremely small for two reasons. Firstly, the upper 95% CI for the mean peak abundances are highly unlikely to occur regularly at the windfarm site. Secondly, mortality rates for displaced birds of 10% are much higher than evidence suggested will actually be the case, and use of the evidence-based displacement (50%) and mortality rate (1%) (and also 70%/2%) would again result in a mortality increase of significantly less than 1%.

2559. **It is concluded that predicted guillemot mortality due to operational phase displacement for the project alone would not adversely affect the integrity of the Lambay Island SPA.**

2560. The confidence in the assessment is high for several reasons. Firstly, the evidence used to inform the evidence-based displacement rates is of high applicability and quality (based on the criteria discussed in **Chapter 12 Offshore Ornithology** of the ES). Whilst there was limited available evidence to inform mortality rates, 1% is considered to be sufficiently precautionary based on expert opinion. This species is not regarded as being highly specialised in its habitat requirements (Bradbury *et al*., 2014; Furness and Wade, 2012; Garthe and Hüppop, 2004), and it is therefore anticipated that displaced birds will find alternative habitat in the vast majority of cases. Finally, the conclusion of the assessment is the same irrespective of whether the mean or 95% upper CI mean peak abundances are used to calculate potential mortality and increases in the baseline mortality rate of the background population, provided the evidence-based displacement and mortality rates are used.

*In-combination*

2561. The in-combination assessment for guillemots from Lambay Island SPA has been undertaken in accordance with the approach presented in **Section 8.1**. The total population apportioned to Lambay Island SPA at risk of displacement is estimated to be 4,623 breeding adults (**Appendix 12.1** of the ES). Annual in-combination displacement and mortality rates for birds from Lambay Island SPA are presented in **Table 8.159**.

2562. Assuming a displacement rate of 70% and a mortality rate of 10% of displaced birds, 324 breeding adult SPA birds would be lost to displacement annually. Including additional apportioned mortality from the Morlais and Holyhead tidal projects (total 5.20 birds), this would increase the existing mortality within the SPA population (3,659 breeding adult birds per year) by 8.99%. Using an evidence-based displacement rate of 50%, and a mortality rate for displaced birds of 1%, the annual in-combination displacement mortality would be 23 birds. This would increase the existing mortality within this population by 0.77%. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that detectable changes in mortality rates will not occur due to the level or mortality predicted if the more realistic rates for mortality are used.

2563. **It is concluded that predicted guillemot mortality due to of operational phase displacement impacts of the Project in-combination with other projects, would not adversely affect the integrity of Lambay Island SPA.**

*Table 8.159 In-combination year-round displacement matrix for guillemot from Lambay Island SPA*

| Annual | Mortality | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| **Displacement** | **1%** | **2%** | **3%** | **4%** | **5%** | **10%** | **20%** | **30%** | **50%** | **80%** | **100%** |
| **10%** | 5 | 9 | 14 | 18 | 23 | 46 | 92 | 139 | 231 | 370 | 462 |
| **20%** | 9 | 18 | 28 | 37 | 46 | 92 | 185 | 277 | 462 | 740 | 925 |
| **30%** | **14** | **28** | **42** | **55** | **69** | **139** | 277 | 416 | 693 | 1110 | 1387 |
| **40%** | **18** | **37** | **55** | **74** | **92** | **185** | 370 | 555 | 925 | 1479 | 1849 |
| **50%** | **23** | **46** | **69** | **92** | **116** | **231** | 462 | 693 | 1156 | 1849 | 2312 |
| **60%** | **28** | **55** | **83** | **111** | **139** | **277** | 555 | 832 | 1387 | 2219 | 2774 |
| **70%** | **32** | **65** | **97** | **129** | **162** | **324** | 647 | 971 | 1618 | 2589 | 3236 |
| **80%** | 37 | 74 | 111 | 148 | 185 | 370 | 740 | 1110 | 1849 | 2959 | 3698 |
| **90%** | 42 | 83 | 125 | 166 | 208 | 416 | 832 | 1248 | 2080 | 3329 | 4161 |
| **100%** | 46 | 92 | 139 | 185 | 231 | 462 | 925 | 1387 | 2312 | 3698 | 4623 |

Note: The cells show the number of birds predicted to die (rounded to the nearest integer) at a given rate of displacement and mortality. Highlighted cells are considered to be the most realistic scenario, in accordance with SNCB advice (SNCBs, 2022).

#### 8.65.3.2 Razorbill

##### Status

2564. The Lambay Island SPA breeding razorbill population stood at 2,906 pairs (or 5,812 breeding adults) in 1999, and 3,805 pairs (or 7,610 breeding adults) in 2004 (NPWS, 2011a). The most recent count is 7,353 individuals in 2015 (JNCC, 2023a); this is used as the reference population for the assessment.

2565. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.105 (1 – 0.895; Horswill and Robinson, 2015), the expected annual mortality from the SPA population would be 772 breeding adults.

##### Functional linkage and seasonal apportionment of potential effects

2566. The mean maximum foraging range of razorbill is 88.7km (±75.9km) and the maximum foraging range is 313km (Woodward *et al*., 2019). The Project is located approximately 156km from Lambay Island SPA, which means the Project is beyond the mean maximum foraging range, but within mean maximum foraging range +1SD and maximum foraging range of razorbills breeding at this SPA.

2567. The NatureScot apportioning tool (NatureScot, 2018) has been used to estimate the proportion of razorbills from each of the relevant SPAs present at the windfarm site during the breeding season. The tool estimates that 24.74% of adult birds present are likely to originate from Lambay Island SPA.

2568. Outside the breeding season, breeding razorbills from the SPA are assumed to range widely and to mix with razorbills of all ages from breeding colonies in Ireland, the UK and beyond. The relevant non-breeding season reference population is the UK Western Waters BDMPS, consisting of 606,914 individuals during autumn and spring passage periods (August to October and January to March), and 341,422 individuals during winter (November and December) (Furness, 2015).

2569. As no published estimate is available, it is assumed that 98% of Lambay Island SPA breeding adults are present within the UK Western Waters BDMPS during the non-breeding season, and 30% during the winter period; the same proportions as 'West Coast UK non-SPA populations' (and other comparable SPAs around the Irish Sea) identified by Furness (2015). It is assumed that the most recent count prior to the publication of Furness (2015) was used to inform the BDMPS total which is 7,610 breeding adults; 98% of this population is 7,458 birds, and 30% is 2,238 birds. This represents 1.2% and 0.7% of the BDMPS population (606,914 during spring/autumn and 341,422 during winter) respectively. These percentages (i.e. 1.2% (spring and autumn migration) and 0.7% (winter)) are the proportions of birds present at the windfarm site that are presumed to originate from the Lambay Island SPA during the relevant seasons.

## Potential effects on the qualifying feature

### Operation and maintenance phase disturbance/displacement/barrier effects

*Project-alone*

2570. The year-round mean peak abundance of razorbills present within the windfarm site and 2km buffer was 1,979 (703-3,552) individuals (refer to **Appendix 12.1** of the ES). Of these, 80 birds (13-179) were likely to be breeding adults from the Lambay Island SPA.

2571. **Table 8.160** sets out the predicted impacts on razorbills from Lambay Island SPA. Displacement rates of 30% to 70% are considered for this species, along with a range of mortality rates of 1% to 10% of displaced birds (UK SNCBs, 2017).

2572. The available evidence suggests that the upper ranges of these displacement and mortality rates may be excessively precautionary. Whilst it is true that guillemots and razorbills tend to be displaced by OWFs they do not avoid them completely, and displacement rates vary between sites. MacArthur Green (2019b) concluded that, on average, densities within OWFs were around half of the density found in the habitats around the OWF. At some OWFs there was also displacement of birds from a buffer zone surrounding it. The size of the buffer zone varied between OWFs and was generally less than 2km, with auk density increasing across the buffer zone as distance from turbines increased, up to the density in the wider area. Another recent review (APEM, 2022) found that the current displacement rates suggested by the SNCBs for guillemot and razorbill (30% to 70%) did not account for the quality of or confidence in the studies which were used to inform this position, and that studies where no significant effects were recorded were not accounted for during the provision of the advice. APEM (2022) suggested that in the case of Hornsea Project Four, an evidence-based displacement rate of 50% was appropriate. However, the study also recognised that larger displacement effects were possible.

2573. Mortality due to displacement could arise due to increased energy costs and/or decreased energy intake, if displacement results in increased flying time to avoid OWFs, and/or increased bird densities and competition for prey in areas of unimpacted habitat outside the OWF. Given that UK OWFs in the Irish and Celtic Seas represent a very small proportion of the available foraging habitat for guillemots and razorbills within UK Western Waters, the increase in density of auks outside OWFs due to displacement is likely to be negligible (MacArthur Green, 2019b). It is considered unlikely that the mortality rate due to displacement would be as high as the empirically estimated 10.5%

annual mortality for adult razorbills that occurs due to the combination of 'natural' factors and anthropogenic activities (Horswill and Robinson, 2015). Indeed, it may be much lower; MacArthur Green (2019b) recommended precautionary rates of 50% displacement for auks and 1% mortality of displaced birds based on a review of available evidence. Modelling undertaken by APEM (2022) found that in the case of Hornsea Project Four, a mortality rate of 1% for displaced auks was considered to be precautionary, and that this may be an overestimate given that Hornsea Project Four is located towards the upper end of the mean maximum foraging range for guillemot from the closest designated sites.

2574. The full ranges of recommended displacement and mortality effects are considered by the assessment, along with evidence-based displacement and mortality rates of 50% and 1%, respectively (APEM, 2022; MacArthur Green, 2019b).

*Table 8.160 Razorbill – predicted operation and maintenance phase displacement and mortality from Lambay Island SPA*

| Mean peak abundance estimate type | Mean peak abundance estimate by season[1] | Number of Lambay Island SPA breeding adults present by season)[2] | Annual mortality range[3] | Annual baseline mortality increase range[4] |
|---|---|---|---|---|
| Upper 95% CI | 605 (b)<br>1,070 (aut)<br>1,297 (win)<br>580 (spr)<br>3,552 (year round) | 150 (b)<br>13 (aut)<br>9 (win)<br>7 (spr)<br>179 (year round) | 1-12 (1) | 0.07-1.62% (0.12%) |
| Mean | 252 (b)<br>694 (aut)<br>651 (win)<br>381 (spr)<br>1,979 (year round) | 62 (b)<br>8 (aut)<br>5 (win)<br>5 (spr)<br>80 (year round) | 0-6 (0) | 0.03-0.72% (0.05%) |
| Lower 95% CI | 21 (b)<br>309 (aut)<br>159 (win)<br>214 (spr)<br>703 (year round) | 5 (b)<br>4 (aut)<br>1 (win)<br>3 (spr)<br>13 (year round) | 0-1 (0) | 0.00-0.11% (0.01%) |

[1] Breeding season = b, autumn migration season = aut, winter season = win, spring migration season = spr
[2] Assumes breeding adult apportioning of 24.5% (breeding season), 5.0% (spring and autumn migration) and 3.6% (winter) to Lambay Island SPA.
[3] Assumes displacement rates of 30-70% and mortality rates of 1-10%. Evidence-based estimates assuming a 50% displacement rate and 1% mortality of displaced birds are presented in parentheses.
[4] Background mortality rate of 10.5% (Horswill and Robinson, 2015)

2575. Based on the mean peak abundances, the annual total of razorbills from the Lambay Island SPA at risk of displacement is 80 birds (**Table 8.160**). At displacement rates of 30% to 70%, and mortality rates of 1% to 10% for displaced birds, 0 to 6 SPA breeding adults would be predicted to die each year due to displacement from the Project.

2576. Assuming a displacement rate of 70% and a mortality rate of 10% of displaced birds, annual mortality within the SPA breeding adult population would increase by 0.72%. Using an evidence-based displacement rate of 50%, and a mortality rate for displaced birds of 1%, annual mortality in the population would instead increase by 0.05% (<1 bird).

2577. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that no detectable changes in mortality rates would occur under almost any combination of displacement and mortality rates when the mean peak abundance estimate assessments are considered.

2578. **It is concluded that predicted razorbill mortality due to operational phase displacement for the project alone would not adversely affect the integrity of the Lambay Island SPA.**

2579. The confidence in the assessment is high for several reasons. Firstly, the evidence used to inform the evidence-based displacement rates is of high applicability and quality (based on the criteria discussed in **Chapter 12 Offshore Ornithology** of the ES). Whilst there was limited available evidence to inform mortality rates, 1% is considered to be sufficiently precautionary based on expert opinion. This species is not regarded as being highly specialised in its habitat requirements (Bradbury *et al*., 2014; Furness and Wade, 2012; Garthe and Hüppop, 2004), and it is therefore anticipated that displaced birds will find alternative habitat in the vast majority of cases. Finally, the conclusion of the assessment is the same irrespective of whether the mean or 95% upper CI mean peak abundances are used to calculate potential mortality and increases in the baseline mortality rate of the background population, provided the evidence-based displacement and mortality rates are used.

*In-combination*

2580. The in-combination assessment for razorbills from Lambay Island SPA has been undertaken in accordance with the approach presented in **Section 8.1**. The total population apportioned to the SPA at risk of displacement is estimated to be 660 breeding adults (**Appendix 12.1** of the ES). Annual in-combination displacement and mortality rates for birds from Lambay Island SPA are presented in **Table 8.161**.

2581. Assuming a displacement rate of 70% and a mortality rate of 10% of displaced birds, 46 breeding adult SPA birds would be lost to displacement annually. Including additional apportioned mortality from the Morlais and Holyhead tidal projects (total 1.24 birds), this would increase the existing mortality within the SPA population (772 breeding adult birds per year) by 6.15%. Using an evidence-based displacement rate of 50%, and a mortality rate for displaced birds of 1%, the annual in-combination mortality would be 3 birds. This would increase the existing mortality within this population by 0.59%. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that detectable changes in mortality rates will not occur due to the level or mortality predicted if the more realistic rates for mortality are used.

2582. **It is concluded that predicted razorbill mortality due to of operational phase displacement impacts of the Project in-combination with other projects, would not adversely affect the integrity of Lambay Island SPA.**

*Table 8.161 In-combination year-round displacement matrix for razorbill from Lambay Island SPA*

| Annual | Mortality | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| **Displacement** | **1%** | **2%** | **3%** | **4%** | **5%** | **10%** | **20%** | **30%** | **50%** | **80%** | **100%** |
| **10%** | 1 | 1 | 2 | 3 | 3 | 7 | 13 | 20 | 33 | 53 | 66 |
| **20%** | 1 | 3 | 4 | 5 | 7 | 13 | 26 | 40 | 66 | 106 | 132 |
| **30%** | **2** | **4** | **6** | **8** | **10** | **20** | 40 | 59 | 99 | 158 | 198 |
| **40%** | **3** | **5** | **8** | **11** | **13** | **26** | 53 | 79 | 132 | 211 | 264 |
| **50%** | **3** | **7** | **10** | **13** | **17** | **33** | 66 | 99 | 165 | 264 | 330 |
| **60%** | **4** | **8** | **12** | **16** | **20** | **40** | 79 | 119 | 198 | 317 | 396 |
| **70%** | **5** | **9** | **14** | **18** | **23** | **46** | 92 | 139 | 231 | 370 | 462 |
| **80%** | 5 | 11 | 16 | 21 | 26 | 53 | 106 | 158 | 264 | 422 | 528 |
| **90%** | 6 | 12 | 18 | 24 | 30 | 59 | 119 | 178 | 297 | 475 | 594 |
| **100%** | 7 | 13 | 20 | 26 | 33 | 66 | 132 | 198 | 330 | 528 | 660 |

Note: The cells show the number of birds predicted to die (rounded to the nearest integer) at a given rate of displacement and mortality. Highlighted cells are considered to be the most realistic scenario, in accordance with SNCB advice (SNCBs, 2022).

#### 8.65.3.3 Puffin

##### Status

2583. The Lambay Island SPA breeding puffin population stood at 265 pairs (or 530 breeding adults) in 1999, and 209 pairs (or 418 breeding adults) in 2004 (NPWS, 2011a). The most recent count is 144 individuals in 2015 (JNCC, 2023a); this is used as the reference population for the assessment.

2584. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.094 (1 – 0.906, Horswill and Robinson (2015), the expected annual mortality from the SPA population would be 14 breeding adults.

##### Functional linkage and seasonal apportionment of potential effects

2585. The mean maximum foraging range of puffin is 137.1km (±128.3km) and the maximum foraging range is 383km (Woodward *et al*., 2019). The Project is located approximately 156km from Lambay Island SPA, which means the Project is beyond the mean maximum foraging range of breeding puffins from the SPA, but within the mean maximum foraging range +1SD, and the maximum foraging range.

2586. The NatureScot apportioning tool (NatureScot, 2018) has been used to estimate the proportion of puffins from each of the relevant SPAs present at the windfarm site during the breeding season. The apportioning to SPA and non-SPA colonies is set out in **Appendix 12.1** of the ES. The tool estimates that 1.1% of adult birds present are likely to originate from Lambay Island SPA.

2587. Outside of the breeding season, breeding puffins, including those from the Lambay Island SPA, are not constrained by requirements to visit nests to incubate eggs or provision chicks. At this time, they are assumed to range more widely and to mix with puffins of all age classes from breeding colonies in Ireland, the UK and further afield. The background population during these seasons is the UK Western Waters BDMPS. This consists of 304,557 individuals during the non-breeding season (August to March).

2588. As no published estimate was available, it is assumed that 18% of Lambay Island SPA breeding adults are present within the UK Western Waters BDMPS during the non-breeding season; the same proportion as 'West Coast UK non-SPA populations' (and other comparable SPAs around the Irish Sea) identified by Furness (2015). It is assumed that the most recent count prior to the publication of Furness (2015) was used to inform the BDMPS total which is 418 breeding adults; 18% of this population is 75 birds. This represents 0.02% of the BDMPS population for this period (304,557). It is therefore assumed that 0.02% of puffins present at the Project site are breeding adults from Lambay Island SPA.

## Potential effects on the qualifying feature

### Operation and maintenance phase disturbance/displacement/barrier effects

*Project-alone*

2589. During the breeding season, the mean peak abundance of puffins present within the windfarm site and 2km buffer was 38.7 (7.7-80.6) individuals (refer to **Appendix 12.1** of the ES). Of these, less than one bird (0.0 (0.0-0.1)) was likely to be a breeding adult from the Lambay Island SPA.

2590. The mean peak abundance of puffins present within the windfarm site and 2km buffer during the non-breeding season was 19.7 (1.9-50.8) individuals. Of these, less than one bird (0.0 (0.0-0.0)) was likely to be a breeding adult from the Lambay Island SPA.

2591. **Table 8.162** sets out the predicted annual impacts on puffins from Lambay Island SPA. This estimates that there would be no measurable increase in mortality from the SPA population, assuming a displacement rate of 30-70% and a mortality of 1-10% for displaced birds.

*Table 8.162 Puffin – predicted operation and maintenance phase displacement and mortality from Lambay Island SPA*

| Mean peak abundance estimate type | Mean peak abundance estimate | Number of Lambay Island SPA breeding adults present [1] | Annual mortality range[2] | Annual baseline mortality increase range[3] |
|---|---|---|---|---|
| Upper 95% CI | 80.6 (breeding)<br>50.8 (non-breeding)<br>131.4 (year round) | 0.92 (breeding)<br>0.01 (non-breeding)<br>0.93 (year round) | 0-0 | 0.02-0.48% |
| Mean | 38.7 (breeding)<br>19.7 (non-breeding)<br>58.4 (year round) | 0.44 (breeding)<br>0.00 (non-breeding)<br>0.44 (year round) | 0-0 | 0.01-0.23% |
| Lower 95% CI | 7.7 (breeding)<br>1.9 (non-breeding)<br>9.5 (year round) | 0.09 (breeding)<br>0.00 (non-breeding)<br>0.09 (year round) | 0-0 | 0.00-0.05% |

[1] Assumes 1.1% of birds present during the breeding season and 0.02% during the non-breeding season are Lambay Island SPA breeding adults
[2] Assumes displacement rates of 30-70% and mortality rates of 1-10%
[3] Background mortality rate of 9.4% (Horswill and Robinson, 2015)

2592. **It is concluded that predicted puffin mortality due to operational phase displacement would not adversely affect the integrity of the Lambay Island SPA.**

2593. The confidence in the assessment is high for several reasons. Firstly, the evidence used to inform the evidence-based displacement rates is of high applicability and quality (based on the criteria discussed in **Chapter 12 Offshore Ornithology** of the ES). Whilst there was limited available evidence to inform mortality rates, 1% is considered to be sufficiently precautionary based on expert opinion. This species is not regarded as being highly specialised in its habitat requirements (Bradbury *et al*., 2014; Furness and Wade, 2012; Garthe and Hüppop, 2004), and it is therefore anticipated that displaced birds will find alternative habitat in the vast majority of cases. Finally, the conclusion of the assessment is the same irrespective of whether the mean or 95% upper CI mean peak abundances are used to calculate potential mortality and increases in the baseline mortality rate of the background population.

*In-combination*

2594. As the Project would have no measurable effect on puffin populations from the Lambay Island SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that predicted puffin mortality due to operational phase displacement due to the Project in-combination with other plans or projects would not adversely affect the integrity of the Lambay Island SPA.**

### 8.65.3.4 Fulmar

#### Status

2595. The Lambay Island SPA breeding fulmar population stood at 585 pairs (or 1,170 breeding adults) in 1999, and 727 pairs (or 1,454 breeding adults) in 2004 (NPWS, 2011a). The most recent count is 375 pairs (AOS), or 750 breeding adults, in 2015 (JNCC, 2023a); this is used as the reference population for the assessment.

2596. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.064 (1 – 0.936, Horswill and Robinson 2015) the expected annual mortality from the SPA population would be 48 breeding adults.

#### Functional linkage and seasonal apportionment of potential effects

2597. The mean maximum foraging range of fulmar is 542.3km (±657.9km) and the maximum foraging range is 2,736km (Woodward *et al*., 2019). The Project is located approximately 156km from Lambay Island SPA, which means that the

Project is within the mean maximum foraging range of fulmars breeding at this SPA.

### Potential effects on the qualifying feature from the Project-alone

2598. Fulmar are considered to have low vulnerability to both collision and displacement impacts from offshore windfarms (Bradbury *et al*., 2014). This species was recorded infrequently within the windfarm site (in only five of 24 surveys, all during the breeding season (January to August; Furness, 2015)) and in low numbers, with densities in four of the five months less than 0.1 birds/km$^2$. A peak density of 0.23 birds/km$^2$ was recorded in May 2022.

2599. Based on the low vulnerability of fulmars to potential impacts, and the low frequency and abundance of birds present at the windfarm site, it is concluded that any effects on breeding populations at Lambay Island SPA are very unlikely, both during and outside of the breeding season.

2600. **It is therefore concluded that there would be no measurable effects on fulmar due to the project alone, and no adverse effect on the integrity of the Lambay Island SPA is predicted.**

### Potential effects on the qualifying feature in-combination with other projects

2601. As the Project would have no measurable effect on fulmar populations from the Lambay Island SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of Lambay Island SPA, when assessed in-combination with other plans or projects.**

#### 8.65.3.5 Lesser black-backed gull

### Status

2602. The Lambay Island SPA breeding lesser black-backed gull population stood at 309 pairs (or 618 breeding adults) in 1999, and 133 pairs (or 266 breeding adults) in 2004 (NPWS, 2011a). The most recent count is 476 pairs (AOT), or 952 breeding adults, in 2010 (JNCC, 2023a); this is used as the reference population for the assessment.

2603. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.115 (1 – 0.885; Horswill and Robinson; 2015), the expected annual mortality from the SPA population would be 109 breeding adults.

### Functional linkage and seasonal apportionment of potential effects

2604. The mean maximum foraging range of lesser black-backed gull is 127km (±109km) and the maximum foraging range is 533km (Woodward *et al*., 2019).

The Project is located approximately 156km from Lambay Island SPA, which means the Project is beyond the mean maximum foraging range of breeding lesser black-backed gulls from the SPA, but within the mean maximum foraging range +1SD, and the maximum foraging range.

2605. The NatureScot apportioning tool (NatureScot, 2018) has been used to estimate the proportion of lesser black-backed gulls from each of the relevant SPAs present at the windfarm site during the breeding season. The apportioning to SPA and non-SPA colonies is set out in **Appendix 12.1** of the ES. 0.39% of impacts at the windfarm site during the breeding season are apportioned to Lambay Island SPA, assuming that only lesser black-backed gulls from coastal colonies are present at the windfarm site (the worst-case scenario, when compared to apportioning both coastal and inland colonies).

2606. Outside the breeding season, breeding lesser black-backed gulls from the SPA are assumed to range widely and to mix with lesser black-backed gulls of all ages from breeding colonies in the UK, Ireland and beyond. The relevant non-breeding season reference population is the UK Western Waters BDMPS, consisting of 163,304 individuals during spring and autumn migration (March and September to October) and 41,159 during winter (November to February) (Furness, 2015).

2607. Furness (2015) estimated that 40% of breeding adults from Ireland colonies are present within the UK Western Waters and Channel BDMPS during the autumn and spring migration periods, and 20% during the winter period. This is equivalent to 381 adults from Lambay Island SPA during the autumn and spring periods, and 190 during winter. This represents 0.23% of the BDMPS population for the autumn and spring periods, and 0.46% during the winter period. Impacts to birds from the SPA during these periods are therefore apportioned accordingly.

## Potential effects on the qualifying feature from the Project-alone

2608. The lesser black-backed gull qualifying feature of the Lambay Island SPA has been screened into the Appropriate Assessment due to the potential risk of collision during the operation and maintenance phase.

### Operation and maintenance phase collision risk

2609. Information for collision risk on breeding adult lesser black-backed gulls belonging to the Lambay Island SPA population is presented in **Table 8.163**. Collision estimates, calculated using Option 2 of the sCRM (McGregor *et al.*, 2018), are presented by biological season. A summary of the annual outputs and the corresponding increase in the annual baseline mortality rate is also presented. Parameters used in the sCRM, together with the avoidance rates applied were agreed with Natural England during the ETG process and are described in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES.

2610. Based on the mean collision rates, the annual total of breeding adult lesser black-backed gulls from Lambay Island SPA at risk of collision as a result of the Project is less than one bird (0.01). This would increase the existing mortality of the SPA breeding population by 0.00%.

*Table 8.163 Predicted seasonal and annual collision mortality (Stochastic model Option 2, avoidance rate 0.994 (±0.0004)), for breeding adult lesser black-backed gulls at the windfarm site, apportioned to Lambay Island SPA, with corresponding increases to baseline mortality of the population*

| Period | Breeding Season Apr-Aug | Autumn Migration Sep-Oct | Non-breeding/winter Nov-Feb | Spring Migration Mar | Annual Jan-Dec |
|---|---|---|---|---|---|
| Total collisions[1] (mean and 95% CIs) | 1.44 (0.00-4.53) | 1.25 (0.00-5.63) | 0.15 (0.00-0.80) | 0.15 (0.00-0.94) | 2.98 (0.00-11.90) |
| % apportioned to the SPA | 0.39% | 0.23% | 0.46% | 0.23% | - |
| Total SPA collisions (mean and 95% CIs) | 0.01 (0.00-0.02) | 0.00 (0.00-0.01) | 0.00 (0.00-0.00) | 0.00 (0.00-0.00) | 0.01 (0.00-0.04%) |
| Mortality increase[2] (mean and 95% CIs) | 0.00% (0.00-0.00%) | 0.00% (0.00-0.00%) | 0.00 (0.00-0.00) | 0.00% (0.00-0.00%) | 0.00% (0.00-0.00%) |

[1] Breeding season collision values reduced to 71.9% of modelled value to reflect proportion of adult birds recorded during site surveys

[2] Assuming predicted annual SPA mortality of 110 birds (952 x 0.115)

2611. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that no detectable changes in mortality rates would occur in this population for the mean monthly collision estimates for the Project, or for the upper 95% CI collision estimate.

2612. **It is concluded that based on predicted lesser black-backed gull mortality due to collision at the windfarm site there is no potential for the Project to have an adverse effect on the integrity of the Lambay Island SPA.**

2613. The confidence in the assessment is high. The evidence used to define the CRM input parameters presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** as part of the ES submitted alongside this RIAA is of high applicability and quality. Whilst there is uncertainty around some of the input parameters (e.g. avoidance rate), the rates selected are considered to be sufficiently precautionary based on expert opinion to provide confidence that collision rates are not underestimated.

### Potential effects on the qualifying feature in-combination with other projects

2614. As the Project would have no measurable effect on lesser black-backed gull populations from the Lambay Island SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of Lambay Island SPA, when assessed in-combination with other plans or projects**.

### 8.65.3.6 Herring gull

### Status

2615. The Lambay Island SPA breeding herring gull population stood at 1,806 pairs (or 3,612 breeding adults) in 1999, and 311 pairs (or 622 breeding adults) in 2004 (NPWS, 2011a). The most recent count is 906 pairs (AON), or 1,812 breeding adults, in 2015 (JNCC, 2023a); this is used as the reference population for the assessment.

2616. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.166 (1 – 0.834; Horswill and Robinson, 2015), the expected annual mortality from the SPA population would be 301 breeding adults.

### Functional linkage and seasonal apportionment of potential effects

2617. The mean maximum foraging range of herring gull is 58.8km (±26.8km) and the maximum foraging range is 92km (Woodward *et al*., 2019). The Project is located approximately 156km from Lambay Island SPA, which means the Project is beyond the maximum foraging range of herring gulls from the SPA.

No impacts during the breeding season from the Project are therefore apportioned to herring gulls breeding at this SPA.

2618. Outside the breeding season, breeding herring gulls from the SPA are assumed to range widely and to mix with herring gulls of all ages from breeding colonies in Ireland, the UK and beyond. The relevant non-breeding season reference population is the UK Western Waters BDMPS, consisting of 173,299 individuals during the non-breeding period (September to February) (Furness, 2015).

2619. Furness (2015) estimated that 30% of the herring gull breeding adults from Ireland colonies are present within the UK Western Waters BDMPS during the non-breeding period, which is the equivalent of 187 birds from the Lambay Island population. This represents 0.11% of the BDMPS population 0.11% of impacts to birds from the SPA are therefore apportioned during the non-breeding season.

## Potential effects on the qualifying feature from the Project-alone

2620. The herring gull qualifying feature of the Lambay Island SPA has been screened into the Appropriate Assessment due to the potential risk of collision during the operation and maintenance phase.

### Operation and maintenance phase collision risk

2621. Information for collision risk on breeding adult herring gulls belonging to the Lambay Island SPA population is presented in **Table 8.164**. Collision estimates, calculated using the sCRM, are presented by biological season. A summary of the annual outputs and the corresponding increase in the annual baseline mortality rate is also presented. Parameters used in the sCRM were agreed with Natural England during the ETG process and are in described **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES.

2622. Based on the mean collision rates, the annual total of breeding adult herring gulls from Lambay Island SPA at risk of collision as a result of the Project is less than one bird (0.00). This would increase the existing mortality of the SPA breeding population by 0.00%.

*Table 8.164 Predicted seasonal and annual collision mortality (Stochastic model Option 2, avoidance rate 0.994 (±0.0004)), for breeding adult herring gulls at the windfarm site, apportioned to Lambay Island SPA, with corresponding increases to baseline mortality of the population*

| Period | Breeding Season Apr-Aug | Autumn Migration Sep-Oct | Non-breeding/winter Nov-Feb | Spring Migration Mar | Annual Jan-Dec |
|---|---|---|---|---|---|
| Total collisions[1] (mean and 95% CIs) | 0.85 (0.00-7.72) | - | 2.38 (0.00-9.70) | - | 3.23 (0.00-13.41) |
| % apportioned to the SPA | 0.00% | - | 0.11% | - | - |
| Total SPA collisions (mean and 95% CIs) | 0.00 (0.00-0.00) | - | 0.00 (0.00-0.01) | - | 0.00 (0.00-0.01) |
| Mortality increase[2] (mean and 95% CIs) | 0.00% (0.00-0.00%) | - | 0.00% (0.00-0.00%) | - | 0.00% (0.00-0.00%) |
| [1] Breeding season collision values reduced to 48.0% of modelled value to reflect proportion of adult birds recorded during site surveys<br>[2] Assuming predicted annual SPA mortality of 301 birds (1,812 x 0.166) | | | | | |

2623. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that no detectable changes in mortality rates would occur on this population for the mean monthly collision estimates for the Project, or for the upper 95% CI collision estimate.

2624. **It is concluded that based on predicted herring gull mortality due to collision at the windfarm site there is no potential for the Project to have an adverse effect on the integrity of the Lambay Island SPA.**

2625. The confidence in the assessment is high. The evidence used to define the CRM input parameters presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** as part of the ES submitted alongside this RIAA is of high applicability and quality. Whilst there is uncertainty around some of the input parameters (e.g. avoidance rate), the rates selected are considered to be sufficiently precautionary based on expert opinion to provide confidence that collision rates are not underestimated.

### Potential effects on the qualifying feature in-combination with other projects

2626. As the Project would have no measurable effect on herring gull populations from the Lambay Island SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of Lambay Island SPA, when assessed in-combination with other plans or projects**.

### 8.65.3.7 Kittiwake

### Status

2627. The Lambay Island SPA breeding kittiwake population stood at 4,091 pairs (or 8,182 breeding adults) in 1999, and 3,947 pairs (or 7,894 breeding adults) in 2004 (NPWS, 2011a). The most recent count is 3,320 pairs (AON), or 6,640 breeding adults, in 2015 (JNCC, 2023a); this is used as the reference population for the assessment.

2628. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.146 (1 – 0.854; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 969 breeding adults.

### Functional linkage and seasonal apportionment of potential effects

2629. The mean maximum foraging range of kittiwake is 156.1km (±144.5km) and the maximum foraging range is 770km (Woodward *et al*., 2019). The Project is located approximately 156km from Lambay Island SPA, which means the Project is beyond the mean maximum foraging range of breeding kittiwakes

from the SPA, but within the mean maximum foraging range +1SD, and the maximum foraging range.

2630. The NatureScot apportioning tool (NatureScot, 2018) has been used to estimate the proportion of kittiwakes from each of the relevant SPAs present at the windfarm site during the breeding season. The apportioning to SPA and non-SPA colonies is set out in **Appendix 12.1** of the ES. 2.32% of impacts at the windfarm site during the breeding season are apportioned to Lambay Island SPA.

2631. Outside the breeding season, breeding kittiwakes from the SPA are assumed to range widely and to mix with kittiwakes of all ages from breeding colonies in the UK, Ireland and beyond. The relevant non-breeding season reference population is the UK Western Waters and Channel BDMPS, consisting of 911,586 individuals during autumn migration (September to December) and 627,816 during spring migration (January to February) (Furness, 2015).

2632. Furness (2015) estimated that 30% of breeding adults from Ireland colonies are present within the UK Western Waters and Channel BDMPS during the autumn and spring migration periods. This proportion of Lambay Island SPA birds is therefore assumed, which is 1,992 birds. This represents 0.22% and 0.32% of the BDMPS population for the autumn and spring periods respectively. During autumn migration and spring migration, 0.22%, and 0.32% of impacts are therefore considered to affect birds from the SPA.

## Potential effects on the qualifying feature from the Project-alone

2633. The kittiwake qualifying feature of the Lambay Island SPA has been screened into the Appropriate Assessment due to the potential risk of collision.

### Operation and maintenance phase collision risk

2634. Information for collision risk on breeding adult kittiwakes belonging to the Lambay Island SPA population is presented in **Table 8.165**. Collision estimates, calculated using the sCRM, are presented by biological season. A summary of the annual outputs and the corresponding increase in the annual baseline mortality rate is also presented. Parameters used in the sCRM were agreed with Natural England during the ETG process and are described in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES.

2635. Based on the mean collision rates, the annual total of breeding adult kittiwakes from Lambay Island SPA at risk of collision as a result of the Project is less than one bird (0.38). This would increase the existing mortality of the SPA breeding population by 0.04%.

*Table 8.165 Predicted seasonal and annual collision mortality (Stochastic model Option 2, avoidance rate 0.993 (±0.0003)), for breeding adult kittiwakes at the windfarm site, apportioned to Lambay Island SPA, with corresponding increases to baseline mortality of the population*

| | Breeding Season | Autumn Migration | Non-breeding/winter | Spring Migration | Annual |
|---|---|---|---|---|---|
| **Period** | **Mar-Aug** | **Sep-Dec** | **-** | **Jan-Feb** | **Jan-Dec** |
| Total collisions[1] (mean and 95% CIs) | 15.36 (4.17-33.87) | 8.49 (2.20-18.85) | - | 0.63 (0.00-1.46) | 24.48 (6.37-54.17) |
| % apportioned to the SPA | 2.35% | 0.22% | - | 0.32% | - |
| Total SPA collisions (mean and 95% CIs) | 0.36 (0.10-0.79) | 0.02 (0.00-0.04) | - | 0.00 (0.00-0.00) | 0.38 (0.10-0.83) |
| Mortality increase[2] (mean and 95% CIs) | 0.04% (0.01-0.08%) | 0.00% (0.00-0.00%) | - | 0.00% (0.00-0.00%) | 0.04% (0.01-0.09%) |

[1] Breeding season collision values reduced to 94.1% of modelled value to reflect proportion of adult birds recorded during site surveys
[2] Assuming predicted annual SPA mortality of 969 birds (6,640 x 0.146)

2636. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that no detectable changes in mortality rates would occur on this population for the mean monthly collision estimates for the Project, or for the upper 95% CI collision estimate.

2637. **It is concluded that based on predicted kittiwake mortality due to collision at the windfarm site there is no potential for the Project to have an adverse effect on the integrity of the Lambay Island SPA.**

2638. The confidence in the assessment is high. The evidence used to define the CRM input parameters presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** as part of the ES submitted alongside this RIAA is of high applicability and quality. Whilst there is uncertainty around some of the input parameters (e.g. avoidance rate), the rates selected are considered to be sufficiently precautionary based on expert opinion to provide confidence that collision rates are not underestimated.

### Potential effects on the qualifying feature in-combination with other projects

2639. As the Project would have no measurable effect on kittiwake populations from the SPA, there would be no contribution to any in-combination effects on this feature. Therefore, it is concluded that there would be **no adverse effect on the integrity of Lambay Island SPA, when assessed in-combination with other plans or projects**.

### 8.65.3.8 Shag

### Status

2640. The Lambay Island SPA breeding shag population stood at 1,122 pairs (or 2,244 breeding adults) in 1999, and 1,734 pairs (or 3,468 breeding adults) in 2004 (NPWS, 2011a). The most recent count is 469 pairs, or 938 breeding adults, in 2015 (JNCC, 2023a); this is used as the reference population for the assessment.

2641. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.142 (1 – 0.858; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 133 breeding adults.

### Functional linkage and seasonal apportionment of potential effects

2642. The mean maximum foraging range of shag is 13.2km (±10.5km) and the maximum foraging range is 46km (Woodward *et al*., 2019). The Project is located approximately 156km from Lambay Island SPA, which means that the Project is beyond the maximum foraging range for shags from the SPA. No impacts during the breeding season from the Project are therefore apportioned to shags breeding at this SPA.

2643. Outside the breeding season, breeding shags from the SPA are not tied to the colony and therefore have the potential to mix with birds of all ages from breeding colonies in the UK and beyond. However, Furness (2015) stated that adult shags showed only limited migration, presenting evidence to suggest that the majority of adults moved less than 50km from their breeding colony. Furthermore, Furness (2015) stated that only 3% of immature birds, and no adult birds, from Irish colonies were present within the UK Wales & SW England waters BDMPS during the non-breeding season. Given the distance of the windfarm site from the SPA (i.e. c.156km) and the low numbers of birds recorded within the Project area (mean peak density 0.02 birds/km$^2$ / <4 birds within the windfarm site and 2km buffer during the non-breeding period), it is concluded that it is very unlikely that breeding adult shags from the Lambay Island SPA will occur at the windfarm site. Accordingly, no impacts during the non-breeding season from the Project are apportioned to shags breeding at this SPA.

### Potential effects on the qualifying feature from the Project-alone

2644. No effects on shags from Lambay Island SPA are predicted. **Therefore, it is concluded that there would be no adverse effect on the integrity of the Lambay Island SPA for the Project-alone**.

### Potential effects on the qualifying feature in-combination with other projects

2645. As the Project would have no measurable effect on shag populations from the Lambay Island SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of Lambay Island SPA, when assessed in-combination with other plans or projects**.

### 8.65.3.9 Cormorant

### Status

2646. The Lambay Island SPA breeding cormorant population stood at 675 pairs (or 1,350 breeding adults) in 1999, and 352 pairs (or 704 breeding adults) in 2004 (NPWS, 2011a). The most recent count was 316 pairs (AON), or 632 breeding adults, in 2015 (JNCC, 2023a); this is used as the reference population for the assessment.

2647. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.132 (1 – 0.868; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 83 breeding adults.

### Functional linkage and seasonal apportionment of potential effects

2648. The mean maximum foraging range of cormorant is 25.6km (±8.3km) and the maximum foraging range is 35km (Woodward *et al*., 2019). The Project is located approximately 156km from Lambay Island SPA, which means the Project is beyond the maximum foraging range of cormorants from the SPA. No impacts during the breeding season from the Project are therefore apportioned to cormorants breeding at this SPA.

2649. Outside the breeding season, breeding cormorants from the SPA are assumed to range widely and to mix with birds of all ages from breeding colonies in Ireland, the UK and beyond. However, as no cormorants were recorded within the windfarm site or 2km buffer, it can be concluded that no birds from Lambay Island SPA are likely to occur at the windfarm site.

### Potential effects on the qualifying feature from the Project-alone

2650. No effects on cormorants from Lambay Island SPA are predicted. **Therefore, it is concluded that there would be no adverse effect on the integrity of the Lambay Island SPA for the Project-alone**.

### Potential effects on the qualifying feature in-combination with other projects

2651. As the Project would have no measurable effect on cormorant populations from the Lambay Island SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of Lambay Island SPA, when assessed in-combination with other plans or projects**.

## 8.66 Howth Head Coast SPA (transboundary site)

2652. Howth Head Coast SPA is located on the east coast of Ireland approximately 159km from the windfarm site.

### 8.66.1 Description of designation

2653. Howth Head Coast SPA is a rocky headland situated on the northern side of Dublin Bay. The site comprises the sea cliffs extending from just east of the Nose of Howth to the tip of the Bailey Lighthouse peninsula. The cliffs vary from between about 60m and 90m in height, and in places comprise fairly sheer, exposed rock face. The marine area to a distance of 500m from the cliff base is included within the site. A range of seabird species breed within the SPA, including a nationally important population of kittiwake.

### 8.66.2 Conservation objectives

2654. The conservation objective of the SPA is 'to maintain or restore the favourable conservation condition of the bird species listed as Special Conservation Interests for this SPA.'

### 8.66.3 Assessment

2655. One qualifying feature of Howth Head Coast SPA has been screened into the Appropriate Assessment (**Table 5.2**): kittiwake.

#### 8.66.3.1 Kittiwake

##### Status

2656. The Howth Head Coast SPA breeding kittiwake population stood at 2,269 pairs, or 4,538 breeding adults, in 1999 (NPWS, 2011b). The most recent count is 3,081 pairs (AON), or 6,162 breeding adults, in 2015 (JNCC, 2023a); this is used as the reference population for the assessment.

2657. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.146 (1 – 0.854; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 900 breeding adults.

##### Functional linkage and seasonal apportionment of potential effects

2658. The mean maximum foraging range of kittiwake is 156.1km (±144.5km) and the maximum foraging range is 770km (Woodward *et al*., 2019). The Project is located approximately 159km from Howth Head Coast SPA, which means the Project is beyond the mean maximum foraging range of breeding kittiwakes from the SPA, but within the mean maximum foraging range +1SD, and the maximum foraging range.

2659. The NatureScot apportioning tool (NatureScot, 2018) has been used to estimate the proportion of kittiwakes from each of the relevant SPAs present at the windfarm site during the breeding season. The apportioning to SPA and non-SPA colonies is set out in **Appendix 12.1** of the ES. 2.38% of impacts at the windfarm site during the breeding season are apportioned to Howth Head Coast SPA.

2660. Outside the breeding season, breeding kittiwakes from the SPA are assumed to range widely and to mix with kittiwakes of all ages from breeding colonies in the UK, Ireland and beyond. The relevant non-breeding season reference population is the UK Western Waters and Channel BDMPS, consisting of 911,586 individuals during autumn migration (September to December) and 627,816 during spring migration (January to February) (Furness, 2015).

2661. Furness (2015) estimated that 30% of breeding adults from Ireland colonies are present within the UK Western Waters and Channel BDMPS during the autumn and spring migration periods. This proportion of Howth Head Coast SPA birds is therefore assumed, which is 1,849 birds. This represents 0.20% and 0.29% of the BDMPS population for the autumn and spring periods respectively. During autumn migration and spring migration, 0.20%, and 0.29% of impacts are therefore considered to affect birds from the SPA.

## Potential effects on the qualifying feature from the Project-alone

2662. The kittiwake qualifying feature of the Howth Head Coast SPA has been screened into the Appropriate Assessment due to the potential risk of collision.

### Operation and maintenance phase collision risk

2663. Information for collision risk on breeding adult kittiwakes belonging to the Howth Head Coast SPA population is presented in **Table 8.166**. Collision estimates, calculated using the sCRM, are presented by biological season. A summary of the annual outputs and the corresponding increase in the annual baseline mortality rate is also presented. Parameters used in the sCRM were agreed with Natural England during the ETG process and are described in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES.

2664. Based on the mean collision rates, the annual total of breeding adult kittiwakes from Howth Head Coast SPA at risk of collision as a result of the Project is less than one bird (0.38). This would increase the existing mortality of the SPA breeding population by 0.04%.

*Table 8.166 Predicted seasonal and annual collision mortality (Stochastic model Option 2, avoidance rate 0.993 (±0.0003)), for breeding adult kittiwakes at the windfarm site, apportioned to Howth Head Coast SPA, with corresponding increases to baseline mortality of the population*

| Period | Breeding Season<br>Mar-Aug | Autumn Migration<br>Sep-Dec | Non-breeding/winter<br>- | Spring Migration<br>Jan-Feb | Annual<br>Jan-Dec |
|---|---|---|---|---|---|
| Total collisions[1] (mean and 95% CIs) | 15.36<br>(4.17-33.87) | 8.49<br>(2.20-18.85) | - | 0.63<br>(0.00-1.46) | 24.48<br>(6.37-54.17) |
| % apportioned to the SPA | 2.38% | 0.20% | - | 0.29% | - |
| Total SPA collisions (mean and 95% CIs) | 0.37<br>(0.10-0.81) | 0.02<br>(0.00-0.04) | - | 0.00<br>(0.00-0.00) | 0.38<br>(0.10-0.85) |
| Mortality increase[2] (mean and 95% CIs) | 0.04%<br>(0.01-0.09%) | 0.00%<br>(0.00-0.00%) | - | 0.00%<br>(0.00-0.00%) | 0.04%<br>(0.01-0.09%) |

[1] Breeding season collision values reduced to 94.1% of modelled value to reflect proportion of adult birds recorded during site surveys

[2] Assuming predicted annual SPA mortality of 900 birds (6,162 x 0.146)

2665. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that no detectable changes in mortality rates would occur on this population for the mean monthly collision estimates for the Project, or for the upper 95% CI collision estimate.

2666. **It is concluded that based on predicted kittiwake mortality due to collision at the windfarm site there is no potential for the Project to have an adverse effect on the integrity of the Howth Head Coast SPA.**

2667. The confidence in the assessment is high. The evidence used to define the CRM input parameters presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** as part of the ES submitted alongside this RIAA is of high applicability and quality. Whilst there is uncertainty around some of the input parameters (e.g. avoidance rate), the rates selected are considered to be sufficiently precautionary based on expert opinion to provide confidence that collision rates are not underestimated.

### Potential effects on the qualifying feature in-combination with other projects

2668. As the Project would have no measurable effect on kittiwake populations from the SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of Howth Head Coast SPA, when assessed in-combination with other plans or projects**.

## 8.67 Ireland's Eye SPA (transboundary site)

2669. Ireland's Eye SPA is located on the east coast of Ireland approximately 159km from the windfarm site.

### 8.67.1 Description of designation

2670. Ireland's Eye is an uninhabited island located about 1.5 km north of Howth in Co. Dublin. The SPA encompasses Ireland's Eye, Rowan Rocks, Thulla, Thulla Rocks, Carrageen Bay and a seaward extension of 200m in the west and 500m to the north and east. The island has an area of c. 24 ha above the high tide mark. Ireland's Eye SPA supports important populations of breeding seabirds.

### 8.67.2 Conservation objectives

2671. The conservation objective of the SPA is 'to maintain or restore the favourable conservation condition of the bird species listed as Special Conservation Interests for this SPA.'

### 8.67.3 Assessment

2672. Three qualifying features of Ireland's Eye SPA have been screened into the Appropriate Assessment (**Table 5.2**): kittiwake, razorbill, and cormorant.

#### 8.67.3.1 Kittiwake

##### Status

2673. The Ireland's Eye SPA breeding kittiwake population stood at 941 pairs (or 1,882 breeding adults) in 1999, and 1,024 pairs (or 2,048 breeding adults) in 2001 (NPWS, 2011c). The most recent count is 1,610 pairs (AON), or 3,220 breeding adults, in 2015 (JNCC, 2023a); this is used as the reference population for the assessment.

2674. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.146 (1 – 0.854; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 470 breeding adults.

##### Functional linkage and seasonal apportionment of potential effects

2675. The mean maximum foraging range of kittiwake is 156.1km (±144.5km) and the maximum foraging range is 770km (Woodward *et al*., 2019). The Project is located approximately 159km from Ireland's Eye SPA, which means the Project is beyond the mean maximum foraging range of breeding kittiwakes

from the SPA, but within the mean maximum foraging range +1SD, and the maximum foraging range.

2676. The NatureScot apportioning tool (NatureScot, 2018) has been used to estimate the proportion of kittiwakes from each of the relevant SPAs present at the windfarm site during the breeding season. The apportioning to SPA and non-SPA colonies is set out in **Appendix 12.1** of the ES. 1.04% of impacts at the windfarm site during the breeding season are apportioned to Ireland's Eye SPA.

2677. Outside the breeding season, breeding kittiwakes from the SPA are assumed to range widely and to mix with kittiwakes of all ages from breeding colonies in the UK, Ireland and beyond. The relevant non-breeding season reference population is the UK Western Waters and Channel BDMPS, consisting of 911,586 individuals during autumn migration (September to December) and 627,816 during spring migration (January to February) (Furness, 2015).

2678. Furness (2015) estimated that 30% of breeding adults from Ireland colonies are present within the UK Western Waters and Channel BDMPS during the autumn and spring migration periods. This proportion of Ireland's Eye SPA birds is therefore assumed, which is 966 birds. This represents 0.11% and 0.15% of the BDMPS population for the autumn and spring periods respectively. During autumn migration and spring migration, 0.11%, and 0.15% of impacts are therefore considered to affect birds from the SPA.

## Potential effects on the qualifying feature from the Project-alone

2679. The kittiwake qualifying feature of the Ireland's Eye SPA has been screened into the Appropriate Assessment due to the potential risk of collision.

### Operation and maintenance phase collision risk

2680. Information for collision risk on breeding adult kittiwakes belonging to the Ireland's Eye SPA population is presented in **Table 8.167**. Collision estimates, calculated using the sCRM, are presented by biological season. A summary of the annual outputs and the corresponding increase in the annual baseline mortality rate is also presented. Parameters used in the sCRM were agreed with Natural England during the ETG process and are described in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES.

2681. Based on the mean collision rates, the annual total of breeding adult kittiwakes from Ireland's Eye SPA at risk of collision as a result of the Project is less than one bird (0.17). This would increase the existing mortality of the SPA breeding population by 0.04%.

*Table 8.167 Predicted seasonal and annual collision mortality (Stochastic model Option 2, avoidance rate 0.993 (±0.0003)), for breeding adult kittiwakes at the windfarm site, apportioned to Ireland's Eye SPA, with corresponding increases to baseline mortality of the population*

| | Breeding Season | Autumn Migration | Non-breeding/winter | Spring Migration | Annual |
|---|---|---|---|---|---|
| **Period** | **Mar-Aug** | **Sep-Dec** | **-** | **Jan-Feb** | **Jan-Dec** |
| Total collisions[1] (mean and 95% CIs) | 15.36 (4.17-33.87) | 8.49 (2.20-18.85) | - | 0.63 (0.00-1.46) | 24.48 (6.37-54.17) |
| % apportioned to the SPA | 1.04% | 0.11% | - | 0.15% | - |
| Total SPA collisions (mean and 95% CIs) | 0.16 (0.04-0.35) | 0.01 (0.00-0.02) | - | 0.00 (0.00-0.00) | 0.17 (0.05-0.37) |
| Mortality increase[2] (mean and 95% CIs) | 0.03% (0.01-0.07%) | 0.00% (0.00-0.00%) | - | 0.00% (0.00-0.00%) | 0.04% (0.01-0.08%) |

[1] Breeding season collision values reduced to 94.1% of modelled value to reflect proportion of adult birds recorded during site surveys
[2] Assuming predicted annual SPA mortality of 470 birds (3,220 x 0.146)

2682. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that no detectable changes in mortality rates would occur on this population for the mean monthly collision estimates for the Project, or for the upper 95% CI collision estimate.

2683. **It is concluded that based on predicted kittiwake mortality due to collision at the windfarm site there is no potential for the Project to have an adverse effect on the integrity of the Ireland's Eye SPA.**

2684. The confidence in the assessment is high. The evidence used to define the CRM input parameters presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** as part of the ES submitted alongside this RIAA is of high applicability and quality. Whilst there is uncertainty around some of the input parameters (e.g. avoidance rate), the rates selected are considered to be sufficiently precautionary based on expert opinion to provide confidence that collision rates are not underestimated.

### Potential effects on the qualifying feature in-combination with other projects

2685. As the Project would have no measurable effect on kittiwake populations from the SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of Ireland's Eye SPA, when assessed in-combination with other plans or projects**.

### 8.67.3.2 Razorbill

### Status

2686. The Ireland's Eye SPA breeding razorbill population stood at 350 pairs (or 700 breeding adults) in 1999, and 460 pairs (or 920 breeding adults) in 2001 (NPWS, 2011c). The most recent count was 1,600 individuals in 2015 (JNCC, 2023a); this is used as the reference population for the assessment.

2687. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.105 (1 – 0.895; Horswill and Robinson, 2015), the expected annual mortality from the SPA population would be 168 breeding adults.

### Functional linkage and seasonal apportionment of potential effects

2688. The mean maximum foraging range of razorbill is 88.7km (±75.9km) and the maximum foraging range is 313km (Woodward *et al*., 2019). The Project is located approximately 159km from Ireland's Eye SPA, which means the Project is beyond the mean maximum foraging range, on the outer limit of mean maximum +1SD (164.6km), but within the maximum foraging range of razorbills breeding at this SPA. The maximum foraging range is a poor indicator of typical foraging behaviour, and it would be expected that few birds or foraging trips will occur at this distance from the colony, and even fewer with any regularity. No impacts during the breeding season are therefore apportioned to birds breeding at this colony.

2689. Outside the breeding season, breeding razorbills from the SPA are assumed to range widely and to mix with razorbills of all ages from breeding colonies in Ireland, the UK and beyond. The relevant non-breeding season reference population is the UK Western Waters BDMPS, consisting of 606,914 individuals during autumn and spring passage periods (August to October and January to March), and 341,422 individuals during winter (November and December) (Furness, 2015).

2690. As no published estimate was available, it is assumed that 98% of Ireland's Eye SPA breeding adults are present within the UK Western Waters BDMPS during the non-breeding season, and 30% during the winter period; the same proportions as 'West Coast UK non-SPA populations' (and other comparable SPAs around the Irish Sea) identified by Furness (2015). It is assumed that the most recent count prior to the publication of Furness (2015) was used to inform the BDMPS total which is 920 breeding adults; 98% of this population is 902 birds, and 30% is 276 birds. This represents 0.1% and 0.1% of the BDMPS population (606,914 during spring/autumn and 341,422 during winter) respectively. These percentages (i.e. 0.1% (spring and autumn migration) and 0.1% (winter)) are the proportions of birds present at the windfarm site that are presumed to originate from the Ireland's Eye SPA during the relevant seasons.

## Potential effects on the qualifying feature

### Operation and maintenance phase disturbance/ displacement/barrier effects

*Project-alone*

2691. The year-round mean peak abundance of razorbills present within the windfarm site and 2km buffer was 1,979 (703-3,552) individuals (refer to **Appendix 12.1** of the ES). Of these, 2 birds (1-3) were likely to be breeding adults from the Ireland's Eye SPA.

2692. **Table 8.168** sets out the predicted impacts on razorbills from Ireland's Eye SPA during the non-breeding season. Displacement rates of 30% to 70% are considered for this species, along with a range of mortality rates of 1% to 10% of displaced birds (UK SNCBs, 2017).

2693. The available evidence suggests that the upper ranges of these displacement and mortality rates may be excessively precautionary. Whilst it is true that guillemots and razorbills tend to be displaced by OWFs they do not avoid them completely, and displacement rates vary between sites. MacArthur Green

(2019b) concluded that, on average, densities within OWFs were around half of the density found in the habitats around the OWF. At some OWFs there was also displacement of birds from a buffer zone surrounding it. The size of the buffer zone varied between OWFs and was generally less than 2km, with auk density increasing across the buffer zone as distance from turbines increased, up to the density in the wider area. Another recent review (APEM, 2022) found that the current displacement rates suggested by the SNCBs for guillemot and razorbill (30% to 70%) did not account for the quality of or confidence in the studies which were used to inform this position, and that studies where no significant effects were recorded were not accounted for during the provision of the advice. APEM (2022) suggested that in the case of Hornsea Project Four, an evidence-based displacement rate of 50% was appropriate. However, the study also recognised that larger displacement effects were possible.

2694. Mortality due to displacement could arise due to increased energy costs and/or decreased energy intake, if displacement results in increased flying time to avoid OWFs, and/or increased bird densities and competition for prey in areas of unimpacted habitat outside the OWF. Given that UK OWFs in the Irish and Celtic Seas represent a very small proportion of the available foraging habitat for guillemots and razorbills within UK Western Waters, the increase in density of auks outside OWFs due to displacement is likely to be negligible (MacArthur Green, 2019b). It is considered unlikely that the mortality rate due to displacement would be as high as the empirically estimated 10.5% annual mortality for adult razorbills that occurs due to the combination of 'natural' factors and anthropogenic activities (Horswill and Robinson, 2015). Indeed, it may be much lower; MacArthur Green (2019b) recommended precautionary rates of 50% displacement for auks and 1% mortality of displaced birds based on a review of available evidence. Modelling undertaken by APEM (2022) found that in the case of Hornsea Project Four, a mortality rate of 1% for displaced auks was considered to be precautionary, and that this may be an overestimate given that Hornsea Project Four is located towards the upper end of the mean maximum foraging range for guillemot from the closest designated sites.

2695. The full ranges of recommended displacement and mortality effects are considered by the assessment, along with evidence-based displacement and mortality rates of 50% and 1%, respectively (APEM, 2022; MacArthur Green, 2019b).

*Table 8.168 Razorbill – predicted operation and maintenance phase displacement and mortality from Ireland's Eye SPA*

| Mean peak abundance estimate type | Mean peak abundance estimate by season[1] | Number of Ireland's Eye SPA breeding adults present by season)[2] | Annual mortality range[3] | Annual baseline mortality increase range[4] |
|---|---|---|---|---|
| Upper 95% CI | 605 (b)<br>1,070 (aut)<br>1,297 (win)<br>580 (spr)<br>3,552 (year round) | 0 (b)<br>1 (aut)<br>1 (win)<br>1 (spr)<br>3 (year round) | 0-0 (0) | 0.01-0.14% (0.01%) |
| Mean | 252 (b)<br>694 (aut)<br>651 (win)<br>381 (spr)<br>1,979 (year round) | 0 (b)<br>1 (aut)<br>1 (win)<br>1 (spr)<br>2 (year round) | 0-0 (0) | 0.00-0.08% (0.01%) |
| Lower 95% CI | 21 (b)<br>309 (aut)<br>159 (win)<br>214 (spr)<br>703 (year round) | 0 (b)<br>0 (aut)<br>0 (win)<br>0 (spr)<br>1 (year round) | 0-0 (0) | 0.00-0.04% (0.00%) |

[1] Breeding season = b, autumn migration season = aut, winter season = win, spring migration season = spr

[2] Assumes breeding adult apportioning of 0.0% (breeding season), 0.1% (spring and autumn migration) and 0.1% (winter) to Ireland's Eye SPA.

[3] Assumes displacement rates of 30-70% and mortality rates of 1-10%. Evidence-based estimates assuming a 50% displacement rate and 1% mortality of displaced birds are presented in parentheses.

[4] Background mortality rate of 10.5% (Horswill and Robinson, 2015)

2696. Based on the mean peak abundances, the annual total of razorbills from the Ireland's Eye SPA at risk of displacement is 2 birds (**Table 8.168**). At displacement rates of 30% to 70%, and mortality rates of 1% to 10% for displaced birds, <1 (0-0) SPA breeding adults would be predicted to die each year due to displacement from the Project.

2697. Assuming a displacement rate of 70% and a mortality rate of 10% of displaced birds, annual mortality within the SPA breeding adult population would increase by 0.08%. Using an evidence-based displacement rate of 50%, and a mortality rate for displaced birds of 1%, annual mortality in the population would instead increase by 0.01% (<1 bird).

2698. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that no detectable changes in mortality rates would occur under almost any combination of displacement and mortality rates when the mean peak abundance estimate assessments are considered. This would be the case even when upper 95% CIs are considered.

2699. **It is concluded that predicted razorbill mortality due to operational phase displacement for the project alone would not adversely affect the integrity of the Ireland's Eye SPA.**

2700. The confidence in the assessment is high for several reasons. Firstly, the evidence used to inform the evidence-based displacement rates is of high applicability and quality (based on the criteria discussed in **Chapter 12 Offshore Ornithology** of the ES). Whilst there was limited available evidence to inform mortality rates, 1% is considered to be sufficiently precautionary based on expert opinion. This species is not regarded as being highly specialised in its habitat requirements (Bradbury *et al*., 2014; Furness and Wade, 2012; Garthe and Hüppop, 2004), and it is therefore anticipated that displaced birds will find alternative habitat in the vast majority of cases. Finally, the conclusion of the assessment is the same irrespective of whether the mean or 95% upper CI mean peak abundances are used to calculate potential mortality and increases in the baseline mortality rate of the background population, provided the evidence-based displacement and mortality rates are used.

*In-combination*

2701. As there would be no measurable increase in mortality on razorbill populations from Ireland's Eye SPA as a result of the Project, there would be no contribution to potential in-combination effects. **Therefore, it is concluded that there would be no adverse effect on the integrity of Ireland's Eye SPA, when assessed in-combination with other plans or projects.**

#### 8.67.3.3 Cormorant

#### Status

2702. The Ireland's Eye SPA breeding cormorant population stood at 306 pairs (or 612 breeding adults) in 1999, and 438 pairs (or 876 breeding adults) in 2001 (NPWS, 2011c). The most recent count was 424 pairs (AON), or 848 breeding adults, in 2015 (JNCC, 2023a); this is used as the reference population for the assessment.

2703. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.132 (1 – 0.868; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 56 breeding adults.

### Functional linkage and seasonal apportionment of potential effects

2704. The mean maximum foraging range of cormorant is 25.6km (±8.3km) and the maximum foraging range is 35km (Woodward *et al*., 2019). The Project is located approximately 159km from Ireland's Eye SPA, which means the Project is beyond the maximum foraging range of cormorants from the SPA. No impacts during the breeding season from the Project are therefore apportioned to cormorants breeding at this SPA.

2705. Outside the breeding season, breeding cormorants from the SPA are assumed to range widely and to mix with birds of all ages from breeding colonies in Ireland, the UK and beyond. However, as no cormorants were recorded within the windfarm site or 2km buffer, it can be concluded that no birds from Ireland's Eye SPA are likely to occur at the windfarm site.

### Potential effects on the qualifying feature from the Project-alone

2706. No effects on cormorants from Ireland's Eye SPA are predicted. **Therefore, it is concluded that there would be no adverse effect on the integrity of the Ireland's Eye SPA for the Project-alone**.

### Potential effects on the qualifying feature in-combination with other projects

2707. As the Project would have no measurable effect on cormorant populations from the Ireland's Eye SPA, there would be no contribution to any in-combination effects on this feature**. Therefore, it is concluded that there would be no adverse effect on the integrity of Ireland's Eye SPA, when assessed in-combination with other plans or projects**.

## 8.68 Wicklow Head SPA (transboundary site)

2708. Wicklow Head SPA is located on the east coast of Ireland approximately 176km from the windfarm site.

### 8.68.1 Description of designation

2709. Wicklow Head is a rocky headland situated approximately 3km south of Wicklow town. The cliffs are highest immediately south of the lighthouse where they rise to about 60m and it is here that most of the seabirds breed. The SPA comprises the cliffs and cliff-top vegetation, as well as some heath vegetation. The marine area to a distance of 500 m from the base of the cliffs is included in the SPA.

### 8.68.2 Conservation objectives

2710. The conservation objective of the SPA is 'to maintain or restore the favourable conservation condition of the bird species listed as Special Conservation Interests for this SPA.'

### 8.68.3 Assessment

2711. One qualifying feature of Wicklow Head SPA has been screened into the Appropriate Assessment (**Table 5.2**): kittiwake.

#### 8.68.3.1 Kittiwake

##### Status

2712. The Wicklow Head SPA breeding kittiwake population stood at 956 pairs, or 1,912 breeding adults, in 2002 (NPWS, 2012a). The most recent count is 674 pairs (AON), or 1,348 breeding adults, in 2022 (JNCC, 2023a); this is used as the reference population for the assessment.

2713. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.146 (1 – 0.854; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 197 breeding adults.

##### Functional linkage and seasonal apportionment of potential effects

2714. The mean maximum foraging range of kittiwake is 156.1km (±144.5km) and the maximum foraging range is 770km (Woodward *et al*., 2019). The Project is located approximately 176km from Wicklow Head SPA, which means the Project is beyond the mean maximum foraging range of breeding kittiwakes from the SPA, but within the mean maximum foraging range +1SD, and the maximum foraging range.

2715. The NatureScot apportioning tool (NatureScot, 2018) has been used to estimate the proportion of kittiwakes from each of the relevant SPAs present at the windfarm site during the breeding season. The apportioning to SPA and non-SPA colonies is set out in **Appendix 12.1** of the ES. 0.40% of impacts at the windfarm site during the breeding season are apportioned to Wicklow Head SPA.

2716. Outside the breeding season, breeding kittiwakes from the SPA are assumed to range widely and to mix with kittiwakes of all ages from breeding colonies in the UK, Ireland and beyond. The relevant non-breeding season reference population is the UK Western Waters and Channel BDMPS, consisting of 911,586 individuals during autumn migration (September to December) and 627,816 during spring migration (January to February) (Furness, 2015).

2717. Furness (2015) estimated that 30% of breeding adults from Ireland colonies are present within the UK Western Waters and Channel BDMPS during the autumn and spring migration periods. This proportion of Wicklow Head SPA birds is therefore assumed, which is 574 birds. This represents 0.06% and 0.09% of the BDMPS population for the autumn and spring periods respectively. During autumn migration and spring migration, 0.06%, and 0.09% of impacts are therefore considered to affect birds from the SPA.

## Potential effects on the qualifying feature from the Project-alone

2718. The kittiwake qualifying feature of the Wicklow Head SPA has been screened into the Appropriate Assessment due to the potential risk of collision.

### Operation and maintenance phase collision risk

2719. Information for collision risk on breeding adult kittiwakes belonging to the Wicklow Head SPA population is presented in **Table 8.169**. Collision estimates, calculated using the sCRM, are presented by biological season. A summary of the annual outputs and the corresponding increase in the annual baseline mortality rate is also presented. Parameters used in the sCRM were agreed with Natural England during the ETG process and are described in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES.

2720. Based on the mean collision rates, the annual total of breeding adult kittiwakes from Wicklow Head SPA at risk of collision as a result of the Project is less than one bird (0.07). This would increase the existing mortality of the SPA breeding population by 0.03%.

*Table 8.169 Predicted seasonal and annual collision mortality (Stochastic model Option 2, avoidance rate 0.993 (±0.0003)), for breeding adult kittiwakes at the windfarm site, apportioned to Wicklow Head SPA, with corresponding increases to baseline mortality of the population*

| | Breeding Season | Autumn Migration | Non-breeding/winter | Spring Migration | Annual |
|---|---|---|---|---|---|
| Period | Mar-Aug | Sep-Dec | - | Jan-Feb | Jan-Dec |
| Total collisions[1] (mean and 95% CIs) | 15.36 (4.17-33.87) | 8.49 (2.20-18.85) | - | 0.63 (0.00-1.46) | 24.48 (6.37-54.17) |
| % apportioned to the SPA | 0.40% | 0.06% | - | 0.09% | - |
| Total SPA collisions (mean and 95% CIs) | 0.06 (0.02-0.14) | 0.01 (0.00-0.01) | - | 0.00 (0.00-0.00) | 0.07 (0.02-0.15) |
| Mortality increase[2] (mean and 95% CIs) | 0.03% (0.01-0.08%) | 0.00% (0.00-0.00%) | - | 0.00% (0.00-0.00%) | 0.03% (0.01-0.08%) |

[1] Breeding season collision values reduced to 94.1% of modelled value to reflect proportion of adult birds recorded during site surveys

[2] Assuming predicted annual SPA mortality of 197 birds (1,348 x 0.146)

2721. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that no detectable changes in mortality rates would occur on this population for the mean monthly collision estimates for the Project, or for the upper 95% CI collision estimate.

2722. **It is concluded that based on predicted kittiwake mortality due to collision at the windfarm site there is no potential for the Project to have an adverse effect on the integrity of the Wicklow Head SPA.**

2723. The confidence in the assessment is high. The evidence used to define the CRM input parameters presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** as part of the ES submitted alongside this RIAA is of high applicability and quality. Whilst there is uncertainty around some of the input parameters (e.g. avoidance rate), the rates selected are considered to be sufficiently precautionary based on expert opinion to provide confidence that collision rates are not underestimated.

### Potential effects on the qualifying feature in-combination with other projects

2724. As the Project would have no measurable effect on kittiwake populations from the SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of Wicklow Head SPA, when assessed in-combination with other plans or projects**.

# 8.69 Saltee Islands SPA (transboundary site)

2725. Saltee Islands SPA is located on the east coast of Ireland approximately 265km from the windfarm site.

## 8.69.1 Description of designation

2726. Saltee Islands SPA is situated between 4km – 5km off the coast of south Co. Wexford and comprises the two islands, Great Saltee and Little Saltee, and the surrounding seas both between them and to a distance of 500m from them. Both islands have exposed rocky cliffs on their south and east – those on Great Saltee being mostly c. 30 m high, those on Little Saltee about half this height. The Saltee Islands are internationally important for their breeding seabird assemblage.

## 8.69.2 Conservation objectives

2727. The conservation objectives for each of the qualifying species of the SPA is 'to maintain the favourable conservation condition of [species] in the Saltee Islands SPA', which is defined by a list of attributes and targets.

## 8.69.3 Assessment

2728. Eight qualifying features of the Saltee Islands SPA have been screened into the Appropriate Assessment (**Table 5.2**): puffin, fulmar, gannet, kittiwake, guillemot, razorbill, shag and cormorant.

### 8.69.3.1 Puffin

#### Status

2729. The Saltee Islands SPA breeding puffin population stood at 1,822 pairs, or 3,644 breeding adults, in the period 1998 – 2000 (NPWS, 2012b). The SMP database (JNCC, 2023a) does not provide a more recent estimate, therefore the 1998 – 2000 count has been used as the reference population for the assessment.

2730. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.094 (1 – 0.906; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 343 breeding adults.

#### Functional linkage and seasonal apportionment of potential effects

2731. The mean maximum foraging range of puffin is 137.1km (±128.3km) and the maximum foraging range is 383km (Woodward *et al*., 2019). The Project is located approximately 265km from Saltee Islands SPA, which means the

Project is beyond the mean maximum foraging range +1SD of puffins breeding at this SPA, but within the maximum foraging range. The maximum foraging range is a poor indicator of typical foraging behaviour. It would be expected that few birds or foraging trips will occur at this distance from the colony, and even fewer with any regularity. No impacts during the breeding season are therefore apportioned to birds breeding at this colony.

2732. Outside of the breeding season, breeding puffins, including those from the Saltee Islands SPA, are not constrained by requirements to visit nests to incubate eggs or provision chicks. At this time, they are assumed to range more widely and to mix with puffins of all age classes from breeding colonies in the UK and further afield. The background population during these seasons is the UK Western Waters BDMPS. This consists of 304,557 individuals during the non-breeding season (August to March).

2733. As no published estimate is available, it is assumed that 18% of Saltee Islands SPA breeding adults are present within the UK Western Waters BDMPS during the non-breeding season; the same proportion as ‘West Coast UK non-SPA populations’ (and other comparable SPAs around the Irish Sea) identified by Furness (2015). It is assumed that the most recent count prior to the publication of Furness (2015) was used to inform the BDMPS total which is 3,644 breeding adults; 18% of this population is 656 birds. This represents 0.2% of the BDMPS population for this period (304,557). It is therefore assumed that 0.2% of puffins present at the Project site are breeding adults from Saltee Islands SPA.

## Potential effects on the qualifying feature

### Operation and maintenance phase disturbance/displacement/barrier effects

*Project-alone*

2734. The mean peak abundance of puffins present within the windfarm site and 2km buffer during the non-breeding season was 19.7 (1.9-50.8) individuals (refer to **Appendix 12.1** of the ES). Of these, less than one bird (0.0 (0.0-0.1)) was likely to be a breeding adult from Saltee Islands SPA.

2735. **Table 8.170** sets out the predicted impacts on puffins from Saltee Islands SPA during the non-breeding season. This estimates that there would be no measurable increase in mortality from the SPA population, assuming a displacement rate of 30-70% and a mortality of 1-10% for displaced birds.

*Table 8.170 Puffin – predicted operation and maintenance phase displacement and mortality from Saltee Islands SPA*

| Mean peak abundance estimate type | Mean peak abundance estimate | Number of Saltee Islands SPA breeding adults present (non-breeding season)[1] | Annual mortality range[2] | Annual baseline mortality increase range[3] |
|---|---|---|---|---|
| Upper 95% CI | 50.8 | 0.1 | 0-0 | 0.00-0.00% |
| Mean | 19.7 | 0.0 | 0-0 | 0.00-0.00% |
| Lower 95% CI | 1.9 | 0.0 | 0-0 | 0.00-0.00% |

[1] Assumes 7.7% of birds present during the non-breeding season are Saltee Islands SPA breeding adults

[2] Assumes displacement rates of 30-70% and mortality rates of 1-10%

[3] Background mortality rate of 9.4% (Horswill and Robinson, 2015)

2736. **It is concluded that predicted puffin mortality due to operational phase displacement would not adversely affect the integrity of the Saltee Islands SPA.**

2737. The confidence in the assessment is high for several reasons. Firstly, the evidence used to inform the evidence-based displacement rates is of high applicability and quality (based on the criteria discussed in **Chapter 12 Offshore Ornithology** of the ES). Whilst there is limited available evidence to inform mortality rates, 1% is considered to be sufficiently precautionary based on expert opinion. This species is not regarded as being highly specialised in its habitat requirements (Bradbury *et al*., 2014; Furness and Wade, 2012; Garthe and Hüppop, 2004), and it is therefore anticipated that displaced birds will find alternative habitat in the vast majority of cases. Finally, the conclusion of the assessment is the same irrespective of whether the mean or 95% upper CI mean peak abundances are used to calculate potential mortality and increases in the baseline mortality rate of the background population.

*In-combination*

2738. As the Project would have no measurable effect on puffin populations from the Saltee Islands SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that predicted puffin mortality due to operational phase displacement due to the Project in-combination with other plans or projects would not adversely affect the integrity of the Saltee Islands SPA.**

#### 8.69.3.2 Fulmar

##### Status

2739. The Saltee Islands SPA breeding fulmar population stood at 520 pairs, or 1,040 breeding adults, in the period 1998 – 2000 (NPWS, 2012b). The most recent combined count from Great Saltee (225 pairs in 2013) and Little Saltee (214 pairs in 2007) is 439 pairs (AON), or 878 breeding adults; this is used as the reference population for the assessment.

2740. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.064 (1 – 0.936; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 56 breeding adults.

##### Functional linkage and seasonal apportionment of potential effects

2741. The mean maximum foraging range of fulmar is 542.3km (±657.9km) and the maximum foraging range is 2,736km (Woodward *et al*., 2019). The Project is located approximately 265km from Saltee Islands SPA, which means that the Project is within the mean maximum foraging range of fulmars breeding at this SPA.

##### Potential effects on the qualifying feature from the Project-alone

2742. Fulmar are considered to have low vulnerability to both collision and displacement impacts from offshore windfarms (Bradbury *et al*., 2014). This species was recorded infrequently within the windfarm site (in only five of 24 surveys, all during the breeding season (January to August; Furness, 2015)) and in low numbers, with densities in four of the five months less than 0.1 birds/km$^2$. A peak density of 0.23 birds/km$^2$ was recorded in May 2022.

2743. On the basis of the low vulnerability of fulmars to potential impacts, and the low frequency and abundance of birds present at the windfarm site, it is concluded that any effects on breeding populations at Saltee Islands SPA are very unlikely, both during and outside of the breeding season.

2744. **It is therefore concluded that there would be no measurable effects on fulmar due to the project alone, and no adverse effect on the integrity of the Saltee Islands SPA is predicted.**

##### Potential effects on the qualifying feature in-combination with other projects

2745. As the Project would have no measurable effect on fulmar populations from the Saltee Islands SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of Saltee Islands SPA, when assessed in-combination with other plans or projects.**

#### 8.69.3.3 Gannet

##### Status

2746. The Saltee Islands SPA breeding gannet population stood at 2,446 pairs, or 4,892 breeding adults, in 2004 (NPWS, 2012b). The most recent count is 4,722 pairs (AOS), or 9,444 breeding adults, in 2013; this is used as the reference population for the assessment.

2747. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.081 (1 – 0.919; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 765 breeding adults.

##### Functional linkage and seasonal apportionment of potential effects

2748. The windfarm site is 265km from Saltee Islands SPA. The mean maximum foraging range of gannet is 315.2km (±194.2km). The windfarm site is therefore within the mean maximum foraging range of gannets from the Saltee Islands SPA.

2749. Modelled at-sea utilisation distributions of breeding adult birds during the breeding season have been published, based on GPS tracking data (Wakefield *et al.*, 2013). These suggest that the windfarm site is located outside of the core foraging range for breeding adult birds from Saltee Islands SPA.

2750. Two UK SPAs designated for gannet are located within the UK Western Waters BDMPS area; Ailsa Craig SPA and Grassholm SPA. These sites are also located within the mean maximum foraging range of this species. Data presented by Wakefield *et al.,* (2013) indicated that the foraging ranges of gannets from different breeding colonies tended not to overlap, and that the windfarm site would be located on the edge of the core foraging area for adult birds from Ailsa Craig SPA, but outside of the foraging area for both Grassholm and Saltee Islands SPA.

2751. Two further UK sites are within the straight-line foraging distance of the windfarm site; Flamborough and Filey Coast SPA (212km) and Forth Islands SPA (239km). However, both sites are on the eastern UK coast, with an across-sea distance of >1000km, and are therefore considered geographically isolated from the windfarm site during the breeding season.

2752. On the basis of the data presented by Wakefield *et al.,* (2013), it assumed that breeding adult gannets present at the windfarm site during the full breeding season (March to September (Furness, 2015)) originate from the Ailsa Craig SPA. Accordingly, no birds present during this period are considered to originate from Saltee Islands SPA.

2753. Outside the breeding season breeding gannets, including those from the Saltee Islands SPA, are not constrained by requirements to visit nests to incubate eggs or provision chicks. At this time, they are assumed to range more widely and to mix with gannets of all age classes from breeding colonies in Ireland, the UK and further afield. The background population during these seasons is the UK Western Waters BDMPS. This consists of 545,954 individuals during autumn migration (September to November), and 661,888 individuals during spring migration (December to March) (Furness, 2015).

2754. Estimates of the proportion of gannets present at the windfarm site which originate from the Saltee Islands SPA during the non-breeding season (and therefore the proportion of predicted mortalities from the SPA population) are based on the most recent SPA population estimate prior to the publication of Furness (2015) (i.e. 9,444 breeding adults in 2013) as a proportion of the UK Western Waters BDMPS during the relevant season. During autumn migration and spring migration, 1.7%, and 1.4% of impacts are considered to affect birds from the SPA respectively.

## Potential effects on the qualifying feature

2755. The gannet qualifying feature of the Saltee Islands SPA has been screened into the assessment due to the potential risk of collision and operational phase displacement/barrier effects during the operation and maintenance phase of the Project.

### Operation and maintenance phase displacement/barrier effects

2756. Displacement effects for gannet for the Project were assessed during the autumn and spring migration periods, based on an unapportioned peak mean population of 124 and eight birds respectively, calculated for the windfarm site and a 2km buffer, in line with recommendations within the SNCB guidance (SNCB 2017). As set out above, no gannets present at the windfarm site have been apportioned to Saltee Islands SPA during the breeding season. The displacement matrices used to calculate potential impacts are presented in **Appendix 12.1** of the ES, and summarised in **Table 8.171**. The inclusion of all birds within the 2km buffer, to determine the total number of birds subject to displacement, is precautionary, as in reality the avoidance rate is likely to fall with distance from the windfarm site.

2757. A displacement rate of 60-80% and mortality rate of 1% has been presented. A maximum 1% mortality value has been selected firstly because gannet is known to possess high habitat flexibility (Furness and Wade, 2012). This suggests that displaced birds will readily find alternative habitats including foraging areas. Secondly, no evidence of displacement-induced mortality has been identified, which means there is limited justification for setting predicted mortality rates at a higher level. Given the extensive foraging range of this species (Woodward *et al.*, 2019), there may be no mortality costs to displacement from the relatively very small footprints of OWFs.

*Table 8.171 Gannet – predicted operation and maintenance phase displacement and mortality from Saltee Islands SPA*

| Mean peak abundance estimate type | Mean peak abundance estimate | Number of SPA breeding adults present by season[1] | Annual mortality range[2] | Annual baseline mortality increase range[3] |
|---|---|---|---|---|
| Upper 95% CI | 809 (breeding)<br>189 (autumn)<br>16 (spring)<br>1,014 (year round) | 0 (breeding)<br>3 (autumn)<br>0 (spring)<br>3 (year round) | 0-0 | 0.00-0.00% |
| Mean | 541 (breeding)<br>124 (autumn)<br>8 (spring)<br>673 (year round) | 0 (breeding)<br>2 (autumn)<br>0 (spring)<br>2 (year round) | 0-0 | 0.00-0.00% |
| Lower 95% CI | 160 (breeding)<br>0 (autumn)<br>0 (spring)<br>160 (year round) | 0 (breeding)<br>0 (autumn)<br>0 (spring)<br>0 (year round) | 0-0 | 0.00-0.00% |

[1] 1.7% and 1.4% of birds are assumed to be breeding adults from the SPA population during the autumn and spring migration periods respectively.

[2] Assumes displacement rates of 60-80% and mortality rate of 1%

[3] Background population is Saltee Islands SPA breeding adults (9,444 individuals), adult age class annual mortality rate of 8.1% (Horswill and Robinson, 2015)

2758. Using the maximum potential mortality value, there would be no measurable increase in gannet mortality. Accordingly, no significant effects on gannet are predicted during the operation and maintenance phase, and **it is concluded that there would be no potential for the Project to have an adverse effect on the integrity of Saltee Islands SPA**.

2759. The confidence in the assessment is high for several reasons. Firstly, the evidence used to set the displacement rates presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES is of high applicability and quality. Whilst there is limited available evidence to inform mortality rates, 1% is considered to be sufficiently precautionary based on expert opinion. Finally, the conclusion of the assessment is the same irrespective of whether the mean or 95% upper CI mean peak abundances are used to calculate potential mortality and increases in the baseline mortality rate of the background population.

### Operation and maintenance phase collision risk

2760. Information to support the Appropriate Assessment for collision risk on breeding adult gannets belonging to the Saltee Islands SPA population is presented in **Table 8.172**. Collision estimates, calculated using the sCRM, are presented by biological season. A summary of the annual outputs and the corresponding increase in the annual baseline mortality rate is also presented. Parameters used in the sCRM were agreed with Natural England during the ETG process and are described in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES. In accordance with Natural England advice, a 70% macro-avoidance correction was applied to gannet abundance data used in the sCRM.

2761. Based on the mean collision rates, no breeding adult gannets from Saltee Islands SPA are considered at risk of collision as a result of the Project. Therefore, there would be no measurable increase the existing mortality of the SPA breeding population.

*Table 8.172 Predicted seasonal and annual collision mortality (Stochastic model Option 2, avoidance rate 0.993 (±0.0003), plus 70% macro-avoidance) for breeding adult gannets at the windfarm site, apportioned to Saltee Islands SPA, with corresponding increases to baseline mortality of the population*

| Period | Breeding Season<br>Apr-Aug | Autumn Migration<br>Sep-Nov | Non-breeding/winter<br>- | Spring Migration<br>Dec-Mar | Annual<br>Jan-Dec |
|---|---|---|---|---|---|
| Total collisions[1] (mean and 95% CIs) | 0.83<br>(0.00-3.35) | 0.14<br>(0.00-0.74) | - | 0.00 | 0.97<br>(0.00-4.10) |
| % apportioned to the SPA | 0.0% | 1.7% | - | 1.4% | - |
| Total SPA collisions (mean and 95% CIs) | 0.00<br>(0.00-0.00) | 0.00<br>(0.00-0.01) | - | 0.00<br>(0.00-0.00) | 0.00<br>(0.00-0.01) |
| Mortality increase[2] (mean and 95% CIs) | 0.00%<br>(0.00-0.00%) | 0.00%<br>(0.00-0.00%) | - | 0.00%<br>(0.00-0.00%) | 0.00%<br>(0.00-0.00%) |

[1] Breeding season collision values reduced to 73.8% of modelled value to reflect proportion of adult birds recorded during site surveys
[2] Assuming predicted annual SPA mortality of 765 birds (9,444 x 0.081)

2762. Accordingly, no significant effects on gannet are predicted during the operation and maintenance phase, and it is concluded that there would be **no potential for the Project to have an adverse effect on the integrity of Saltee Islands SPA**. Comments received from RSPB during the ETG process, indicating that they do not accept the 70% macro-avoidance rate for collision risk recommended by Natural England, are noted. However, even in the absence of this correction factor, the net increase in mortality would be unchanged (i.e. zero).

2763. The confidence in the assessment is high. The evidence used to define the CRM input parameters presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES submitted alongside this RIAA is of high applicability and quality. Whilst there is uncertainty around some of the input parameters (e.g. avoidance rate), the rates selected are considered to be sufficiently precautionary based on expert opinion to provide confidence that collision rates are not underestimated.

**Combined displacement/barrier effects and collision risk**

2764. As no measurable increase in mortality is predicted for both displacement and collision risk, the mean combined displacement and collision rates for breeding adult gannet from the Saltee Islands SPA would be zero. Therefore, there would be no net increase in existing mortality rates.

2765. **It is concluded that based on predicted gannet mortality due to the combined effects of operational phase displacement and collision there is no potential for the Project-alone or in-combination to have an adverse effect on the integrity of the Saltee Islands SPA**.

2766. The confidence in the assessment is high, for the reasons provided in the individual displacement and collision assessments.

### Potential effects in-combination with other projects

2767. As no measurable effects of displacement/barrier and collision on gannet are predicted as a result of the Project-alone, there would be no contribution to other plans or projects in-combination**. It is therefore concluded that there would be no potential for the Project-alone or in-combination to have an adverse effect on the integrity of Saltee Islands SPA**.

### 8.69.3.4 Kittiwake

### Status

2768. The Saltee Islands SPA breeding kittiwake population stood at 2,125 pairs, or 4,250 breeding adults, in the period 1998 – 2000 (NPWS, 2012b). The most recent count is 845 pairs (AON), or 1,690 breeding adults; this is used as the reference population for the assessment.

2769. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.146 (1 – 0.854; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 247 breeding adults.

### Functional linkage and seasonal apportionment of potential effects

2770. The mean maximum foraging range of kittiwake is 156.1km (±144.5km) and the maximum foraging range is 770km (Woodward *et al*., 2019). The Project is located approximately 265km from Saltee Islands SPA, which means the Project is beyond the mean maximum foraging range of breeding kittiwakes from the SPA, but within the mean maximum foraging range +1SD, and the maximum foraging range.

2771. The NatureScot apportioning tool (NatureScot, 2018) has been used to estimate the proportion of kittiwakes from each of the relevant SPAs present at the windfarm site during the breeding season. The apportioning to SPA and non-SPA colonies is set out in **Appendix 12.1** of the ES. 0.31% of impacts at the windfarm site during the breeding season are apportioned to Saltee Islands SPA.

2772. Outside the breeding season, breeding kittiwakes from the SPA are assumed to range widely and to mix with kittiwakes of all ages from breeding colonies in the UK, Ireland and beyond. The relevant non-breeding season reference population is the UK Western Waters and Channel BDMPS, consisting of 911,586 individuals during autumn migration (September to December) and 627,816 during spring migration (January to February) (Furness, 2015).

2773. Furness (2015) estimated that 30% of breeding adults from Ireland colonies are present within the UK Western Waters and Channel BDMPS during the autumn and spring migration periods. This proportion of Saltee Islands SPA birds is therefore assumed, which is 507 birds. This represents 0.06% and 0.08% of the BDMPS population for the autumn and spring periods respectively. During autumn migration and spring migration, 0.06%, and 0.08% of impacts are therefore considered to affect birds from the SPA.

### Potential effects on the qualifying feature from the Project-alone

2774. The kittiwake qualifying feature of the Saltee Islands SPA has been screened into the Appropriate Assessment due to the potential risk of collision.

#### Operation and maintenance phase collision risk

2775. Information for collision risk on breeding adult kittiwakes belonging to the Saltee Islands SPA population is presented in **Table 8.173**. Collision estimates, calculated using the sCRM, are presented by biological season. A summary of the annual outputs and the corresponding increase in the annual baseline mortality rate is also presented. Parameters used in the sCRM were agreed with Natural England during the ETG process and are described in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES.

2776. Based on the mean collision rates, the annual total of breeding adult kittiwakes from Saltee Islands SPA at risk of collision as a result of the Project is less than one bird (0.05). This would increase the existing mortality of the SPA breeding population by 0.02%.

*Table 8.173 Predicted seasonal and annual collision mortality (Stochastic model Option 2, avoidance rate 0.993 (±0.0003)), for breeding adult kittiwakes at the windfarm site, apportioned to Saltee Islands SPA, with corresponding increases to baseline mortality of the population*

| | **Breeding Season** | **Autumn Migration** | **Non-breeding/winter** | **Spring Migration** | **Annual** |
|---|---|---|---|---|---|
| **Period** | **Mar-Aug** | **Sep-Dec** | **-** | **Jan-Feb** | **Jan-Dec** |
| Total collisions[1] (mean and 95% CIs) | 15.36 (4.17-33.87) | 8.49 (2.20-18.85) | - | 0.63 (0.00-1.46) | 24.48 (6.37-54.17) |
| % apportioned to the SPA | 0.31% | 0.06% | - | 0.08% | - |
| Total SPA collisions (mean and 95% CIs) | 0.05 (0.01-0.10) | 0.00 (0.00-0.01) | - | 0.00 (0.00-0.00) | 0.05 (0.01-0.12) |
| Mortality increase[2] (mean and 95% CIs) | 0.02% (0.01-0.04%) | 0.00% (0.00-0.00%) | - | 0.00% (0.00-0.00%) | 0.02% (0.01-0.05%) |

[1] Breeding season collision values reduced to 94.1% of modelled value to reflect proportion of adult birds recorded during site surveys

[2] Assuming predicted annual SPA mortality of 247 birds (1,690 x 0.146)

2777. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that no detectable changes in mortality rates would occur on this population for the mean monthly collision estimates for the Project, or for the upper 95% CI collision estimate.

2778. **It is concluded that based on predicted kittiwake mortality due to collision at the windfarm site there is no potential for the Project to have an adverse effect on the integrity of the Saltee Islands SPA**.

2779. The confidence in the assessment is high. The evidence used to define the CRM input parameters presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** as part of the ES submitted alongside this RIAA is of high applicability and quality. Whilst there is uncertainty around some of the input parameters (e.g. avoidance rate), the rates selected are considered to be sufficiently precautionary based on expert opinion to provide confidence that collision rates are not underestimated.

### Potential effects on the qualifying feature in-combination with other projects

2780. As the Project would have no measurable effect on kittiwake populations from the SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of Saltee Islands SPA, when assessed in-combination with other plans or projects**.

### 8.69.3.5 Guillemot

### Status

2781. The Saltee Islands SPA breeding guillemot population stood at 14,362 pairs, or 28,724 breeding adults, in the period 1998 – 2000 (NPWS, 2012b). The most recent count is 17,501 individuals in 2013 (JNCC, 2023a); this is used as the reference population for the assessment.

2782. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.061 (1 – 0.939; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 1,068 breeding adults.

### Functional linkage and seasonal apportionment of potential effects

2783. The mean maximum foraging range of guillemot is 73.2km (±80.5km) and the maximum foraging range is 338km (Woodward *et al*., 2019). The Project is located approximately 265km from Saltee Islands SPA, which means that the Project is beyond the mean maximum foraging range and mean maximum foraging range +1SD of guillemots breeding at this SPA, but within the maximum foraging range. The maximum foraging range is a poor indicator of typical foraging behaviour. It would be expected that few birds or foraging trips will occur at this distance from the colony, and even fewer with any regularity. No impacts during the breeding season are therefore apportioned to birds breeding at this colony.

2784. Outside the breeding season, breeding guillemots from the SPA are assumed to range widely and to mix with guillemots of all ages from breeding colonies in Ireland, the UK and beyond. The relevant non-breeding season reference population is the UK Western Waters BDMPS, consisting of 1,139,220 individuals (August to February) (Furness, 2015).

2785. As no published estimate was available, it is assumed that 95% of Saltee Islands SPA breeding adults are present within the UK Western Waters BDMPS during the non-breeding season; the same proportion as ‘West Coast UK non-SPA populations’ (and other comparable SPAs around the Irish Sea) identified by Furness (2015). It is assumed that the most recent count prior to the publication of Furness (2015) was used to inform the BDMPS total which is 17,501 breeding adults; 95% of this population is 16,626 birds. This represents 1.5% of the BDMPS population for this period (1,139,220). It is therefore assumed that 1.5% of guillemots present at the Project site are breeding adults from Saltee Islands SPA.

## Potential effects on the qualifying feature

### Operation and maintenance phase disturbance/displacement/barrier effects

*Project-alone*

2786. The mean peak abundance of guillemots present within the windfarm site and 2km buffer during the non-breeding season was 8,315 (6,085-12,047) individuals (refer to **Appendix 12.1** of the ES). Of these, 540 birds (396-783) were likely to be breeding adults from the Saltee Islands SPA.

2787. **Table 8.174** sets out the predicted impacts on guillemots from Saltee Islands SPA during the non-breeding season. Displacement rates of 30% to 70% are considered for this species, along with a range of mortality rates of 1% to 10% of displaced birds (UK SNCBs, 2017).

2788. The available evidence suggests that the upper ranges of these displacement and mortality rates may be excessively precautionary. Whilst it is true that guillemots and razorbills tend to be displaced by OWFs they do not avoid them completely, and displacement rates vary between sites. MacArthur Green (2019b) concluded that, on average, densities within OWFs were around half of the density found in the habitats around the OWF. At some OWFs there was also displacement of birds from a buffer zone surrounding it. The size of the buffer zone varied between OWFs and was generally less than 2km, with auk density increasing across the buffer zone as distance from turbines increased, up to the density in the wider area. Another recent review (APEM, 2022) found that the current displacement rates suggested by the SNCBs for guillemot and razorbill (30% to 70%) did not account for the quality of or confidence in the studies which were used to inform this position, and that studies where no significant effects were recorded were not accounted for during the provision of the advice. APEM (2022) suggested that in the case of Hornsea Project Four, an evidence-based displacement rate of 50% was appropriate. However, the study also recognised that larger displacement effects were possible.

2789. Mortality due to displacement could arise due to increased energy costs and/or decreased energy intake, if displacement results in increased flying time to avoid OWFs, and/or increased bird densities and competition for prey in areas of unimpacted habitat outside the OWF. Given that UK OWFs in the Irish and Celtic Seas represent a very small proportion of the available foraging habitat for guillemots and razorbills within UK Western Waters, the increase in density of auks outside OWFs due to displacement is likely to be negligible (MacArthur Green, 2019b). It is considered unlikely that the mortality rate due to displacement would be as high as the empirically estimated 6.1% annual mortality for adult guillemots that occurs due to the combination of 'natural' factors and anthropogenic activities (Horswill and Robinson, 2015). Indeed, it may be much lower; MacArthur Green (2019b) recommended precautionary rates of 50% displacement for guillemot and 1% mortality of displaced birds based on a review of available evidence. Modelling undertaken by APEM (2022) found that in the case of Hornsea Project Four, a mortality rate of 1% for displaced auks was considered to be precautionary, and that this may be an overestimate given that Hornsea Project Four is located towards the upper end of the mean maximum foraging range for guillemot from the closest designated sites. It is also noted that in the recent decision on Hornsea Project Four, the SoS determined that a rate of 70% displacement/2% mortality was appropriate to inform the assessment of effects on guillemots from Flamborough and Filey Coast SPA (DESNZ, 2023b).

2790. The full ranges of recommended displacement and mortality effects are considered by the assessment, along with evidence-based displacement and mortality rates of 50% and 1%, respectively (APEM, 2022; MacArthur Green, 2019b).

*Table 8.174 Guillemot – predicted operation and maintenance phase displacement and mortality from Saltee Islands SPA*

| Mean peak abundance estimate type | Mean peak abundance estimate | Number of Saltee Islands SPA breeding adults present (non-breeding season)[1] | Annual mortality range[2] | Annual baseline mortality increase range[3] |
|---|---|---|---|---|
| Upper 95% CI | 12,047 | 181 | 1-13 | 0.05-1.18% |
| Mean | 8,315 | 125 | 0-9 | 0.04-0.82% |
| Lower 95% CI | 6,085 | 91 | 0-6 | 0.03-0.60% |

[1] Assumes 1.5% of birds present during the non-breeding season are Saltee Islands SPA breeding adults

[2] Assumes displacement rates of 30-70% and mortality rates of 1-10%

[3] Background mortality rate of 6.1% (Horswill and Robinson, 2015)

2791. Based on the mean peak abundances, the annual total of guillemots from the Saltee Islands SPA at risk of displacement is 120 birds (**Table 8.174**). At displacement rates of 30% to 70%, and mortality rates of 1% to 10% for displaced birds, 0 to 9 SPA breeding adults would be predicted to die each year due to displacement from the Project.

2792. Assuming a displacement rate of 70% and a mortality rate of 10% of displaced birds, annual mortality within the SPA breeding adult population would increase by 0.82%. Using an evidence-based displacement rate of 50%, and a mortality rate for displaced birds of 1%, annual mortality in the population would instead increase by 0.06% (1 bird).

2793. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that no detectable changes in mortality rates would occur when the mean peak abundance estimate assessments are considered.

2794. Mortality rate increases of over 1% are predicted for mean peak abundance estimate assessments if the upper 95% CIs for mean peak abundances are used as inputs to the assessment alongside a 10% mortality rate for displaced birds. Use of the evidence-based displacement (50%) and mortality rate (1%) would result in a mortality increase of significantly less than 1%, as would a rate of 70%/2% agreed by the SoS in respect of Hornsea Project Four (DESNZ, 2023b). The upper 95% CI for the mean peak abundances are highly unlikely to occur regularly at the windfarm site.

2795. **It is concluded that predicted guillemot mortality due to operational phase displacement for the project alone would not adversely affect the integrity of the Saltee Islands SPA.**

2796. The confidence in the assessment is high for several reasons. Firstly, the evidence used to inform the evidence-based displacement rates is of high applicability and quality (based on the criteria discussed in **Chapter 12 Offshore Ornithology** of the ES). Whilst there was limited available evidence to inform mortality rates, 1% is considered to be sufficiently precautionary based on expert opinion. This species is not regarded as being highly specialised in its habitat requirements (Bradbury *et al*., 2014; Furness and Wade, 2012; Garthe and Hüppop, 2004), and it is therefore anticipated that displaced birds will find alternative habitat in the vast majority of cases. Finally, the conclusion of the assessment is the same irrespective of whether the mean or 95% upper CI mean peak abundances are used to calculate potential mortality and increases in the baseline mortality rate of the background population, provided the evidence-based displacement and mortality rates are used.

*In-combination*

2797. The in-combination assessment for guillemots from Saltee Islands SPA has been undertaken in accordance with the approach presented in **Section 8.1**. The total population apportioned to Saltee Islands SPA at risk of displacement is estimated to be 1,487 breeding adults (**Appendix 12.1** of the ES). Annual in-combination displacement and mortality rates for birds from Saltee Islands SPA are presented in **Table 8.175**.

2798. Assuming a displacement rate of 70% and a mortality rate of 10% of displaced birds, 104 breeding adult SPA birds would be lost to displacement annually. Including additional apportioned mortality from the Morlais and Holyhead tidal projects (total 0.47 birds), this would increase the existing mortality within the SPA population (1,068 breeding adult birds per year) by 9.80%. Using an evidence-based displacement rate of 50%, and a mortality rate for displaced birds of 1%, the annual in-combination displacement mortality would be 7 birds. This would increase the existing mortality within this population by 0.74%. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that detectable changes in mortality rates will not occur due to the level or mortality predicted if the more realistic rates for mortality are used.

2799. **It is concluded that predicted guillemot mortality due to of operational phase displacement impacts of the Project in-combination with other projects, would not adversely affect the integrity of Saltee Islands SPA.**

*Table 8.175 In-combination year-round displacement matrix for guillemot from Saltee Islands SPA*

| Annual | Mortality | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| **Displacement** | **1%** | **2%** | **3%** | **4%** | **5%** | **10%** | **20%** | **30%** | **50%** | **80%** | **100%** |
| **10%** | 1 | 3 | 4 | 6 | 7 | 15 | 30 | 45 | 74 | 119 | 149 |
| **20%** | 3 | 6 | 9 | 12 | 15 | 30 | 59 | 89 | 149 | 238 | 297 |
| **30%** | 4 | 9 | 13 | 18 | 22 | 45 | 89 | 134 | 223 | 357 | 446 |
| **40%** | 6 | 12 | 18 | 24 | 30 | 59 | 119 | 178 | 297 | 476 | 595 |
| **50%** | 7 | 15 | 22 | 30 | 37 | 74 | 149 | 223 | 372 | 595 | 744 |
| **60%** | 9 | 18 | 27 | 36 | 45 | 89 | 178 | 268 | 446 | 714 | 892 |
| **70%** | 10 | 21 | 31 | 42 | 52 | 104 | 208 | 312 | 520 | 833 | 1041 |
| **80%** | 12 | 24 | 36 | 48 | 59 | 119 | 238 | 357 | 595 | 952 | 1190 |
| **90%** | 13 | 27 | 40 | 54 | 67 | 134 | 268 | 402 | 669 | 1071 | 1338 |
| **100%** | 15 | 30 | 45 | 59 | 74 | 149 | 297 | 446 | 744 | 1190 | 1487 |

Note: The cells show the number of birds predicted to die (rounded to the nearest integer) at a given rate of displacement and mortality. Highlighted cells are considered to be the most realistic scenario, in accordance with SNCB advice (SNCBs, 2022).

#### 8.69.3.6 Razorbill

##### Status

2800. The Saltee Islands SPA breeding razorbill population stood at 2,505 pairs, or 5,010 breeding adults, in the period 1998 – 2000 (NPWS, 2012b). The most recent count from Great Saltee was 2,931 individuals in 2013 (JNCC, 2023a) however no estimate has been provided for Little Saltee since 2000. The 1998 – 2000 count is therefore used as the reference population for the assessment.

2801. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.105 (1 – 0.895; Horswill and Robinson, 2015), the expected annual mortality from the SPA population would be 526 breeding adults.

##### Functional linkage and seasonal apportionment of potential effects

2802. The mean maximum foraging range of razorbill is 88.7km (±75.9km) and the maximum foraging range is 313km (Woodward *et al*., 2019). The Project is located approximately 265km from Saltee Islands SPA, which means the Project is beyond the mean maximum foraging range and mean maximum +1SD, but within the maximum foraging range of razorbills breeding at this SPA. The maximum foraging range is a poor indicator of typical foraging behaviour, and it would be expected that few birds or foraging trips will occur at this distance from the colony, and even fewer with any regularity. No impacts during the breeding season are therefore apportioned to birds breeding at this colony.

2803. Outside the breeding season, breeding razorbills from the SPA are assumed to range widely and to mix with razorbills of all ages from breeding colonies in Ireland, the UK and beyond. The relevant non-breeding season reference population is the UK Western Waters BDMPS, consisting of 606,914 individuals during autumn and spring passage periods (August to October and January to March), and 341,422 individuals during winter (November and December) (Furness, 2015).

2804. As no published estimate was available, it is assumed that 98% of Saltee Islands SPA breeding adults are present within the UK Western Waters BDMPS during the non-breeding season, and 30% during the winter period; the same proportions as 'West Coast UK non-SPA populations' (and other comparable SPAs around the Irish Sea) identified by Furness (2015). It is assumed that the most recent count prior to the publication of Furness (2015) was used to inform the BDMPS total which is 5,010 breeding adults; 98% of this population is 4,910 birds, and 30% is 1,503 birds. This represents 0.8% and 0.4% of the BDMPS population (606,914 during spring/autumn and 341,422 during winter) respectively. These percentages (i.e. 0.8% (spring and autumn migration) and 0.4% (winter)) are the proportions of birds present at the windfarm site that are presumed to originate from the Saltee Islands SPA during the relevant seasons.

## Potential effects on the qualifying feature

### Operation and maintenance phase disturbance/displacement/barrier effects

*Project-alone*

2805. The year-round mean peak abundance of razorbills present within the windfarm site and 2km buffer was 1,979 (703-3,552) individuals (refer to **Appendix 12.1** of the ES). Of these, 2 birds (1-3) were likely to be breeding adults from the Saltee Islands SPA.

2806. **Table 8.176** sets out the predicted impacts on razorbills from Saltee Islands SPA. Displacement rates of 30% to 70% are considered for this species, along with a range of mortality rates of 1% to 10% of displaced birds (UK SNCBs, 2017).

2807. The available evidence suggests that the upper ranges of these displacement and mortality rates may be excessively precautionary. Whilst it is true that guillemots and razorbills tend to be displaced by OWFs they do not avoid them completely, and displacement rates vary between sites. MacArthur Green (2019b) concluded that, on average, densities within OWFs were around half of the density found in the habitats around the OWF. At some OWFs there was also displacement of birds from a buffer zone surrounding it. The size of the buffer zone varied between OWFs and was generally less than 2km, with auk density increasing across the buffer zone as distance from turbines increased, up to the density in the wider area. Another recent review (APEM, 2022) found that the current displacement rates suggested by the SNCBs for guillemot and razorbill (30% to 70%) did not account for the quality of or confidence in the studies which were used to inform this position, and that studies where no significant effects were recorded were not accounted for during the provision of the advice. APEM (2022) suggested that in the case of Hornsea Project Four, an evidence-based displacement rate of 50% was appropriate. However, the study also recognised that larger displacement effects were possible.

2808. Mortality due to displacement could arise due to increased energy costs and/or decreased energy intake, if displacement results in increased flying time to avoid OWFs, and/or increased bird densities and competition for prey in areas of unimpacted habitat outside the OWF. Given that UK OWFs in the Irish and Celtic Seas represent a very small proportion of the available foraging habitat for guillemots and razorbills within UK Western Waters, the increase in density of auks outside OWFs due to displacement is likely to be negligible (MacArthur Green, 2019b). It is considered unlikely that the mortality rate due to displacement would be as high as the empirically estimated 10.5% annual mortality for adult razorbills that occurs due to the combination of 'natural' factors and anthropogenic activities (Horswill and Robinson, 2015). Indeed, it may be much lower; MacArthur Green (2019b) recommended precautionary rates of 50% displacement for auks and 1% mortality of displaced birds based on a review of available evidence. Modelling undertaken by APEM (2022) found that in the case of Hornsea Project Four, a mortality rate of 1% for displaced auks was considered to be precautionary, and that this may be an overestimate given that Hornsea Project Four is located towards the upper end of the mean maximum foraging range for guillemot from the closest designated sites.

2809. The full ranges of recommended displacement and mortality effects are considered by the assessment, along with evidence-based displacement and mortality rates of 50% and 1%, respectively (APEM, 2022; MacArthur Green, 2019b).

*Table 8.176 Razorbill – predicted operation and maintenance phase displacement and mortality from Saltee Islands SPA*

| Mean peak abundance estimate type | Mean peak abundance estimate by season[1] | Number of Saltee Islands SPA breeding adults present by season)[2] | Annual mortality range[3] | Annual baseline mortality increase range[4] |
|---|---|---|---|---|
| Upper 95% CI | 605 (b)<br>1,070 (aut)<br>1,297 (win)<br>580 (spr)<br>3,552 (year round) | 0 (b)<br>9 (aut)<br>5 (win)<br>5 (spr)<br>18 (year round) | 0-1 (0) | 0.01-0.24% (0.02%) |
| Mean | 252 (b)<br>694 (aut)<br>651 (win)<br>381 (spr)<br>1,979 (year round) | 0 (b)<br>6 (aut)<br>3 (win)<br>3 (spr)<br>11 (year round) | 0-1 (0) | 0.01-0.15% (0.01%) |
| Lower 95% CI | 21 (b)<br>309 (aut)<br>159 (win)<br>214 (spr)<br>703 (year round) | 0 (b)<br>2 (aut)<br>1 (win)<br>2 (spr)<br>5 (year round) | 0-0 (0) | 0.00-0.06% (0.00%) |

[1] Breeding season = b, autumn migration season = aut, winter season = win, spring migration season = spr
[2] Assumes breeding adult apportioning of 0.0% (breeding season), 0.8% (spring and autumn migration) and 0.4% (winter) to Saltee Islands SPA.
[3] Assumes displacement rates of 30-70% and mortality rates of 1-10%. Evidence-based estimates assuming a 50% displacement rate and 1% mortality of displaced birds are presented in parentheses.
[4] Background mortality rate of 10.5% (Horswill and Robinson, 2015)

2810. Based on the mean peak abundances, the annual total of razorbills from the Saltee Islands SPA at risk of displacement is 11 birds (**Table 8.176**). At displacement rates of 30% to 70%, and mortality rates of 1% to 10% for displaced birds, 0 to 1 SPA breeding adults would be predicted to die each year due to displacement from the Project.

2811. Assuming a displacement rate of 70% and a mortality rate of 10% of displaced birds, annual mortality within the SPA breeding adult population would increase by 0.15%. Using an evidence-based displacement rate of 50%, and a mortality rate for displaced birds of 1%, annual mortality in the population would instead increase by 0.01% (<1 bird).

2812. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that no detectable changes in mortality rates would occur under almost any combination of displacement and mortality rates when the mean peak abundance estimate assessments are considered. This would be the case even when upper 95% CIs are considered.

2813. **It is concluded that predicted razorbill mortality due to operational phase displacement for the project alone would not adversely affect the integrity of the Saltee Islands SPA.**

2814. The confidence in the assessment is high for several reasons. Firstly, the evidence used to inform the evidence-based displacement rates is of high applicability and quality (based on the criteria discussed in **Chapter 12 Offshore Ornithology** of the ES). Whilst there was limited available evidence to inform mortality rates, 1% is considered to be sufficiently precautionary based on expert opinion. This species is not regarded as being highly specialised in its habitat requirements (Bradbury *et al*., 2014; Furness and Wade, 2012; Garthe and Hüppop, 2004), and it is therefore anticipated that displaced birds will find alternative habitat in the vast majority of cases. Finally, the conclusion of the assessment is the same irrespective of whether the mean or 95% upper CI mean peak abundances are used to calculate potential mortality and increases in the baseline mortality rate of the background population.

*In-combination*

2815. The in-combination assessment for razorbills from Saltee Islands SPA has been undertaken in accordance with the approach presented in **Section 8.1**. The total population apportioned to the SPA at risk of displacement is estimated to be 118 breeding adults (**Appendix 12.1** of the ES). Annual in-combination displacement and mortality rates for birds from Saltee Islands SPA are presented in **Table 8.177**.

2816. Assuming a displacement rate of 70% and a mortality rate of 10% of displaced birds, 8 breeding adult SPA birds would be lost to displacement annually.

Including additional apportioned mortality from the Morlais and Holyhead tidal projects (total 0.11 birds), this would increase the existing mortality within the SPA population (526 breeding adult birds per year) by 1.59%. Using an evidence-based displacement rate of 50%, and a mortality rate for displaced birds of 1%, the annual in-combination mortality would be one bird. This would increase the existing mortality within this population by 0.13%. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that detectable changes in mortality rates will not occur due to the level or mortality predicted if the more realistic rates for mortality are used.

2817. **It is concluded that predicted razorbill mortality due to of operational phase displacement impacts of the Project in-combination with other projects, would not adversely affect the integrity of Saltee Islands SPA.**

*Table 8.177 In-combination year-round displacement matrix for razorbill from Saltee Islands SPA*

| Annual | Mortality | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| **Displacement** | **1%** | **2%** | **3%** | **4%** | **5%** | **10%** | **20%** | **30%** | **50%** | **80%** | **100%** |
| **10%** | 0 | 0 | 0 | 0 | 1 | 1 | 2 | 4 | 6 | 9 | 12 |
| **20%** | 0 | 0 | 1 | 1 | 1 | 2 | 5 | 7 | 12 | 19 | 24 |
| **30%** | 0 | 1 | 1 | 1 | 2 | 4 | 7 | 11 | 18 | 28 | 35 |
| **40%** | 0 | 1 | 1 | 2 | 2 | 5 | 9 | 14 | 24 | 38 | 47 |
| **50%** | 1 | 1 | 2 | 2 | 3 | 6 | 12 | 18 | 30 | 47 | 59 |
| **60%** | 1 | 1 | 2 | 3 | 4 | 7 | 14 | 21 | 35 | 57 | 71 |
| **70%** | 1 | 2 | 2 | 3 | 4 | 8 | 17 | 25 | 41 | 66 | 83 |
| **80%** | 1 | 2 | 3 | 4 | 5 | 9 | 19 | 28 | 47 | 76 | 95 |
| **90%** | 1 | 2 | 3 | 4 | 5 | 11 | 21 | 32 | 53 | 85 | 106 |
| **100%** | 1 | 2 | 4 | 5 | 6 | 12 | 24 | 35 | 59 | 95 | 118 |

Note: The cells show the number of birds predicted to die (rounded to the nearest integer) at a given rate of displacement and mortality. Highlighted cells are considered to be the most realistic scenario, in accordance with SNCB advice (SNCBs, 2022).

### 8.69.3.7 Shag

#### Status

2818. The Saltee Islands SPA breeding shag population stood at 268 pairs, or 536 breeding adults, in the period 1998 – 2000 (NPWS, 2012b). The combined total of the most recent counts on Great Saltee (138 pairs in 2003) and Little Saltee (28 pairs in 2000) is 166 pairs, or 332 breeding adults (JNCC, 2023a); this is used as the reference population for the assessment.

2819. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.142 (1 – 0.858; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 47 breeding adults.

#### Functional linkage and seasonal apportionment of potential effects

2820. The mean maximum foraging range of shag is 13.2km (±10.5km) and the maximum foraging range is 46km (Woodward *et al.*, 2019). The Project is located approximately 265km from Saltee Islands SPA, which means that the Project is beyond the maximum foraging range for shags from the SPA. No impacts during the breeding season from the Project are therefore apportioned to shags breeding at this SPA.

2821. Outside the breeding season, breeding shags from the SPA are not tied to the colony and therefore have the potential to mix with birds of all ages from breeding colonies in the UK and beyond. However, Furness (2015) stated that adult shags showed only limited migration, presenting evidence to suggest that the majority of adults moved less than 50km from their breeding colony. Furthermore, Furness (2015) stated that only 3% of immature birds, and no adult birds, from Irish colonies were present within the UK Wales & SW England waters BDMPS during the non-breeding season. Given the distance of the windfarm site from the SPA (i.e. c.265km) and the low numbers of birds recorded within the Project area (mean peak density 0.02 birds/km$^2$ / <4 birds within the windfarm site and 2km buffer during the non-breeding period), it is concluded that it is very unlikely that breeding adult shags from the Saltee Islands SPA will occur at the windfarm site. Accordingly, no impacts during the non-breeding season from the Project are apportioned to shags breeding at this SPA.

#### Potential effects on the qualifying feature from the Project-alone

2822. No effects on shags from Saltee Islands SPA are predicted. **Therefore, it is concluded that there would be no adverse effect on the integrity of the Saltee Islands SPA for the Project-alone**.

### Potential effects on the qualifying feature in-combination with other projects

2823. As the Project would have no measurable effect on shag populations from the Saltee Islands SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of Saltee Islands SPA, when assessed in-combination with other plans or projects**.

### 8.69.3.8 Cormorant

### Status

2824. The Saltee Islands SPA breeding cormorant population stood at 273 pairs, or 546 breeding adults, in the period 1998 – 2000 (NPWS, 2012b). The combined total of the most recent counts on Great Saltee (115 pairs in 2012) and Little Saltee (273 pairs in 2000) is 388 pairs, or 776 breeding adults (JNCC, 2023a); this is used as the reference population for the assessment.

2825. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.132 (1 – 0.868; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 102 breeding adults.

### Functional linkage and seasonal apportionment of potential effects

2826. The mean maximum foraging range of cormorant is 25.6km (±8.3km) and the maximum foraging range is 35km (Woodward *et al*., 2019). The Project is located approximately 265km from Saltee Islands SPA, which means the Project is beyond the maximum foraging range of cormorants from the SPA. No impacts during the breeding season from the Project are therefore apportioned to cormorants breeding at this SPA.

2827. Outside the breeding season, breeding cormorants from the SPA are assumed to range widely and to mix with birds of all ages from breeding colonies in Ireland, the UK and beyond. However, as no cormorants were recorded within the windfarm site or 2km buffer, it can be concluded that no birds from Saltee Islands SPA are likely to occur at the windfarm site.

### Potential effects on the qualifying feature from the Project-alone

2828. No effects on cormorants from Saltee Islands SPA are predicted. **Therefore, it is concluded that there would be no adverse effect on the integrity of the Saltee Islands SPA for the Project-alone**.

### Potential effects on the qualifying feature in-combination with other projects

2829. As the Project would have no measurable effect on cormorant populations from the Saltee Islands SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would** be **no adverse effect on the integrity of Saltee Islands SPA, when assessed in-combination with other plans or projects**.

# 8.70 Horn Head to Fanad Head SPA (transboundary site)

2830. Horn Head to Fanad Head SPA is located on the north coast of Ireland approximately 291km from the windfarm site (straight-line distance) or 334km (across sea).

## 8.70.1 Description of designation

2831. Horn Head to Fanad Head SPA comprises a number of separate sections of the north Co. Donegal coastline stretching some 70km eastwards from Dooros Point, south-west of Horn Head to just south of Saldanha Head, south of Fanad Head. The site includes the high coast areas and sea cliffs, land adjacent to the cliff edge and the sand dunes and lake at Dunfanaghy/Rinclevan. The high water mark forms the seaward boundary, except at Horn Head where the adjacent sea area to a distance of 500 m from the cliff base is included. Sea cliffs are present along virtually all the site; almost all are greater than 10m in height, but often over 30m and up to 200m in places. The qualifying seabird species of the SPA comprise fulmar, cormorant, shag, kittiwake, guillemot and razorbill.

## 8.70.2 Conservation objectives

2832. The conservation objective of the SPA is 'to maintain or restore the favourable conservation condition of the bird species listed as Special Conservation Interests for this SPA.'

## 8.70.3 Assessment

2833. Four qualifying features of Horn Head to Fanad Head SPA have been screened into the Appropriate Assessment (**Table 5.2**): fulmar, kittiwake, shag and cormorant.

### 8.70.3.1 Fulmar

#### Status

2834. The Horn Head to Fanad Head SPA breeding fulmar population stood at 1,974 pairs, or 3,948 breeding adults, in 1999 (NPWS, 2014a). The most recent estimate is 540 pairs, or 1,080 breeding adults, in 2015 (JNCC, 2023a); however, it is unclear if this count covered the full extent of the SPA. The 1999 population is, therefore, used as the reference population for the assessment.

2835. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.064 (1 – 0.936; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 273 breeding adults.

### Functional linkage and seasonal apportionment of potential effects

2836. The mean maximum foraging range of fulmar is 542.3km (±657.9km) and the maximum foraging range is 2,736km (Woodward *et al*., 2019). The Project is located approximately 334km from Horn Head to Fanad Head SPA, which means that the Project is within the mean maximum foraging range of fulmars breeding at this SPA.

### Potential effects on the qualifying feature from the Project-alone

2837. Fulmar are considered to have low vulnerability to both collision and displacement impacts from offshore windfarms (Bradbury *et al*., 2014). This species was recorded infrequently within the windfarm site (in only five of 24 surveys, all during the breeding season (January to August; Furness, 2015)) and in low numbers, with densities in four of the five months less than 0.1 birds/km$^2$. A peak density of 0.23 birds/km$^2$ was recorded in May 2022.

2838. On the basis of the low vulnerability of fulmars to potential impacts, and the low frequency and abundance of birds present at the windfarm site, it is concluded that any effects on breeding populations at Horn Head to Fanad Head SPA are very unlikely, both during and outside of the breeding season.

2839. **It is therefore concluded that there would be no measurable effects on fulmar due to the project alone, and no adverse effect on the integrity of the Horn Head to Fanad Head SPA is predicted.**

### Potential effects on the qualifying feature in-combination with other projects

2840. As the Project would have no measurable effect on fulmar populations from the Horn Head to Fanad Head SPA, there would be no contribution to any in-combination effects on this feature. Therefore, it is concluded that there would be no adverse effect on the integrity of Horn Head to Fanad Head SPA, when assessed in-combination with other plans or projects.

### 8.70.3.2 Kittiwake

### Status

2841. The Horn Head to Fanad Head SPA breeding kittiwake population stood at 3,853 pairs, or 7,706 breeding adults, in 1999 (NPWS, 2014a). The most recent estimate is 2,015 pairs, or 4,030 breeding adults, in 2015 (JNCC, 2023a); however, it is unclear if this count covered the full extent of the SPA. The 1999 population is, therefore, used as the reference population for the assessment.

2842. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.146 (1 – 0.854; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 1,125 breeding adults.

### Functional linkage and seasonal apportionment of potential effects

2843. The mean maximum foraging range of kittiwake is 156.1km (±144.5km) and the maximum foraging range is 770km (Woodward *et al*., 2019). The Project is located approximately 297km from Horn Head to Fanad Head SPA. Although theoretically within the mean maximum +1SD mean maximum foraging range of the SPA, the component kittiwake colonies of the SPA are all located beyond this distance from the Project site (i.e. >300.6km). The Project site is located within the maximum foraging range for kittiwake, but this is considered a poor indicator of typical foraging behaviour. It would be expected that few birds or foraging trips will occur at this distance from the colony, and even fewer with any regularity. No impacts during the breeding season are therefore apportioned to birds breeding at this colony.

2844. Outside the breeding season, breeding kittiwakes from the SPA are assumed to range widely and to mix with kittiwakes of all ages from breeding colonies in the UK, Ireland and beyond. The relevant non-breeding season reference population is the UK Western Waters and Channel BDMPS, consisting of 911,586 individuals during autumn migration (September to December) and 627,816 during spring migration (January to February) (Furness, 2015).

2845. Furness (2015) estimated that 30% of breeding adults from Ireland colonies are present within the UK Western Waters and Channel BDMPS during the autumn and spring migration periods. This proportion of Horn Head to Fanad Head SPA birds is therefore assumed, which is 2,312 birds. This represents 0.25% and 0.37% of the BDMPS population for the autumn and spring periods respectively. During autumn migration and spring migration, 0.25%, and 0.37% of impacts are therefore considered to affect birds from the SPA.

### Potential effects on the qualifying feature from the Project-alone

2846. The kittiwake qualifying feature of the Horn Head to Fanad Head SPA has been screened into the Appropriate Assessment due to the potential risk of collision.

#### Operation and maintenance phase collision risk

2847. Information for collision risk on breeding adult kittiwakes belonging to the Horn Head to Fanad Head SPA population is presented in **Table 8.178**. Collision estimates, calculated using the sCRM, are presented by biological season. A summary of the annual outputs and the corresponding increase in the annual baseline mortality rate is also presented. Parameters used in the sCRM were agreed with Natural England during the ETG process and are described in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES.

2848. Based on the mean collision rates, the annual total of breeding adult kittiwakes from Horn Head to Fanad Head SPA at risk of collision as a result of the Project is less than one bird (0.02). This would increase the existing mortality of the SPA breeding population by 0.00%.

*Table 8.178 Predicted seasonal and annual collision mortality (Stochastic model Option 2, avoidance rate 0.993 (±0.0003)), for breeding adult kittiwakes at the windfarm site, apportioned to Horn Head to Fanad Head SPA, with corresponding increases to baseline mortality of the population*

| Period | Breeding Season Mar-Aug | Autumn Migration Sep-Dec | Non-breeding/winter - | Spring Migration Jan-Feb | Annual Jan-Dec |
|---|---|---|---|---|---|
| Total collisions[1] (mean and 95% CIs) | 15.36 (4.17-33.87) | 8.49 (2.20-18.85) | - | 0.63 (0.00-1.46) | 24.48 (6.37-54.17) |
| % apportioned to the SPA | 0.00% | 0.25% | - | 0.37% | - |
| Total SPA collisions (mean and 95% CIs) | 0.00 (0.00-0.00) | 0.02 (0.01-0.05) | - | 0.00 (0.00-0.01) | 0.02 (0.01-0.05) |
| Mortality increase[2] (mean and 95% CIs) | 0.00% (0.00-0.00%) | 0.00% (0.00-0.00%) | - | 0.00% (0.00-0.00%) | 0.00% (0.00-0.00%) |

[1] Breeding season collision values reduced to 94.1% of modelled value to reflect proportion of adult birds recorded during site surveys

[2] Assuming predicted annual SPA mortality of 1,125 birds (7,706 x 0.146)

2849. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that no detectable changes in mortality rates would occur on this population for the mean monthly collision estimates for the Project, or for the upper 95% CI collision estimate.

2850. **It is concluded that based on predicted kittiwake mortality due to collision at the windfarm site there is no potential for the Project to have an adverse effect on the integrity of the Horn Head to Fanad Head SPA**.

2851. The confidence in the assessment is high. The evidence used to define the CRM input parameters presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** as part of the ES submitted alongside this RIAA is of high applicability and quality. Whilst there is uncertainty around some of the input parameters (e.g. avoidance rate), the rates selected are considered to be sufficiently precautionary based on expert opinion to provide confidence that collision rates are not underestimated.

### Potential effects on the qualifying feature in-combination with other projects

2852. As the Project would have no measurable effect on kittiwake populations from the SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of Horn Head to Fanad Head SPA, when assessed in-combination with other plans or projects**.

#### 8.70.3.3 Shag

### Status

2853. The Horn Head to Fanad Head SPA breeding shag population stood at 110 pairs, or 220 breeding adults, in 1999 (NPWS, 2014a). The most recent count is 92 pairs, or 184 breeding adults, in 2015 (JNCC, 2023a); this is used as the reference population for the assessment.

2854. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.142 (1 – 0.858; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 26 breeding adults.

### Functional linkage and seasonal apportionment of potential effects

2855. The mean maximum foraging range of shag is 13.2km (±10.5km) and the maximum foraging range is 46km (Woodward *et al*., 2019). The Project is located approximately 334km from Horn Head to Fanad Head SPA, which means that the Project is beyond the maximum foraging range for shags from the SPA. No impacts during the breeding season from the Project are therefore apportioned to shags breeding at this SPA.

2856. Outside the breeding season, breeding shags from the SPA are not tied to the colony and therefore have the potential to mix with birds of all ages from breeding colonies in the UK and beyond. However, Furness (2015) stated that adult shags showed only limited migration, presenting evidence to suggest that the majority of adults moved less than 50km from their breeding colony. Furthermore, Furness (2015) stated that only 3% of immature birds, and no adult birds, from Irish colonies were present within the UK Wales & SW England waters BDMPS during the non-breeding season. Given the distance of the windfarm site from the SPA (i.e. c.334km) and the low numbers of birds recorded within the Project area (mean peak density 0.02 birds/km$^2$ / <4 birds within the windfarm site and 2km buffer during the non-breeding period), it is concluded that it is very unlikely that breeding adult shags from the Horn Head to Fanad Head SPA will occur at the windfarm site. Accordingly, no impacts during the non-breeding season from the Project are apportioned to shags breeding at this SPA.

### Potential effects on the qualifying feature from the Project-alone

2857. No effects on shags from Horn Head to Fanad Head SPA are predicted. Therefore, it is concluded that there would be **no adverse effect on the integrity of the Horn Head to Fanad Head SPA for the Project-alone**.

### Potential effects on the qualifying feature in-combination with other projects

2858. As the Project would have no measurable effect on shag populations from the Horn Head to Fanad Head SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of Horn Head to Fanad Head SPA, when assessed in-combination with other plans or projects**.

### 8.70.3.4 Cormorant

### Status

2859. The Horn Head to Fanad Head SPA breeding cormorant population stood at 79 pairs, or 158 breeding adults, in 1999 (NPWS, 2014a). The SMP database (JNCC, 2023a) did not provide a more recent estimate, therefore the 1999 count has been used as the reference population for the assessment.

2860. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.132 (1 – 0.868; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 21 breeding adults.

### Functional linkage and seasonal apportionment of potential effects

2861. The mean maximum foraging range of cormorant is 25.6km (±8.3km) and the maximum foraging range is 35km (Woodward *et al*., 2019). The Project is located approximately 334km from Horn Head to Fanad Head SPA, which means the Project is beyond the maximum foraging range of cormorants from the SPA. No impacts during the breeding season from the Project are therefore apportioned to cormorants breeding at this SPA.

2862. Outside the breeding season, breeding cormorants from the SPA are assumed to range widely and to mix with birds of all ages from breeding colonies in Ireland, the UK and beyond. However, as no cormorants were recorded within the windfarm site or 2km buffer, it can be concluded that no birds from Horn Head to Fanad Head SPA are likely to occur at the windfarm site.

### Potential effects on the qualifying feature from the Project-alone

2863. No effects on cormorants from Horn Head to Fanad Head SPA are predicted. **Therefore, it is concluded that there would be no adverse effect on the integrity of the Horn Head to Fanad Head SPA for the Project-alone**.

### Potential effects on the qualifying feature in-combination with other projects

2864. As the Project would have no measurable effect on cormorant populations from the Horn Head to Fanad Head SPA, there would be no contribution to any in-combination effects on this feature**. Therefore, it is concluded that there would be no adverse effect on the integrity of Horn Head to Fanad Head SPA, when assessed in-combination with other plans or projects**.

## 8.71 West Donegal Coast SPA (transboundary site)

2865. West Donegal Coast SPA is located on the north coast of Ireland approximately 327km from the windfarm site (straight line) or 376km (across sea).

### 8.71.1 Description of designation

2866. The West Donegal Coast SPA comprises separate sections of the Co. Donegal coastline and extends from Muckros Head in the south, northwards to Slieve League, Malin Beg, Rocky Point, Glen Head, Slieve Tooey, Maghera, Loughros Point, Dunmore Head, Aran Island, Magheradrumman, Carrickfin, Carnboy, Bunbeg, Magheragallan, Lunniagh, as far as Carrick, to the south of Bloody Foreland. The site includes the high coast areas and sea cliffs of the mainland and Aran Island, the land adjacent to the cliff, areas of sand dunes/machair at Maghera, Mullaghderg, Braade/Carrickfin/Carnboy, Magheragallan and Lunniagh/Carrick, and also several areas further inland of the coast at Croaghmuckros and Slieve League, north of Glencolumbkille and south of Dunmore Head. The qualifying seabird species of the SPA comprise fulmar, cormorant, shag, herring gull, kittiwake and razorbill.

### 8.71.2 Conservation objectives

2867. The conservation objective of the SPA is 'to maintain or restore the favourable conservation condition of the bird species listed as Special Conservation Interests for this SPA.'

### 8.71.3 Assessment

2868. Three qualifying features of West Donegal Coast SPA have been screened into the Appropriate Assessment (**Table 5.2**): fulmar, shag and cormorant.

#### 8.71.3.1 Fulmar

##### Status

2869. The West Donegal Coast SPA breeding fulmar population stood at 1,879 pairs, or 3,758 breeding adults, in 1999 (NPWS, 2015j). The SMP database (JNCC, 2023a) did not provide a more recent estimate, therefore the 1999 count has been used as the reference population for the assessment.

2870. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.064 (1 – 0.936; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 241 breeding adults.

### Functional linkage and seasonal apportionment of potential effects

2871. The mean maximum foraging range of fulmar is 542.3km (±657.9km) and the maximum foraging range is 2,736km (Woodward *et al*., 2019). The Project is located approximately 376km from West Donegal Coast SPA, which means that the Project is within the mean maximum foraging range of fulmars breeding at this SPA.

### Potential effects on the qualifying feature from the Project-alone

2872. Fulmar are considered to have low vulnerability to both collision and displacement impacts from offshore windfarms (Bradbury *et al*., 2014). This species was recorded infrequently within the windfarm site (in only five of 24 surveys, all during the breeding season (January to August; Furness, 2015)) and in low numbers, with densities in four of the five months less than 0.1 birds/km$^2$. A peak density of 0.23 birds/km$^2$ was recorded in May 2022.

2873. On the basis of the low vulnerability of fulmars to potential impacts, and the low frequency and abundance of birds present at the windfarm site, it is concluded that any effects on breeding populations at West Donegal Coast SPA are very unlikely, both during and outside of the breeding season.

2874. **It is therefore concluded that there would be no measurable effects on fulmar due to the project alone, and no adverse effect on the integrity of the West Donegal Coast SPA is predicted.**

### Potential effects on the qualifying feature in-combination with other projects

2875. As the Project would have no measurable effect on fulmar populations from the West Donegal Coast SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of West Donegal Coast SPA, when assessed in-combination with other plans or projects.**

### 8.71.3.2 Shag

### Status

2876. The West Donegal Coast SPA breeding shag population stood at 86 pairs, or 172 breeding adults, in 1999 (NPWS, 2015a). The SMP database (JNCC, 2023a) did not provide a more recent estimate, therefore the 1999 count has been used as the reference population for the assessment.

2877. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.142 (1 – 0.858; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 24 breeding adults.

### Functional linkage and seasonal apportionment of potential effects

2878. The mean maximum foraging range of shag is 13.2km (±10.5km) and the maximum foraging range is 46km (Woodward *et al*., 2019). The Project is located approximately 376km from West Donegal Coast SPA, which means that the Project is beyond the maximum foraging range for shags from the SPA. No impacts during the breeding season from the Project are therefore apportioned to shags breeding at this SPA.

2879. Outside the breeding season, breeding shags from the SPA are not tied to the colony and therefore have the potential to mix with birds of all ages from breeding colonies in the UK and beyond. However, Furness (2015) stated that adult shags show only limited migration, with evidence to suggest that the majority of adults moved less than 50km from their breeding colony. Furthermore, Furness (2015) stated that only 3% of immature birds, and no adult birds, from Irish colonies are present within the UK Wales & SW England waters BDMPS during the non-breeding season. Given the distance of the windfarm site from the SPA (i.e. c.376km) and the low numbers of birds recorded within the Project area (mean peak density 0.02 birds/km$^2$ / <4 birds within the windfarm site and 2km buffer during the non-breeding period), it is concluded that it is very unlikely that breeding adult shags from the West Donegal Coast SPA will occur at the windfarm site. Accordingly, no impacts during the non-breeding season from the Project are apportioned to shags breeding at this SPA.

### Potential effects on the qualifying feature from the Project-alone

2880. No effects on shags from West Donegal Coast SPA are predicted. **Therefore, it is concluded that there would be no adverse effect on the integrity of the West Donegal Coast SPA for the Project-alone**.

### Potential effects on the qualifying feature in-combination with other projects

2881. As the Project would have no measurable effect on shag populations from the West Donegal Coast SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of West Donegal Coast SPA, when assessed in-combination with other plans or projects**.

## 8.71.3.3 Cormorant

### Status

2882. The West Donegal Coast SPA breeding cormorant population stood at 71 pairs, or 142 breeding adults, in 1999 and 2006 (NPWS, 2015a). The SMP database (JNCC, 2023a) did not provide a more recent estimate, therefore the

1999 and 2006 counts have been used as the reference population for the assessment.

2883. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.132 (1 – 0.868; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 19 breeding adults.

### Functional linkage and seasonal apportionment of potential effects

2884. The mean maximum foraging range of cormorant is 25.6km (±8.3km) and the maximum foraging range is 35km (Woodward *et al*., 2019). The Project is located approximately 376km from West Donegal Coast SPA, which means the Project is beyond the maximum foraging range of cormorants from the SPA. No impacts during the breeding season from the Project are therefore apportioned to cormorants breeding at this SPA.

2885. Outside the breeding season, breeding cormorants from the SPA are assumed to range widely and to mix with birds of all ages from breeding colonies in Ireland, the UK and beyond. However, as no cormorants were recorded within the windfarm site or 2km buffer, it can be concluded that no birds from West Donegal Coast SPA are likely to occur at the windfarm site.

### Potential effects on the qualifying feature from the Project-alone

2886. No effects on cormorants from West Donegal Coast SPA are predicted. Therefore, it is concluded that there would be **no adverse effect on the integrity of the West Donegal Coast SPA for the Project-alone**.

### Potential effects on the qualifying feature in-combination with other projects

2887. As the Project would have no measurable effect on cormorant populations from the West Donegal Coast SPA, there would be no contribution to any in-combination effects on this feature**. Therefore, it is concluded that there would be no adverse effect on the integrity of West Donegal Coast SPA, when assessed in-combination with other plans or projects**.

## 8.72 Tory Island SPA (transboundary site)

2888. Tory Island SPA is located on the north coast of Ireland approximately 329km from the windfarm site (straight line) or 373km (across sea).

### 8.72.1 Description of designation

2889. Tory Island is a remote, rocky island lying some 11km to the north of Bloody Foreland in County Donegal. The island is around 4km long by 1km wide. Steep cliffs occur along the northern and eastern coastlines, whereas the southern shore is low-lying. A marine area, extending 500 m from the base of the cliffs along the eastern and north-east side of the island, is included within the site. The qualifying seabird species of the SPA comprise fulmar, razorbill and puffin.

### 8.72.2 Conservation objectives

2890. The conservation objective of the SPA is 'to maintain or restore the favourable conservation condition of the bird species listed as Special Conservation Interests for this SPA.'

### 8.72.3 Assessment

2891. One qualifying feature of Tory Island SPA has been screened into the Appropriate Assessment (**Table 5.2**): breeding fulmar.

#### 8.72.3.1 Fulmar

##### Status

2892. The Tory Island SPA breeding fulmar population stood at 641 pairs, or 1,282 breeding adults, in 1999 (NPWS, 2015b). The most recent count is 468 pairs (AON), or 936 breeding adults, in 2015 (JNCC, 2023a); this is used as the reference population for the assessment.

2893. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.064 (1 – 0.936; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 60 breeding adults.

##### Functional linkage and seasonal apportionment of potential effects

2894. The mean maximum foraging range of fulmar is 542.3km (±657.9km) and the maximum foraging range is 2,736km (Woodward *et al*., 2019). The Project is located approximately 373km from Tory Island SPA, which means that the Project is within the mean maximum foraging range of fulmars breeding at this SPA.

### Potential effects on the qualifying feature from the Project-alone

2895. Fulmar are considered to have low vulnerability to both collision and displacement impacts from offshore windfarms (Bradbury *et al*., 2014). This species was recorded infrequently within the windfarm site (in only five of 24 surveys, all during the breeding season (January to August; Furness, 2015)) and in low numbers, with densities in four of the five months less than 0.1 birds/km$^2$. A peak density of 0.23 birds/km$^2$ was recorded in May 2022.

2896. On the basis of the low vulnerability of fulmars to potential impacts, and the low frequency and abundance of birds present at the windfarm site, it is concluded that any effects on breeding populations at Tory Island SPA are very unlikely, both during and outside of the breeding season.

2897. **It is therefore concluded that there would be no measurable effects on fulmar due to the project alone, and no adverse effect on the integrity of the Tory Island SPA is predicted.**

### Potential effects on the qualifying feature in-combination with other projects

2898. As the Project would have no measurable effect on fulmar populations from the Tory Island SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of Tory Island SPA, when assessed in-combination with other plans or projects.**

# 8.73 Cliffs of Moher SPA (transboundary site)

2899. Cliffs of Moher SPA is located on the west coast of Ireland approximately 387km from the windfarm site (straight line) or 697km (across sea).

## 8.73.1 Description of designation

2900. Cliffs of Moher SPA extends a distance of some 9.5 km along the north Co. Clare coast from Faunmore in the north to just south of Cancregga Point in the south. The site includes the cliffs, which rise to over 200m, the land adjacent to the cliff edge (inland for 300 m) as well as the adjacent sea area to a distance of up to 500 m from the cliff base. The qualifying bird species of the SPA comprise fulmar, kittiwake, guillemot, razorbill and puffin.

## 8.73.2 Conservation objectives

2901. The conservation objective of the SPA is 'to maintain or restore the favourable conservation condition of the bird species listed as Special Conservation Interests for this SPA.'

## 8.73.3 Assessment

2902. Four qualifying features of Cliffs of Moher SPA have been screened into the Appropriate Assessment (**Table 5.2**): fulmar, guillemot, razorbill and kittiwake.

### 8.73.3.1 Fulmar

#### Status

2903. The Cliffs of Moher SPA breeding fulmar population stood at 3,566 pairs, or 7,132 breeding adults, in 1998/99 (NPWS, 2015c). The most recent count is 4,801 pairs, or 9,602 breeding adults, in 2015 (JNCC, 2023a); this is used as the reference population for the assessment.

2904. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.064 (1 – 0.936; Horswill and Robinson, 2015), the expected annual mortality from the SPA population would be 307 breeding adults.

#### Functional linkage and seasonal apportionment of potential effects

2905. The mean maximum foraging range of fulmar is 542.3km (±657.9km) and the maximum foraging range is 2,736km (Woodward *et al*., 2019). The Project is located approximately 697km from Cliffs of Moher SPA, which means that the Project is beyond the mean maximum foraging range of fulmars breeding at this SPA, but within the mean maximum foraging range +1SD, and the maximum foraging range.

### Potential effects on the qualifying feature from the Project-alone

2906. Fulmar are considered to have low vulnerability to both collision and displacement impacts from offshore windfarms (Bradbury *et al.*, 2014). This species was recorded infrequently within the windfarm site (in only five of 24 surveys, all during the breeding season (January to August; Furness, 2015) and in low numbers, with densities in four of the five months less than 0.1 birds/km$^2$. A peak density of 0.23 birds/km$^2$ was recorded in May 2022.

2907. On the basis of the low vulnerability of fulmars to potential impacts, and the low frequency and abundance of birds present at the windfarm site, it is concluded that any effects on breeding populations at Cliffs of Moher SPA are very unlikely, both during and outside of the breeding season.

2908. **It is therefore concluded that there would be no measurable effects on fulmar due to the project alone, and no adverse effect on the integrity of the Cliffs of Moher SPA is predicted.**

### Potential effects on the qualifying feature in-combination with other projects

2909. As the Project would have no measurable effect on fulmar populations from the Cliffs of Moher SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of Cliffs of Moher SPA, when assessed in-combination with other plans or projects.**

### 8.73.3.2 Guillemot

### Status

2910. The Cliffs of Moher SPA breeding guillemot population stood at 13,375 pairs, or 26,750 breeding adults, in 1998/99 (NPWS, 2015c). The most recent count is 34,827 individuals in 2015 (JNCC, 2023a); this is used as the reference population for the assessment.

2911. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.061 (1 – 0.939; Horswill and Robinson, 2015), the expected annual mortality rate from the SPA population would be 2,124 breeding adults.

### Functional linkage and seasonal apportionment of potential effects

2912. The mean maximum foraging range of guillemot is 73.2km (±80.5km) and the maximum foraging range is 338km (Woodward *et al*., 2019). The Project is located approximately 697km from Cliffs of Moher SPA, which means that the Project is beyond the mean maximum foraging range, mean maximum foraging range +1SD and the maximum foraging range of guillemots breeding at this SPA. No impacts during the breeding season are therefore apportioned to birds breeding at this colony.

2913. Outside the breeding season, Furness (2015) apportioned no guillemots from breeding colonies from the west coast of Ireland to the UK Western Waters BDMPS. On that basis, it is concluded that significant numbers of birds from Cliffs of Moher SPA are unlikely to occur at the windfarm site, and no impacts during the non-breeding season are therefore apportioned to birds breeding at this colony.

### Potential effects on the qualifying feature

#### Operation and maintenance phase disturbance/displacement/barrier effects

*Project-alone*

2914. As no guillemots present at the windfarm site are apportioned to the Cliffs of Moher SPA, it is concluded that predicted guillemot mortality due to operational phase displacement for the project alone would not adversely affect the integrity of the Cliffs of Moher SPA.

*In-combination*

2915. As the Project would have no measurable effect on guillemot populations from the Cliffs of Moher SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of Cliffs of Moher SPA, when assessed in-combination with other plans or projects.**

## 8.73.3.3 Razorbill

### Status

2916. The Cliffs of Moher SPA breeding razorbill population stood at 5,159 pairs, or 10,318 breeding adults, in 1998/99 (NPWS, 2015c). The most recent count was 4,046 individuals in 2015 (JNCC, 2023a); this is used as the reference population for the assessment.

2917. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.105 (1 – 0.895; Horswill and Robinson, 2015), the expected annual mortality from the SPA population would be 452 breeding adults

### Functional linkage and seasonal apportionment of potential effects

2918. The mean maximum foraging range of razorbill is 88.7km (±75.9km) and the maximum foraging range is 313km (Woodward *et al*., 2019). The Project is located approximately 697km from Cliffs of Moher SPA, which means the Project is beyond the maximum foraging range of razorbills breeding at this

SPA. No impacts during the breeding season are therefore apportioned to birds breeding at this colony.

2919. Outside the breeding season, breeding razorbills from the SPA are assumed to range widely and to mix with razorbills of all ages from breeding colonies in Ireland, the UK and beyond. The relevant non-breeding season reference population is the UK Western Waters BDMPS, consisting of 606,914 individuals during autumn and spring passage periods (August to October and January to March), and 341,422 individuals during winter (November and December) (Furness, 2015).

2920. As no published estimate is available, it is assumed that 10% of Cliffs of Moher SPA breeding adults are present within the UK Western Waters BDMPS during the non-breeding season, and 10% during the winter period. This is the rate for 'Ireland' populations identified by Furness (2015), and is considered appropriate given the geographic isolation between the west coast of Ireland (where the SPA is located) and the windfarm site. It is assumed that the most recent count prior to the publication of Furness (2015) was used to inform the BDMPS total which is 10,318 breeding adults; 10% of this population is 1,032 birds. This represents 0.2% and 0.3% of the BDMPS population (606,914 during spring/autumn and 341,422 during winter) respectively. These percentages (i.e. 0.2% (spring and autumn migration) and 0.3% (winter)) are the proportions of birds present at the windfarm site that are presumed to originate from the Cliffs of Moher SPA during the relevant seasons.

## Potential effects on the qualifying feature

### Operation and maintenance phase disturbance/displacement/barrier effects

*Project-alone*

2921. The year-round mean peak abundance of razorbills present within the windfarm site and 2km buffer was 1,979 (703-3,552) individuals (refer to **Appendix 12.1** of the ES). Of these, 4 birds (2-7) were likely to be breeding adults from the Cliffs of Moher SPA.

2922. **Table 8.179** sets out the predicted impacts on razorbills from Cliffs of Moher SPA. Displacement rates of 30% to 70% are considered for this species, along with a range of mortality rates of 1% to 10% of displaced birds (UK SNCBs, 2017).

2923. The available evidence suggests that the upper ranges of these displacement and mortality rates may be excessively precautionary. Whilst it is true that guillemots and razorbills tend to be displaced by OWFs they do not avoid them completely, and displacement rates vary between sites. MacArthur Green (2019b) concluded that, on average, densities within OWFs were around half of the density found in the habitats around the OWF. At some OWFs there was also displacement of birds from a buffer zone surrounding it. The size of the buffer zone varied between OWFs and was generally less than 2km, with auk density increasing across the buffer zone as distance from turbines increased, up to the density in the wider area. Another recent review (APEM, 2022) found that the current displacement rates suggested by the SNCBs for guillemot and razorbill (30% to 70%) did not account for the quality of or confidence in the studies which were used to inform this position, and that studies where no significant effects were recorded were not accounted for during the provision of the advice. APEM (2022) suggested that in the case of Hornsea Project Four, an evidence-based displacement rate of 50% was appropriate. However, the study also recognised that larger displacement effects were possible.

2924. Mortality due to displacement could arise due to increased energy costs and/or decreased energy intake, if displacement results in increased flying time to avoid OWFs, and/or increased bird densities and competition for prey in areas of unimpacted habitat outside the OWF. Given that UK OWFs in the Irish and Celtic Seas represent a very small proportion of the available foraging habitat for guillemots and razorbills within UK Western Waters, the increase in density of auks outside OWFs due to displacement is likely to be negligible (MacArthur Green, 2019b). It is considered unlikely that the mortality rate due to displacement would be as high as the empirically estimated 10.5% annual mortality for adult razorbills that occurs due to the combination of ‘natural’ factors and anthropogenic activities (Horswill and Robinson, 2015). Indeed, it may be much lower; MacArthur Green (2019b) recommended precautionary rates of 50% displacement for auks and 1% mortality of displaced birds based on a review of available evidence. Modelling undertaken by APEM (2022) found that in the case of Hornsea Project Four, a mortality rate of 1% for displaced auks was considered to be precautionary, and that this may be an overestimate given that Hornsea Project Four is located towards the upper end of the mean maximum foraging range for guillemot from the closest designated sites.

2925. The full ranges of recommended displacement and mortality effects are considered by the assessment, along with evidence-based displacement and mortality rates of 50% and 1%, respectively (APEM, 2022; MacArthur Green, 2019b).

*Table 8.179 Razorbill – predicted operation and maintenance phase displacement and mortality from Cliffs of Moher SPA*

| Mean peak abundance estimate type | Mean peak abundance estimate by season[1] | Number of Cliffs of Moher SPA breeding adults present by season)[2] | Annual mortality range[3] | Annual baseline mortality increase range[4] |
|---|---|---|---|---|
| Upper 95% CI | 605 (b)<br>1,070 (aut)<br>1,297 (win)<br>580 (spr)<br>3,552 (year round) | 0 (b)<br>2 (aut)<br>4 (win)<br>1 (spr)<br>7 (year round) | 0-1 (0) | 0.01-0.12% (0.01%) |
| Mean | 252 (b)<br>694 (aut)<br>651 (win)<br>381 (spr)<br>1,979 (year round) | 0 (b)<br>1 (aut)<br>2 (win)<br>1 (spr)<br>4 (year round) | 0-0 (0) | 0.00-0.07% (0.00%) |
| Lower 95% CI | 21 (b)<br>309 (aut)<br>159 (win)<br>214 (spr)<br>703 (year round) | 0 (b)<br>1 (aut)<br>0 (win)<br>0 (spr)<br>2 (year round) | 0-0 (0) | 0.00-0.03% (0.00%) |

[1] Breeding season = b, autumn migration season = aut, winter season = win, spring migration season = spr

[2] Assumes breeding adult apportioning of 0.0% (breeding season), 0.1% (spring and autumn migration) and 0.1% (winter) to Cliffs of Moher SPA.

[3] Assumes displacement rates of 30-70% and mortality rates of 1-10%. Evidence-based estimates assuming a 50% displacement rate and 1% mortality of displaced birds are presented in parentheses.

[4] Background mortality rate of 10.5% (Horswill and Robinson, 2015)

2926. Based on the mean peak abundances, the annual total of razorbills from the Cliffs of Moher SPA at risk of displacement is 4 birds (**Table 8.179**). At displacement rates of 30% to 70%, and mortality rates of 1% to 10% for displaced birds, <1 (0-0) SPA breeding adults would be predicted to die each year due to displacement from the Project.

2927. Assuming a displacement rate of 70% and a mortality rate of 10% of displaced birds, annual mortality within the SPA breeding adult population would increase by 0.07%. Using an evidence-based displacement rate of 50%, and a mortality rate for displaced birds of 1%, annual mortality in the population would instead increase by <0.01% (<1 bird).

2928. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that no detectable changes in mortality rates would occur under almost any combination of displacement and mortality rates when the mean peak abundance estimate assessments are considered. This would be the case even when upper 95% CIs are considered.

2929. **It is concluded that predicted razorbill mortality due to operational phase displacement for the project alone would not adversely affect the integrity of the Cliffs of Moher SPA.**

2930. The confidence in the assessment is high for several reasons. Firstly, the evidence used to inform the evidence-based displacement rates is of high applicability and quality (based on the criteria discussed in **Chapter 12 Offshore Ornithology** of the ES). Whilst there was limited available evidence to inform mortality rates, 1% is considered to be sufficiently precautionary based on expert opinion. This species is not regarded as being highly specialised in its habitat requirements (Bradbury *et al*., 2014; Furness and Wade, 2012; Garthe and Hüppop, 2004), and it is therefore anticipated that displaced birds will find alternative habitat in the vast majority of cases. Finally, the conclusion of the assessment is the same irrespective of whether the mean or 95% upper CI mean peak abundances are used to calculate potential mortality and increases in the baseline mortality rate of the background population.

*In-combination*

2931. As there would be no measurable increase in mortality on razorbill populations from Cliffs of Moher SPA as a result of the Project, there would be no contribution to potential in-combination effects. **Therefore, it is concluded that there would be no adverse effect on the integrity of Cliffs of Moher SPA, when assessed in-combination with other plans or projects.**

#### 8.73.3.4 Kittiwake

#### Status

2932. The Cliffs of Moher SPA breeding kittiwake population stood at 7,698 pairs, or 15,396 breeding adults, in 1998/99 (NPWS, 2015c). The most recent count is 3,981 pairs (AON), plus a further 590 individuals, giving a combined total of 8,552 assumed breeding adults; this is used as the reference population for the assessment.

2933. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.146 (1 – 0.854; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 1,249 breeding adults.

## Functional linkage and seasonal apportionment of potential effects

2934. The mean maximum foraging range of kittiwake is 156.1km (±144.5km) and the maximum foraging range is 770km (Woodward *et al*., 2019). The Project is located approximately 387km from Cliffs of Moher SPA, which means the Project is beyond the mean maximum and mean maximum +1SD foraging range of breeding kittiwakes from the SPA, but within the maximum foraging range. The maximum foraging range is a poor indicator of typical foraging behaviour. It would be expected that few birds or foraging trips will occur at this distance from the colony, and even fewer with any regularity. No impacts during the breeding season are therefore apportioned to birds breeding at this colony.

2935. Outside the breeding season, breeding kittiwakes from the SPA are assumed to range widely and to mix with kittiwakes of all ages from breeding colonies in the UK, Ireland and beyond. The relevant non-breeding season reference population is the UK Western Waters and Channel BDMPS, consisting of 911,586 individuals during autumn migration (September to December) and 627,816 during spring migration (January to February) (Furness, 2015).

2936. Furness (2015) estimated that 30% of breeding adults from Ireland colonies are present within the UK Western Waters and Channel BDMPS during the autumn and spring migration periods. This proportion of Cliffs of Moher SPA birds is therefore assumed, which is 2,566 birds. This represents 0.28% and 0.41% of the BDMPS population for the autumn and spring periods respectively. During autumn migration and spring migration, 0.28%, and 0.41% of impacts are therefore considered to affect birds from the SPA.

## Potential effects on the qualifying feature from the Project-alone

2937. The kittiwake qualifying feature of the Cliffs of Moher SPA has been screened into the Appropriate Assessment due to the potential risk of collision.

### Operation and maintenance phase collision risk

2938. Information for collision risk on breeding adult kittiwakes belonging to the Cliffs of Moher SPA population is presented in **Table 8.180**. Collision estimates, calculated using the sCRM, are presented by biological season. A summary of the annual outputs and the corresponding increase in the annual baseline mortality rate is also presented. Parameters used in the sCRM were agreed with Natural England during the ETG process and are described in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES.

2939. Based on the mean collision rates, the annual total of breeding adult kittiwakes from Cliffs of Moher SPA at risk of collision as a result of the Project is less than one bird (0.03). This would increase the existing mortality of the SPA breeding population by 0.00%.

*Table 8.180 Predicted seasonal and annual collision mortality (Stochastic model Option 2, avoidance rate 0.993 (±0.0003)), for breeding adult kittiwakes at the windfarm site, apportioned to Cliffs of Moher SPA, with corresponding increases to baseline mortality of the population*

| | Breeding Season | Autumn Migration | Non-breeding/winter | Spring Migration | Annual |
|---|---|---|---|---|---|
| **Period** | **Mar-Aug** | **Sep-Dec** | **-** | **Jan-Feb** | **Jan-Dec** |
| Total collisions[1] (mean and 95% CIs) | 15.36 (4.17-33.87) | 8.49 (2.20-18.85) | - | 0.63 (0.00-1.46) | 24.48 (6.37-54.17) |
| % apportioned to the SPA | 0.00% | 0.22% | - | 0.32% | - |
| Total SPA collisions (mean and 95% CIs) | 0.00 (0.00-0.00) | 0.02 (0.01-0.05) | - | 0.00 (0.00-0.01) | 0.03 (0.01-0.06) |
| Mortality increase[2] (mean and 95% CIs) | 0.00% (0.00-0.00%) | 0.00% (0.00-0.00%) | - | 0.00% (0.00-0.00%) | 0.00-0.00% (0.00%) |

[1] Breeding season collision values reduced to 94.1% of modelled value to reflect proportion of adult birds recorded during site surveys
[2] Assuming predicted annual SPA mortality of 1,249 birds (8,552 x 0.146)

2940. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that no detectable changes in mortality rates would occur on this population for the mean monthly collision estimates for the Project, or for the upper 95% CI collision estimate.

2941. **It is concluded that based on predicted kittiwake mortality due to collision at the windfarm site there is no potential for the Project to have an adverse effect on the integrity of the Cliffs of Moher SPA.**

2942. The confidence in the assessment is high. The evidence used to define the CRM input parameters presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** as part of the ES submitted alongside this RIAA is of high applicability and quality. Whilst there is uncertainty around some of the input parameters (e.g. avoidance rate), the rates selected are considered to be sufficiently precautionary based on expert opinion to provide confidence that collision rates are not underestimated.

### Potential effects on the qualifying feature in-combination with other projects

2943. As the Project would have no measurable effect on kittiwake populations from the Cliffs of Moher SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of Cliffs of Moher SPA, when assessed in-combination with other plans or projects.**

## 8.74 Stags of Broad Haven SPA (transboundary site)

2944. Stags of Broad Haven SPA is located on the west coast of Ireland approximately 400km from the windfarm site (straight line) or 507km (across sea).

### 8.74.1 Description of designation

2945. The Stags of Broad Haven are a group of four precipitous rocky islets, totalling 4ha, rising to almost 100m, located about 2km north of Benwee Head, Co. Mayo. The surrounding seas to a distance of 500 m are included in the SPA. The qualifying seabird species of the SPA comprise storm petrel and Leach's petrel.

### 8.74.2 Conservation objectives

2946. The conservation objective of the SPA is 'to maintain or restore the favourable conservation condition of the bird species listed as Special Conservation Interests for this SPA.'

### 8.74.3 Assessment

2947. One qualifying feature of Stags of Broad Haven SPA has been screened into the Appropriate Assessment (**Table 5.2**): breeding Leach's storm-petrel.

#### 8.74.3.1 Leach's storm-petrel

##### Status

2948. The Stags of Broad Haven breeding Leach's storm-petrel population was estimated at 310 pairs, or 620 breeding adults, in 2001 (NPWS, 2015d). The SMP database (JNCC, 2023a) did not provide a more recent count, therefore the 2001 estimate has been used as the reference population for the assessment.

##### Functional linkage and seasonal apportionment of potential effects

2949. The mean foraging range of Leach's storm-petrel is 657km (Woodward *et al*., 2019); estimates for maximum and mean maximum foraging ranges are not available. The Project is located approximately 507km from Stags of Broad Haven SPA, which means that the Project is within the mean foraging range of Leach's storm-petrels breeding at this SPA.

##### Potential effects on the qualifying feature from the Project-alone

2950. Leach's storm-petrel was not recorded during baseline surveys of the windfarm site (including buffer areas). It is therefore concluded that this species does not occur regularly in this area. It is noted that Leach's storm-petrel is considered to have low vulnerability to collision risk and very low vulnerability to displacement impacts (Bradbury *et al*., 2014), and therefore the risk of significant effects would be low, even if this species occurred at the windfarm site.

2951. **It is therefore concluded that there would be no measurable effects on Leach's storm-petrel due to the project alone, and no adverse effect on the integrity of the Stags of Broad Haven SPA is predicted**.

### Potential effects on the qualifying feature in-combination with other projects

2952. As the Project would have no measurable effect on Leach's storm-petrel populations from the Stags of Broad Haven SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of Stags of Broad Haven SPA, when assessed in-combination with other plans or projects.**

# 8.75 Clare Island SPA (transboundary site)

2953. Clare Island SPA is located on the west coast of Ireland approximately 408km from the windfarm site (straight line) or 588km (across sea).

## 8.75.1 Description of designation

2954. Clare Island SPA lies at the entrance to Clew Bay, in Co. Mayo, and some 5km from the mainland. The site comprises all of the cliffs on the island, a length of approximately 10km, as well as the land adjacent to the cliff edge (inland for 300m) and the adjacent marine waters (to distances of 200m or 500m, depending on auk distribution). The qualifying seabird species for the SPA comprise fulmar, shag, common gull, kittiwake, guillemot, and razorbill.

## 8.75.2 Conservation objectives

2955. The conservation objective of the SPA is 'to maintain or restore the favourable conservation condition of the bird species listed as Special Conservation Interests for this SPA.'

## 8.75.3 Assessment

2956. One qualifying feature of Clare Island SPA has been screened into the Appropriate Assessment (**Table 5.2**): breeding fulmar.

### 8.75.3.1 Fulmar

#### Status

2957. The Clare Island SPA breeding fulmar population stood at 4,029 pairs, or 8,058 breeding adults, in 1999 (NPWS, 2014b). The most recent count is 667 pairs in 2015, however it is unclear if this count covered the full extent of the SPA. The 1999 population is, therefore, used as the reference population for the assessment.

2958. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.064 (1 – 0.936; Horswill and Robinson, 2015), the expected annual mortality from the SPA population would be 516 breeding adults.

#### Functional linkage and seasonal apportionment of potential effects

2959. The mean maximum foraging range of fulmar is 542.3km (±657.9km) and the maximum foraging range is 2,736km (Woodward *et al*., 2019). The Project is located approximately 588km from Clare Island SPA, which means that the Project is beyond the mean maximum foraging range of fulmars breeding at

this SPA, but within the mean maximum foraging range +1SD, and the maximum foraging range.

### Potential effects on the qualifying feature from the Project-alone

2960. Fulmar are considered to have low vulnerability to both collision and displacement impacts from offshore windfarms (Bradbury *et al*., 2014). This species was recorded infrequently within the windfarm site (in only five of 24 surveys, all during the breeding season (January to August; Furness, 2015)) and in low numbers, with densities in four of the five months less than 0.1 birds/km$^2$. A peak density of 0.23 birds/km$^2$ was recorded in May 2022.

2961. On the basis of the low vulnerability of fulmars to potential impacts, and the low frequency and abundance of birds present at the windfarm site, it is concluded that any effects on breeding populations at Clare Island SPA are very unlikely, both during and outside of the breeding season.

2962. **It is therefore concluded that there would be no measurable effects on fulmar due to the project alone, and no adverse effect on the integrity of the Clare Island SPA is predicted.**

### Potential effects on the qualifying feature in-combination with other projects

2963. As the Project would have no measurable effect on fulmar populations from the Clare Island SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of Clare Island SPA, when assessed in-combination with other plans or projects.**

## 8.76 Duvillaun Islands SPA (transboundary site)

2964. Duvillaun Islands SPA is located on the west coast of Ireland approximately 420km from the windfarm site (straight line) or 547km (across sea).

### 8.76.1 Description of designation

2965. Duvillaun Islands SPA comprises a group of marine islands, rocks and reefs, located between 1km and 5km off the southern tip of the Mullet Peninsula in Co. Mayo. The surrounding seas to a distance of 200m and the area of water between the islands are included in the site. The qualifying bird species of the SPA comprise fulmar and storm petrel.

### 8.76.2 Conservation objectives

2966. The conservation objective of the SPA is 'to maintain or restore the favourable conservation condition of the bird species listed as Special Conservation Interests for this SPA.'

### 8.76.3 Assessment

2967. One qualifying feature of Duvillaun Islands SPA has been screened into the Appropriate Assessment (**Table 5.2**): breeding fulmar.

#### 8.76.3.1 Fulmar

##### Status

2968. The Duvillaun Islands SPA breeding fulmar population stood at 638 pairs, or 1,276 breeding adults, in 2000 (NPWS, 2014c). The most recent count is 547 pairs, or 1,094 breeding adults, in 2015 (JNCC, 2023a); this is used as the reference population for the assessment.

2969. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.064 (1 – 0.936; Horswill and Robinson, 2015), the expected annual mortality from the SPA population would be 70 breeding adults.

##### Functional linkage and seasonal apportionment of potential effects

2970. The mean maximum foraging range of fulmar is 542.3km (±657.9km) and the maximum foraging range is 2,736km (Woodward *et al.*, 2019). The Project is located approximately 547km from Duvillaun Islands SPA, which means that the Project is beyond the mean maximum foraging range of fulmars breeding at this SPA, but within the mean maximum foraging range +1SD, and the maximum foraging range.

### Potential effects on the qualifying feature from the Project-alone

2971. Fulmar are considered to have low vulnerability to both collision and displacement impacts from offshore windfarms (Bradbury *et al*., 2014). This species was recorded infrequently within the windfarm site (in only five of 24 surveys, all during the breeding season (January to August; Furness, 2015)) and in low numbers, with densities in four of the five months less than 0.1 birds/km$^2$. A peak density of 0.23 birds/km$^2$ was recorded in May 2022.

2972. On the basis of the low vulnerability of fulmars to potential impacts, and the low frequency and abundance of birds present at the windfarm site, it is concluded that any effects on breeding populations at Duvillaun Islands SPA are very unlikely, both during and outside of the breeding season.

2973. **It is therefore concluded that there would be no measurable effects on fulmar due to the project alone, and no adverse effect on the integrity of the Duvillaun Islands SPA is predicted.**

### Potential effects on the qualifying feature in-combination with other projects

2974. As the Project would have no measurable effect on fulmar populations from the Duvillaun Islands SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of Duvillaun Islands SPA, when assessed in-combination with other plans or projects.**

# 8.77 High Island, Inishshark and Davillaun SPA (transboundary site)

2975. High Island, Inishshark and Davillaun SPA is located on the west coast of Ireland approximately 421km from the windfarm site (straight line) or 599km (across sea).

## 8.77.1 Description of designation

2976. High Island, Inishshark and Davillaun are small, uninhabited islands lying some 3-5 km north and west of Aughrus Point on the Co. Galway coast. Grassland is the main vegetation type found, with vegetated sea cliffs, dry heath, exposed rock and some freshwater marsh also present. The surrounding sea to a distance of 200 m from each island is included within the SPA. The qualifying seabird species of the SPA comprise fulmar and Arctic tern.

## 8.77.2 Conservation objectives

2977. The conservation objective of the SPA is 'to maintain or restore the favourable conservation condition of the bird species listed as Special Conservation Interests for this SPA.'

## 8.77.3 Assessment

2978. One qualifying feature of High Island, Inishshark and Davillaun SPA has been screened into the Appropriate Assessment (**Table 5.2**): breeding fulmar.

### 8.77.3.1 Fulmar

#### Status

2979. The High Island, Inishshark and Davillaun SPA breeding fulmar population stood at 830 pairs, or 1,660 breeding adults, in 2000 (NPWS, 2010a). The most recent combined counts from High Island (358 pairs/AOS in 2015) Inishshark (1,160 pairs/AOS in 2015) and Davillaun (43 pairs/AOS in 2016) give a total of 1,561 pairs, or 3,122 breeding adults (JNCC, 2023a); this is used as the reference population for the assessment.

2980. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.064 (1 – 0.936; Horswill and Robinson, 2015), the expected annual mortality from the SPA population would be 200 breeding adults.

### Functional linkage and seasonal apportionment of potential effects

2981. The mean maximum foraging range of fulmar is 542.3km (±657.9km) and the maximum foraging range is 2,736km (Woodward *et al*., 2019). The Project is located approximately 599km from High Island, Inishshark and Davillaun SPA, which means that the Project is beyond the mean maximum foraging range of fulmars breeding at this SPA, but within the mean maximum foraging range +1SD, and the maximum foraging range.

### Potential effects on the qualifying feature from the Project-alone

2982. Fulmar are considered to have low vulnerability to both collision and displacement impacts from offshore windfarms (Bradbury *et al*., 2014). This species was recorded infrequently within the windfarm site (in only five of 24 surveys, all during the breeding season (January to August; Furness, 2015)) and in low numbers, with densities in four of the five months less than 0.1 birds/km$^2$. A peak density of 0.23 birds/km$^2$ was recorded in May 2022.

2983. On the basis of the low vulnerability of fulmars to potential impacts, and the low frequency and abundance of birds present at the windfarm site, it is concluded that any effects on breeding populations at High Island, Inishshark and Davillaun SPA are very unlikely, both during and outside of the breeding season.

2984. **It is therefore concluded that there would be no measurable effects on fulmar due to the project alone, and no adverse effect on the integrity of the High Island, Inishshark and Davillaun SPA is predicted.**

### Potential effects on the qualifying feature in-combination with other projects

2985. As the Project would have no measurable effect on fulmar populations from the High Island, Inishshark and Davillaun SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of High Island, Inishshark and Davillaun SPA, when assessed in-combination with other plans or projects.**

## 8.78 Kerry Head SPA (transboundary site)

2986. Kerry Head SPA is located on the west coast of Ireland approximately 426km from the windfarm site (straight line) or 661km (across sea).

### 8.78.1 Description of designation

2987. Kerry Head SPA is situated on the south side of the mouth of the River Shannon in north Co. Kerry. It encompasses the sea cliffs from just west of Ballyheigue, around the end of Kerry Head to the west and north-eastwards as far as Kilmore. The site includes the sea cliffs and land adjacent to the cliff edge. The high water mark forms the seaward boundary There is one qualifying seabird species for this SPA; fulmar.

### 8.78.2 Conservation objectives

2988. The conservation objective of the SPA is 'to maintain or restore the favourable conservation condition of the bird species listed as Special Conservation Interests for this SPA.'

### 8.78.3 Assessment

2989. One qualifying feature of Kerry Head SPA has been screened into the Appropriate Assessment (**Table 5.2**): fulmar.

#### 8.78.3.1 Fulmar

##### Status

2990. The Kerry Head SPA breeding fulmar population stood at 421 pairs, or 842 breeding adults, in 2000 (NPWS, 2015e). The most recent count is 128 pairs, or 256 breeding adults (JNCC, 2023a) however it is unclear if this count covered the full extent of the SPA. The 2000 count is, therefore, used as the reference population for the assessment.

2991. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.064 (1 – 0.936; Horswill and Robinson, 2015), the expected annual mortality from the SPA population would be 54 breeding adults.

##### Functional linkage and seasonal apportionment of potential effects

2992. The mean maximum foraging range of fulmar is 542.3km (±657.9km) and the maximum foraging range is 2,736km (Woodward *et al*., 2019). The Project is located approximately 661km from Kerry Head SPA, which means that the Project is beyond the mean maximum foraging range of fulmars breeding at

this SPA, but within the mean maximum foraging range +1SD, and the maximum foraging range.

### Potential effects on the qualifying feature from the Project-alone

2993. Fulmar are considered to have low vulnerability to both collision and displacement impacts from offshore windfarms (Bradbury *et al*., 2014). This species was recorded infrequently within the windfarm site (in only five of 24 surveys, all during the breeding season (January to August; Furness, 2015)) and in low numbers, with densities in four of the five months less than 0.1 birds/km$^2$. A peak density of 0.23 birds/km$^2$ was recorded in May 2022.

2994. On the basis of the low vulnerability of fulmars to potential impacts, and the low frequency and abundance of birds present at the windfarm site, it is concluded that any effects on breeding populations at Kerry Head SPA are very unlikely, both during and outside of the breeding season.

2995. **It is therefore concluded that there would be no measurable effects on fulmar due to the project alone, and no adverse effect on the integrity of the Kerry Head SPA is predicted.**

### Potential effects on the qualifying feature in-combination with other projects

2996. As the Project would have no measurable effect on fulmar populations from the Kerry Head SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of Kerry Head SPA, when assessed in-combination with other plans or projects.**

# 8.79 Cruagh Island SPA (transboundary site)

2997. Cruagh Island SPA is located on the west coast of Ireland approximately 428km from the windfarm site (straight line) or 607km (across sea).

## 8.79.1 Description of designation

2998. Cruagh Island is located approximately 2km west of Omey Island, off the Connemara coast in Co. Galway. It is a low-lying island (maximum height 62m) and is uninhabited. The island is dominated by a maritime grassy sward with some exposed rock. The sea area to a distance of 500m is included in the site to accommodate 'rafting' Manx shearwaters.

## 8.79.2 Conservation objectives

2999. The conservation objective of the SPA is 'to maintain or restore the favourable conservation condition of the bird species listed as Special Conservation Interests for this SPA.'

## 8.79.3 Assessment

3000. One qualifying feature of Cruagh Island SPA has been screened into the Appropriate Assessment (**Table 5.2**): Manx shearwater.

### 8.79.3.1 Manx shearwater

#### Status

3001. The Cruagh Island SPA breeding Manx shearwater population was estimated at 3,286 pairs, or 6,572 breeding adults, in 2001 (NPWS, 2010b). The SMP database (JNCC, 2023a) did not provide a more recent count, therefore the 2001 estimate has been used as the reference population for the assessment.

3002. Based on the most recent SPA population of breeding adults, and an annual breeding adult baseline mortality rate of 0.130 (1 – 0.870; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 854 breeding adults.

#### Functional linkage and seasonal apportionment of potential effects

3003. The mean maximum foraging range of Manx shearwater is 1346.8km (±1018.7km) and the maximum foraging range is 2890km. The Project is located approximately 607km from Cruagh Island SPA, which means that the Project is within the mean maximum foraging range of Manx shearwaters breeding at this SPA.

3004. A number of SPA and non-SPA Manx shearwater colonies are located in and around the UK Western Waters BDMPS area, all of which are within the mean maximum foraging range of this species. For a review of these sites see **Section 8.21.3.1**.

3005. The NatureScot apportioning tool (NatureScot, 2018) has been used to estimate the proportion of Manx shearwaters from each of the relevant SPAs present at the windfarm site during the breeding season. The apportioning to SPA and non-SPA colonies is set out in **Table 8.181**; refer also to **Appendix 12.1** of the ES for further information on the apportioning approach and results. Accordingly, 0.1% of impacts at the windfarm site during the breeding season are apportioned to Cruagh Island SPA.

*Table 8.181 Manx shearwater breeding season apportioning*

| Site | Apportioning rate |
|---|---|
| Glannau Aberdaron ac Ynys Enlli/Aberdaron Coast and Bardsey Island SPA | 8.63% |
| Copeland Islands SPA | 2.22% |
| Skomer, Skokholm and the Seas off Pembrokeshire / Sgomer, Sgogwm a Moroedd Penfro SPA | 76.54% |
| Rum SPA | 8.44% |
| St Kilda SPA | 0.20% |
| **Cruagh Island SPA (transboundary site)** | **0.10%** |
| Blasket Islands SPA (transboundary site) | 0.61% |
| Deenish Island and Scariff Island SPA (transboundary site) | 0.08% |
| Puffin Island SPA (transboundary site) | 0.22% |
| Skelligs SPA (transboundary site) | 0.03% |
| Non-SPA colonies | 2.91% |

3006. During the pre- and post-breeding periods, breeding Manx shearwaters from the Cruagh Island SPA migrate through UK waters. The relevant reference population is considered to be the UK Western Waters BDMPS. This consists of 1,580,895 individuals during the post-breeding (August-early October) and return migration (late march-May) periods.

3007. Estimates of the proportion of Manx shearwaters present at the windfarm site which originate from the Cruagh Island SPA during the post-breeding and return migration periods (and therefore the proportion of predicted mortalities from the SPA population) are based on the SPA population (i.e. 6,572 adults) as a proportion of the UK Western Waters BDMPS during the relevant season.

During the post-breeding and return migration periods, 0.4% of impacts are considered to affect birds from the SPA (Furness, 2015).

### Potential effects on the qualifying feature from the Project-alone

3008. The Manx shearwater qualifying feature of the Cruagh Island SPA has been screened into the assessment due to the potential risk of disturbance, displacement and barrier effects during the construction and decommissioning, and operation and maintenance phases of the Project.

#### Construction and decommissioning phase disturbance/displacement/barrier effects

3009. Effects during the construction and decommissioning phases of the Project are considered unlikely, given the transient presence of the species and low susceptibility to disturbance related impacts; refer to **Paragraph 3011** below. However, in accordance with feedback received from Natural England and NRW, a precautionary estimation of construction and decommissioning phase disturbance, displacement and barrier effects has been undertaken assuming 50% of the operational phase effect.

3010. Applying 50% reduction to the operational values presented in **Table 8.182**, and based on mean density, predicted mortality would be between zero and one bird (30-70% displacement and 1-10% mortality of displaced birds). Using realistic values (i.e. mean density, 50% displacement and 1% mortality), there would be an annual increase in mortality of less than one (0.1) birds, which is equivalent to a 0.01% increase in background mortality for the SPA population. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. Accordingly, no significant effects on Manx shearwater are predicted during the construction and decommissioning phases, and **it is concluded that there would be no adverse effect on the integrity of Cruagh Island SPA**.

#### Operation and maintenance phase disturbance/displacement/barrier effects

3011. Manx shearwater is generally considered to have a low susceptibility to disturbance and displacement (Furness *et al.*, 2013). See **Section 8.21.3.1** for summary of effects from Dierschke *et al*., (2016) and Bradbury *et al.,* (2014).

3012. Displacement effects for Manx shearwater for the Project were assessed during the breeding, autumn migration and spring migration periods, based on an unapportioned peak mean population of 4,705, 2,650 and 1,617 birds respectively, calculated for the windfarm site and a 2km buffer, in line with recommendations within the SNCB guidance (SNCBs, 2017). The displacement matrices used to calculate potential impacts are presented in **Appendix 12.1** of the ES, and summarised in **Table 8.182.** The application of the same displacement rate to the OWF and the 2km buffer, to determine the total number of birds subject to displacement, is precautionary, as in reality the displacement rate is likely to fall with distance from the windfarm site.

3013. A displacement rate of 30-70% and mortality rate of 1-10% has been presented. Given that 10% would represent a rate close to the expected 'natural' annual mortality (0.13), this rate is considered very unlikely. Accordingly, a 1% mortality rate is considered to be most appropriate, with the upper end of this range likely to be precautionary. Given the very extensive foraging range of this species (Woodward *et al.*, 2019), there may be no mortality costs to displacement from the relatively very small footprints of OWFs.

*Table 8.182 Manx shearwater – predicted operation and maintenance phase displacement and mortality from Cruagh Island SPA*

| Mean peak abundance estimate type | Mean peak abundance estimate | Number of SPA breeding adults present by season[1] | Annual mortality range[2] | Annual baseline mortality increase range[3] |
|---|---|---|---|---|
| Upper 95% CI | 10,010 (breeding)<br>4,447 (autumn)<br>4,711 (spring)<br>19,168 (year round) | 10 (breeding)<br>18 (autumn)<br>20 (spring)<br>48 (year round) | 0-3 | 0.02-0.39% |
| Mean | 4,705 (breeding)<br>2,650 (autumn)<br>1,617 (spring)<br>8,972 (year round) | 5 (breeding)<br>11 (autumn)<br>7 (spring)<br>22 (year round) | 0-2 | 0.01-0.18% |
| Lower 95% CI | 783 (breeding)<br>1,308 (autumn)<br>0 (spring)<br>2,092 (year round) | 1 (breeding)<br>5 (autumn)<br>0 (spring)<br>6 (year round) | 0-0 | 0.00-0.05% |

[1] During the breeding season, assumes 0.1% of recorded birds are adults from the SPA population (6,572), and 0.4% during the autumn and spring migration periods

[2] Assumes displacement rates of 30-70% and mortality rates of 1-10%

[3] Background population is Cruagh Island SPA breeding adults (6,572 individuals), adult age class annual mortality rate of 13% (Horswill and Robinson, 2015)

3014. Using realistic values (i.e. mean density, 50% displacement and 1% mortality), there would be an annual increase in mortality of <1 (0.11) bird, representing a 0.01% increase in mortality rate. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. Accordingly, no significant effects on Manx shearwater are predicted during the operation and maintenance phase, and **it is concluded that there is no potential for the Project to have an adverse effect on the integrity of Cruagh Island SPA**.

3015. A review of the potential effects of artificial light on Manx shearwaters is presented in Section 12.6.3.1 of **Chapter 12 Offshore Ornithology**. This concludes that lighting associated with the Project is very unlikely to significantly affect disturbance and displacement effects on Manx shearwater, and therefore the conclusions of the assessment are unchanged.

3016. The confidence in the assessment is high. The evidence used to set the displacement rates presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES is of high applicability and quality. Whilst there is limited available evidence to inform mortality rates, 1% is considered to be sufficiently precautionary based on expert opinion.

### Potential effects in-combination with other projects

3017. No in-combination effects are predicted during the construction and decommissioning phases. This is because it is unlikely that there would be significant temporal and/or spatial overlap with other plans or projects, and due to the negligible effects predicted from the project alone.

3018. Neither the HRA for the Round 4 offshore wind leasing (NIRAS, 2021), nor the RIAA of the recently consented Awel y Môr OWF application (Awel y Môr Offshore Wind Farm Ltd, 2022) have assessed the in-combination effects on Manx shearwater from Cruagh Island SPA. In the case of the Round 4 HRA (which includes the Project), no effect on site integrity (for all SPAs) was concluded on the basis of the low vulnerability to OWFs and low density of this species within Round 4 areas. The Awel y Môr RIAA screened out in-combination effects on the basis of the small contribution of the Awel y Môr OWF and absence of linkage to populations from the SPA.

3019. Given the very low numbers of Manx shearwaters from Cruagh Island SPA predicted to occur at the windfarm site, and consequent low predicted mortality increase (<1 bird), it is considered reasonable to conclude that no significant effects on the SPA population are predicted during the operation and maintenance phase, and that there would be **no potential for the Project-alone or in-combination to have an adverse effect on the integrity of Cruagh Island SPA**.

# 8.80 Dingle Peninsula SPA (transboundary site)

3020. Dingle Peninsula SPA is located on the west coast of Ireland approximately 453km from the windfarm site (straight line) or 611km (across sea).

## 8.80.1 Description of designation

3021. Dingle Peninsula SPA is a large site situated on the west coast of Co. Kerry. It encompasses the high coast and sea cliff sections of the peninsula from just south of Brandon Point in the north, around to the end of the peninsula at Slea Head, and as far east as Inch in the south. The site includes the sea cliffs, the land adjacent to the cliff edge, areas of sand dune on the Magharees Peninsula and near Murreagh, and also several upland areas further inland of the coast. The high water mark forms the seaward boundary There is one qualifying seabird species for the SPA; fulmar.

## 8.80.2 Conservation objectives

3022. The conservation objective of the SPA is 'to maintain or restore the favourable conservation condition of the bird species listed as Special Conservation Interests for this SPA.'

## 8.80.3 Assessment

3023. One qualifying feature of Dingle Peninsula SPA has been screened into the Appropriate Assessment (**Table 5.2**): breeding fulmar.

### 8.80.3.1 Fulmar

#### Status

3024. The Dingle Peninsula SPA breeding fulmar population stood at 1,016 pairs, or 2,032 breeding adults, in 1999 – 2000 (NPWS, 2014d). The most recent count is 852 pairs (AOS), or 1,704 breeding adults, in 2016 – 2018 (JNCC, 2023a); this is used as the reference population for the assessment.

3025. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.064 (1 – 0.936; Horswill and Robinson, 2015), the expected annual mortality from the SPA population would be 109 breeding adults.

#### Functional linkage and seasonal apportionment of potential effects

3026. The mean maximum foraging range of fulmar is 542.3km (±657.9km) and the maximum foraging range is 2,736km (Woodward *et al*., 2019). The Project is located approximately 611km from the Dingle Peninsula SPA, which means that the Project is beyond the mean maximum foraging range of fulmars breeding at this SPA, but within the mean maximum foraging range +1SD, and the maximum foraging range.

### Potential effects on the qualifying feature from the Project-alone

3027. Fulmar are considered to have low vulnerability to both collision and displacement impacts from offshore windfarms (Bradbury *et al*., 2014). This species was recorded infrequently within the windfarm site (in only five of 24 surveys, all during the breeding season (January to August; Furness, 2015)) and in low numbers, with densities in four of the five months less than 0.1 birds/km$^2$. A peak density of 0.23 birds/km$^2$ was recorded in May 2022.

3028. On the basis of the low vulnerability of fulmars to potential impacts, and the low frequency and abundance of birds present at the windfarm site, it is concluded that any effects on breeding populations at Dingle Peninsula SPA are very unlikely, both during and outside of the breeding season.

3029. **It is therefore concluded that there would be no measurable effects on fulmar due to the project alone, and no adverse effect on the integrity of the Dingle Peninsula SPA is predicted.**

### Potential effects on the qualifying feature in-combination with other projects

3030. As the Project would have no measurable effect on fulmar populations from the Dingle Peninsula SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of Dingle Peninsula SPA, when assessed in-combination with other plans or projects.**

## 8.81 Iveragh Peninsula SPA (transboundary site)

3031. Iveragh Peninsula SPA is located on the west coast of Ireland approximately 463km from the windfarm site (straight line) or 568km (across sea).

### 8.81.1 Description of designation

3032. Iveragh Peninsula SPA is a large site situated on the west coast of Co. Kerry. The site encompasses the high coast and sea cliff sections of the peninsula from just west of Rossbehy in the north, around to the end of the peninsula at Valencia Island and Bolus Head, and as far east as Lamb's Head in the south. The site includes the sea cliffs, the land adjacent to the cliff edge and also areas of sand dunes at Derrynane and Beginish. The high water mark forms the seaward boundary except at Doulus Head/Killelan Mountain where the adjacent sea area to a distance of 500 m from the cliff base is included. The qualifying seabird species of the SPA comprise fulmar, kittiwake and guillemot.

### 8.81.2 Conservation objectives

3033. The conservation objective of the SPA is 'to maintain or restore the favourable conservation condition of the bird species listed as Special Conservation Interests for this SPA.'

### 8.81.3 Assessment

3034. One qualifying feature of Iveragh Peninsula SPA has been screened into the Appropriate Assessment (**Table 5.2**): fulmar.

#### 8.81.3.1 Fulmar

#### Status

3035. The Iveragh Peninsula SPA breeding fulmar population stood at 766 pairs, or 1,532 breeding adults, in 1999 – 2000 (NPWS, 2015f). The SMP database (JNCC, 2023a) did not provide a more recent estimate, therefore the 1999 – 2000 count has been used as the reference population for the assessment.

3036. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.064 (1 – 0.936; Horswill and Robinson, 2015), the expected annual mortality from the SPA population would be 98 breeding adults.

#### Functional linkage and seasonal apportionment of potential effects

3037. The mean maximum foraging range of fulmar is 542.3km (±657.9km) and the maximum foraging range is 2,736km (Woodward *et al.*, 2019). The Project is located approximately 568km from Iveragh SPA, which means that the Project is beyond the mean maximum foraging range of fulmars breeding at this SPA, but within the mean maximum foraging range +1SD, and the maximum foraging range.

### Potential effects on the qualifying feature from the Project-alone

3038. Fulmar are considered to have low vulnerability to both collision and displacement impacts from offshore windfarms (Bradbury *et al*., 2014). This species was recorded infrequently within the windfarm site (in only five of 24 surveys, all during the breeding season (January to August; Furness, 2015)) and in low numbers, with densities in four of the five months less than 0.1 birds/km$^2$. A peak density of 0.23 birds/km$^2$ was recorded in May 2022.

3039. On the basis of the low vulnerability of fulmars to potential impacts, and the low frequency and abundance of birds present at the windfarm site, it is concluded that any effects on breeding populations at Iveragh Peninsula SPA are very unlikely, both during and outside of the breeding season.

3040. **It is therefore concluded that there would be no measurable effects on fulmar due to the project alone, and no adverse effect on the integrity of the Iveragh Peninsula SPA is predicted.**

### Potential effects on the qualifying feature in-combination with other projects

3041. As the Project would have no measurable effect on fulmar populations from the Iveragh Peninsula SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of Iveragh Peninsula SPA, when assessed in-combination with other plans or projects.**

## 8.82 Blasket Islands SPA (transboundary site)

3042. Blasket Islands SPA is located on the west coast of Ireland approximately 491km from the windfarm site (straight line) or 611km (across sea).

### 8.82.1 Description of designation

3043. The Blasket Islands are situated at the end of the Dingle peninsula in Co. Kerry. The SPA comprises all of the main islands in the group, as well as the various islets and rocks, and also the seas which surround the islands to a distance of 500m. The qualifying bird species of the SPA comprise fulmar, Manx shearwater, storm petrel, shag, lesser black-backed gull, herring gull, kittiwake, Arctic tern, razorbill and puffin.

### 8.82.2 Conservation objectives

3044. The conservation objective of the SPA is 'to maintain or restore the favourable conservation condition of the bird species listed as Special Conservation Interests for this SPA.'

### 8.82.3 Assessment

3045. Four qualifying features of Blasket Islands SPA have been screened into the Appropriate Assessment (**Table 5.2**): fulmar, Manx shearwater, puffin, and lesser black-backed gull.

#### 8.82.3.1 Fulmar

##### Status

3046. The Blasket Islands SPA breeding fulmar population stood at 2,179 pairs, or 4,358 breeding adults, in 1988 (NPWS, 2015g). The most recent combined count from Great Blasket Island (452 pairs/AOS), Inishnabro (672 pairs/AOS) and Inishtooskert (133 pairs/AOS) is 1,257 pairs, or 2,514 breeding adults in 2015 (JNCC, 2023a); this is used as the reference population for the assessment.

3047. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.064 (1 – 0.936; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 161 breeding adults.

##### Functional linkage and seasonal apportionment of potential effects

3048. The mean maximum foraging range of fulmar is 542.3km (±657.9km) and the maximum foraging range is 2,736km (Woodward *et al*., 2019). The Project is located approximately 611km from Blasket Islands SPA, which means that the

Project is beyond the mean maximum foraging range of fulmars breeding at this SPA, but within the mean maximum foraging range +1SD, and the maximum foraging range.

### Potential effects on the qualifying feature from the Project-alone

3049. Fulmar are considered to have low vulnerability to both collision and displacement impacts from offshore windfarms (Bradbury *et al*., 2014). This species was recorded infrequently within the windfarm site (in only five of 24 surveys, all during the breeding season (January to August; Furness, 2015) and in low numbers, with densities in four of the five months less than 0.1 birds/km$^2$. A peak density of 0.23 birds/km$^2$ was recorded in May 2022.

3050. On the basis of the low vulnerability of fulmars to potential impacts, and the low frequency and abundance of birds present at the windfarm site, it is concluded that any effects on breeding populations at Blasket Islands SPA are very unlikely, both during and outside of the breeding season.

3051. **It is therefore concluded that there would be no measurable effects on fulmar due to the project alone, and no adverse effect on the integrity of the Blasket Islands SPA is predicted.**

### Potential effects on the qualifying feature in-combination with other projects

3052. As the Project would have no measurable effect on fulmar populations from the Blasket Islands SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of Blasket Islands SPA, when assessed in-combination with other plans or projects.**

#### 8.82.3.2 Manx shearwater

### Status

3053. The Blasket Islands SPA breeding Manx shearwater population was estimated at 19,534 pairs, or 39,068 breeding adults, in 2000 – 2001 (NPWS, 2015g). The SMP database (JNCC, 2023a) did not provide a more recent count, therefore the 2000 – 2001 estimate has been used as the reference population for the assessment.

3054. Based on the most recent SPA population of breeding adults, and an annual breeding adult baseline mortality rate of 0.130 (1 – 0.870; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 5,079 breeding adults.

### Functional linkage and seasonal apportionment of potential effects

3055. The mean maximum foraging range of Manx shearwater is 1346.8km (±1018.7km) and the maximum foraging range is 2890km. The Project is located approximately 611km from Blasket Islands SPA, which means that the Project is within the mean maximum foraging range of Manx shearwaters breeding at this SPA.

3056. A number of SPA and non-SPA Manx shearwater colonies are located in and around the UK Western Waters BDMPS area, all of which are within the mean maximum foraging range of this species. For a review of these sites see **Section 8.21.3.1**.

3057. The NatureScot apportioning tool (NatureScot, 2018) has been used to estimate the proportion of Manx shearwaters from each of the relevant SPAs present at the windfarm site during the breeding season. The apportioning to SPA and non-SPA colonies is set out in **Table 8.183**; refer also to **Appendix 12.1** of the ES for further information on the apportioning approach and results. Accordingly, 0.6% of impacts at the windfarm site during the breeding season are apportioned to Blasket Islands SPA.

*Table 8.183 Manx shearwater breeding season apportioning*

| Site | Apportioning rate |
|---|---|
| Glannau Aberdaron ac Ynys Enlli/Aberdaron Coast and Bardsey Island SPA | 8.63% |
| Copeland Islands SPA | 2.22% |
| Skomer, Skokholm and the Seas off Pembrokeshire / Sgomer, Sgogwm a Moroedd Penfro SPA | 76.54% |
| Rum SPA | 8.44% |
| St Kilda SPA | 0.20% |
| Cruagh Island SPA (transboundary site) | 0.10% |
| **Blasket Islands SPA (transboundary site)** | **0.61%** |
| Deenish Island and Scariff Island SPA (transboundary site) | 0.08% |
| Puffin Island SPA (transboundary site) | 0.22% |
| Skelligs SPA (transboundary site) | 0.03% |
| Non-SPA colonies | 2.91% |

3058. During the pre- and post-breeding periods, breeding Manx shearwaters from the Blasket Islands SPA migrate through UK waters. The relevant reference population is considered to be the UK Western Waters BDMPS. This consists of 1,580,895 individuals during the post-breeding (August-early October) and return migration (late march-May) periods.

3059. Estimates of the proportion of Manx shearwaters present at the windfarm site which originate from the Blasket Islands SPA during the post-breeding and return migration periods (and therefore the proportion of predicted mortalities from the SPA population) are based on the SPA population (i.e. 39,068 adults) as a proportion of the UK Western Waters BDMPS during the relevant season. During the post-breeding and return migration periods, 2.5% of impacts are considered to affect birds from the SPA (Furness, 2015).

### Potential effects on the qualifying feature from the Project-alone

3060. The Manx shearwater qualifying feature of the Blasket Islands SPA has been screened into the assessment due to the potential risk of disturbance, displacement and barrier effects during the construction and decommissioning, and operation and maintenance phases of the Project.

#### Construction and decommissioning phase disturbance/displacement/barrier effects

3061. Effects during the construction and decommissioning phases of the Project are considered unlikely, given the transient presence of the species and low susceptibility to disturbance related impacts; refer to **Paragraph 3063** below. However, in accordance with feedback received from Natural England and NRW, a precautionary estimation of construction and decommissioning phase disturbance, displacement and barrier effects has been undertaken assuming 50% of the operational phase effect.

3062. Applying 50% reduction to the operational values presented in 3063, and based on mean density, predicted mortality would be between zero and five birds (30-70% displacement and 1-10% mortality of displaced birds). Using realistic values (i.e. mean density, 50% displacement and 1% mortality), there would be an annual increase in mortality of less than one (0.3) birds, which is equivalent to a 0.01% increase in background mortality for the SPA population. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. Accordingly, no significant effects on Manx shearwater are predicted during the construction and decommissioning phases, and **it is concluded that there would be no adverse effect on the integrity of Blasket Islands SPA**.

#### Operation and maintenance phase disturbance/displacement/ barrier effects

3063. Manx shearwater are generally considered to have a low susceptibility to disturbance and displacement (Furness *et al.*, 2013). See **Section 8.21.3.1** for summary of effects from Dierschke *et al*., (2016) and Bradbury *et al.,* (2014).

3064. Displacement effects for Manx shearwater for the Project were assessed during the breeding, autumn migration and spring migration periods, based on an unapportioned peak mean population of 4,705, 2,650 and 1,617 birds respectively, calculated for the windfarm site and a 2km buffer, in line with recommendations within the SNCB guidance (SNCBs, 2017). The displacement matrices used to calculate potential impacts are presented in **Appendix 12.1** of the ES and summarised in **Table 8.184.** The application of the same displacement rate to the OWF and the 2km buffer, to determine the total number of birds subject to displacement, is precautionary, as in reality the displacement rate is likely to fall with distance from the windfarm site.

3065. A displacement rate of 30-70% and mortality rate of 1-10% has been presented. Given that 10% would represent a rate close to the expected ‘natural’ annual mortality (0.13), this rate is considered very unlikely. Accordingly, a 1% mortality rate is considered to be most appropriate, with the upper end of this range likely to be precautionary. Given the very extensive foraging range of this species (Woodward *et al.*, 2019), there may be no mortality costs to displacement from the relatively very small footprints of OWFs.

*Table 8.184 Manx shearwater – predicted operation and maintenance phase displacement and mortality from Blasket Islands SPA*

| Mean peak abundance estimate type | Mean peak abundance estimate | Number of SPA breeding adults present by season[1] | Annual mortality range[2] | Annual baseline mortality increase range[3] |
|---|---|---|---|---|
| Upper 95% CI | 10,010 (breeding)<br>4,447 (autumn)<br>4,711 (spring)<br>19,168 (year round) | 60 (breeding)<br>110 (autumn)<br>116 (spring)<br>286 (year round) | 1-20 | 0.02-0.39% |
| Mean | 4,705 (breeding)<br>2,650 (autumn)<br>1,617 (spring)<br>8,972 (year round) | 28 (breeding)<br>66 (autumn)<br>40 (spring)<br>134 (year round) | 0-9 | 0.01-0.18% |
| Lower 95% CI | 783 (breeding)<br>1,308 (autumn)<br>0 (spring)<br>2,092 (year round) | 5 (breeding)<br>32 (autumn)<br>0 (spring)<br>37 (year round) | 0-3 | 0.00-0.05% |

[1] During the breeding season, assumes 0.6% of recorded birds are adults from the SPA population (39,068), and 2.5% during the autumn and spring migration periods

[2] Assumes displacement rates of 30-70% and mortality rates of 1-10%

[3] Background population Blasket Islands SPA breeding adults (39,068 individuals), adult age class annual mortality rate of 13% (Horswill and Robinson, 2015)

3066. Using realistic values (i.e. mean density, 50% displacement and 1% mortality), there would be an annual increase in mortality of <1 (0.7) bird, representing a 0.01% increase in mortality rate. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. Accordingly, no significant effects on Manx shearwater are predicted during the operation and maintenance phase, and **it is concluded that there is no potential for the Project to have an adverse effect on the integrity of Blasket Islands SPA**.

3067. A review of the potential effects of artificial light on Manx shearwaters is presented in Section 12.6.3.1 of **Chapter 12 Offshore Ornithology**. This concludes that lighting associated with the Project is very unlikely to significantly affect disturbance and displacement effects on Manx shearwater, and therefore the conclusions of the assessment are unchanged.

3068. The confidence in the assessment is high. The evidence used to set the displacement rates presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES is of high applicability and quality. Whilst there is limited available evidence to inform mortality rates, 1% is considered to be sufficiently precautionary based on expert opinion.

### Potential effects in-combination with other projects

3069. No in-combination effects are predicted during the construction and decommissioning phases. This is because it is unlikely that there would be significant temporal and/or spatial overlap with other plans or projects, and due to the negligible effects predicted from the project alone.

3070. Neither the HRA for the Round 4 offshore wind leasing (NIRAS, 2021), nor the RIAA of the recently submitted Awel y Môr OWF application (Awel y Môr Offshore Wind Farm Ltd, 2022) have assessed the in-combination effects on Manx shearwater from Blasket Islands SPA. In the case of the Round 4 HRA (which includes the Project), no effect on site integrity (for all SPAs) was concluded on the basis of the low vulnerability to OWFs and low density of this species within Round 4 areas. The Awel y Môr RIAA screened out in-combination effects on the basis of the small contribution of the Awel y Môr OWF and absence of linkage to populations from the SPA.

3071. Given the very low numbers of Manx shearwaters from Blasket Islands SPA predicted to occur at the windfarm site, and consequent low predicted mortality increase (<1 bird), it is considered reasonable to conclude that no significant effects on the SPA population are predicted during the operation and maintenance phase, and that there would be **no potential for the Project-alone or in-combination to have an adverse effect on the integrity of Blasket Islands SPA**.

#### 8.82.3.3 Puffin

##### Status

3072. The Blasket Islands SPA breeding puffin population stood at 4,924 pairs, or 9,848 breeding adults, in 1988 (NPWS, 2015g). The SMP database (JNCC, 2023a) did not provide a more recent estimate, therefore the 1988 count has been used as the reference population for the assessment.

3073. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.094 (1 – 0.906; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 926 breeding adults.

##### Functional linkage and seasonal apportionment of potential effects

3074. The mean maximum foraging range of puffin is 137.1km (±128.3km) and the maximum foraging range is 383km (Woodward *et al*., 2019). The Project is located approximately 611km from Blasket Islands SPA, which means the Project is beyond the maximum foraging range for this species. No impacts during the breeding season are therefore apportioned to birds breeding at this colony.

3075. Outside of the breeding season, breeding puffins, including those from the Blasket Islands SPA, are not constrained by requirements to visit nests to incubate eggs or provision chicks. At this time, they are assumed to range more widely and to mix with puffins of all age classes from breeding colonies in Ireland, the UK and further afield. The background population during these seasons is the UK Western Waters BDMPS. This consists of 304,557 individuals during the non-breeding season (August to March).

3076. As no published estimate was available, it is assumed that 10% of Blasket Islands SPA breeding adults are present within the UK Western Waters BDMPS during the non-breeding season. This is the rate for 'Ireland' populations identified by Furness (2015), and is considered appropriate given the geographic isolation between the west coast of Ireland (where the SPA is located) and the windfarm site. It is assumed that the most recent count prior to the publication of Furness (2015) was used to inform the BDMPS total which is 9,848 breeding adults; 10% of this population is 985 birds. This represents 0.3% of the BDMPS population for this period (304,557). It is therefore assumed that 0.3% of puffins present at the Project site are breeding adults from Blasket Islands SPA.

## Potential effects on the qualifying feature

### Operation and maintenance phase disturbance/displacement/barrier effects

*Project-alone*

3077. The mean peak abundance of puffins present within the windfarm site and 2km buffer during the non-breeding season was 19.7 (1.9-50.8) individuals (refer to **Appendix 12.1** of the ES). Of these, less than one bird (0.1 (0.0-0.2)) was likely to be a breeding adult from Blasket Islands SPA.

3078. **Table 8.185** sets out the predicted impacts on puffins from Blasket Islands SPA during the non-breeding season. This estimates that there would be no measurable increase in mortality from the SPA population, assuming a displacement rate of 30-70% and a mortality of 1-10% for displaced birds.

*Table 8.185 Puffin – predicted operation and maintenance phase displacement and mortality from Blasket Islands SPA*

| Mean peak abundance estimate type | Mean peak abundance estimate | Number of Blasket Islands SPA breeding adults present (non-breeding season)[1] | Annual mortality range[2] | Annual baseline mortality increase range[3] |
|---|---|---|---|---|
| Upper 95% CI | 50.8 | 0.2 | 0-0 | 0.00-0.00% |
| Mean | 19.7 | 0.1 | 0-0 | 0.00-0.00% |
| Lower 95% CI | 1.9 | 0.0 | 0-0 | 0.00-0.00% |

[1] Assumes 0.3% of birds present during the non-breeding season are Blasket Islands SPA breeding adults

[2] Assumes displacement rates of 30-70% and mortality rates of 1-10%

[3] Background mortality rate of 9.4% (Horswill and Robinson, 2015)

3079. **It is concluded that predicted puffin mortality due to operational phase displacement would not adversely affect the integrity of the Blasket Islands SPA.**

3080. The confidence in the assessment is high for several reasons. Firstly, the evidence used to inform the evidence-based displacement rates is of high applicability and quality (based on the criteria discussed in **Chapter 12 Offshore Ornithology** of the ES). Whilst there was limited available evidence to inform mortality rates, 1% is considered to be sufficiently precautionary based on expert opinion. This species is not regarded as being highly specialised in its habitat requirements (Bradbury *et al*., 2014; Furness and Wade, 2012; Garthe and Hüppop, 2004), and it is therefore anticipated that displaced birds will find alternative habitat in the vast majority of cases. Finally, the conclusion of the assessment is the same irrespective of whether the mean or 95% upper CI mean peak abundances are used to calculate potential mortality and increases in the baseline mortality rate of the background population.

*In-combination*

3081. As the Project would have no measurable effect on puffin populations from the Blasket Islands SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that predicted puffin mortality due to operational phase displacement due to the Project in-combination with other plans or projects would not adversely affect the integrity of the Blasket Islands SPA.**

#### 8.82.3.4 Lesser black-backed gull

##### Status

3082. The Blasket Islands SPA breeding lesser black-backed gull population was estimated to be at least 333 pairs, or 666 breeding adults, in 1988 (NPWS, 2015g). More recent counts are available on the SMP database (JNCC, 2023a), however, it is unclear whether these counts covered the full extent of the SPA. The 1998 estimate is therefore used as the reference population for the assessment.

3083. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.115 (1 – 0.885; Horswill and Robinson; 2015), the expected annual mortality from the SPA population would be 77 breeding adults.

##### Functional linkage and seasonal apportionment of potential effects

3084. The mean maximum foraging range of lesser black-backed gull is 127km (±109km) and the maximum foraging range is 533km (Woodward *et al*., 2019). The Project is located approximately 611km from Blasket Islands SPA, which means the Project is beyond the maximum foraging range for this species. No breeding season effects are therefore apportioned to this species.

3085. Outside the breeding season, breeding lesser black-backed gulls from the SPA are assumed to range widely and to mix with lesser black-backed gulls of all ages from breeding colonies in the UK, Ireland and beyond. The relevant non-breeding season reference population is the UK Western Waters BDMPS, consisting of 163,304 individuals during spring and autumn migration (March and September to October) and 41,159 during winter (November to February) (Furness, 2015).

3086. Furness (2015) estimated that 40% of breeding adults from Ireland colonies are present within the UK Western Waters and Channel BDMPS during the autumn and spring migration periods, and 20% during the winter period. This is equivalent to 381 adults from Blasket Islands SPA during the autumn and spring periods, and 190 during winter. This represents 0.23% of the BDMPS population for the autumn and spring periods, and 0.46% during the winter period. Impacts to birds from the SPA during these periods are therefore apportioned accordingly.

### Potential effects on the qualifying feature from the Project-alone

3087. The lesser black-backed gull qualifying feature of the Blasket Islands SPA has been screened into the Appropriate Assessment due to the potential risk of collision.

#### Operation and maintenance phase collision risk

3088. Information for collision risk on breeding adult lesser black-backed gulls belonging to the Blasket Islands SPA population is presented in **Table 8.186**. Collision estimates, calculated using the sCRM, are presented by biological season. A summary of the annual outputs and the corresponding increase in the annual baseline mortality rate is also presented. Parameters used in the sCRM were agreed with Natural England during the ETG process and are described in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES.

3089. Based on the mean collision rates, the annual total of breeding adult lesser black-backed gulls from Blasket Islands SPA at risk of collision as a result of the Project is less than one bird (0.00). This would increase the existing mortality of the SPA breeding population by 0.01%.

*Table 8.186 Predicted seasonal and annual collision mortality (Stochastic model Option 2, avoidance rate 0.994 (±0.0004)), for breeding adult lesser black-backed gulls at the windfarm site, apportioned to Blasket Islands SPA, with corresponding increases to baseline mortality of the population*

| | Breeding Season | Autumn Migration | Non-breeding/winter | Spring Migration | Annual |
|---|---|---|---|---|---|
| **Period** | **Apr-Aug** | **Sep-Oct** | **Nov-Feb** | **Mar** | **Jan-Dec** |
| Total collisions[1] (mean and 95% CIs) | 1.44 (0.00-4.53) | 1.25 (0.00-5.63) | 0.15 (0.00-0.80) | 0.15 (0.00-0.94) | 2.98 (0.00-11.90) |
| % apportioned to the SPA | 0.00% | 0.16% | 0.32% | 0.16% | - |
| Total SPA collisions (mean and 95% CIs) | 0.00 (0.00-0.00) | 0.00 (0.00-0.01) | 0.00 (0.00-0.00) | 0.00 (0.00-0.00) | 0.00 (0.00-0.01) |
| Mortality increase[2] (mean and 95% CIs) | 0.00% (0.00-0.00%) | 0.00% (0.00-0.02%) | 0.00 (0.00-0.01) | 0.00% (0.00-0.00%) | 0.01% (0.00-0.03%) |

[1] Breeding season collision values reduced to 71.9% of modelled value to reflect proportion of adult birds recorded during site surveys

[2] Assuming predicted annual SPA mortality of 77 birds (666 x 0.115)

3090. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. This means that no detectable changes in mortality rates would occur on this population for the mean monthly collision estimates for the Project, or for the upper 95% CI collision estimate.

3091. **It is concluded that based on predicted lesser black-backed gull mortality due to collision at the windfarm site there is no potential for the Project to have an adverse effect on the integrity of the Blasket Islands SPA.**

3092. The confidence in the assessment is high. The evidence used to define the CRM input parameters presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** as part of the ES submitted alongside this RIAA is of high applicability and quality. Whilst there is uncertainty around some of the input parameters (e.g. avoidance rate), the rates selected are considered to be sufficiently precautionary based on expert opinion to provide confidence that collision rates are not underestimated.

### Potential effects on the qualifying feature in-combination with other projects

3093. As the Project would have no measurable effect on lesser black-backed gull populations from the Blasket Islands SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of Blasket Islands SPA, when assessed in-combination with other plans or projects**.

# 8.83 Deenish Island and Scariff Island SPA (transboundary site)

3094. Deenish Island and Scariff Island SPA is located on the west coast of Ireland approximately 493km from the windfarm site (straight line) or 568km (across sea).

## 8.83.1 Description of designation

3095. Deenish Island and Scariff Island are situated between 5km and 7km west of Lamb's Head off the Co. Kerry coast. Scariff is the larger of the two; it is steep-sided all the way around and rises to a peak of 252 m. The highest cliffs are on the south side. Deenish is less rugged than Scariff, and rises to 144m in its southern half; the northern half is lower and flatter. The surrounding seas to a distance of 500m around the islands are included within the SPA. qualifying seabird species of the SPA comprise fulmar, Manx shearwater, storm petrel, lesser black-backed gull and Arctic tern.

## 8.83.2 Conservation objectives

3096. The conservation objective of the SPA is 'to maintain or restore the favourable conservation condition of the bird species listed as Special Conservation Interests for this SPA.'

## 8.83.3 Assessment

3097. Two qualifying features of Deenish Island and Scariff Island SPA have been screened into the Appropriate Assessment (**Table 5.2**): fulmar and manx shearwater.

### 8.83.3.1 Fulmar

#### Status

3098. The Deenish Island and Scariff Island SPA breeding fulmar population stood at 385 pairs, or 770 breeding adults, in 2000 (NPWS, 2015h). The SMP database (JNCC, 2023a) did not provide a more recent estimate, therefore the 2000 count has been used as the reference population for the assessment.

3099. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.064 (1 – 0.936; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 25 breeding adults.

### Functional linkage and seasonal apportionment of potential effects

3100. The mean maximum foraging range of fulmar is 542.3km (±657.9km) and the maximum foraging range is 2,736km (Woodward *et al*., 2019). The Project is located approximately 568km from Deenish Island and Scariff Island SPA, which means that the Project is beyond the mean maximum foraging range of fulmars breeding at this SPA, but within the mean maximum foraging range +1SD, and the maximum foraging range.

### Potential effects on the qualifying feature from the Project-alone

3101. Fulmar are considered to have low vulnerability to both collision and displacement impacts from offshore windfarms (Bradbury *et al*., 2014). This species was recorded infrequently within the windfarm site (in only five of 24 surveys, all during the breeding season (January to August; Furness, 2015)) and in low numbers, with densities in four of the five months less than 0.1 birds/km$^2$. A peak density of 0.23 birds/km$^2$ was recorded in May 2022.

3102. On the basis of the low vulnerability of fulmars to potential impacts, and the low frequency and abundance of birds present at the windfarm site, it is concluded that any effects on breeding populations at Deenish Island and Scariff Island SPA are very unlikely, both during and outside of the breeding season.

3103. **It is therefore concluded that there would be no measurable effects on fulmar due to the project alone, and no adverse effect on the integrity of the Deenish Island and Scariff Island SPA is predicted.**

### Potential effects on the qualifying feature in-combination with other projects

3104. As the Project would have no measurable effect on fulmar populations from the Deenish Island and Scariff Island SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of Deenish Island and Scariff Island SPA, when assessed in-combination with other plans or projects.**

### 8.83.3.2 Manx shearwater

### Status

3105. The Deenish Island and Scariff Island SPA breeding Manx shearwater population stood at 2,311 pairs, or 4,622 breeding adults, in 2000 (NPWS, 2015h). The SMP database (JNCC, 2023a) did not provide a more recent estimate, therefore the 2000 count has been used as the reference population for the assessment.

3106. Based on the most recent SPA population of breeding adults, and an annual breeding adult baseline mortality rate of 0.130 (1 – 0.870; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 601 breeding adults.

### Functional linkage and seasonal apportionment of potential effects

3107. The mean maximum foraging range of Manx shearwater is 1346.8km (±1018.7km) and the maximum foraging range is 2890km. The Project is located approximately 568km from Deenish Island and Scariff Island SPA, which means that the Project is within the mean maximum foraging range of Manx shearwaters breeding at this SPA.

3108. A number of SPA and non-SPA Manx shearwater colonies are located in and around the UK Western Waters BDMPS area, all of which are within the mean maximum foraging range of this species. For a review of these sites see **Section 8.21.3.1**.

3109. The NatureScot apportioning tool (NatureScot, 2018) has been used to estimate the proportion of Manx shearwaters from each of the relevant SPAs present at the windfarm site during the breeding season. The apportioning to SPA and non-SPA colonies is set out in **Table 8.187**; refer also to **Appendix 12.1** of the ES for further information on the apportioning approach and results. Accordingly, 0.08% of impacts at the windfarm site during the breeding season are apportioned to Deenish Island and Scariff Island SPA.

*Table 8.187 Manx shearwater breeding season apportioning*

| Site | Apportioning rate |
|---|---|
| Glannau Aberdaron ac Ynys Enlli/Aberdaron Coast and Bardsey Island SPA | 8.63% |
| Copeland Islands SPA | 2.22% |
| Skomer, Skokholm and the Seas off Pembrokeshire / Sgomer, Sgogwm a Moroedd Penfro SPA | 76.54% |
| Rum SPA | 8.44% |
| St Kilda SPA | 0.20% |
| Cruagh Island SPA (transboundary site) | 0.10% |
| Blasket Islands SPA (transboundary site) | 0.61% |
| **Deenish Island and Scariff Island SPA (transboundary site)** | **0.08%** |
| Puffin Island SPA (transboundary site) | 0.22% |
| Skelligs SPA (transboundary site) | 0.03% |
| Non-SPA colonies | 2.91% |

3110. During the pre- and post-breeding periods, breeding Manx shearwaters from the Deenish Island and Scariff Island SPA migrate through UK waters. The relevant reference population is considered to be the UK Western Waters BDMPS. This consists of 1,580,895 individuals during the post-breeding (August-early October) and return migration (late march-May) periods.

3111. Estimates of the proportion of Manx shearwaters present at the windfarm site which originate from the Deenish Island and Scariff Island SPA during the post-breeding and return migration periods (and therefore the proportion of predicted mortalities from the SPA population) are based on the SPA population (i.e. 4,622 adults) as a proportion of the UK Western Waters BDMPS during the relevant season. During the post-breeding and return migration periods, 0.3% of impacts are considered to affect birds from the SPA (Furness, 2015).

## Potential effects on the qualifying feature from the Project-alone

3112. The Manx shearwater qualifying feature of the Deenish Island and Scariff Island SPA has been screened into the assessment due to the potential risk of disturbance, displacement and barrier effects during the construction and decommissioning, and operation and maintenance phases of the Project.

### Construction and decommissioning phase disturbance/displacement/barrier effects

3113. Effects during the construction and decommissioning phases of the Project are considered unlikely, given the transient presence of the species and low susceptibility to disturbance related impacts; refer to **Paragraph 3115** below. However, in accordance with feedback received from Natural England and NRW, a precautionary estimation of construction and decommissioning phase disturbance, displacement and barrier effects has been undertaken assuming 50% of the operational phase effect.

3114. Applying 50% reduction to the operational values presented in **Table 8.188**, and based on mean density, predicted mortality would be between zero and one bird (30-70% displacement and 1-10% mortality of displaced birds). Using realistic values (i.e. mean density, 50% displacement and 1% mortality), there would be an annual increase in mortality of less than one (0.04) birds, which is equivalent to a 0.01% increase in background mortality for the SPA population. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. Accordingly, no significant effects on Manx shearwater are predicted during the construction and decommissioning phases, and **it is concluded that there would be no adverse effect on the integrity of Deenish Island and Scariff Island SPA**.

### Operation and maintenance phase disturbance/displacement/barrier effects

3115. Manx shearwater are generally considered to have a low susceptibility to disturbance and displacement (Furness *et al.*, 2013). See **Section 8.21.3.1** for summary of effects from Dierschke *et al*., (2016) and Bradbury *et al.,* (2014).

3116. Displacement effects for Manx shearwater for the Project were assessed during the breeding, autumn migration and spring migration periods, based on an unapportioned peak mean population of 4,705, 2,650 and 1,617 birds respectively, calculated for the windfarm site and a 2km buffer, in line with recommendations within the SNCB guidance (SNCBs, 2017). The displacement matrices used to calculate potential impacts are presented in **Appendix 12.1** of the ES, and summarised in **Table 8.188.** The application of the same displacement rate to the OWF and the 2km buffer, to determine the total number of birds subject to displacement, is precautionary, as in reality the displacement rate is likely to fall with distance from the windfarm site.

3117. A displacement rate of 30-70% and mortality rate of 1-10% has been presented. Given that 10% would represent a rate close to the expected 'natural' annual mortality (0.13), this rate is considered very unlikely. Accordingly, a 1% mortality rate is considered to be most appropriate, with the upper end of this range likely to be precautionary. Given the very extensive foraging range of this species (Woodward *et al.*, 2019), there may be no mortality costs to displacement from the relatively very small footprints of OWFs.

*Table 8.188 Manx shearwater – predicted operation and maintenance phase displacement and mortality from Deenish Island and Scariff Island SPA*

| Mean peak abundance estimate type | Mean peak abundance estimate | Number of SPA breeding adults present by season[1] | Annual mortality range[2] | Annual baseline mortality increase range[3] |
|---|---|---|---|---|
| Upper 95% CI | 10,010 (breeding)<br>4,447 (autumn)<br>4,711 (spring)<br>19,168 (year round) | 8 (breeding)<br>13 (autumn)<br>14 (spring)<br>35 (year round) | 0-2 | 0.02-0.41% |
| Mean | 4,705 (breeding)<br>2,650 (autumn)<br>1,617 (spring)<br>8,972 (year round) | 4 (breeding)<br>8 (autumn)<br>5 (spring)<br>16 (year round) | 0-1 | 0.01-0.19% |
| Lower 95% CI | 783 (breeding)<br>1,308 (autumn)<br>0 (spring)<br>2,092 (year round) | 1 (breeding)<br>4 (autumn)<br>0 (spring)<br>4 (year round) | 0-0 | 0.00-0.05% |

[1] During the breeding season, assumes 0.1% of recorded birds are adults from the SPA population (4,622), and 0.3% during the autumn and spring migration periods

[2] Assumes displacement rates of 30-70% and mortality rates of 1-10%

[3] Background population Deenish Island and Scariff Island SPA breeding adults (4,622 individuals), adult age class annual mortality rate of 13% (Horswill and Robinson, 2015)

3118. Using realistic values (i.e. mean density, 50% displacement and 1% mortality), there would be an annual increase in mortality of <1 (0.1) bird, representing a 0.01% increase in mortality rate. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. Accordingly, no significant effects on Manx shearwater are predicted during the operation and maintenance phase, and **it is concluded that there is no potential for the Project to have an adverse effect on the integrity of Deenish Island and Scariff Island SPA**.

3119. A review of the potential effects of artificial light on Manx shearwaters is presented in Section 12.6.3.1 of **Chapter 12 Offshore Ornithology**. This concludes that lighting associated with the Project is very unlikely to significantly affect disturbance and displacement effects on Manx shearwater, and therefore the conclusions of the assessment are unchanged.

3120. The confidence in the assessment is high. The evidence used to set the displacement rates presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES is of high applicability and quality. Whilst there is limited available evidence to inform mortality rates, 1% is considered to be sufficiently precautionary based on expert opinion.

### Potential effects in-combination with other projects

3121. No in-combination effects are predicted during the construction and decommissioning phases. This is because it is unlikely that there would be significant temporal and/or spatial overlap with other plans or projects, and due to the negligible effects predicted from the project alone.

3122. Neither the HRA for the Round 4 offshore wind leasing (NIRAS, 2021), nor the RIAA of the recently consented Awel y Môr OWF application (Awel y Môr Offshore Wind Farm Ltd, 2022) have assessed the in-combination effects on Manx shearwater from Deenish Island and Scariff Island SPA. In the case of the Round 4 HRA (which includes the Project), no effect on site integrity (for all SPAs) was concluded on the basis of the low vulnerability to OWFs and low density of this species within Round 4 areas. The Awel y Môr RIAA screened out in-combination effects on the basis of the small contribution of the Awel y Môr OWF and absence of linkage to populations from the SPA.

3123. Given the very low numbers of Manx shearwaters from Deenish Island and Scariff Island SPA predicted to occur at the windfarm site, and consequent low predicted mortality increase (<1 bird), it is considered reasonable to **conclude that no significant effects on the SPA population are predicted during the operation and maintenance phase, and that there would be no potential for the Project-alone or in-combination to have an adverse effect on the integrity of Deenish Island and Scariff Island SPA**.

# 8.84 Puffin Island SPA (transboundary site)

3124. Puffin Island SPA is located on the west coast of Ireland approximately 498km from the windfarm site (straight line) or 568km (across sea).

## 8.84.1 Description of designation

3125. Puffin Island lies approximately 0.5 km off the northern side of St Finan's bay in south-west Co. Kerry. The island is almost divided into two halves – the southern half is a long narrow, rocky ridge, rising to 130 m, while the northern half broadens into a grassy plateau though has a high point of 159 m. The island is surrounded by mostly steep cliffs and slopes. The qualifying bird species of the SPA comprise fulmar, Manx shearwater, storm petrel, lesser black-backed gull, razorbill and puffin.

## 8.84.2 Conservation objectives

3126. The conservation objective of the SPA is 'to maintain or restore the favourable conservation condition of the bird species listed as Special Conservation Interests for this SPA.'

## 8.84.3 Assessment

3127. Three qualifying features of Puffin Island SPA have been screened into the Appropriate Assessment (**Table 5.2**): fulmar, Manx shearwater, and puffin.

### 8.84.3.1 Fulmar

#### Status

3128. The Puffin Island SPA breeding fulmar population stood at 447 pairs, or 894 breeding adults, in 2000 (NPWS, 2015i). The most recent count was 670 pairs (AOS), or 1,340 breeding adults, in 2018 (JNCC, 2023a); this is used as the reference population for the assessment.

3129. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.064 (1 – 0.936; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 86 breeding adults.

#### Functional linkage and seasonal apportionment of potential effects

3130. The mean maximum foraging range of fulmar is 542.3km (±657.9km) and the maximum foraging range is 2,736km (Woodward *et al*., 2019). The Project is located approximately 568km from Puffin Island SPA, which means that the Project is beyond the mean maximum foraging range of fulmars breeding at

this SPA, but within the mean maximum foraging range +1SD, and the maximum foraging range.

### Potential effects on the qualifying feature from the Project-alone

3131. Fulmar are considered to have low vulnerability to both collision and displacement impacts from offshore windfarms (Bradbury *et al.*, 2014). This species was recorded infrequently within the windfarm site (in only five of 24 surveys, all during the breeding season (January to August; Furness, 2015)) and in low numbers, with densities in four of the five months less than 0.1 birds/km$^2$. A peak density of 0.23 birds/km$^2$ was recorded in May 2022.

3132. On the basis of the low vulnerability of fulmars to potential impacts, and the low frequency and abundance of birds present at the windfarm site, it is concluded that any effects on breeding populations at Puffin Island SPA are very unlikely, both during and outside of the breeding season.

3133. **It is therefore concluded that there would be no measurable effects on fulmar due to the project alone, and no adverse effect on the integrity of the Puffin Island SPA is predicted.**

### Potential effects on the qualifying feature in-combination with other projects

3134. As the Project would have no measurable effect on fulmar populations from the Puffin Island SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of Puffin Island SPA, when assessed in-combination with other plans or projects.**

### 8.84.3.2 Manx shearwater

### Status

3135. The Puffin Island SPA breeding Manx shearwater population stood at 6,329 pairs, or 12,658 breeding adults, in 2000 (NPWS, 2015i). The SMP database (JNCC, 2023a) did not provide a more recent estimate, therefore the 2000 count has been used as the reference population for the assessment.

3136. Based on the most recent SPA population of breeding adults, and an annual breeding adult baseline mortality rate of 0.130 (1 – 0.870; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 1,646 breeding adults.

### Functional linkage and seasonal apportionment of potential effects

3137. The mean maximum foraging range of Manx shearwater is 1,346.8km (±1018.7km) and the maximum foraging range is 2,890km. The Project is located approximately 568km from Puffin Island SPA, which means that the

Project is within the mean maximum foraging range of Manx shearwaters breeding at this SPA.

3138. A number of SPA and non-SPA Manx shearwater colonies are located in and around the UK Western Waters BDMPS area, all of which are within the mean maximum foraging range of this species. For a review of these sites see **Section 8.21.3.1**.

3139. The NatureScot apportioning tool (NatureScot, 2018) has been used to estimate the proportion of Manx shearwaters from each of the relevant SPAs present at the windfarm site during the breeding season. The apportioning to SPA and non-SPA colonies is set out in **Table 8.189**; refer also to **Appendix 12.1** of the ES for further information on the apportioning approach and results. Accordingly, 0.22% of impacts at the windfarm site during the breeding season are apportioned to Puffin Island SPA.

*Table 8.189 Manx shearwater breeding season apportioning*

| Site | Apportioning rate |
|---|---|
| Glannau Aberdaron ac Ynys Enlli/Aberdaron Coast and Bardsey Island SPA | 8.63% |
| Copeland Islands SPA | 2.22% |
| Skomer, Skokholm and the Seas off Pembrokeshire / Sgomer, Sgogwm a Moroedd Penfro SPA | 76.54% |
| Rum SPA | 8.44% |
| St Kilda SPA | 0.20% |
| Cruagh Island SPA (transboundary site) | 0.10% |
| Blasket Islands SPA (transboundary site) | 0.61% |
| Deenish Island and Scariff Island SPA (transboundary site) | 0.08% |
| **Puffin Island SPA (transboundary site)** | **0.22%** |
| Skelligs SPA (transboundary site) | 0.03% |
| Non-SPA colonies | 2.91% |

3140. During the pre- and post-breeding periods, breeding Manx shearwaters from the Puffin Island SPA migrate through UK waters. The relevant reference population is considered to be the UK Western Waters BDMPS. This consists of 1,580,895 individuals during the post-breeding (August-early October) and return migration (late march-May) periods.

3141. Estimates of the proportion of Manx shearwaters present at the windfarm site which originate from the Puffin Island SPA during the post-breeding and return migration periods (and therefore the proportion of predicted mortalities from the SPA population) are based on the SPA population (i.e. 4,622 adults) as a proportion of the UK Western Waters BDMPS during the relevant season. During the post-breeding and return migration periods, 0.8% of impacts are considered to affect birds from the SPA (Furness, 2015).

## Potential effects on the qualifying feature from the Project-alone

3142. The Manx shearwater qualifying feature of the Puffin Island SPA has been screened into the assessment due to the potential risk of disturbance, displacement and barrier effects during the construction and decommissioning, and operation and maintenance phases of the Project.

### Construction and decommissioning phase disturbance/displacement/barrier effects

3143. Effects during the construction and decommissioning phases of the Project are considered unlikely, given the transient presence of the species and low susceptibility to disturbance related impacts; refer to **Paragraph 3145** below. However, in accordance with feedback received from Natural England and NRW, a precautionary estimation of construction and decommissioning phase disturbance, displacement and barrier effects has been undertaken assuming 50% of the operational phase effect.

3144. Applying 50% reduction to the operational values presented in **Table 8.190**, and based on mean density, predicted mortality would be between zero and two birds (30-70% displacement and 1-10% mortality of displaced birds). Using realistic values (i.e. mean density, 50% displacement and 1% mortality), there would be an annual increase in mortality of less than one (0.1) birds, which is equivalent to a 0.01% increase in background mortality for the SPA population. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. Accordingly, no significant effects on Manx shearwater are predicted during the construction and decommissioning phases, and **it is concluded that there would be no adverse effect on the integrity of Puffin Island SPA**.

### Operation and maintenance phase disturbance/displacement/barrier effects

3145. Manx shearwater is generally considered to have a low susceptibility to disturbance and displacement (Furness *et al.*, 2013). See **Section 8.21.3.1** for summary of effects from Dierschke *et al*., (2016) and Bradbury *et al.,* (2014).

3146. Displacement effects for Manx shearwater for the Project were assessed during the breeding, autumn migration and spring migration periods, based on an unapportioned peak mean population of 4,705, 2,650 and 1,617 birds respectively, calculated for the windfarm site and a 2km buffer, in line with recommendations within the SNCB guidance (SNCBs, 2017). The displacement matrices used to calculate potential impacts are presented in **Appendix 12.1** of the ES and summarised in **Table 8.190.** The application of the same displacement rate to the OWF and the 2km buffer, to determine the total number of birds subject to displacement, is precautionary, as in reality the displacement rate is likely to fall with distance from the windfarm site.

3147. A displacement rate of 30-70% and mortality rate of 1-10% has been presented. Given that 10% would represent a rate close to the expected 'natural' annual mortality (0.13), this rate is considered very unlikely. Accordingly, a 1% mortality rate is considered to be most appropriate, with the upper end of this range likely to be precautionary. Given the very extensive foraging range of this species (Woodward *et al.*, 2019), there may be no mortality costs to displacement from the relatively very small footprints of OWFs.

*Table 8.190 Manx shearwater – predicted operation and maintenance phase displacement and mortality from Puffin Island SPA*

| Mean peak abundance estimate type | Mean peak abundance estimate | Number of SPA breeding adults present by season[1] | Annual mortality range[2] | Annual baseline mortality increase range[3] |
|---|---|---|---|---|
| Upper 95% CI | 10,010 (breeding)<br>4,447 (autumn)<br>4,711 (spring)<br>19,168 (year round) | 22 (breeding)<br>36 (autumn)<br>38 (spring)<br>95 (year round) | 0-7 | 0.02-0.41% |
| Mean | 4,705 (breeding)<br>2,650 (autumn)<br>1,617 (spring)<br>8,972 (year round) | 10 (breeding)<br>21 (autumn)<br>13 (spring)<br>45 (year round) | 0-3 | 0.01-0.19% |
| Lower 95% CI | 783 (breeding)<br>1,308 (autumn)<br>0 (spring)<br>2,092 (year round) | 2 (breeding)<br>10 (autumn)<br>0 (spring)<br>12 (year round) | 0-1 | 0.00-0.05% |

[1] During the breeding season, assumes 0.2% of recorded birds are adults from the SPA population (4,622), and 0.8% during the autumn and spring migration periods

[2] Assumes displacement rates of 30-70% and mortality rates of 1-10%

[3] Background population Puffin Island SPA breeding adults (12,658 individuals), adult age class annual mortality rate of 13% (Horswill and Robinson, 2015)

3148. Using realistic values (i.e. mean density, 50% displacement and 1% mortality), there would be an annual increase in mortality of <1 (0.2) bird, representing a 0.01% increase in mortality rate. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. Accordingly, no significant effects on Manx shearwater are predicted during the operation and maintenance phase, and **it is concluded that there is no potential for the Project to have an adverse effect on the integrity of Puffin Island SPA**.

3149. A review of the potential effects of artificial light on Manx shearwaters is presented in Section 12.6.3.1 of **Chapter 12 Offshore Ornithology**. This concludes that lighting associated with the Project is very unlikely to significantly affect disturbance and displacement effects on Manx shearwater, and therefore the conclusions of the assessment are unchanged.

3150. The confidence in the assessment is high. The evidence used to set the displacement rates presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES is of high applicability and quality. Whilst there is limited available evidence to inform mortality rates, 1% is considered to be sufficiently precautionary based on expert opinion.

### Potential effects in-combination with other projects

3151. No in-combination effects are predicted during the construction and decommissioning phases. This is because it is unlikely that there would be significant temporal and/or spatial overlap with other plans or projects, and due to the negligible effects predicted from the project alone.

3152. Neither the HRA for the Round 4 offshore wind leasing (NIRAS, 2021), nor the RIAA of the recently consented Awel y Môr OWF application (Awel y Môr Offshore Wind Farm Ltd, 2022) have assessed the in-combination effects on Manx shearwater from Puffin Island SPA. In the case of the Round 4 HRA (which includes the Project), no effect on site integrity (for all SPAs) was concluded on the basis of the low vulnerability to OWFs and low density of this species within Round 4 areas. The Awel y Môr RIAA screened out in-combination effects on the basis of the small contribution of the Awel y Môr OWF and absence of linkage to populations from the SPA.

3153. Given the very low numbers of Manx shearwaters from Puffin Island SPA predicted to occur at the windfarm site, and consequent low predicted mortality increase (<1 bird), **it is considered reasonable to conclude that no significant effects on the SPA population are predicted during the operation and maintenance phase, and that there would be no potential for the Project-alone or in-combination to have an adverse effect on the integrity of Puffin Island SPA**.

#### 8.84.3.3 Puffin

##### Status

3154. The Puffin Island SPA breeding puffin population stood at 5,125 pairs, or 10,250 breeding adults, in 2000 (NPWS, 2015i). The most recent count is 1,360 individuals in 2011 (JNCC, 2023a); this is used as the reference population for the assessment.

3155. Based on the SPA population of assumed breeding adults, and an annual adult baseline mortality rate of 0.094 (1 – 0.906; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 128 breeding adults.

##### Functional linkage and seasonal apportionment of potential effects

3156. The mean maximum foraging range of puffin is 137.1km (±128.3km) and the maximum foraging range is 383km (Woodward *et al*., 2019). The Project is located approximately 568km from Puffin Island SPA, which means the Project is beyond the maximum foraging range for this species. No impacts during the breeding season are therefore apportioned to birds breeding at this colony.

3157. Outside of the breeding season, breeding puffins, including those from the Puffin Island SPA, are not constrained by requirements to visit nests to incubate eggs or provision chicks. At this time, they are assumed to range more widely and to mix with puffins of all age classes from breeding colonies in Ireland, the UK and further afield. The background population during these seasons is the UK Western Waters BDMPS. This consists of 304,557 individuals during the non-breeding season (August to March).

3158. As no published estimate was available, it is assumed that 10% of Puffin Island SPA breeding adults are present within the UK Western Waters BDMPS during the non-breeding season. This is the rate for 'Ireland' populations identified by Furness (2015) and is considered appropriate given the geographic isolation between the west coast of Ireland (where the SPA is located) and the windfarm site. It is assumed that the most recent count prior to the publication of Furness (2015) was used to inform the BDMPS total which is 1,360 breeding adults; 10% of this population is 136 birds. This represents 0.04% of the BDMPS population for this period (304,557). It is therefore assumed that 0.04% of puffins present at the Project site are breeding adults from Puffin Island SPA.

## Potential effects on the qualifying feature

### Operation and maintenance phase disturbance/displacement/barrier effects

*Project-alone*

3159. The mean peak abundance of puffins present within the windfarm site and 2km buffer during the non-breeding season was 19.7 (1.9-50.8) individuals (refer to **Appendix 12.1** of the ES). Of these, less than one bird (0.0 (0.0-0.0)) was likely to be a breeding adult from Puffin Island SPA.

3160. **Table 8.191** sets out the predicted impacts on puffins from Puffin Island SPA during the non-breeding season. This estimates that there would be no measurable increase in mortality from the SPA population, assuming a displacement rate of 30-70% and a mortality of 1-10% for displaced birds.

*Table 8.191 Puffin – predicted operation and maintenance phase displacement and mortality from Puffin Island SPA*

| Mean peak abundance estimate type | Mean peak abundance estimate | Number of Puffin Island SPA breeding adults present (non-breeding season)[1] | Annual mortality range[2] | Annual baseline mortality increase range[3] |
|---|---|---|---|---|
| Upper 95% CI | 50.8 | 0.0 | 0-0 | 0.00-0.00% |
| Mean | 19.7 | 0.0 | 0-0 | 0.00-0.00% |
| Lower 95% CI | 1.9 | 0.0 | 0-0 | 0.00-0.00% |

[1] Assumes 0.04% of birds present during the non-breeding season are Puffin Island SPA breeding adults

[2] Assumes displacement rates of 30-70% and mortality rates of 1-10%

[3] Background mortality rate of 9.4% (Horswill and Robinson, 2015)

3161. **It is concluded that predicted puffin mortality due to operational phase displacement would not adversely affect the integrity of the Puffin Island SPA.**

3162. The confidence in the assessment is high for several reasons. Firstly, the evidence used to inform the evidence-based displacement rates is of high applicability and quality (based on the criteria discussed in **Chapter 12 Offshore Ornithology** of the ES). Whilst there is limited available evidence to inform mortality rates, 1% is considered to be sufficiently precautionary based on expert opinion. This species is not regarded as being highly specialised in its habitat requirements (Bradbury *et al*., 2014; Furness and Wade, 2012; Garthe and Hüppop, 2004), and it is therefore anticipated that displaced birds will find alternative habitat in the vast majority of cases. Finally, the conclusion of the assessment is the same irrespective of whether the mean or 95% upper

CI mean peak abundances are used to calculate potential mortality and increases in the baseline mortality rate of the background population.

*In-combination*

3163. As the Project would have no measurable effect on puffin populations from the Puffin Island SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that predicted puffin mortality due to operational phase displacement due to the Project in-combination with other plans or projects would not adversely affect the integrity of the Puffin Island SPA.**

# 8.85 The Bull and The Cow Rocks SPA (transboundary site)

3164. The Bull and The Cow Rocks SPA is located on the west coast of Ireland approximately 505km from the windfarm site (straight line) or 549km (across sea).

## 8.85.1 Description of designation

3165. The site comprises two very small rocky islands, The Cow and The Bull, situated 2.5 km and 4 km respectively from Dursey Head off the coast of Co. Cork. The islands, which are composed of vertically stratified sandstone, rise to over 60m and are generally precipitous. A few rocky islets occur off the main islands. The surrounding water, between and to a distance of 500 m around each island, is included within the site for breeding seabirds.

## 8.85.2 Conservation objectives

3166. The conservation objective of the SPA is 'to maintain or restore the favourable conservation condition of the bird species listed as Special Conservation Interests for this SPA.'

## 8.85.3 Assessment

3167. One qualifying feature of The Bull and The Cow Rocks SPA has been screened into the Appropriate Assessment (**Table 5.2**): gannet.

### 8.85.3.1 Gannet

#### Status

3168. The Bull and The Cow Rocks SPA breeding gannet population stood at 3,694 pairs, or 7,388 breeding adults, in 2004 (NPWS, 2014e). The most recent count is 6,388 pairs (AON), or 12,776 breeding adults, in 2014 (JNCC, 2023a); this is used as the reference population for the assessment.

3169. Based on the most recent SPA population of breeding adults, and an annual breeding adult baseline mortality rate of 0.081 (1 – 0.919; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 1,035 breeding adults.

#### Functional linkage and seasonal apportionment of potential effects

3170. The windfarm site is 549km from The Bull and The Cow Rocks SPA. The mean maximum foraging range of gannet is 315.2km (±194.2km), which means that the Project is beyond the mean maximum foraging range +1SD for gannets from the SPA, but within the maximum foraging range. The maximum foraging range is a poor indicator of typical foraging behaviour. It would be expected that few birds or foraging trips will occur at this distance from the colony, and even fewer with any regularity. No impacts during the breeding season are therefore apportioned to birds breeding at this colony.

3171. Outside the breeding season breeding gannets, including those from The Bull and The Cow Rocks SPA, are not constrained by requirements to visit nests to incubate eggs or provision chicks. At this time, they are assumed to range more widely and to mix with gannets of all age classes from breeding colonies in Ireland, the UK and further afield. The background population during these seasons is the UK Western Waters BDMPS. This consists of 545,954 individuals during autumn migration (September to November), and 661,888 individuals during spring migration (December to March) (Furness, 2015).

3172. As no published estimate was available, it is assumed that 20% of The Bull and The Cow Rocks SPA breeding adults (12,766; the most recent count prior to the publication of Furness, 2015) are present within the UK Western Waters BDMPS during the autumn migration period, which is 2,553 birds, and that 30% the SPA population (i.e. 3,830 birds) is present during spring migration. These are the rates for 'Ireland' populations identified by Furness (2015), and is considered appropriate given the geographic isolation between the west coast of Ireland (where the SPA is located) and the windfarm site.

3173. Estimates of the proportion of gannets present at the windfarm site which originate from The Bull and The Cow Rocks SPA during the non-breeding season (and therefore the proportion of predicted mortalities from the SPA population) are based on these population estimates as a proportion of the UK Western Waters BDMPS during the relevant season. During autumn migration and spring migration, 0.5%, and 0.6% of impacts are considered to affect birds from the SPA respectively.

## Potential effects on the qualifying feature from the Project-alone

3174. The gannet qualifying feature of The Bull and The Cow Rocks SPA has been screened into the assessment due to the potential risk of collision and operational phase displacement/barrier effects during the operation and maintenance phase of the Project.

### Operation and maintenance phase displacement/barrier effects

3175. Displacement effects for gannet for the Project were assessed during the autumn and spring migration periods, based on an unapportioned peak mean population of 124 and eight birds respectively, calculated for the windfarm site and a 2km buffer, in line with recommendations within the SNCB guidance (SNCB 2017). As set out above, no gannets present at the windfarm site have been apportioned to The Bull and The Cow Rocks SPA during the breeding season. The displacement matrices used to calculate potential impacts are presented in **Appendix 12.1** of the ES, and summarised in **Table 8.192**. The inclusion of all birds within the 2km buffer, to determine the total number of birds subject to displacement, is precautionary, as in reality the avoidance rate is likely to fall with distance from the windfarm site.

3176. A displacement rate of 60-80% and mortality rate of 1% has been presented. A maximum 1% mortality value has been selected firstly because gannet is known to possess high habitat flexibility (Furness and Wade, 2012). This suggests that displaced birds will readily find alternative habitats including foraging areas. Secondly, no evidence of displacement-induced mortality has been identified, which means there is limited justification for setting predicted mortality rates at a higher level. Given the extensive foraging range of this species (Woodward *et al.*, 2019), there may be no mortality costs to displacement from the relatively very small footprints of OWFs.

*Table 8.192 Gannet – predicted operation and maintenance phase displacement and mortality from The Bull and The Cow Rocks SPA*

| Mean peak abundance estimate type | Mean peak abundance estimate | Number of SPA breeding adults present by season[1] | Annual mortality range[2] | Annual baseline mortality increase range[3] |
|---|---|---|---|---|
| Upper 95% CI | 809 (breeding)<br>189 (autumn)<br>16 (spring)<br>1,014 (year round) | 0 (breeding)<br>1 (autumn)<br>0 (spring)<br>1 (year round) | 0-0 | 0.00-0.00% |
| Mean | 541 (breeding)<br>124 (autumn)<br>8 (spring)<br>673 (year round) | 0 (breeding)<br>1 (autumn)<br>0 (spring)<br>1 (year round) | 0-0 | 0.00-0.00% |
| Lower 95% CI | 160 (breeding)<br>0 (autumn)<br>0 (spring)<br>160 (year round) | 0 (breeding)<br>0 (autumn)<br>0 (spring)<br>0 (year round) | 0-0 | 0.00-0.00% |

[1]0.5% and 0.6% of birds are assumed to be breeding adults from the SPA population during the autumn and spring migration periods respectively.

[2] Assumes displacement rates of 60-80% and mortality rate of 1%

[3] Background population is The Bull and The Cow Rocks SPA breeding adults (12,766 individuals), adult age class annual mortality rate of 8.1% (Horswill and Robinson, 2015)

3177. Using the maximum potential mortality value, there would be no measurable increase in gannet mortality. Accordingly, no significant effects on gannet are predicted during the operation and maintenance phase, and **it is concluded that there would be no potential for the Project to have an adverse effect on the integrity of The Bull and The Cow Rocks SPA**.

3178. The confidence in the assessment is high for several reasons. Firstly, the evidence used to set the displacement rates presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES is of high applicability and quality. Whilst there is limited available evidence to inform mortality rates, 1% is considered to be sufficiently precautionary based on expert opinion. Finally, the conclusion of the assessment is the same irrespective of whether the mean or 95% upper CI mean peak abundances are used to calculate potential mortality and increases in the baseline mortality rate of the background population.

**Operation and maintenance phase collision risk**

3179. Information to support the Appropriate Assessment for collision risk on breeding adult gannets belonging to The Bull and The Cow Rocks SPA population is presented in **Table 8.193**. Collision estimates, calculated using the sCRM, are presented by biological season. A summary of the annual outputs and the corresponding increase in the annual baseline mortality rate is also presented. Parameters used in the sCRM were agreed with Natural England during the ETG process and are described in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES. In accordance with Natural England advice, a 70% macro-avoidance correction was applied to gannet abundance data used in the sCRM.

3180. Based on the mean collision rates, no breeding adult gannets from The Bull and The Cow Rocks SPA are considered at risk of collision as a result of the Project. Therefore, there would be no measurable increase in the existing mortality of the SPA breeding population.

*Table 8.193 Predicted seasonal and annual collision mortality (Stochastic model Option 2, avoidance rate 0.993 (±0.0003), plus 70% macro-avoidance) for breeding adult gannets at the windfarm site, apportioned to The Bull and The Cow Rocks SPA, with corresponding increases to baseline mortality of the population*

| | Breeding Season | Autumn Migration | Non-breeding/winter | Spring Migration | Annual |
|---|---|---|---|---|---|
| **Period** | **Apr-Aug** | **Sep-Nov** | **-** | **Dec-Mar** | **Jan-Dec** |
| Total collisions[1] (mean and 95% CIs) | 0.83 (0.00-3.35) | 0.14 (0.00-0.74) | - | 0.00 | 0.97 (0.00-4.10) |
| % apportioned to the SPA | 0.0% | 0.5% | - | 0.6% | - |
| Total SPA collisions (mean and 95% CIs) | 0.00 (0.00-0.00) | 0.00 (0.00-0.00) | - | 0.00 (0.00-0.00) | 0.00 (0.00-0.00) |
| Mortality increase[2] (mean and 95% CIs) | 0.00% (0.00-0.00%) | 0.00% (0.00-0.00%) | - | 0.00% (0.00-0.00%) | 0.00% (0.00-0.00%) |

[1] Breeding season collision values reduced to 73.8% of modelled value to reflect proportion of adult birds recorded during site surveys
[2] Assuming predicted annual SPA mortality of 1,035 birds (12,766 x 0.081)

3181. Accordingly, no significant effects on gannet are predicted during the operation and maintenance phase, and **it is concluded that there would be no potential for the Project to have an adverse effect on the integrity of The Bull and The Cow Rocks SPA**. Comments received from RSPB during the ETG process, indicating that they do not accept the 70% macro-avoidance rate for collision risk recommended by Natural England, are noted. However, even in the absence of this correction factor, the net increase in mortality would be unchanged (i.e. zero).

3182. The confidence in the assessment is high. The evidence used to define the CRM input parameters presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES submitted alongside this RIAA is of high applicability and quality. Whilst there is uncertainty around some of the input parameters (e.g. avoidance rate), the rates selected are considered to be sufficiently precautionary based on expert opinion to provide confidence that collision rates are not underestimated.

**Combined displacement/barrier effects and collision risk**

3183. As no measurable increase in mortality is predicted for both displacement and collision risk, the mean combined displacement and collision rates for breeding adult gannet from The Bull and The Cow Rocks SPA would be zero. Therefore, there would be no net increase in existing mortality rates.

3184. **It is concluded that based on predicted gannet mortality due to the combined effects of operational phase displacement and collision there is no potential for the Project-alone or in-combination to have an adverse effect on the integrity of The Bull and The Cow Rocks SPA**.

3185. The confidence in the assessment is high, for the reasons provided in the individual displacement and collision assessments.

### Potential effects in-combination with other projects

3186. As no measurable effects of displacement /barrier and collision on gannet are predicted as a result of the Project-alone, there would be no contribution to other plans or projects in-combination. **It is therefore concluded that there would be no potential for the Project-alone or in-combination to have an adverse effect on the integrity of The Bull and The Cow Rocks SPA**.

# 8.86 Skelligs SPA (transboundary site)

3187. Skelligs SPA is located on the west coast of Ireland approximately 508km from the windfarm site (straight line) or 579km (across sea).

## 8.86.1 Description of designation

3188. The site comprises Great Skellig and Little Skellig islands. These highly exposed and isolated islands, which are separated by a distance of 3km, are located in the Atlantic some 14km and 11km (respectively) off the Co. Kerry mainland. Both islands are precipitous rocky sea stacks, Great Skellig rising to 218m and Little Skellig to 134m. The qualifying seabird species of the SPA comprise fulmar, Manx shearwater, storm petrel, gannet, kittiwake, guillemot and puffin.

## 8.86.2 Conservation objectives

3189. The conservation objective of the SPA is 'to maintain or restore the favourable conservation condition of the bird species listed as Special Conservation Interests for this SPA.'

## 8.86.3 Assessment

3190. Four qualifying features of Skelligs SPA have been screened into the Appropriate Assessment (**Table 5.2**): gannet, Manx shearwater, fulmar and puffin.

### 8.86.3.1 Gannet

#### Status

3191. The Skelligs SPA breeding gannet population stood at 29,683 pairs, or 59,366 breeding adults, in 2004 (NPWS, 2015j). The most recent count was 35,294 pairs (AON), or 70,588 breeding adults, in 2014 (JNCC, 2023a); this is used as the reference population for the assessment.

3192. Based on the most recent SPA population of breeding adults, and an annual breeding adult baseline mortality rate of 0.081 (1 – 0.919; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 5,718 breeding adults.

#### Functional linkage and seasonal apportionment of potential effects

3193. The windfarm site is 579km from Skelligs SPA. The mean maximum foraging range of gannet is 315.2km (±194.2km), which means that the Project is beyond the mean maximum foraging range +1SD for gannets from the SPA, but within the maximum foraging range. The maximum foraging range is a poor indicator of typical foraging behaviour. It would be expected that few birds or foraging trips will occur at this distance from the colony, and even fewer with any regularity. No impacts during the breeding season are therefore apportioned to birds breeding at this colony.

3194. Outside the breeding season breeding gannets, including those from the Skelligs SPA, are not constrained by requirements to visit nests to incubate eggs or provision chicks. At this time, they are assumed to range more widely and to mix with gannets of all age classes from breeding colonies in Ireland, the UK and further afield. The background population during these seasons is the UK Western Waters BDMPS. This consists of 545,954 individuals during autumn migration (September to November), and 661,888 individuals during spring migration (December to March) (Furness, 2015).

3195. As no published estimate was available, it is assumed that 20% of the Skelligs SPA breeding adults (70,588; the most recent count prior to the publication of Furness, 2015) are present within the UK Western Waters BDMPS during the autumn migration period, which is 14,118 birds, and that 30% the SPA population (i.e. 21,176 birds) is present during spring migration. These are the rates for 'Ireland' populations identified by Furness (2015), and is considered appropriate given the geographic isolation between the west coast of Ireland (where the SPA is located) and the windfarm site.

3196. Estimates of the proportion of gannets present at the windfarm site which originate from the Skelligs SPA during the non-breeding season (and therefore the proportion of predicted mortalities from the SPA population) are based on these population estimates as a proportion of the UK Western Waters BDMPS during the relevant season. During autumn migration and spring migration, 2.6%, and 3.2% of impacts are considered to affect birds from the SPA respectively

## Potential effects on the qualifying feature from the Project-alone

3197. The gannet qualifying feature of the Skelligs SPA has been screened into the assessment due to the potential risk of collision and operational phase displacement/barrier effects during the operation and maintenance phase of the Project.

### Operation and maintenance phase displacement/barrier effects

3198. Displacement effects for gannet for the Project were assessed during the autumn and spring migration periods, based on an unapportioned peak mean population of 124 and eight birds respectively, calculated for the windfarm site and a 2km buffer, in line with recommendations within the SNCB guidance (SNCB 2017). As set out above, no gannets present at the windfarm site have been apportioned to Skelligs SPA during the breeding season. The displacement matrices used to calculate potential impacts are presented in

**Appendix 12.1** of the ES, and summarised in **Table 8.194**. The inclusion of all birds within the 2km buffer, to determine the total number of birds subject to displacement, is precautionary, as in reality the avoidance rate is likely to fall with distance from the windfarm site.

3199. A displacement rate of 60-80% and mortality rate of 1% has been presented. A maximum 1% mortality value has been selected firstly because gannet are known to possess high habitat flexibility (Furness and Wade, 2012). This suggests that displaced birds will readily find alternative habitats including foraging areas. Secondly, no evidence of displacement-induced mortality has been identified, which means there is limited justification for setting predicted mortality rates at a higher level. Given the extensive foraging range of this species (Woodward *et al.*, 2019), there may be no mortality costs to displacement from the relatively very small footprints of OWFs.

*Table 8.194 Gannet – predicted operation and maintenance phase displacement and mortality from Skelligs SPA*

| Mean peak abundance estimate type | Mean peak abundance estimate | Number of SPA breeding adults present by season[1] | Annual mortality range[2] | Annual baseline mortality increase range[3] |
|---|---|---|---|---|
| Upper 95% CI | 809 (breeding)<br>189 (autumn)<br>16 (spring)<br>1,014 (year round) | 0 (breeding)<br>5 (autumn)<br>1 (spring)<br>5 (year round) | 0-0 | 0.00-0.00% |
| Mean | 541 (breeding)<br>124 (autumn)<br>8 (spring)<br>673 (year round) | 0 (breeding)<br>3 (autumn)<br>0 (spring)<br>4 (year round) | 0-0 | 0.00-0.00% |
| Lower 95% CI | 160 (breeding)<br>0 (autumn)<br>0 (spring)<br>160 (year round) | 0 (breeding)<br>0 (autumn)<br>0 (spring)<br>0 (year round) | 0-0 | 0.00-0.00% |

[1]2.6% and 3.2% of birds are assumed to be breeding adults from the SPA population during the autumn and spring migration periods respectively.
[2] Assumes displacement rates of 60-80% and mortality rate of 1%
[3] Background population is Skelligs SPA breeding adults (70,588 individuals), adult age class annual mortality rate of 8.1% (Horswill and Robinson, 2015)

3200. Using the maximum potential mortality value, there would be no measurable increase in gannet mortality. Accordingly, no significant effects on gannet are predicted during the operation and maintenance phase, and **it is concluded that there would be no potential for the Project to have an adverse effect on the integrity of Skelligs SPA**.

3201. The confidence in the assessment is high for several reasons. Firstly, the evidence used to set the displacement rates presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES is of high applicability and quality. Whilst there is limited available evidence to inform mortality rates, 1% is considered to be sufficiently precautionary based on expert opinion. Finally, the conclusion of the assessment is the same irrespective of whether the mean or 95% upper CI mean peak abundances are used to calculate potential mortality and increases in the baseline mortality rate of the background population.

**Operation and maintenance phase collision risk**

3202. Information to support the Appropriate Assessment for collision risk on breeding adult gannets belonging to the Skelligs SPA population is presented in **Table 8.195**. Collision estimates, calculated using the sCRM, are presented by biological season. A summary of the annual outputs and the corresponding increase in the annual baseline mortality rate is also presented. Parameters used in the sCRM were agreed with Natural England during the ETG process and are described in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES. In accordance with Natural England advice, a 70% macro-avoidance correction was applied to gannet abundance data used in the sCRM.

3203. Based on the mean collision rates, no breeding adult gannets from Skelligs SPA are considered at risk of collision as a result of the Project. Therefore, there would be no measurable increase the existing mortality of the SPA breeding population.

*Table 8.195 Predicted seasonal and annual collision mortality (Stochastic model Option 2, avoidance rate 0.993 (±0.0003), plus 70% macro-avoidance) for breeding adult gannets at the windfarm site, apportioned to Skelligs SPA, with corresponding increases to baseline mortality of the population*

| Period | Breeding Season<br>Apr-Aug | Autumn Migration<br>Sep-Nov | Non-breeding/winter<br>- | Spring Migration<br>Dec-Mar | Annual<br>Jan-Dec |
|---|---|---|---|---|---|
| Total collisions[1] (mean and 95% CIs) | 0.83<br>(0.00-3.35) | 0.14<br>(0.00-0.74) | - | 0.00 | 0.97<br>(0.00-4.10) |
| % apportioned to the SPA | 0.0% | - | - | - | - |
| Total SPA collisions (mean and 95% CIs) | 0.00<br>(0.00-0.00) | 0.00<br>(0.00-0.02) | - | 0.00<br>(0.00-0.00) | 0.00<br>(0.00-0.02) |
| Mortality increase[2] (mean and 95% CIs) | 0.00%<br>(0.00-0.00%) | 0.00%<br>(0.00-0.00%) | - | 0.00%<br>(0.00-0.00%) | 0.00%<br>(0.00-0.00%) |

[1] Breeding season collision values reduced to 73.8% of modelled value to reflect proportion of adult birds recorded during site surveys
[2] Assuming predicted annual SPA mortality of 5,718 birds (70,588 x 0.081)

3204. Accordingly, no significant effects on gannet are predicted during the operation and maintenance phase, and **it is concluded that there would be no potential for the Project to have an adverse effect on the integrity of Skelligs SPA**.

3205. The confidence in the assessment is high. The evidence used to define the CRM input parameters presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES submitted alongside this RIAA is of high applicability and quality. Whilst there was uncertainty around some of the input parameters (e.g. avoidance rate), the rates selected are considered to be sufficiently precautionary based on expert opinion to provide confidence that collision rates are not underestimated.

**Combined displacement/barrier effects and collision risk**

3206. As no measurable increase in mortality is predicted for both displacement and collision risk, the mean combined displacement and collision rates for breeding adult gannet from the Skelligs SPA would be zero. Therefore, there would be no net increase in existing mortality rates. Comments received from RSPB during the ETG process, indicating that they do not accept the 70% macro-avoidance rate for collision risk recommended by Natural England, are noted. However, even in the absence of this correction factor, the net increase in mortality would be unchanged (i.e. zero).

3207. **It is concluded that based on predicted gannet mortality due to the combined effects of operational phase displacement and collision there is no potential for the Project-alone to have an adverse effect on the integrity of the Skelligs SPA**.

3208. The confidence in the assessment is high, for the reasons provided in the individual displacement and collision assessments.

### Potential effects in-combination with other projects

3209. As no measurable effects of displacement /barrier and collision on gannet are predicted as a result of the Project-alone, there would be no contribution to other plans or projects in-combination. **It is therefore concluded that there would be no potential for the Project-alone or in-combination to have an adverse effect on the integrity of Skelligs SPA**.

### 8.86.3.2 Manx shearwater

### Status

3210. The Skelligs SPA breeding Manx shearwater population stood at 902 pairs, or 1,804 breeding adults, in 2002 (NPWS, 2015j). The SMP database (JNCC, 2023a) did not provide a more recent estimate, therefore the 2002 count has been used as the reference population for the assessment.

3211. Based on the most recent SPA population of breeding adults, and an annual breeding adult baseline mortality rate of 0.130 (1 – 0.870; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 235 breeding adults.

### Functional linkage and seasonal apportionment of potential effects

3212. The mean maximum foraging range of Manx shearwater is 1346.8km (±1018.7km) and the maximum foraging range is 2890km. The Project is located approximately 579km from Skelligs SPA, which means that the Project is within the mean maximum foraging range of Manx shearwaters breeding at this SPA.

3213. A number of SPA and non-SPA Manx shearwater colonies are located in and around the UK Western Waters BDMPS area, all of which are within the mean maximum foraging range of this species. For a review of these sites see **Section 8.21.3.1**.

3214. The NatureScot apportioning tool (NatureScot, 2018) has been used to estimate the proportion of Manx shearwaters from each of the relevant SPAs present at the windfarm site during the breeding season. The apportioning to SPA and non-SPA colonies is set out in **Table 8.196**; refer also to **Appendix 12.1** of the ES for further information on the apportioning approach and results. Accordingly, 0.03% of impacts at the windfarm site during the breeding season are apportioned to Skelligs SPA.

*Table 8.196 Manx shearwater breeding season apportioning*

| Site | Apportioning rate |
|---|---|
| Glannau Aberdaron ac Ynys Enlli/Aberdaron Coast and Bardsey Island SPA | 8.63% |
| Copeland Islands SPA | 2.22% |
| Skomer, Skokholm and the Seas off Pembrokeshire / Sgomer, Sgogwm a Moroedd Penfro SPA | 76.54% |
| Rum SPA | 8.44% |
| St Kilda SPA | 0.20% |
| Cruagh Island SPA (transboundary site) | 0.10% |
| Blasket Islands SPA (transboundary site) | 0.61% |
| Deenish Island and Scariff Island SPA (transboundary site) | 0.08% |
| Puffin Island SPA (transboundary site) | 0.22% |
| **Skelligs SPA (transboundary site)** | **0.03%** |
| Non-SPA colonies | 2.91% |

3215. During the pre- and post-breeding periods, breeding Manx shearwaters from the Skelligs SPA migrate through UK waters. The relevant reference population is considered to be the UK Western Waters BDMPS. This consists of 1,580,895 individuals during the post-breeding (August-early October) and return migration (late march-May) periods.

3216. Estimates of the proportion of Manx shearwaters present at the windfarm site which originate from the Skelligs SPA during the post-breeding and return migration periods (and therefore the proportion of predicted mortalities from the SPA population) are based on the SPA population (i.e. 1,804 adults) as a proportion of the UK Western Waters BDMPS during the relevant season. During the post-breeding and return migration periods, 0.1% of impacts are considered to affect birds from the SPA (Furness, 2015).

## Potential effects on the qualifying feature from the Project-alone

3217. The Manx shearwater qualifying feature of the Skelligs SPA has been screened into the assessment due to the potential risk of disturbance, displacement and barrier effects during the construction and decommissioning, and operation and maintenance phases of the Project.

### Construction and decommissioning phase disturbance/displacement/barrier effects

3218. Effects during the construction and decommissioning phases of the Project are considered unlikely, given the transient presence of the species and low susceptibility to disturbance related impacts; refer to **Paragraph 3220** below. However, in accordance with feedback received from Natural England and NRW, a precautionary estimation of construction and decommissioning phase disturbance, displacement and barrier effects has been undertaken assuming 50% of the operational phase effect.

3219. Applying 50% reduction to the operational values presented in **Table 8.197**, and based on mean density, predicted mortality would be between zero and zero birds (30-70% displacement and 1-10% mortality of displaced birds). Using realistic values (i.e. mean density, 50% displacement and 1% mortality), there would be an annual increase in mortality of less than one (0.02) birds, which is equivalent to a 0.01% increase in background mortality for the SPA population. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. Accordingly, no significant effects on Manx shearwater are predicted during the construction and decommissioning phases, and **it is concluded that there would be no adverse effect on the integrity of the Skelligs SPA**.

### Operation and maintenance phase disturbance/displacement/barrier effects

3220. Manx shearwater are generally considered to have a low susceptibility to disturbance and displacement (Furness *et al.*, 2013). See **Section 8.21.3.1** for summary of effects from Dierschke *et al*., (2016) and Bradbury *et al.,* (2014).

3221. Displacement effects for Manx shearwater for the Project were assessed during the breeding, autumn migration and spring migration periods, based on an unapportioned peak mean population of 4,705, 2,650 and 1,617 birds respectively, calculated for the windfarm site and a 2km buffer, in line with recommendations within the SNCB guidance (SNCBs, 2017). The displacement matrices used to calculate potential impacts are presented in **Appendix 12.1** of the ES, and summarised in **Table 8.197.** The application of the same displacement rate to the OWF and the 2km buffer, to determine the total number of birds subject to displacement, is precautionary, as in reality the displacement rate is likely to fall with distance from the windfarm site.

3222. A displacement rate of 30-70% and mortality rate of 1-10% has been presented. Given that 10% would represent a rate close to the expected ‘natural’ annual mortality (0.13), this rate is considered very unlikely. Accordingly, a 1% mortality rate is considered to be most appropriate, with the upper end of this range likely to be precautionary. Given the very extensive foraging range of this species (Woodward *et al.*, 2019), there may be no mortality costs to displacement from the relatively very small footprints of OWFs.

*Table 8.197 Manx shearwater – predicted operation and maintenance phase displacement and mortality from Skelligs SPA*

| Mean peak abundance estimate type | Mean peak abundance estimate | Number of SPA breeding adults present by season[1] | Annual mortality range[2] | Annual baseline mortality increase range[3] |
|---|---|---|---|---|
| Upper 95% CI | 10,010 (breeding)<br>4,447 (autumn)<br>4,711 (spring)<br>19,168 (year round) | 22 (breeding)<br>36 (autumn)<br>38 (spring)<br>95 (year round) | 0-1 | 0.02-0.40% |
| Mean | 4,705 (breeding)<br>2,650 (autumn)<br>1,617 (spring)<br>8,972 (year round) | 10 (breeding)<br>21 (autumn)<br>13 (spring)<br>45 (year round) | 0-0 | 0.01-0.19% |
| Lower 95% CI | 783 (breeding)<br>1,308 (autumn)<br>0 (spring)<br>2,092 (year round) | 2 (breeding)<br>10 (autumn)<br>0 (spring)<br>12 (year round) | 0-0 | 0.00-0.05% |

[1] During the breeding season, assumes 0.03% of recorded birds are adults from the SPA population (1,804), and 0.1% during the autumn and spring migration periods

[2] Assumes displacement rates of 30-70% and mortality rates of 1-10%

[3] Background population Skelligs SPA breeding adults (1,804 individuals), adult age class annual mortality rate of 13% (Horswill and Robinson, 2015)

3223. Using realistic values (i.e. mean density, 50% displacement and 1% mortality), there would be an annual increase in mortality of <1 (0.03) bird, representing a 0.01% increase in mortality rate. Increases in the existing mortality rate of less than 1% are likely to be undetectable against natural variation. Accordingly, no significant effects on Manx shearwater are predicted during the operation and maintenance phase, and **it is concluded that there is no potential for the Project to have an adverse effect on the integrity of the Skelligs SPA**.

3224. A review of the potential effects of artificial light on Manx shearwaters is presented in Section 12.6.3.1 of **Chapter 12 Offshore Ornithology**. This concludes that lighting associated with the Project is very unlikely to significantly affect disturbance and displacement effects on Manx shearwater, and therefore the conclusions of the assessment are unchanged.

3225. The confidence in the assessment is high. The evidence used to set the displacement rates presented in **Chapter 12 Offshore Ornithology** and **Appendix 12.1** of the ES is of high applicability and quality. Whilst there is limited available evidence to inform mortality rates, 1% is considered to be sufficiently precautionary based on expert opinion.

### Potential effects in-combination with other projects

3226. No in-combination effects are predicted during the construction and decommissioning phases. This is because it is unlikely that there would be significant temporal and/or spatial overlap with other plans or projects, and due to the negligible effects predicted from the project alone.

3227. Neither the HRA for the Round 4 offshore wind leasing (NIRAS, 2021), nor the RIAA of the recently consented Awel y Môr OWF application (Awel y Môr Offshore Wind Farm Ltd, 2022) have assessed the in-combination effects on Manx shearwater from the Skelligs SPA. In the case of the Round 4 HRA (which includes the Project), no effect on site integrity (for all SPAs) was concluded on the basis of the low vulnerability to OWFs and low density of this species within Round 4 areas. The Awel y Môr RIAA screened out in-combination effects on the basis of the small contribution of the Awel y Môr OWF and absence of linkage to populations from the SPA.

3228. Given the very low numbers of Manx shearwaters from the Skelligs SPA predicted to occur at the windfarm site, and consequent low predicted mortality increase (<1 bird**), it is considered reasonable to conclude that no significant effects on the SPA population are predicted during the operation and maintenance phase, and that there would be no potential for the Project-alone or in-combination to have an adverse effect on the integrity of the Skelligs SPA**.

#### 8.86.3.3 Fulmar

**Status**

3229. The Skelligs SPA breeding fulmar population stood at 830 pairs, or 1,660 breeding adults, in 2002 (NPWS, 2015j). The most recent count was 795 pairs, or 1,590 breeding adults, in 2021 (JNCC, 2023a); this is used as the reference population for the assessment.

3230. Based on the most recent SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.064 (1 – 0.936; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 102 breeding adults.

**Functional linkage and seasonal apportionment of potential effects**

3231. The mean maximum foraging range of fulmar is 542.3km (±657.9km) and the maximum foraging range is 2,736km (Woodward *et al*., 2019). The Project is located approximately 579km from the Skelligs SPA, which means that the Project is beyond the mean maximum foraging range of fulmars breeding at this SPA, but within the mean maximum foraging range +1SD, and the maximum foraging range.

**Potential effects on the qualifying feature from the Project-alone**

3232. Fulmar are considered to have low vulnerability to both collision and displacement impacts from offshore windfarms (Bradbury *et al*., 2014). This species was recorded infrequently within the windfarm site (in only five of 24 surveys, all during the breeding season (January to August; Furness, 2015)) and in low numbers, with densities in four of the five months less than 0.1 birds/km$^2$. A peak density of 0.23 birds/km$^2$ was recorded in May 2022.

3233. On the basis of the low vulnerability of fulmars to potential impacts, and the low frequency and abundance of birds present at the windfarm site, it is concluded that any effects on breeding populations at Skelligs SPA are very unlikely, both during and outside of the breeding season.

3234. **It is therefore concluded that there would be no measurable effects on fulmar due to the project alone, and no adverse effect on the integrity of the Skelligs SPA is predicted.**

**Potential effects on the qualifying feature in-combination with other projects**

3235. As the Project would have no measurable effect on fulmar populations from the Skelligs SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that there would be no adverse effect on the integrity of Skelligs SPA, when assessed in-combination with other plans or projects.**

### 8.86.3.4 Puffin

#### Status

3236. The Skelligs SPA breeding puffin population was estimated at 6,000 pairs, or 12,000 breeding adults, in 2002 (NPWS, 2015j). More recent counts were available on the SMP database (JNCC, 2023a), however, it was unclear whether these counts covered the full extent of the SPA. The 2002 estimate is therefore used as the reference population for the assessment.

3237. Based on the SPA population of assumed breeding adults, and an annual breeding adult baseline mortality rate of 0.094 (1 – 0.906; Horswill and Robinson 2015), the expected annual mortality from the SPA population would be 1,128 breeding adults.

#### Functional linkage and seasonal apportionment of potential effects

3238. The mean maximum foraging range of puffin is 137.1km (±128.3km) and the maximum foraging range is 383km (Woodward *et al*., 2019). The Project is located approximately 579km from Skelligs SPA, which means the Project is beyond the maximum foraging range for this species. No impacts during the breeding season are therefore apportioned to birds breeding at this colony.

3239. Outside of the breeding season, breeding puffins, including those from the Skelligs SPA, are not constrained by requirements to visit nests to incubate eggs or provision chicks. At this time, they are assumed to range more widely and to mix with puffins of all age classes from breeding colonies in Ireland, the UK and further afield. The background population during these seasons is the UK Western Waters BDMPS. This consists of 304,557 individuals during the non-breeding season (August to March).

3240. As no published estimate was available, it is assumed that 10% of the Skelligs SPA breeding adults are present within the UK Western Waters BDMPS during the non-breeding season. This is the rate for ‘Ireland’ populations identified by Furness (2015), and is considered appropriate given the geographic isolation between the west coast of Ireland (where the SPA is located) and the windfarm site. It is assumed that the most recent count prior to the publication of Furness (2015) was used to inform the BDMPS total which is 12,000 breeding adults; 10% of this population is 1,200 birds. This represents 0.4% of the BDMPS population for this period (304,557). It is therefore assumed that 0.4% of puffins present at the Project site are breeding adults from Skelligs SPA.

Potential effects on the qualifying feature

**Operation and maintenance phase disturbance/displacement/barrier effects**

*Project-alone*

3241. The mean peak abundance of puffins present within the windfarm site and 2km buffer during the non-breeding season was 19.7 (1.9-50.8) individuals (refer to **Appendix 12.1** of the ES). Of these, less than one bird (0.1 (0.0-0.2)) was likely to be a breeding adult from Puffin Island SPA.

3242. **Table 8.198** sets out the predicted impacts on puffins from Skelligs SPA during the non-breeding season. This estimates that there would be no measurable increase in mortality from the SPA population, assuming a displacement rate of 30-70% and a mortality of 1-10% for displaced birds.

*Table 8.198 Puffin – predicted operation and maintenance phase displacement and mortality from Skelligs SPA*

| Mean peak abundance estimate type | Mean peak abundance estimate | Number of Skelligs SPA breeding adults present (non-breeding season)[1] | Annual mortality range[2] | Annual baseline mortality increase range[3] |
|---|---|---|---|---|
| Upper 95% CI | 50.8 | 0.2 | 0-0 | 0.00-0.00% |
| Mean | 19.7 | 0.1 | 0-0 | 0.00-0.00% |
| Lower 95% CI | 1.9 | 0.0 | 0-0 | 0.00-0.00% |

[1] Assumes 0.4% of birds present during the non-breeding season are Skelligs SPA breeding adults

[2] Assumes displacement rates of 30-70% and mortality rates of 1-10%

[3] Background mortality rate of 9.4% (Horswill and Robinson, 2015)

3243. **It is concluded that predicted puffin mortality due to operational phase displacement would not adversely affect the integrity of the Skelligs SPA.**

3244. The confidence in the assessment is high for several reasons. Firstly, the evidence used to inform the evidence-based displacement rates is of high applicability and quality (based on the criteria discussed in **Chapter 12 Offshore Ornithology** of the ES). Whilst there was limited available evidence to inform mortality rates, 1% is considered to be sufficiently precautionary based on expert opinion. This species is not regarded as being highly specialised in its habitat requirements (Bradbury *et al*., 2014; Furness and Wade, 2012; Garthe and Hüppop, 2004), and it is therefore anticipated that displaced birds will find alternative habitat in the vast majority of cases. Finally, the conclusion of the assessment is the same irrespective of whether the mean or 95% upper CI mean peak abundances are used to calculate potential mortality and increases in the baseline mortality rate of the background population.

*In-combination*

3245. As the Project would have no measurable effect on puffin populations from the Skelligs SPA, there would be no contribution to any in-combination effects on this feature. **Therefore, it is concluded that predicted puffin mortality due to operational phase displacement due to the Project in-combination with other plans or projects would not adversely affect the integrity of the Skelligs SPA.**

## 8.87 Summary of potential effects

3246. **Table 8.199** below summarises the conclusions of the potential effects arising from the Project. No adverse effects on site integrity have been identified, either Project-alone or in-combination. This accords with the conclusions of the Round 4 offshore wind leasing HRA (NIRAS, 2021), which considered the effects of the Project and proposed Mona and Morgan Offshore Wind Projects, together with other existing OWFs in the Irish Sea.

*Table 8.199 Summary of potential effects*

| Site | Qualifying feature | Potential effects | Potential for adverse effect on site integrity, alone and in-combination |
|---|---|---|---|
| Liverpool Bay / Bae Lerpwl SPA | Red-throated diver | Disturbance/displacement/barrier effects (construction and decommissioning, operation and maintenance) | No adverse effect on site integrity. |
| | Black (common) scoter | Disturbance/displacement/barrier effects (construction and decommissioning, operation and maintenance) | No adverse effect on site integrity. |
| | Little gull | Collision risk (operation and maintenance phase) | No adverse effect on site integrity. |
| | Common tern | Collision risk (operation and maintenance phase) | No adverse effect on site integrity. |
| Morecambe Bay and Duddon Estuary SPA and Ramsar sites | Little egret | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Whooper swan | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Pink-footed goose | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Common shelduck | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Northern pintail | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Eurasian oystercatcher | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Ringed plover | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |

| Site | Qualifying feature | Potential effects | Potential for adverse effect on site integrity, alone and in-combination |
|---|---|---|---|
| | European golden plover | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Grey plover | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Ruff | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Red knot | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Sanderling | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Bar-tailed godwit | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Eurasian curlew | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Common redshank | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Ruddy turnstone | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Mediterranean gull | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Lesser black-backed gull | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Black-tailed godwit | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |

| Site | Qualifying feature | Potential effects | Potential for adverse effect on site integrity, alone and in-combination |
|---|---|---|---|
| | Dunlin | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Lesser black-backed gull | Collision risk (operation and maintenance phase) | No adverse effect on site integrity. |
| | Herring gull | Collision risk (operation and maintenance phase) | No adverse effect on site integrity. |
| | Sandwich tern | Collision risk (operation and maintenance phase) | No adverse effect on site integrity. |
| | Common tern | Collision risk (operation and maintenance phase) | No adverse effect on site integrity. |
| | Seabird assemblage | Collision risk (operation and maintenance phase) | No adverse effect on site integrity. |
| | Waterbird assemblage | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| Ribble and Alt Estuaries SPA and Ramsar | Tundra swan | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Whooper swan | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Pink-footed goose | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Common shelduck | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Eurasian wigeon | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |

| Site | Qualifying feature | Potential effects | Potential for adverse effect on site integrity, alone and in-combination |
|---|---|---|---|
| | Eurasian teal | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Northern pintail | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Eurasian oystercatcher | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Ringed plover | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | European golden plover | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Grey plover | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Red knot | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Sanderling | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Sanderling | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Bar-tailed godwit | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Common redshank | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Common redshank | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |

| Site | Qualifying feature | Potential effects | Potential for adverse effect on site integrity, alone and in-combination |
|---|---|---|---|
| | Black-tailed godwit | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Dunlin | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Ruff | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Lesser black-backed gull | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Common tern | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Seabird assemblage | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Waterbird assemblage | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| Mersey Narrows and North Wirral Foreshore SPA and Ramsar | Bar-tailed godwit | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Little gull | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Black-tailed godwit | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Red knot | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Common tern | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |

| Site | Qualifying feature | Potential effects | Potential for adverse effect on site integrity, alone and in-combination |
|---|---|---|---|
| | Waterbird assemblage | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| Martin Mere SPA and Ramsar | Tundra swan | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Whooper swan | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Pink-footed goose | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Eurasian teal | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Northern pintail | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Eurasian wigeon | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Waterbird assemblage | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| The Dee Estuary SPA and Ramsar | Common shelduck | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Eurasian teal | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Northern pintail | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Eurasian oystercatcher | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Grey plover | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |

| Site | Qualifying feature | Potential effects | Potential for adverse effect on site integrity, alone and in-combination |
|---|---|---|---|
| | Red knot | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Bar-tailed godwit | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Eurasian curlew | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Common redshank | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Common redshank | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Sandwich tern | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Black-tailed godwit | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Dunlin | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Common tern | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Waterbird assemblage | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| Anglesey Terns / Morwenoliaid Ynys Môn SPA | Sandwich tern | Collision risk (operation and maintenance phase) | No adverse effect on site integrity. |
| | Common tern | Collision risk (operation and maintenance phase) | No adverse effect on site integrity. |
| | Arctic tern | Collision risk (operation and maintenance phase) | No adverse effect on site integrity. |

| Site | Qualifying feature | Potential effects | Potential for adverse effect on site integrity, alone and in-combination |
|---|---|---|---|
| Bowland Fells SPA | Hen harrier | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Merlin | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Lesser black-backed gull | Collision risk (operation and maintenance phase) | No adverse effect on site integrity. |
| Mersey Estuary SPA and Ramsar | Great crested grebe | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Common shelduck | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Eurasian wigeon | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Eurasian teal | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Northern pintail | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Ringed plover | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | European golden plover | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Grey plover | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Northern lapwing | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Eurasian curlew | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |

| Site | Qualifying feature | Potential effects | Potential for adverse effect on site integrity, alone and in-combination |
|---|---|---|---|
| | Common redshank | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Common redshank | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Black-tailed godwit | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Dunlin | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Waterbird assemblage | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| Ynys Seiriol / Puffin Island SPA | Great cormorant | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| Leighton Moss Ramsar | Waterbird assemblage | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Wetland bird assemblage | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| Traeth Lafan/ Lavan Sands, Conway Bay SPA | Great crested grebe | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Red-breasted merganser | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Eurasian oystercatcher | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Eurasian curlew | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Common redshank | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |

| Site | Qualifying feature | Potential effects | Potential for adverse effect on site integrity, alone and in-combination |
|---|---|---|---|
| Solway Firth SPA | Red-throated diver | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Great cormorant | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Whooper swan | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Pink-footed goose | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Barnacle goose | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Common shelduck | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Eurasian teal | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Northern pintail | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Northern shoveler | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Greater scaup | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Black (common) scoter | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Common goldeneye | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Goosander | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |

| Site | Qualifying feature | Potential effects | Potential for adverse effect on site integrity, alone and in-combination |
|---|---|---|---|
| | Eurasian oystercatcher | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Ringed plover | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | European golden plover | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Grey plover | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Northern lapwing | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Red knot | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Sanderling | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Bar-tailed godwit | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Eurasian curlew | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Common redshank | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Ruddy turnstone | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Black-headed gull | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Mew gull | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |

| Site | Qualifying feature | Potential effects | Potential for adverse effect on site integrity, alone and in-combination |
|---|---|---|---|
| | Herring gull | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Dunlin | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| Migneint-Arenig-Dduallt SPA | Hen harrier | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Merlin | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Peregrine falcon | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| Berwyn SPA | Red kite | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Hen harrier | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Merlin | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Peregrine falcon | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| South Pennine Moors Phase 2 SPA | Merlin | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | European golden plover | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Short-eared owl | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| North Pennine Moors SPA | Hen harrier | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |

| Site | Qualifying feature | Potential effects | Potential for adverse effect on site integrity, alone and in-combination |
|---|---|---|---|
| | Merlin | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | Peregrine falcon | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| | European golden plover | Potential risk of collision during migratory flights to and from the designated site | No adverse effect on site integrity. |
| Glannau Aberdaron ac Ynys Enlli/ Aberdaron Coast and Bardsey Island SPA | Manx shearwater | Disturbance/displacement/barrier effects (operation and maintenance) | No adverse effect on site integrity. |
| Strangford Lough SPA and Ramsar | Sandwich tern | Collision risk (operation and maintenance phase) | No adverse effect on site integrity. |
| | Common tern | Collision risk (operation and maintenance phase) | No adverse effect on site integrity. |
| Copeland Islands SPA | Manx shearwater | Disturbance/displacement/barrier effects (operation and maintenance) | No adverse effect on site integrity. |
| Larne Lough SPA and Ramsar | Sandwich tern | Collision risk (operation and maintenance phase) | No adverse effect on site integrity. |
| Ailsa Craig SPA | Northern gannet | Disturbance/displacement/barrier effects and collision risk (operation and maintenance) | No adverse effect on site integrity. |
| | Lesser black-backed gull | Collision risk (operation and maintenance phase) | No adverse effect on site integrity. |
| | Black-legged kittiwake | Collision risk (operation and maintenance phase) | No adverse effect on site integrity. |

| Site | Qualifying feature | Potential effects | Potential for adverse effect on site integrity, alone and in-combination |
|---|---|---|---|
| | Herring gull | Collision risk (operation and maintenance phase) | No adverse effect on site integrity. |
| | Common guillemot | Disturbance/displacement/barrier effects (operation and maintenance) | No adverse effect on site integrity. |
| | Seabird assemblage | Collision risk (operation and maintenance phase) | No adverse effect on site integrity. |
| Coquet Island SPA | Common tern | Collision risk (operation and maintenance phase) | No adverse effect on site integrity. |
| | Seabird assemblage | Collision risk (operation and maintenance phase) | No adverse effect on site integrity. |
| Flamborough and Filey Coast SPA | Northern gannet | Disturbance/displacement/barrier effects and collision risk (operation and maintenance) | No adverse effect on site integrity. |
| | Black-legged kittiwake | Collision risk (operation and maintenance phase) | No adverse effect on site integrity. |
| | Seabird assemblage | Collision risk (operation and maintenance phase) | No adverse effect on site integrity. |
| Rathlin Island SPA | Black-legged kittiwake | Collision risk (operation and maintenance phase) | No adverse effect on site integrity. |
| | Common guillemot | Disturbance/displacement/barrier effects (operation and maintenance) | No adverse effect on site integrity. |
| | Razorbill | Disturbance/displacement/barrier effects (operation and maintenance) | No adverse effect on site integrity. |
| | Seabird assemblage | n/a | No adverse effect on site integrity. |

| Site | Qualifying feature | Potential effects | Potential for adverse effect on site integrity, alone and in-combination |
|---|---|---|---|
| Sheep Island SPA | Great cormorant | Disturbance/displacement/barrier effects (operation and maintenance) | No adverse effect on site integrity. |
| Farne Islands SPA | Seabird assemblage | n/a | No adverse effect on site integrity. |
| Forth Islands SPA | Northern gannet | Disturbance/displacement/barrier effects and collision risk (operation and maintenance) | No adverse effect on site integrity. |
| | Atlantic puffin | Disturbance/displacement/barrier effects (operation and maintenance) | No adverse effect on site integrity. |
| | Seabird assemblage | n/a | No adverse effect on site integrity. |
| Skomer, Skokholm and the Seas off Pembrokeshire / Sgomer, Sgogwm a Moroedd Penfro SPA | Manx shearwater | Disturbance/displacement/barrier effects (operation and maintenance) | No adverse effect on site integrity. |
| | European storm-petrel | Low risk of collision and/or disturbance/displacement/barrier effects | No adverse effect on site integrity. |
| | Common guillemot | Disturbance/displacement/barrier effects (operation and maintenance) | No adverse effect on site integrity. |
| | Razorbill | Disturbance/displacement/barrier effects (operation and maintenance) | No adverse effect on site integrity. |
| | Atlantic puffin | Disturbance/displacement/barrier effects (operation and maintenance) | No adverse effect on site integrity. |
| | Lesser black-backed gull | Collision risk (operation and maintenance phase) | No adverse effect on site integrity. |
| | Seabird assemblage | n/a | No adverse effect on site integrity. |

| Site | Qualifying feature | Potential effects | Potential for adverse effect on site integrity, alone and in-combination |
|---|---|---|---|
| Grassholm SPA | Northern gannet | Disturbance/displacement/barrier effects and collision risk (operation and maintenance) | No adverse effect on site integrity. |
| North Colonsay and Western Cliffs SPA | Black-legged kittiwake | Collision risk (operation and maintenance phase) | No adverse effect on site integrity. |
| | Common guillemot | Disturbance/displacement/barrier effects (operation and maintenance) | No adverse effect on site integrity. |
| | Seabird assemblage | n/a | No adverse effect on site integrity. |
| Treshnish Isles SPA | European storm-petrel | Low risk of collision and/or disturbance/displacement/barrier effects | No adverse effect on site integrity. |
| Fowlsheugh SPA | Northern fulmar | Low risk of collision and/or disturbance/displacement/barrier effects | No adverse effect on site integrity. |
| | Black-legged kittiwake | Collision risk (operation and maintenance phase) | No adverse effect on site integrity. |
| | Seabird assemblage | n/a | No adverse effect on site integrity. |
| Rum SPA | Manx shearwater | Disturbance/displacement/barrier effects (operation and maintenance) | No adverse effect on site integrity. |
| | Seabird assemblage | n/a | No adverse effect on site integrity. |
| Canna and Sanday SPA | Common guillemot | Disturbance/displacement/barrier effects (operation and maintenance) | No adverse effect on site integrity. |
| Buchan Ness to Collieston Coast SPA | Northern fulmar | Low risk of collision and/or disturbance/displacement/barrier effects | No adverse effect on site integrity. |

| Site | Qualifying feature | Potential effects | Potential for adverse effect on site integrity, alone and in-combination |
|---|---|---|---|
| | Black-legged kittiwake | Collision risk (operation and maintenance phase) | No adverse effect on site integrity. |
| | Seabird assemblage | n/a | No adverse effect on site integrity. |
| Mingulay and Berneray SPA | Northern fulmar | Low risk of collision and/or disturbance/displacement/barrier effects | No adverse effect on site integrity. |
| | Common guillemot | Disturbance/displacement/barrier effects (operation and maintenance) | No adverse effect on site integrity. |
| | Razorbill | Disturbance/displacement/barrier effects (operation and maintenance) | No adverse effect on site integrity. |
| | Seabird assemblage | n/a | No adverse effect on site integrity. |
| Troup, Pennan and Lion's Heads SPA | Northern fulmar | Low risk of collision and/or disturbance/displacement/barrier effects | No adverse effect on site integrity. |
| | Black-legged kittiwake | Collision risk (operation and maintenance phase) | No adverse effect on site integrity. |
| | Seabird assemblage | n/a | No adverse effect on site integrity. |
| Isles of Scilly SPA | European shag | Disturbance/displacement/barrier effects (operation and maintenance) | No adverse effect on site integrity. |
| | Lesser black-backed gull | Collision risk (operation and maintenance phase) | No adverse effect on site integrity. |
| | Great black-backed gull | Collision risk (operation and maintenance phase) | No adverse effect on site integrity. |

| Site | Qualifying feature | Potential effects | Potential for adverse effect on site integrity, alone and in-combination |
|---|---|---|---|
| | Seabird assemblage | n/a | No adverse effect on site integrity. |
| East Caithness Cliffs SPA | Northern fulmar | Low risk of collision and/or disturbance/displacement/barrier effects | No adverse effect on site integrity. |
| | Black-legged kittiwake | Collision risk (operation and maintenance phase) | No adverse effect on site integrity. |
| | Seabird assemblage | n/a | No adverse effect on site integrity. |
| Shiant Isles SPA | Northern fulmar | Low risk of collision and/or disturbance/displacement/barrier effects | No adverse effect on site integrity. |
| | Common guillemot | Disturbance/displacement/barrier effects (operation and maintenance) | No adverse effect on site integrity. |
| | Razorbill | Disturbance/displacement/barrier effects (operation and maintenance) | No adverse effect on site integrity. |
| | Atlantic puffin | Disturbance/displacement/barrier effects (operation and maintenance) | No adverse effect on site integrity. |
| | Seabird assemblage | n/a | No adverse effect on site integrity. |
| Handa SPA | Northern fulmar | Low risk of collision and/or disturbance/displacement/barrier effects | No adverse effect on site integrity. |
| | Great skua | Collision risk (operation and maintenance phase) | No adverse effect on site integrity. |
| | Black-legged kittiwake | Collision risk (operation and maintenance phase) | No adverse effect on site integrity. |
| | Common guillemot | Disturbance/displacement/barrier effects (operation and maintenance) | No adverse effect on site integrity. |

| Site | Qualifying feature | Potential effects | Potential for adverse effect on site integrity, alone and in-combination |
|---|---|---|---|
| | Razorbill | Disturbance/displacement/barrier effects (operation and maintenance) | No adverse effect on site integrity. |
| | Seabird assemblage | n/a | No adverse effect on site integrity. |
| North Caithness Cliffs SPA | Northern fulmar | Low risk of collision and/or disturbance/displacement/barrier effects | No adverse effect on site integrity. |
| | Black-legged kittiwake | Collision risk (operation and maintenance phase) | No adverse effect on site integrity. |
| | Seabird assemblage | n/a | No adverse effect on site integrity. |
| St Kilda SPA | Northern fulmar | Low risk of collision and/or disturbance/displacement/barrier effects | No adverse effect on site integrity. |
| | Manx shearwater | Disturbance/displacement/barrier effects (operation and maintenance) | No adverse effect on site integrity. |
| | Leach's storm-petrel | Low risk of collision and/or disturbance/displacement/barrier effects | No adverse effect on site integrity. |
| | Great skua | Collision risk (operation and maintenance phase) | No adverse effect on site integrity. |
| | Common guillemot | Disturbance/displacement/barrier effects (operation and maintenance) | No adverse effect on site integrity. |
| | Atlantic puffin | Disturbance/displacement/barrier effects (operation and maintenance) | No adverse effect on site integrity. |
| | Northern gannet | Disturbance/displacement/barrier effects and collision risk (operation and maintenance) | No adverse effect on site integrity. |

| Site | Qualifying feature | Potential effects | Potential for adverse effect on site integrity, alone and in-combination |
|---|---|---|---|
| | Seabird assemblage | n/a | No adverse effect on site integrity. |
| Cape Wrath SPA | Northern fulmar | Low risk of collision and/or disturbance/displacement/barrier effects | No adverse effect on site integrity. |
| | Black-legged kittiwake | Collision risk (operation and maintenance phase) | No adverse effect on site integrity. |
| | Common guillemot | Disturbance/displacement/barrier effects (operation and maintenance) | No adverse effect on site integrity. |
| | Razorbill | Disturbance/displacement/barrier effects (operation and maintenance) | No adverse effect on site integrity. |
| | Seabird assemblage | n/a | No adverse effect on site integrity. |
| Flannan Isles SPA | Northern fulmar | Low risk of collision and/or disturbance/displacement/barrier effects | No adverse effect on site integrity. |
| | Leach's storm-petrel | Low risk of collision and/or disturbance/displacement/barrier effects | No adverse effect on site integrity. |
| | Common guillemot | Disturbance/displacement/barrier effects (operation and maintenance) | No adverse effect on site integrity. |
| | Atlantic puffin | Disturbance/displacement/barrier effects (operation and maintenance) | No adverse effect on site integrity. |
| | Seabird assemblage | n/a | No adverse effect on site integrity. |
| Hoy SPA | Red-throated diver | Disturbance/displacement/barrier effects (operation and maintenance) | No adverse effect on site integrity. |
| | Northern fulmar | Low risk of collision and/or disturbance/displacement/barrier effects | No adverse effect on site integrity. |

| Site | Qualifying feature | Potential effects | Potential for adverse effect on site integrity, alone and in-combination |
|---|---|---|---|
| | Great skua | Collision risk (operation and maintenance phase) | No adverse effect on site integrity. |
| | Seabird assemblage | n/a | No adverse effect on site integrity. |
| Copinsay SPA | Northern fulmar | Low risk of collision and/or disturbance/displacement/barrier effects | No adverse effect on site integrity. |
| | Seabird assemblage | n/a | No adverse effect on site integrity. |
| Sule Skerry and Sule Stack SPA | Leach's storm-petrel | Low risk of collision and/or disturbance/displacement/barrier effects | No adverse effect on site integrity. |
| | Northern gannet | Disturbance/displacement/barrier effects and collision risk (operation and maintenance) | No adverse effect on site integrity. |
| | Common guillemot | Disturbance/displacement/barrier effects (operation and maintenance) | No adverse effect on site integrity. |
| | Atlantic puffin | Disturbance/displacement/barrier effects (operation and maintenance) | No adverse effect on site integrity. |
| | Seabird assemblage | n/a | No adverse effect on site integrity. |
| Rousay SPA | Northern fulmar | Low risk of collision and/or disturbance/displacement/barrier effects | No adverse effect on site integrity. |
| | Seabird assemblage | n/a | No adverse effect on site integrity. |
| North Rona and Sula Sgeir SPA | Northern fulmar | Low risk of collision and/or disturbance/displacement/barrier effects | No adverse effect on site integrity. |
| | Leach's storm-petrel | Low risk of collision and/or disturbance/displacement/barrier effects | No adverse effect on site integrity. |

| Site | Qualifying feature | Potential effects | Potential for adverse effect on site integrity, alone and in-combination |
|---|---|---|---|
| | Northern gannet | Disturbance/displacement/barrier effects and collision risk (operation and maintenance) | No adverse effect on site integrity. |
| | Common guillemot | Disturbance/displacement/barrier effects (operation and maintenance) | No adverse effect on site integrity. |
| | Seabird assemblage | n/a | No adverse effect on site integrity. |
| Calf of Eday SPA | Northern fulmar | Low risk of collision and/or disturbance/displacement/barrier effects | No adverse effect on site integrity. |
| | Seabird assemblage | n/a | No adverse effect on site integrity. |
| West Westray SPA | Northern fulmar | Low risk of collision and/or disturbance/displacement/barrier effects | No adverse effect on site integrity. |
| | Black-legged kittiwake | Collision risk (operation and maintenance phase) | No adverse effect on site integrity. |
| | Seabird assemblage | n/a | No adverse effect on site integrity. |
| Fair Isle SPA | Northern fulmar | Low risk of collision and/or disturbance/displacement/barrier effects | No adverse effect on site integrity. |
| | Great skua | Collision risk (operation and maintenance phase) | No adverse effect on site integrity. |
| | Seabird assemblage | n/a | No adverse effect on site integrity. |
| Sumburgh Head SPA | Northern fulmar | Low risk of collision and/or disturbance/displacement/barrier effects | No adverse effect on site integrity. |
| | Seabird assemblage | n/a | No adverse effect on site integrity. |

| Site | Qualifying feature | Potential effects | Potential for adverse effect on site integrity, alone and in-combination |
|---|---|---|---|
| Foula SPA | Northern fulmar | Low risk of collision and/or disturbance/displacement/barrier effects | No adverse effect on site integrity. |
| | Great skua | Collision risk (operation and maintenance phase) | No adverse effect on site integrity. |
| | Red-throated diver | Disturbance/displacement/barrier effects (operation and maintenance) | No adverse effect on site integrity. |
| | Atlantic puffin | Disturbance/displacement/barrier effects (operation and maintenance) | No adverse effect on site integrity. |
| | Seabird assemblage | n/a | No adverse effect on site integrity. |
| Noss SPA | Northern fulmar | Low risk of collision and/or disturbance/displacement/barrier effects | No adverse effect on site integrity. |
| | Great skua | Collision risk (operation and maintenance phase) | No adverse effect on site integrity. |
| | Northern gannet | Disturbance/displacement/barrier effects and collision risk (operation and maintenance) | No adverse effect on site integrity. |
| | Seabird assemblage | n/a | No adverse effect on site integrity. |
| Ronas Hill - North Roe and Tingon SPA and Ramsar | Red-throated diver | Disturbance/displacement/barrier effects (operation and maintenance) | No adverse effect on site integrity. |
| | Great skua | Collision risk (operation and maintenance phase) | No adverse effect on site integrity. |
| Fetlar SPA | Northern fulmar | Low risk of collision and/or disturbance/displacement/barrier effects | No adverse effect on site integrity. |
| | Great skua | Collision risk (operation and maintenance phase) | No adverse effect on site integrity. |

| Site | Qualifying feature | Potential effects | Potential for adverse effect on site integrity, alone and in-combination |
|---|---|---|---|
| | Seabird assemblage | n/a | No adverse effect on site integrity. |
| Hermaness, Saxa Vord and Valla Field SPA | Northern fulmar | Low risk of collision and/or disturbance/displacement/barrier effects | No adverse effect on site integrity. |
| | Great skua | Collision risk (operation and maintenance phase) | No adverse effect on site integrity. |
| | Northern gannet | Disturbance/displacement/barrier effects and collision risk (operation and maintenance) | No adverse effect on site integrity. |
| | Red-throated diver | Disturbance/displacement/barrier effects (operation and maintenance) | No adverse effect on site integrity. |
| | Atlantic puffin | Disturbance/displacement/barrier effects (operation and maintenance) | No adverse effect on site integrity. |
| | Seabird assemblage | n/a | No adverse effect on site integrity. |
| Lambay Island SPA (transboundary site) | Guillemot | Disturbance/displacement/barrier effects (operation and maintenance) | No adverse effect on site integrity. |
| | Puffin | Disturbance/displacement/barrier effects (operation and maintenance) | No adverse effect on site integrity. |
| | Fulmar | Low risk of collision and/or disturbance/displacement/barrier effects | No adverse effect on site integrity. |
| | Lesser black-backed gull | Collision risk (operation and maintenance phase) | No adverse effect on site integrity. |
| | Kittiwake | Collision risk (operation and maintenance phase) | No adverse effect on site integrity. |
| | Razorbill | Disturbance/displacement/barrier effects (operation and maintenance) | No adverse effect on site integrity. |

| Site | Qualifying feature | Potential effects | Potential for adverse effect on site integrity, alone and in-combination |
|---|---|---|---|
| | Herring gull | Collision risk (operation and maintenance phase) | No adverse effect on site integrity. |
| | Shag | Disturbance/displacement/barrier effects (operation and maintenance) | No adverse effect on site integrity. |
| | Cormorant | Disturbance/displacement/barrier effects (operation and maintenance) | No adverse effect on site integrity. |
| Howth Head Coast SPA (transboundary site) | Kittiwake | Collision risk (operation and maintenance phase) | No adverse effect on site integrity. |
| Ireland's Eye SPA (transboundary site) | Kittiwake | Collision risk (operation and maintenance phase) | No adverse effect on site integrity. |
| | Razorbill | Disturbance/displacement/barrier effects (operation and maintenance) | No adverse effect on site integrity. |
| | Cormorant | Disturbance/displacement/barrier effects (operation and maintenance) | No adverse effect on site integrity. |
| Wicklow Head SPA (transboundary site) | Kittiwake | Collision risk (operation and maintenance phase) | No adverse effect on site integrity. |
| Saltee Islands SPA (transboundary site) | Puffin | Disturbance/displacement/barrier effects (operation and maintenance) | No adverse effect on site integrity. |
| | Fulmar | Low risk of collision and/or disturbance/displacement/barrier effects | No adverse effect on site integrity. |
| | Gannet | Disturbance/displacement/barrier effects and collision risk (operation and maintenance) | No adverse effect on site integrity. |
| | Kittiwake | Collision risk (operation and maintenance phase) | No adverse effect on site integrity. |

| Site | Qualifying feature | Potential effects | Potential for adverse effect on site integrity, alone and in-combination |
|---|---|---|---|
| | Guillemot | Disturbance/displacement/barrier effects (operation and maintenance) | No adverse effect on site integrity. |
| | Shag | Disturbance/displacement/barrier effects (operation and maintenance) | No adverse effect on site integrity. |
| | Cormorant | Disturbance/displacement/barrier effects (operation and maintenance) | No adverse effect on site integrity. |
| | Razorbill | Disturbance/displacement/barrier effects (operation and maintenance) | No adverse effect on site integrity. |
| Horn Head to Fanad Head SPA (transboundary site) | Fulmar | Low risk of collision and/or disturbance/displacement/barrier effects | No adverse effect on site integrity. |
| | Kittiwake | Collision risk (operation and maintenance phase) | No adverse effect on site integrity. |
| | Shag | Disturbance/displacement/barrier effects (operation and maintenance) | No adverse effect on site integrity. |
| | Cormorant | Disturbance/displacement/barrier effects (operation and maintenance) | No adverse effect on site integrity. |
| West Donegal Coast SPA (transboundary site) | Fulmar | Low risk of collision and/or disturbance/displacement/barrier effects | No adverse effect on site integrity. |
| | Shag | Disturbance/displacement/barrier effects (operation and maintenance) | No adverse effect on site integrity. |
| | Cormorant | Disturbance/displacement/barrier effects (operation and maintenance) | No adverse effect on site integrity. |
| Tory Island SPA (transboundary site) | Fulmar | Low risk of collision and/or disturbance/displacement/barrier effects | No adverse effect on site integrity. |
| Cliffs of Moher SPA (transboundary site) | Fulmar | Low risk of collision and/or disturbance/displacement/barrier effects | No adverse effect on site integrity. |

| Site | Qualifying feature | Potential effects | Potential for adverse effect on site integrity, alone and in-combination |
|---|---|---|---|
| | Guillemot | Disturbance/displacement/barrier effects (operation and maintenance) | No adverse effect on site integrity. |
| | Kittiwake | Collision risk (operation and maintenance phase) | No adverse effect on site integrity. |
| | Razorbill | Disturbance/displacement/barrier effects (operation and maintenance) | No adverse effect on site integrity. |
| Stags of Broad Haven SPA (transboundary site) | Leach's petrel | Low risk of collision and/or disturbance/displacement/barrier effects | No adverse effect on site integrity. |
| Clare Island SPA (transboundary site) | Fulmar | Low risk of collision and/or disturbance/displacement/barrier effects | No adverse effect on site integrity. |
| Duvillaun Islands SPA (transboundary site) | Fulmar | Low risk of collision and/or disturbance/displacement/barrier effects | No adverse effect on site integrity. |
| High Island, Inishshark and Davillaun SPA (transboundary site) | Fulmar | Low risk of collision and/or disturbance/displacement/barrier effects | No adverse effect on site integrity. |
| Kerry Head SPA (transboundary site) | Fulmar | Low risk of collision and/or disturbance/displacement/barrier effects | No adverse effect on site integrity. |
| Cruagh Island SPA (transboundary site) | Manx shearwater | Disturbance/displacement/barrier effects (operation and maintenance) | No adverse effect on site integrity. |
| Dingle Peninsula SPA (transboundary site) | Fulmar | Low risk of collision and/or disturbance/displacement/barrier effects | No adverse effect on site integrity. |
| Iveragh Peninsula SPA (transboundary site) | Fulmar | Low risk of collision and/or disturbance/displacement/barrier effects | No adverse effect on site integrity. |

| Site | Qualifying feature | Potential effects | Potential for adverse effect on site integrity, alone and in-combination |
|---|---|---|---|
| Blasket Islands SPA (transboundary site) | Fulmar | Low risk of collision and/or disturbance/displacement/barrier effects | No adverse effect on site integrity. |
| | Manx shearwater | Disturbance/displacement/barrier effects (operation and maintenance) | No adverse effect on site integrity. |
| | Puffin | Disturbance/displacement/barrier effects (operation and maintenance) | No adverse effect on site integrity. |
| | Lesser black-backed gull | Collision risk (operation and maintenance phase) | No adverse effect on site integrity. |
| Deenish Island and Scariff Island SPA (transboundary site) | Fulmar | Low risk of collision and/or disturbance/displacement/barrier effects | No adverse effect on site integrity. |
| | Manx shearwater | Disturbance/displacement/barrier effects (operation and maintenance) | No adverse effect on site integrity. |
| Puffin Island SPA (transboundary site) | Fulmar | Low risk of collision and/or disturbance/displacement/barrier effects | No adverse effect on site integrity. |
| | Manx shearwater | Disturbance/displacement/barrier effects (operation and maintenance) | No adverse effect on site integrity. |
| | Puffin | Disturbance/displacement/barrier effects (operation and maintenance) | No adverse effect on site integrity. |
| The Bull and The Cow Rocks SPA (transboundary site) | Gannet | Disturbance/displacement/barrier effects and collision risk (operation and maintenance) | No adverse effect on site integrity. |
| Skelligs SPA (transboundary site) | Gannet | Disturbance/displacement/barrier effects and collision risk (operation and maintenance) | No adverse effect on site integrity. |
| | Manx shearwater | Disturbance/displacement/barrier effects (operation and maintenance) | No adverse effect on site integrity. |

| Site | Qualifying feature | Potential effects | Potential for adverse effect on site integrity, alone and in-combination |
|---|---|---|---|
| | Fulmar | Low risk of collision and/or disturbance/displacement/barrier effects | No adverse effect on site integrity. |
| | Puffin | Disturbance/displacement/barrier effects (operation and maintenance) | No adverse effect on site integrity. |

# 9 Offshore Annex II sites designated for marine mammals

## 9.1 Approach to assessment

3247. This section provides information in order to determine the potential for the Project to have AEoI on designated sites for marine mammals. HRA Screening (Morecambe Offshore Windfarm Ltd, 2023a; Document Reference 4.10) has been conducted and, as set out in **Section 5.4**, the following sites have been screened in with potential for LSE (**Figure 9.1** and **Figure 9.2**):

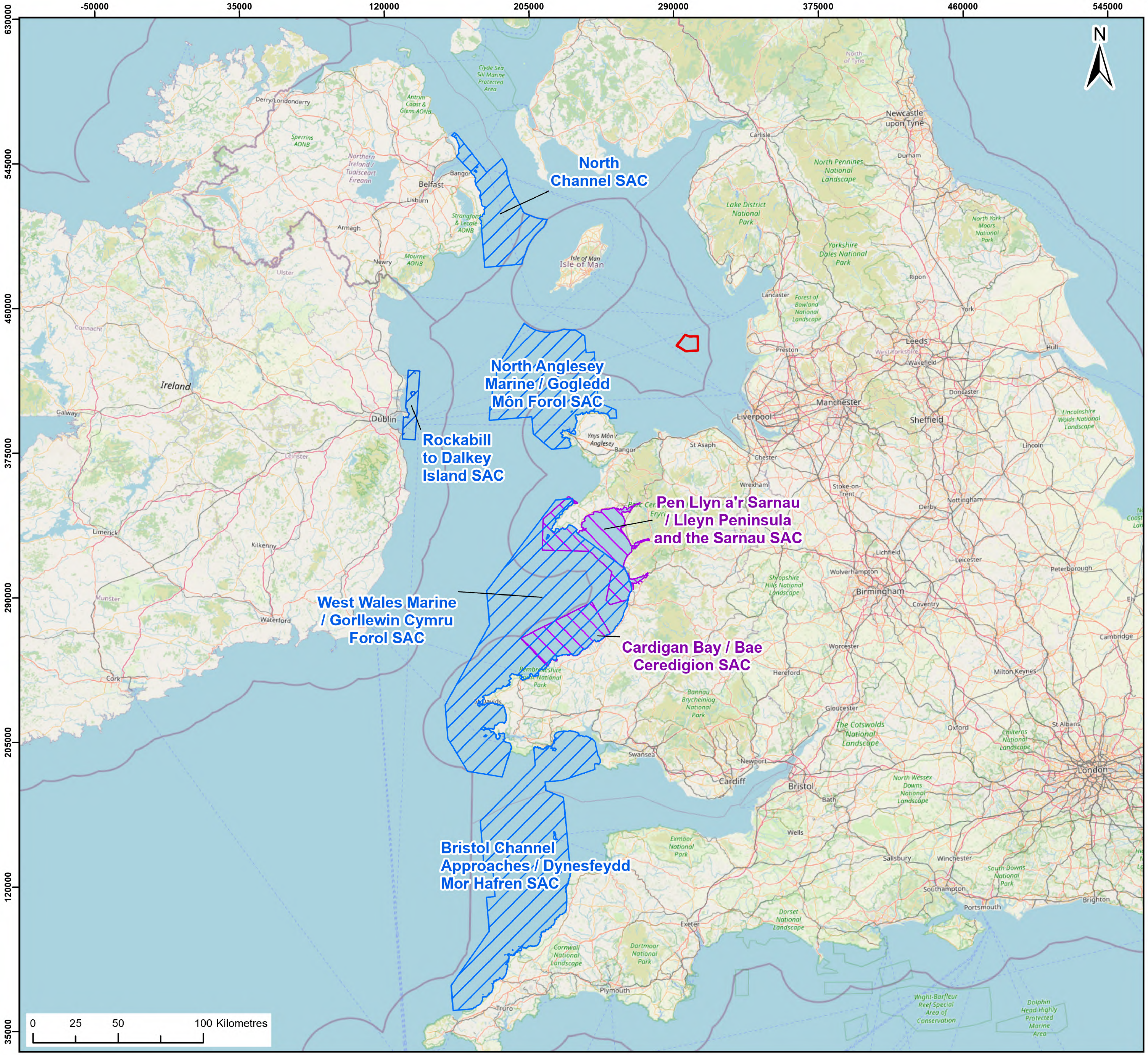
- Sites where harbour porpoise are a qualifying feature:
  - North Anglesey Marine SAC
  - North Channel SAC
  - West Wales Marine SAC
  - Rockabill to Dalkey Island SAC
  - Bristol Channel Approaches SAC
- Sites where bottlenose dolphin are a qualifying feature:
  - Pen Llŷn a`r Sarnau SAC
  - Cardigan Bay SAC
- Sites where grey seal are a qualifying feature:
  - Cardigan Bay SAC
  - Pen Llŷn a`r Sarnau SAC
  - Pembrokeshire Marine SAC
- Sites where harbour seal are a qualifying feature:
  - Strangford Lough SAC

3248. For each species the following has been provided:

- A summary of each designated site, population, and conservation status
- An assessment of the potential effects during the construction, operation and maintenance, and decommissioning, and assessment on whether the Project-alone could adversely affect the integrity of screened in European sites in view of their conservation objectives
- An assessment of the potential for in-combination effects alongside the Transmission Assets, and assessment on whether the Project-alone or

in-combination could adversely affect the integrity of screened in European sites in view of their conservation objectives

- An assessment of the potential for in-combination effects alongside other relevant developments and projects, including the Transmission Assets, and assessment on whether the Project-alone or in-combination could adversely affect the integrity of screened in European sites in view of their conservation objectives





Legend:

- Morecambe Offshore Windfarm Site
- Harbour Porpoise Special Area of Conservation (SAC)
- Bottlenose Dolphin Special Area of Conservation (SAC)



Report:
Morecambe Offshore Windfarm: Generation Assets
Habitat Regulations Report to Inform Appropriate Assessment

Title:
SACs screened in for Annex II sites designated for cetaceans

Figure: 9.1 | Drawing No: PC1165-RHD-ES-OF-DR-Z-0068

| Revision: | Date: | Drawn: | Checked: | Size: | Scale: |
|---|---|---|---|---|---|
| P01 | 22/04/2024 | JH | SB | A3 | 1:2,250,000 |

Co-ordinate system: WGS 1984 UTM Zone 30N

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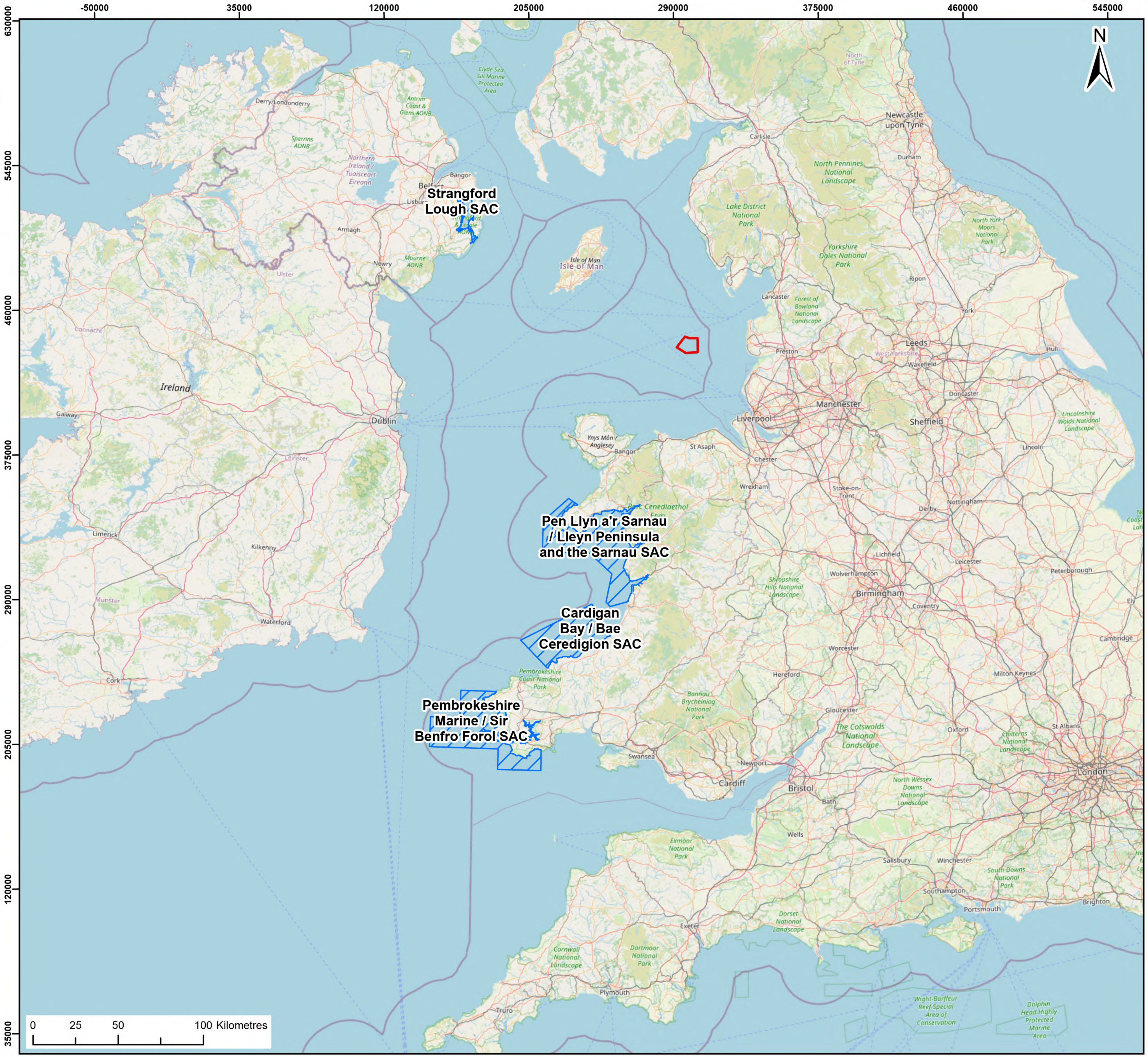



Legend:

- Morecambe Offshore Windfarm Site
- Special Area of Conservation (SAC)



Report: Morecambe Offshore Windfarm: Generation Assets Habitat Regulations Report to Inform Appropriate Assessment

Title: SACs screened in for Annex II sites designated for grey and harbour seals

Figure: 9.2 | Drawing No: PC1165-RHD-ES-OF-DR-Z-0111

| Revision: | Date: | Drawn: | Checked: | Size: | Scale: |
|---|---|---|---|---|---|
| P01 | 19/04/2024 | JH | SB | A3 | 1:2,250,000 |

Co-ordinate system: WGS 1984 UTM Zone 30N

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3250. As the Project windfarm site would be outwith any direct overlap with any SAC (the nearest SAC was North Anglesey Marine SAC at a distance of 49km) there would be no direct effects on any SAC. Within the plan level HRA (NIRAS, 2021), no adverse effects on integrity were found based on the distance of projects from SACs (considering a 26km Effective Deterrence Radius (EDR) for all marine mammal species).

3251. On a precautionary basis, an assessment was undertaken for each species for all relevant sites which may have connectivity to the Project windfarm site in relation to the relevant reference population (given that any potential impacts would occur outside any SAC). The assessment has then determined how many individuals may be affected by each impact as a percentage of the reference population. In the absence of any guidance on thresholds for significance of effect, the following have been proposed as thresholds above which further assessment on the consequence of the effect would be undertaken:

- For permanent effects, further assessment may be required if there was an effect to 1% or more of the population (based on Agreement on the Conservation of Small Cetaceans of the Baltic, North East Atlantic, Irish and North Seas (ASCOBANS) and Defra advice (Defra, 2003; ASCOBANS, 2015))
- For temporary effects, further assessment may be required if there was an effect to 5% or more of the population (based on JNCC *et al.,* (2010) draft guidance)

3252. Further assessment included population modelling using interim Population Consequences of Disturbance (iPCoD)) which has been undertaken for piling at the cetacean Management Unit (MU) and SAC level where relevant.

## 9.2 Consultation

3253. Consultation with regard to marine mammals has been undertaken in line with the general process described in **Section 4.2**. The feedback received through the EPP has been considered in preparing this RIAA. **Table 9.1** provides a summary of how the consultation responses received in relation to the HRA Screening Report and draft RIAA have influenced the approach that has been taken.

*Table 9.1 Consultation comments*

| Consultee | Date/ Document | Comment | Project response/where addressed |
|---|---|---|---|
| Natural England | 14th September 2022<br>Advice HRA Screening Report Morecambe Offshore Windfarm Generation Assets (Morecambe Offshore Windfarm Ltd, 2023a) | We welcome continued engagement on the underwater noise modelling undertaken for the project. The applicant lists underwater noise associated only with construction, however we anticipate assessment of the noise during other phases too. | For each species, Project-alone assessments have been made for the construction, operation, and decommissioning phase. |
| | | The title of this impact pathway does not reflect that disturbance to seals at-sea will be assessed. | Disturbance to seals has been assessed in **Section 9.6** (grey seal) and **Section 1.1** (harbour seal), for which the SAC-specific Carter *et al.,* (2022) seals-at-sea data has been used. |
| | | The value added by the results of AyM OWF is questionable given that most animals observed were unidentified species. Explore the relevance of data collected by other OWFs, including Round 4 projects. | Data from other OWFs that were screened in were based on the best available information (PEIRs and ESs). |
| | | The group size (and confidence value) presented here for bottlenose dolphin in Block E is incorrect. | For the assessment, the data was updated to the more recent Small Cetaceans in the European Atlantic and North Sea (SCANS)-IV (Gilles *et al*., 2023), in the Project relevant block CS-E. |
| | | For reference, we advise that use of Carter *et al.,* (2022) that is the peer-reviewed version (with minor updates) of Carter *et al.,* (2020). To illustrate, Carter *et al.,* (2022) should be used to determine density estimates in the project area. Carter *et al.,* (2022) also provides additional useful information such as an example of apportioning impacts to sites, and SAC-specific seal usage patterns. Consideration should be given as to how this information could be used in the assessment. | SAC-specific Carter *et al.,* (2022) seals-at-sea data has been applied for grey seal (**Section 9.6**), and harbour seal (**Section 1.1**). |

| Consultee | Date/ Document | Comment | Project response/where addressed |
|---|---|---|---|
| | | From this figure it is not clear whether there is connectivity between the project area and MU 11 (SW England) for grey seals. | |
| | | For reference, the latest SCOS (Special Committee on Seals) Report is now SCOS 2021. | The latest SCOS report was published in Q4 2023, which has been applied. |
| | | Carter *et al.,* (2022) report that the maximum foraging trip of harbour seal was 273 km, which is substantially greater than the quoted “typical and average” foraging range. | The Strangford Lough SAC for harbour seal is 135km from the windfarm site, which was just above the “average” yet below the maximum foraging range reported by Carter *et al.,* (2022). Only one harbour seal was observed at the Project and as there were no harbour seal haul-outs in the vicinity of the Project, there is the possibility that it could have been travelling from this Irish SAC. |
| | | We acknowledge the Applicant’s proposal to potentially screen out harbour seal SACs at a later date, in agreement with the ETG. However, we advise that if SACs in MUs are screened out based on a lack of connectivity, the inclusion of those MUs in the reference population should be re-assessed. It is important that connectivity between the project and seal MUs is treated consistently.<br>This paragraph references Section 7.3.4 for further details however this paragraph is already in Section 7.3.4. It is therefore unclear what further details are being referred to.<br>We require that the Applicant provide further evidence to support their consideration that other sites are “too far” and can be screened out; it is insufficient to simply state that the sites are “too far”. For example, we note that the Bristol Channel Approaches SAC has been | Approach and text have been amended.<br>In the HRA Screening Report (Document Reference 4.10), SACs have been screened in on the basis that they were within the boundaries of the species-specific cumulative screening area:<br>▪ Harbour porpoise: Celtic and Irish Sea (CIS) MU<br>▪ Bottlenose dolphin: IS MU<br>▪ Grey seal: NW England + Isle of Man + SW Scotland + Wales MU + NI MU + E RoI + SE RoI<br>▪ Harbour seal: NW England MU + NI MU<br><br>For the screened-in sites, refer to **Section 9.1.** |

| Consultee | Date/ Document | Comment | Project response/where addressed |
|---|---|---|---|
| | | screened out in this way, despite being in the Celtic and IS MU (same as the project) and being close to the West Wales Marine SAC (which is screened in). • Please note that, should it be determined that LSE cannot be excluded for English sites, we would require a full assessment of AEoI on that site. The approach advised in Welsh waters, to assess the nearest site only, is not applicable to sites in English waters. | |
| | | Carter *et al.,* (2022) report that the maximum foraging trip of grey seal was 448 km, which is substantially greater than the quoted "typical and average" foraging range. | The screened-in SACs were within distance of the maximum grey seal foraging range and within areas where connectivity has been evident. |
| | | The North Anglesey Marine SAC has been omitted from this figure (though we note that it is referenced in the text). | The SAC has been included in **Figure 9.1.** |
| Natural England | 2nd June 2023<br>Section 42 comments on the PEIR Appendix 11.1 Underwater noise modelling report/draft RIAA | Paragraph 3.2.2 (also draft RIAA, Table 9.4): Natural England understands that sequential or concurrent piling of monopiles is not being considered. Also, that concurrent pin piles are not being considered. The only option for multiple piling events in one day is sequential piling of up to 4 pin piles. This will need to be secured as a licence condition. The piling WCS should be secured as a licence condition in the submitted Deemed Marine Licence (DML). | Due to updates to the PDE there is the potential for up to three mono-piles and four pin piles to be installed sequentially in 24 hours.<br>Underwater noise modelling and impact assessments have been updated accordingly, **Section 9.3**.<br>The final piling parameters would be confirmed post-consent and secured through consultation on the final Marine Mammal Mitigation Protocol (MMMP) process. |
| Natural England | 2nd June 2023<br>Section 42 comments on the draft RIAA | Appendix B (also draft RIAA Table 9.4): Natural England notes that Appendix B to Appendix 11.1 refers to hammer energies of 6,600kJ (for monopiles) which have been used for sensitivity testing. This hammer energy is notably higher than 5,000kJ used in the | Assessment has been updated for confirmed worst-case hammer energy (6,600kJ) as outlined in **Section 9.3.2**. |

| Consultee | Date/ Document | Comment | Project response/where addressed |
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| | | assessment. We therefore seek clarity on what the WCS is. It is imperative that the WCS is assessed given NE will advise that the WCS is conditioned through the deemed Marine Licence.<br>Clarify the worst-case scenario hammer energy. The piling WCS should be secured as a licence condition in the submitted dML. | |
| Natural England | 2nd June 2023<br>Section 42 comments on the draft RIAA | Table 9.4: The Applicant states that jetting will produce the highest noise of the cable laying activities (more so than rock placement and cable laying). However, jetting does not appear to have been included in the underwater noise modelling.<br>Present the underwater noise levels and impact zones associated with jetting | As per the Project description in **Chapter 5 Project Description** of the ES, "cable burial can be achieved using […] trenching (including jetting and mechanical cutting)", thus has not been modelled separately but has been covered under 'trenching'. |
| Natural England | | Paragraph 1.588: Natural England considers that all relevant SACs with marine mammal features in English waters have been screened in. | Noted. |
| Natural England | | Paragraph 1.638: The relevant SNCB for the Republic of Ireland has not signed up to the JNCC *et al.*, 2019 guidance on harbour porpoise SACs. Therefore, the approach to determine the site population for Rockabill to Dalkey Island SAC should be checked with the relevant SNCB.<br>Check the approach to determine the site population with the relevant SNCB. | Noted. The site-specific conservation objectives have been taken into account for the assessment in **Section 9.3**. Consultation with NPWS has also been sought. |
| Natural England | | Section 9.4.1.5: Please note that it is Natural England's remit to provide advice on the assessment in so much as it relates to SACs in English waters. We defer to the relevant SNCBs on the appropriate approach for assessing SACs outside English waters. For clarity, we | Noted. |

| Consultee | Date/ Document | Comment | Project response/where addressed |
|---|---|---|---|
| | | have only reviewed the assessment of SACs for harbour porpoise. | |
| Natural England | | Paragraph 1.647: Natural England considers that the winter density of harbour porpoise would be more appropriate to use when assessing impacts to the Bristol Channel Approaches SAC. This specific SAC is only in effect during winter, therefore there is only an impact pathway with the site during the winter months.<br>The submitted ES should use a winter-specific density when assessing impacts to the Bristol Channel Approaches SAC. | The highest density for harbour porpoise (summer average) has been applied to the assessments as the worst-case for potential effects and evaluated at the management unit level for the CIS.<br>Therefore, any potential effects during the winter season would be expected to be less than assessed. |
| Natural England | | Paragraph 1.658: The conclusion of no significant effect references the mitigation to be detailed in the piling MMMP. A draft piling MMMP will be submitted with the DCO Application. Natural England cannot provide a view on the assessment conclusion for the pathway of “physical and permanent auditory injury” until the draft MMMP has been provided.<br>Provide the draft piling MMMP with the DCO Application (already proposed by the Applicant). | The draft MMMP (Document Reference 6.5) has been provided as part of the DCO Application. |
| Natural England | | Paragraph 1.675 and Table 9.9; also Paragraphs 1.686-1.687: Paragraph 1.675 states that both 2km and 4km has been used for disturbance from construction vessels. Based on the text here, it appears 4km would be an appropriate WCS for disturbance from construction vessels. The areas of disturbance in the assessment should be reviewed to ensure they reflect 4km rather than 2km. This is also applicable to the similar assessment of disturbance from vessels during operation. Note that our assessment is based on the | Benhemma-Le Gall *et al.,* (2021) indicated that at 4km distance to a vessel, harbour porpoise presence was nearly constant at a probability of 40% at all vessel intensity levels, indicating that the vessel did not affect the animals. However, at 2km distance from the vessel, the probability of occurrence decreased (with vessel intensity) by ~34%, inferring that the animals were responding to the vessel disturbance and avoided the area. |

| Consultee | Date/ Document | Comment | Project response/where addressed |
|---|---|---|---|
| | | number of vessels that could be on site at any one time, as this is the WCS.<br>Use 4km for harbour porpoise disturbance from construction and operation vessels, and revise the final assessment accordingly. | Therefore, as a precautionary approach, 4km has been used in assessing disturbance from vessels. |
| Natural England | 2nd June 2023<br>Section 42 comments on the draft RIAA | Paragraph 1.708: The conclusion of no significant effect references the mitigation to be detailed in the PEMP. A draft piling MMMP will be submitted with the DCO Application. Natural England cannot provide a view on the assessment conclusion for the pathway of "vessel interactions" until the PEMP has been provided.<br>Provide the PEMP with the DCO Application | An outline PEMP (Document Reference 6.2) has been provided with the DCO Application. |
| Natural England | | Paragraph 1.740 and 1.741: We consider that the terminology in the in-combination assessments section should be clarified, to make it clearer what is being concluded. For example, the Applicant concludes "that there would be no significant in-combination effect on the harbour porpoise CIS MU population during construction" from Permanent Threshold Shift (PTS), and that "the potential risk of PTS is not considered further". This conclusion has not been presented in standard HRA terms - it does not reference LSE or AEoI, nor does it present the conclusion relative to the SAC – which means it is difficult to agree with the conclusions. A new table, or an expansion on Table 9.44, that presents the conclusions for each pathway could help for clarity. Note that the RIAA, once revised for clarity, should be checked against the CEA to ensure that the approach is consistent, on what pathways do have potential for a cumulative/in-combination effect for example.<br>Clarify the wording in the submitted RIAA | Wording has been clarified within in-combination sections of assessments. |

| Consultee | Date/ Document | Comment | Project response/where addressed |
|---|---|---|---|
| Natural England | | Paragraph 1.743 (as an example): Please review our earlier advice regarding the ES Chapter 11 to determine those relevant to the RIAA. Any changes made in light of our advice on the Cumulative Effects assessment should be tracked through to the in-combination assessment in the RIAA, where relevant.<br>Ensure relevant changes made to the submitted ES are also made in the RIAA. | Changes made to the ES have been reflected in the RIAA. |
| Natural England | | Paragraph 1.766, 1.796: The Applicant has identified that up to 13% of the CIS MU population of harbour porpoise may be disturbed at any one time from all projects in-combination. Whilst we acknowledge no spatial overlap between the Project and the Bristol Channel Approaches SAC, our concern is whether this level of in combination disturbance could impact the ability of harbour porpoise to remain a viable component of the site (Conservation objective 1). We welcome further engagement on potential further assessment/ mitigation to demonstrate/ensure that no adverse effect on site integrity could occur.<br>Continue engagement on potential further assessment/mitigation of in-combination disturbance effects to demonstrate no AEoI to harbour porpoise SACs | Population modelling has been undertaken to assess a population level effect and if there would be any AEoI. |
| Natural England | | The Applicant has identified that up to 13% of the CIS MU population of harbour porpoise may be disturbed at any one time from all projects in-combination. Whilst we acknowledge no spatial overlap between the Project and the Bristol Channel Approaches SAC, our concern is whether this level of in-combination disturbance could impact the ability of harbour porpoise to remain a viable component of the site (Conservation Objective 1). We | The nearest designated site to the Project for harbour porpoise is the North Anglesey Marine/Gogledd Môn Forol SAC (**Section 9.4.1.1**). The harbour porpoise population has been assessed based on the MU and was considered in relation to the conservation objectives for all the relevant SACs. |

| Consultee | Date/ Document | Comment | Project response/where addressed |
|---|---|---|---|
| | | welcome further engagement on potential further assessment/mitigation to demonstrate/ensure that no adverse effect on site integrity could occur.<br>Continue engagement on potential further assessment/mitigation of in-combination disturbance effects to demonstrate no AEoI to harbour porpoise SACs. | However, since the worst-case activities (such as underwater noise from piling) are expected to be scheduled for the summer season, while the Bristol Channel Approaches SAC is designated for the winter season (when the harbour porpoise presence was higher) (see **Section 9.4.1.5**), it is anticipated that the Project would have a lesser impact on the associated population.<br>Population modelling has been undertaken to determine whether there was a risk at a population level through Project-alone (see **Sections 9.4.2, 9.5.2, 9.6.2** and **9.7.2**) and in-combination effects (see **Sections 9.4.3**, **9.5.30, 9.6.30** and **9.7.30**) and if there could be any potential for AEoI. |
| Natural Resources Wales (NRW) | 21st May 2023<br>Section 42 comments on the draft RIAA | In Table 5.1 Summary of European sites and features screened in, NRW (A) advise that Cardigan Bay SAC is designated for both Bottlenose dolphin and Grey seal.<br>Furthermore, Pembrokeshire Marine SAC designated for Grey seal has not been screened in for assessment in this table. NRW (A) recommend that Pembrokeshire Marine SAC is included in line with NRW's position statement on the use of marine mammal management units (MMMUs) in HRA (NRW, 2022). | Both Cardigan Bay and Pembrokeshire Marine SACs are designated for grey seal, which have been screened in and assessed in **Section 9.6.1.2** and **9.6.1.3**, respectively. |
| NRW | | Regarding the reference to population extent for Grey seal in Section 9.7 Grey Seal, Paragraph 1.1105, reference should be made to the OSPAR Region III interim MU and the relevant NRW position statement (NRW, 2022). | The Applicant acknowledges the provided evidence supporting the knowledge of wide ranges exhibited by grey seals. For the ES the assessment therefore included the relevant MUs (including Republic of Ireland) that were understood to be the most representative of this behaviour and supported by tagging data. |

| Consultee | Date/ Document | Comment | Project response/where addressed |
|---|---|---|---|
| | | | The ES assessment did not use the OSPAR region III as the baseline population in the Cumulative Effect Assessment (CEA), only projects within the associated MUs have been screened in and assessed. This approach has been carried forward to the RIAA. |
| NRW | | In Table 9.4 Realistic worst-case scenarios for marine mammal assessments, it is stated in the row 'Underwater noise from other construction activities' that jetting is the worst-case cable installation method. However, this noise source has not been included in the underwater noise modelling. | As per **Chapter 5 Project Description** of the ES, "cable burial can be achieved using […] trenching (including jetting and mechanical cutting)". Jetting has not been modelled separately but it would be covered under 'trenching' in the underwater noise modelling. |
| NRW | | In Section 9.4.2.1 Underwater noise and disturbance from other sources, Paragraph 1.675, as stated in Paragraph 22 of the current document, NRW (A) advise a more precautionary 4km vessel disturbance range assessment is conducted around the vessel rather than the stated 2km, as per Benhemma-le Gall *et al.,* (2021). | Benhemma-Le Gall *et al.,* (2021) indicated that at 4km distance to a vessel, harbour porpoise presence was nearly constant at a probability of 40% at all vessel intensity levels, indicating that the vessel did not affect the animals. However, at 2km distance from the vessel, the probability of occurrence decreased (with vessel intensity) by ~34%, inferring that the animals were responding to the vessel disturbance and avoided the area.<br>Therefore, as a precautionary approach, 4km has been used in assessing disturbance from vessels. |
| NRW | | In Section 9.4.2 Project-alone Assessment, Paragraph 1.658, please refer to our comments in Paragraph 11 of the current document regarding the use of noise mitigation strategies/attenuation technology such as bubble curtains, timing of piling (given North Anglesey Marine is a summer site) and piling methods as potential mitigation methods. | Embedded mitigation measures have been described in **Section 9.3.1** which includes piling schedules and soft-start and ramp up procedures. Mitigation including potential measures under consideration have been further discussed in the Draft MMMP |

| Consultee | Date/ Document | Comment | Project response/where addressed |
|---|---|---|---|
| | | | (Document Reference 6.5) submitted with the DCO Application. |
| NRW | | With reference to Section 9.4.2.2 Barrier effects caused by underwater noise, Paragraph 1.695, as noted in Paragraph 23 of the current document, NRW (A) recommend that further evidence is provided to support the statement that "the windfarm site is not located on any known migration routes of marine mammals".<br>Given the presence of a haul-out site in the Dee estuary, NRW (A) advise that the potential for barrier effects to impact grey seal movement towards the haul-out site needs to be considered and adequately assessed. | Barrier effects have been assessed both for Project-alone and in-combination.<br>The potential for barrier effects from underwater noise for the Project-alone during operation and maintenance has been assessed in **Chapter 11 Marine Mammals** of the ES (Section 11.6.4.4 and Section 11.6.4.5).<br>The evidence in **Chapter 11 Marine Mammals** of the ES (Section 11.6.3.5) regarding migration routes and barrier effects has been reviewed, and the assessment adjusted where appropriate.<br>The potential for effects to haul out sites has been assessed in **Section 9.6.2.6.** |
| NRW | | Regarding Section 9.4 Harbour porpoise, Paragraphs 1.609–10, as noted above, NRW (A) advise the use of Evans and Waggitt (2023) over Waggitt *et al.,* (2019). | Both data sources (Evans and Waggitt (2023) and Waggitt *et al.,* (2019)) have been considered, with the survey site specific data presenting the worst case. |

## 9.3 Assessment of potential effects

3254. The HRA Screening Report (Document Reference 4.10) identified the following potential effects that should be taken forward for further assessment in relation to the construction, operation and maintenance and decommissioning phases of the Project:

- Underwater noise
  - Permanent auditory injury/permanent loss of hearing sensitivity (referred to as Permanent Threshold Shift (PTS))
  - Disturbance
  - Barrier effects
- Vessel interactions
- Changes to prey resources
- Changes to water quality
- Disturbance to seals at haul-out sites

3255. Assessments for temporary change in hearing sensitivity (Temporary Threshold Shift (TTS)) have not been included in the assessment, as TTS does not result in permanent injury. TTS assessments have been included in **Chapter 11 Marine Mammals** and **Appendix 11.1** of the ES.

3256. The embedded mitigation and worst-case scenario presented in **Sections 9.3.1**, **9.3.2**, and **9.3.3** relate to these effects.

### 9.3.1 Embedded mitigation

3257. This section outlines the embedded mitigation relevant to the marine mammal assessments, which have been incorporated into the design of the Project.

*Table 9.2 Embedded mitigation measures relevant to marine mammals*

| Parameter | Mitigation measures embedded into the design of the Project |
|---|---|
| Piling schedule | No concurrent Project piling would be undertaken |
| Soft-start and ramp-up | Each piling event would commence with a soft-start at a lower hammer energy, followed by a gradual ramp-up to the maximum hammer energy required.<br>The soft-start and ramp-up would allow mobile species to move away from the area before the maximum hammer energy with the greatest noise impact area was reached. |

| Parameter | Mitigation measures embedded into the design of the Project |
|---|---|
| Pollution prevention | As outlined in **Chapter 8 Marine Sediment and Water Quality**, the Applicant is committed to the use of best practice techniques and due diligence regarding the potential for pollution throughout all construction, operation and maintenance, and decommissioning activities. An Outline Project Environment Management Plan (PEMP) (Document Reference 6.2) has been included with the Application. The PEMP, in line with international and national regulations, would set out all procedures and measures (including a Marine Pollution Contingency Plan (MPCP) and chemical risk assessment) to be followed during construction, operation and maintenance, and decommissioning phases to minimise the risk of, and effects in the event of an accidental spill. The final PEMP would be agreed with the Marine Management Organisation (MMO) prior to construction. |
| Cables and cable burial | Cables would be buried where possible. The cable burial range would be between 0.5m and 3.0m below the seabed (with a target depth of 1.5m where ground conditions allow (recognised industry good practice which would reduce effects of EMF)).<br>A Cable Burial Risk Assessment (CBRA) would also be required to confirm the extent to which cable burial can be achieved. Where it is not reasonably practicable to achieve cable burial, additional cable protection (e.g., rock placement, concrete mattresses or grout bags) would be required. An Outline Scour Protection and Cable Protection Plan (Document Reference 6.8) has been included with the Application.<br>Cables would be specified to reduce EMF emissions as per industry standards and best practice measures, such as, the relevant IEC (International Electrotechnical Commission) specifications. |

### 9.3.2 Commitment to additional mitigation measures

3258. In addition to the embedded mitigation measures outlined above, the Applicant has also committed to the production of an MMMP for piling and to apply best practice measures to reduce collision risk (**Table 9.3**). The Applicant has also committed to producing a MMMP for UXO clearance, should UXO clearance activities be required. This would be submitted as a separate Marine Licence, if UXO clearance is required, and does not form part of the DCO Application.

Table 9.3 Additional measures

| Document | Measures |
|---|---|
| MMMP for piling activities | The MMMP for piling would be developed in the pre-construction period and based upon best available information, methodologies, industry best practice, latest scientific understanding, current guidance and detailed project design. The MMMP for piling would be developed in consultation with the relevant SNCBs and the MMO, detailing the proposed mitigation measures to reduce the risk of any physical or permanent auditory injury PTS) to marine mammals during all piling operations.<br>This would include details of the embedded mitigation, for the soft-start and ramp-up, as well as details of the proposed mitigation zone and any additional mitigation measures required to minimise potential impacts of any physical or PTS, for example, the activation of an Acoustic Deterrent Device (ADD) prior to the soft-start.<br>The Draft MMMP (Document Reference 6.5) has been submitted with the DCO Application. |
| MMMP for UXO | A detailed MMMP would be prepared for UXO clearance during the pre-construction phase. The MMMP for UXO clearance would ensure there are adequate mitigation measures to minimise the risk of any physical injury or PTS to marine mammals as a result of UXO clearance.<br>The MMMP for UXO clearance would be developed in the pre-construction period when there would be more detailed information on the UXO clearance that may be required, and the most suitable mitigation measures, based upon best available information and methodologies at that time. The MMMP for UXO clearance would be prepared in consultation with the MMO and relevant SNCBs.<br>The MMMP for UXO clearance would include details of all the required mitigation measures to minimise the potential risk of PTS as a result of underwater noise during UXO clearance. |
| PEMP and as part of the Vessel Traffic Management Plan. | Best practice to reduce vessel collision risk:<br>Where reasonably practicable, vessel movements would follow set routes (and hence areas where marine mammals would be accustomed to vessels) to reduce collision risk. In line with efficient programming of tasks and utilisation of vessels, all vessel movements associated with the Project would be kept to a minimum. This, in turn, minimises the residual risk of collision.<br>Additionally, vessel operators would use good practice to reduce any risk of collisions with marine mammals. Consideration would also be given to minimum operating distances from seal haul-out sites, outside main shipping channels, particularly during sensitive periods for breeding and molting.<br>The Outline PEMP (Document Reference 6.2) and Vessel Traffic Management Plan (Document Reference 6.9) have been submitted with the DCO Application. |

### 9.3.3 Realistic worst-case scenario

3259. The final design of the Project would be confirmed through detailed engineering design studies that would be undertaken post-consent to enable the commencement of construction. To provide a precautionary, but robust impact assessment at this stage of the development process, realistic worst-case scenarios have been defined. The realistic worst-case scenario (having the most impact) for each individual impact was derived from the PDE to ensure that all other design scenarios would have less or the same impact. Further details have been provided in **Chapter 6 EIA Methodology** of the ES. This approach has been common practice for developments of this nature, as set out in PINS Advice Note Nine: Rochdale Envelope (PINS, 2018).

3260. The realistic worst-case scenarios for each potential impact have been outlined in **Table 9.4**.

*Table 9.4 Realistic worst-case scenario for Annex II sites designated for marine mammals*

| Impact | Worst-case scenario | Notes and rationale |
|---|---|---|
| **Construction phase** | | |
| Underwater noise during foundation installation (piling) | **Number of piles for WTG foundations:**<br>▪ Maximum of 35 WTGs<br>  ○ Up to 35 monopiles or<br>  ○ Up to 140 jacket pin-piles<br><br>**Number of piles for OSP foundations:**<br>▪ Maximum of two OSPs<br>  ○ Up to 2 monopiles or<br>  ○ Up to 8 jacket pin-piles<br><br>**Total number of piles for WTG and OSP foundations:**<br>▪ Maximum of 37 foundations<br>  ○ Up to 37 monopiles or<br>  ○ Up to 148 jacket pin-piles | The worst-case scenario for number of piles assumes the maximum number of WTGs (35) and OSPs (2) and assumes 100% of foundations are piled.<br>The worst-case scenario for number of piles assumes either one monopile per WTG and OSP, or four jacket pin-piles per WTG and OSP. The worst-case for sequential piling is three monopiles or four pin-piles installed sequentially in 24 hours.<br>The worst-case underwater noise modelling locations are as described in **Appendix 11.1** of the ES.<br>Hammer (impact) piled foundations represent the worst-case scenario for underwater noise.<br>Alternative foundation types are also considered, but do not represent the worst-case for underwater noise. |
| | **Maximum hammer energy for monopiles**:<br>▪ Up to 6,600kJ<br>**Maximum hammer energy for jacket pin-piles:**<br>▪ Up to 2,500kJ | The worst-case scenario assumes the maximum hammer energy would be required for each piling event after the completion of the soft start and ramp up.<br>However, in reality this is not expected to be required for all piles and would not be required for the entire duration while installing a pile. |
| | **Duration of WTG/OSP foundation installation**:<br>▪ Approximately 9 -12 months | Piling would not take place over the entire 9 -12 month period expected to be required for WTG and OSP installation. |

<table>
<tr><th>Impact</th><th>Worst-case scenario</th><th>Notes and rationale</th></tr>
<tr><td rowspan="3"></td><td>Maximum piling time for WTG foundations:<br>▪ Monopiles (including soft-start and ramp-up):<br>○ 3 hours 48 minutes per WTG<br>○ Up to 133 hours for 35 WTGs<br>or<br>▪ Jacket pin-piles (including soft-start and ramp-up):<br>○ 3 hours 13 minutes per jacket pin-pile<br>○ Up to 12 hours 53 minutes per foundation (4 pin-piles per foundation)<br>○ Up to 452 hours for 35 WTGs</td><td rowspan="2">Maximum piling time includes soft-start and ramp-up.<br>The maximum duration listed here reflects the worst-case scenario for underwater noise which considers the highest strike rate. It is noted that the duration of piling could be up to 4 hours 30 minutes per pile (monopile and each pin pile) if a lower strike rate was used but this does not present the worst-case for underwater noise ranges. The minor difference between piling duration in the high strike rate scenario and the lower strike rate scenario is not considered to be material, and as such the high strike rate is carried throughout the assessment.</td></tr>
<tr><td>Maximum piling time for OSP foundations:<br>▪ Monopiles (including soft-start and ramp-up):<br>○ 3 hours 48 minutes per OSP<br>○ Up to 7 hours 36 minutes for 2 OSPs<br>or<br>▪ Jacket pin-piles (including soft-start and ramp-up):<br>○ 3 hours 13 minutes per jacket pin-pile<br>○ Up to 12 hours 53 minutes per foundation (4 pin-piles per foundation)<br>○ Up to 25 hours 47 minutes for two OSPs</td></tr>
<tr><td>Maximum total piling time for WTGs and OSPs (including soft-start and ramp-up):<br>▪ Monopiles for WTGs and OSPs:<br>○ 190 hours<br>▪ Monopiles for WTGs and jacket pin-piles for OSPs:<br>○ 213 hours and 12 minutes</td><td>Worst-case scenario for total active piling time was assumed to be jacket piles for all WTGs plus OSP(s) (including soft-start and ramp-up).</td></tr>
</table>

| Impact | Worst-case scenario | Notes and rationale |
| --- | --- | --- |
| | - Jacket pin-piles for WTGs and OSPs:<br>  - Up to 619 hours and 36 minutes | |
| | **Activation of ADD:**<br>- For example: 80 minutes per monopile or 58 minutes for four sequential jacket pin-piles. | Indicative only, as this would be confirmed based on the final design and defined in the MMMP post-consent. |
| | **No concurrent piling for**:<br>- Installation of WTG/OSP foundations (monopiles or jacket piles)<br>- Installation of OSP foundations (monopiles or jacket piles)<br>- Installation of WTG and OSP foundations (monopiles or jacket piles) | The Project has not included any option for concurrent piling.<br>[concurrent piling = two or more piles installed at the same time at different locations from different vessels]. |
| | **Potential for sequential piling**:<br>- Monopiles = yes<br>  - Up to 3 monopiles could be installed sequentially in same 24-hour period<br>- Jacket piles = yes<br>  - Up to 4 jacket pin-piles could be installed sequentially in same 24-hour period | Assessments based on a worst-case scenario of three monopiles installed sequentially in the same 24-hour period, or up to four jacket piles installed sequentially in the same 24-hour period.<br>[sequential piling = one pile is installed after another pile in the same 24-hour period].<br>Cumulative sound exposure levels ($SEL_{cum}$) have been modelled for each piling event under consideration: single monopiles, single pin-piles, three monopiles piled sequentially and four pin-piles piled sequentially. Three sequential monopiles provided the worst-case in terms of $SEL_{cum}$. |
| | Underwater noise modelling undertaken for worst-case scenarios for piling. See **Appendix 11.1** (Document Reference 5.2.11.1) of the ES for parameters and scenarios. | |

<table>
<tr><th>Impact</th><th>Worst-case scenario</th><th>Notes and rationale</th></tr>
<tr><td rowspan="4">Underwater noise during other construction activities (such as seabed preparations, cable installation and rock placement)</td><td>Seabed clearance methods could include:<br>▪ Pre-lay grapnel run, boulder grab, plough, sandwave levelling (pre-sweeping) and dredging</td><td>Dredging was considered to be the worst-case in terms of underwater noise levels.</td></tr>
<tr><td>Cable & cable protection installation methods:<br>▪ Trenching (e.g. jetting or mechanical cutting)<br>▪ Dredging<br>▪ Ploughing<br>▪ Cable laying<br>▪ Rock placement</td><td>Underwater noise modelling undertaken for dredging, trenching, cable laying and rock placement. These activities have been considered the worst-case in terms of underwater noise for construction activities other than piling (see Appendix 11.1 of the ES).</td></tr>
<tr><td>Windfarm site: 87km$^2$</td><td>Maximum windfarm area.</td></tr>
<tr><td>Duration of offshore construction: 2.5 years</td><td>Offshore construction works could require up to 2.5 years (excluding pre-construction activities such as UXO clearance and geophysical surveys).</td></tr>
<tr><td>Underwater noise, presence and movements of vessels</td><td>Vessels:<br>▪ 2,583 return trips per year vessels including deliveries, installation vessels and support vessels<br>▪ Maximum total number of construction vessels on site at any one time = up to 37 vessels</td><td>Construction port(s) would be confirmed prior to the start of construction.<br>Not all construction vessels would be on site at same time, number of vessels will vary depending on activities taking place within windfarm site.<br>For example, the piling vessel for the OSP(s) would not be on site at same time as the piling vessel for the WTGs, as no concurrent piling would take place.<br>Assessments based on worst-case scenario for maximum number of vessels on site at any one-time during construction period.<br>Assessments based on worst-case scenario for maximum number of return vessel trips during construction period.</td></tr>
</table>

| Impact | Worst-case scenario | Notes and rationale |
|---|---|---|
| Barrier effect from underwater noise | Maximum impact range for all potential noise sources from underwater noise assessments (worst-case parameters described above).<br>Windfarm site located approximately 30km from the nearest point on the coast. | The maximum spatial area of potential impact, and duration of impacts, are considered to cause the worst-case barrier effect for underwater noise. |
| Changes to prey resources | Impacts to prey species and habitat as described in **Chapter 9 Benthic Ecology** and **Chapter 10 Fish and Shellfish Ecology** of the ES: Temporary habitat loss/physical disturbance; increased SSCs) and sediment re-deposition; remobilisation of contaminated sediments; underwater noise and vibration; and changes in fishing activity. | |
| | **Temporary habitat loss/seabed disturbance**<br><br>**WTG & OSP foundations:**<br>▪ 35 x WTGs with GBS foundations (including jack-up footprint) = 303,625m$^2$<br>▪ Two x OSPs with GBS foundations (including jack-up footprint) = 17,350m$^2$<br>▪ Anchoring for 35 WTGs and two OSPs = 26,640m$^2$<br><br>**Inter-array and platform link cables:**<br>▪ Inter-array cables = 1,750,000m$^2$<br>▪ Platform link cables = 250,000m$^2$<br><br>**Total area of seabed disturbance: 2,347,615m$^2$ (approximately 2.4km$^2$)** | Given the seabed preparation is the same per foundation for smaller and larger WTGs, the worst-case assumes 35 x smaller WTGs with GBS foundations. GBS foundations are assumed to have a diameter of 65m + 10m disturbance either side.<br>The worst-case scenario is for two jack-up visits per WTG/OSP foundation in different positions over the construction period (each jack-up with 6 legs, each with a 250m$^2$ footprint). This equates to a total footprint of 1,500m$^2$ per jack-up vessel visit and 3,000m$^2$ over the construction period per WTG/OSP foundation.<br>The worst-case scenario is for two anchor positions per foundation (including resetting), with up to 12 anchors per location. Each anchor width is estimated to be 6m, with an approximate seabed footprint of 30m$^2$ per anchor. |
| | **Sediment displaced during seabed preparation:**<br>▪ 35 x WTGs with GBS foundations = 455,438m$^3$<br>▪ Two x OSPs with GBS foundations = 26,025m$^3$<br>▪ Inter-array cables = 70,000m$^3$ | Seabed preparation (e.g. dredging using a trailing The seabed preparation area parameters are outlined in Impact 1 above. The seabed preparation area would be dredged to a depth of up to 1.5m. |

| Impact | Worst-case scenario | Notes and rationale |
|---|---|---|
| | ▪ Platform link cables = 10,000m³<br><br>**Sediment displaced during cable installation:**<br>▪ Inter-array cables = 472,500m³<br>▪ Platform link cables = 67,500m³<br><br>**Total volume of sediment disturbed: 1,101,463m³ (approximately 1.1km³)** | Drill arisings from drive-drill-drive installation methodology would result in a lower volume of sediment being disturbed (55,865m³ – based on monopile foundations).<br>The worst-case length of inter-array cables is 70km and platform link cables is 10km.<br>The worst-case assumes that 10% of the length of inter-array and platform link cables would require sandwave clearance/levelling. A clearance width of 10m and height of 1m is used. The worst-case assumes sediment would be released at the water surface.<br>The worst-case for cable installation assumes that 50% of inter-array and platform link cables are buried at 3m and 50% length is buried at 1.5m by jetting in a box-shaped trench, with a 3m trench width.<br>See **Chapter 9 Benthic Ecology** of the ES for more details. |
| | Underwater noise and vibration: Underwater noise modelling in **Appendix 11.1** of the ES.<br>Assessments for prey species in **Chapter 10 Fish and Shellfish Ecology** of the ES.<br>Barrier effects to prey species from underwater noise: as assessed in **Chapter 10 Fish and Shellfish Ecology** of the ES. | As above for underwater noise parameters. |
| | Changes in fish activity: as assessed in **Chapter 13 Commercial Fisheries**. | |
| Changes to water quality | Changes to water quality: as assessed **Chapter 8 Marine Sediment and Water Quality** of the ES. | Worst-case for any potential changes to water quality that could affect marine mammals directly. |
| Disturbance at seal haul-out sites | Distance of the windfarm site to seal haul-out sites: | Construction port(s) would be confirmed prior to the start of construction, however the assessment |

| Impact | Worst-case scenario | Notes and rationale |
|---|---|---|
| | ▪ Dee Estuary/ Hilbre Island: approximately 45km<br>▪ South Walney: approximately 30km<br>Windfarm site located approximately 30km from the nearest point on the coast.<br>Number of vessel trips as outlined above. | considered the potential for vessels in transit in proximity to the seal haul out sites in the study area.<br>Movements of construction vessels could occur throughout the year. |
| **Operation and maintenance phase** | | |
| Underwater noise from operational turbines | WTG parameters (e.g. size and number) as outlined above and underwater noise parameters described in **Appendix 11.1** of the ES.<br>Operational life of windfarm = 35 years | |
| Underwater noise from maintenance activities | **Estimated inter-array cable repair/replacement or reburial works:**<br>▪ Average length of inter-array/platform link cable repair/replacement every year = up to 200m<br>▪ Average length of inter-array/platform link cable reburial every year = up to 100m | Disturbance is shown on average per year; however, repair/replacement, cable lengths and reburial activities could vary across years during the operation and maintenance phase.<br>Underwater noise modelling undertaken for dredging, trenching, cable laying and rock placement (see above and **Appendix 11.1**). |
| Underwater noise, presence and movements of vessels | **Vessels:**<br>▪ Types of vessels: cable laying and burial, rock placement, support vessels, crew transfer vessels, jack-up vessels<br>▪ Maximum number of vessels on site at any one time:<br>○ Three vessels during a standard year and 10 vessels on a 'heavy maintenance' year (every 5 years)<br>▪ Maximum annual number of vessel return trips to port:<br>▪ 384 vessels during a standard year and 832 vessels on a 'heavy maintenance' year | Operation and maintenance port(s) have still to be determined.<br>Assessments based on worst-case scenario for maximum number of operation and maintenance vessels on site at any one-time and maximum number of return vessel trips during operation and maintenance period. |

| Impact | Worst-case scenario | Notes and rationale |
|---|---|---|
| Barrier effect from underwater noise | Maximum impact range for all potential noise sources from underwater noise assessments (as above) during operation and maintenance phase.<br><br>**WTG spacing:**<br>▪ Minimum in row spacing: 1,060m<br>▪ Minimum inter row spacing: 1,410m | The maximum spatial area of potential impact, and duration of impacts, were considered to cause the worst-case barrier effect for underwater noise. |
| Changes to prey resources | Impacts to prey species and habitat as described in **Chapter 9 Benthic Ecology** and **Chapter 10 Fish and Shellfish** of the ES: Permanent habitat loss; temporary habitat loss/physical disturbance of the seabed, increased SSCs and sediment deposition; underwater noise; EMF; barrier effects; introduction of hard substrates; and changes in fishing activity. | |
| | **Worst-case for total habitat loss to the footprint of infrastructure:**<br>▪ 35 x GBS WTGs with scour protection = 248,080$m^2$<br>▪ Two GBS OSPs with scour protection = 14,176$m^2$<br>▪ Inter-array cables = 91,000$m^2$<br>▪ Platform link cables = 13,000$m^2$<br>▪ Cable protection at the entry to WTGs and OSPs = 45,500$m^2$<br>▪ Cable crossings (at inter-array and platform link cables): 66,750$m^2$<br>▪ Replacement scour protection = 13,950$m^2$<br><br>**Total worst-case habitat loss: 514,081$m^2$ (approximately 0.51$km^2$)** | The worst-case scenario based on maximum area of infrastructure on the seabed. |
| | Temporary habitat loss, physical disturbance of the seabed, increases in SSCs and sediment deposition due to maintenance activities could result from periodic jack-up vessel deployment, and cable repair, replacement and reburial activities. These activities are likely to be lower in magnitude than for construction. | |

| Impact | Worst-case scenario | Notes and rationale |
|---|---|---|
| | Underwater noise parameters as outlined for operation noise-related impacts above and **Appendix 11.1** of the ES (operational WTGs, maintenance activities, vessels). | As above for underwater noise. |
| | **EMF from offshore cables**<br>Up to 70km of inter-array and 10km platform link cables:<br>▪ Cable operating voltage of 220/275kV AC<br>▪ Burial range of 0.5m-3m where possible with a target burial depth of 1.5m | Cable burial would substantially reduce the levels of EMF in the surrounding area. Where cable burial was not possible, protection would be added which would reduce the levels of EMF. |
| | Barrier effects from underwater noise or EMF: As above | |
| | Introduction of hard substrate: As above for WTGs, OSP(s), scour protection, inter-array and platform link cable protection, cable protection at the entry to WTGs and OSP(s) and cable crossings (approximately 0.51km$^2$) | As above for total habitat loss to the footprint of infrastructure. |
| Changes to water quality | Changes to water quality as assessed in **Chapter 8 Marine Sediment and Water Quality**. | |
| Disturbance at seal haul-out sites | Distance of the windfarm site and vessel routes to seal haul-out sites:<br>▪ Dee Estuary/ Hilbre Island: approximately 45km<br>▪ South Walney: approximately 30km<br><br>Windfarm site located approximately 30km from the nearest point on the coast.<br>Number of vessel trips as outlined above. | Operation and maintenance port(s) to be confirmed post-consent, at this stage assumed within a 50km range and considered in transit in regard to the seal haul-out sites in the study area.<br>Movements of vessels could occur throughout the year. |

| Impact | Worst-case scenario | Notes and rationale |
|---|---|---|
| **Decommissioning phase** | | |
| As for construction | The decommissioning policy for the Project infrastructure is not yet defined however it is anticipated that structures above the seabed would be removed.<br>The following infrastructure is likely be removed, reused, or recycled where practicable:<br>▪ WTGs and foundations<br>▪ OSP(s) including topsides and foundations.<br>The following infrastructure is likely to be decommissioned and could be left *in situ*, depending on regulator advice and available information at the time of decommissioning:<br>▪ Inter array and platform link cables<br>▪ Scour protection<br>▪ Cable crossings and cable protection<br>▪ Part of the foundations (e.g. some foundation material below the seabed may be left *in situ*) | The detail and scope of the decommissioning works would be determined by the relevant legislation and guidance at the time.<br>Decommissioning arrangements would be detailed in a Decommissioning Programme, which would be drawn up and agreed with the relevant authority at the time, prior to decommissioning.<br>For the purposes of the worst-case scenario, it is anticipated that the impacts would be comparable to those identified for the construction phase. |

## 9.4 Harbour porpoise

### 9.4.1 Relevant sites

#### 9.4.1.1 North Anglesey Marine SAC

**Description of designation**

3261. North Anglesey Marine SAC has been recognised as an area with persistent high densities of harbour porpoise and covers an area of 3,249km$^2$ (JNCC *et al*., 2019a).

3262. The North Anglesey Marine SAC is 49km from the Project, when measured as a straight line distance.

3263. North Anglesey Marine SAC has been designated because of its importance to harbour porpoise in the summer months (April to September). The selection was primarily based on the long-term, relatively higher densities of porpoise in contrast to other areas of the MU. The implication is that the SAC provides relatively good foraging habitat and may also be used for breeding and calving (JNCC *et al*., 2019a).

**Harbour porpoise population and density**

3264. For conservation and management purposes, it is practical to divide the UK harbour porpoise population into smaller units, termed Management Units (MUs). These MUs were developed to take account of biological populations of animals but were also determined by political boundaries and are at an appropriate scale at which to assess human activities. In the UK, three MUs have been defined for harbour porpoise: West of Scotland, Celtic and ISs, and North Sea (Inter-Agency Marine Mammal Working Group (IAMMWG), 2023). The relevant MU for this assessment was the Celtic and ISs (CIS) MU. The estimate of harbour porpoise abundance in the CIS MU was 62,517 (Coefficient of Variation (CV) = 0.13; 95% Confidence Interval (CI) = 48,324 – 80,877; IAMMWG, 2023).

3265. It has been estimated that the North Anglesey Marine SAC supports approximately 1,088 harbour porpoise and represents approximately 2.4% of the population within the UK part of the CIS MU (NRW and JNCC, 2017). However, NRW and JNCC advised that because this estimate was from a one month survey in a single year (July 2005) it could not be considered as a specific population number for the site. It was therefore not appropriate to assign a site population estimate because of the daily and seasonal movements of the animals (NRW and JNCC, 2017). JNCC *et al.,* (2019a) advised that for the purpose of assessment the reference population was the MU population in which the SAC was situated.

3266. The reference population for harbour porpoise used in the assessments was the CIS MU (62,517).

3267. Density estimates for the Project windfarm site were reviewed, including distribution and abundance maps developed by Waggitt *et al.,* (2019); results from the SCANS-IV survey, undertaken in summer 2022, for survey block CS-E in which the windfarm site is located (Gilles *et al*., 2023); and data from the two year (March 2021 to February 2023) Project site-specific surveys (see **Chapter 11 Marine Mammals** and **Appendix 11.2 Marine Mammal Information and Survey Data** (Document Reference 5.2.11.2) of the ES for further information).

3268. The average summer density estimate of 1.621 harbour porpoise per km$^2$ from the two year site-specific surveys has been used in the assessments.

### Conservation status

3269. Based on the most recent 2013-2018 reporting by JNCC (2019), the overall assessments of Conservation Status for harbour porpoise population in UK waters is ‘unknown’.

### Conservation objectives

3270. The relevant conservation objective for the North Anglesey Marine SAC is (JNCC *et al*., 2019a):

- To ensure that the integrity of the site is maintained and that it makes the best possible contribution to maintaining Favourable Conservation Status (FCS) for Harbour Porpoise in UK waters

3271. In the context of natural change, this could be achieved by ensuring that:

- Harbour porpoise is a viable component of the site
- There is no significant disturbance of the species
- The condition of supporting habitats and processes, and the availability of prey is maintained

*Conservation objective 1: The species is a viable component of the site*

3272. This conservation objective has been designed to minimise the risk of injury and killing or other factors that could restrict the survivability and reproductive potential of harbour porpoise using the SAC. Specifically, this objective is primarily concerned with operations that would result in unacceptable levels of those impacts on harbour porpoise using the SAC. Unacceptable levels can be defined as those having an impact on the FCS of the population of the species in their natural range. The reference population for assessments against this objective was the MU population in which the SAC was situated.

*Conservation objective 2: There is no significant disturbance of the species*

3273. The disturbance of harbour porpoise typically, but not exclusively, originates from operations that cause underwater noise, including activities such as seismic surveys, pile driving and sonar.

3274. Disturbance is considered to be significant if it leads to the exclusion of harbour porpoise from a significant portion of the site for a significant period of time. The latest SNCB guidance for the assessment of significant noise disturbance on harbour porpoise in the North Anglesey Marine SAC (JNCC *et al*., 2019a; JNCC *et al*., 2020) was that:

- "Noise disturbance within an SAC21 from a plan/project individually or in-combination is considered to be significant if it excludes harbour porpoise from more than:
  - 20% of the relevant area[22] of the site in any given day[23], or
  - An average of 10% of the relevant area[24] of the site over a season[25]"

*Conservation objective 3: The condition of supporting habitats and processes, and the availability of their prey is maintained*

3275. Supporting habitats, in this context, mean the characteristics of the seabed and water column. Supporting processes encompass the movements and physical properties of the habitat. The maintenance of these supporting habitats and processes contributes to ensuring prey would be maintained within the site and available to harbour porpoise using the SAC. Harbour porpoise are strongly reliant on the availability of prey species year round due to their high energy demands, and their distribution and condition may strongly reflect the availability and energy density of prey.

3276. This conservation objective has been designed to ensure that harbour porpoise are able to access food resources year round, and that activities occurring in the North Anglesey Marine SAC would not affect this.

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[21] It is noted that the Project would be over 26km from the SAC and therefore there would be no spatial overlap of effect upon the SAC itself.

[22] The relevant area has been defined as that part of the SAC that was designated on the basis of higher persistent densities for that season (summer defined as April to September inclusive, winter as October to March inclusive).

[23] To be considered within the HRA and, if needed, licence conditions should ensure that daily thresholds would not be exceeded.

[24] For example, a daily footprint of 19% for 95 days would result in an average of 19x95/183 days (summer) =9.86%

[25] Summer defined as April to September inclusive, winter as October to March inclusive.

3277. For the purposes of the assessment, the potential effects have been considered in relation to the North Anglesey Marine SAC conservation objectives, as outlined in **Table 9.5.**

*Table 9.5 Potential effects in relation to the conservation objectives for the North Anglesey Marine SAC for harbour porpoise*

| Conservation objective | Potential effect |
|---|---|
| Harbour porpoise is a viable component of the site | Physical and permanent auditory injury from piling would be mitigated, however this has been considered in detail in line with current advice. |
| | Significant disturbance and displacement as a result of increased underwater noise levels (e.g. piling) has the potential to affect harbour porpoise from the SAC and has been assessed. |
| | Increased collision risk with vessels has the potential to affect harbour porpoise from the SAC which has been assessed. |
| There is no significant disturbance of the species | The conservation objective strictly refers to disturbance within the SAC, given the lack of overlap of underwater noise ranges this was not considered relevant.<br>Disturbance outside the SAC has been assessed in relation to the harbour porpoise being a viable component of the site |
| The condition of supporting habitats and processes, and the availability of prey is maintained | Changes in water quality and prey availability have the potential to affect the harbour porpoise from the North Anglesey Marine SAC and have been assessed. |

#### 9.4.1.2 North Channel SAC

##### Description of designation

3278. North Channel SAC covers an area of 1,604km$^2$, and has been designated because of its importance to harbour porpoise in the winter months (October – March) (JNCC and Department of Agriculture, Environment and Rural Affairs (DAERA), 2019; JNCC *et al*., 2020).

3279. The North Channel SAC is 103km from the Project windfarm site (measured as a straight line distance) and 108km (measured as a coastline distance).

##### Harbour porpoise population and density

3280. It has been estimated (based on the SCANS-II survey which took place in July 2005 only) that the site supports approximately 537 individuals (95% CI 276 – 1,046) (for at least part of the year as seasonal differences are likely to occur) and represented approximately 1.2% of the population within the UK part of the Celtic and IS MU (DAERA and JNCC, 2017).

3281. As per JNCC *et al.,* (2019a) advice (see **Section 9.4.1.1**) the reference population for harbour porpoise used in the assessments was the CIS MU.

3282. The average summer density estimate of 1.621 harbour porpoise per km$^2$ from the Project site-specific surveys has been used in the assessments.

### Conservation status

3283. Unknown (see North Anglesey Marine SAC **Section 9.4.1.1**).

### Conservation objectives

3284. The conservation objectives for the North Channel SAC were the same as those for the North Anglesey Marine SAC (see **Section 9.4.1.1**) and have not been repeated here.

## 9.4.1.3 West Wales Marine SAC

### Description of designation

3285. The West Wales Marine SAC coverer an area of 7,376km$^2$, off the coast of Wales from the Llŷn peninsula in the north, to Pembrokeshire in the south-west (NRW and JNCC, 2019).

3286. The West Wales Marine SAC is 109km from the windfarm site, measured as a straight line distance, and 129km, measured as a coastline distance.

3287. The West Wales Marine SAC has been designated because of its importance to harbour porpoise in both the summer and winter months (NRW and JNCC, 2019). The summer period is April to September (inclusive) and the area used in summer is 7,379km$^2$. The winter period is October to March (inclusive) and the area used in winter is 1,460km$^2$ (JNCC *et al*., 2020).

### Harbour porpoise population and density

3288. It has been estimated (based on the SCANS-II survey which took place in July 2005) that the site supported approximately 5,222 individuals (95% CI: 1419 - 4484) (for at least part of the year as seasonal differences were likely to occur). This represents approximately 5.4% of the population within the UK part of the CIS MU. Revised “population in the site” estimates based on the 2016 survey (Hammond *et al.,* 2021) were a minimum of 964 (lower 95% CI) and maximum of 2558 (higher 95% CI). All these estimates have been derived from one-month summer surveys and should not be considered as specific population sizes for the site and as such the widest population is still listed as the supporting population of the site based on the 2005 survey.

3289. As per JNCC *et al.*, (2019a) advice the reference population for harbour porpoise used in the assessments was the CIS MU.

3290. The average annual density estimate of 1.621 harbour porpoise per km$^2$ from the two year site-specific surveys has been used in the assessments.

### Conservation status

3291. Unknown (see North Anglesey Marine SAC).

### Conservation objectives

3292. The conservation objectives for the West Wales Marine SAC are the same as those for the North Anglesey Marine SAC (see **Section 9.4.1.1**) and have not been repeated here.

## 9.4.1.4 Rockabill to Dalkey Island SAC

### Description of designation

3293. The Rockabill to Dalkey Island SAC in the western IS represents a key habitat for harbour porpoise within the IS. The species has been observed year-round within the site and comparatively high group sizes have been recorded (NPWS, 2013). The Natura 2000 data form was updated in 2019 with a population of between 138-349.

3294. The Rockabill to Dalkey Island SAC was 156km from the windfarm site, measured as a straight line distance.

### Conservation status

3295. Unknown (see North Anglesey Marine SAC **Section 9.4.1.1**).

### Harbour porpoise population and density

3296. As per JNCC *et al.*, (2019a) advice (see **Section 9.4.1.1**) the reference population for harbour porpoise used in the assessments was the CIS MU[26].

3297. The average summer density estimate of 1.621 harbour porpoise per km$^2$ from the two year Project site-specific surveys has been used in the assessments.

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[26] Although the Rockabill to Dalkey Island SAC has not signed up to the JNCC *et al.*, 2019 guidance on harbour porpoise SACs, the JNCC approach has been followed for consistency of assessing the SACs.

## Conservation objectives

3298. The conservation objectives for the Rockabill to Dalkey Island SAC are (NPWS, 2013):

- To maintain the favourable conservation condition of harbour porpoise in Rockabill to Dalkey Island SAC, which is defined by the following list of attributes and targets:
  - Species range within the site should not be restricted by artificial barriers to site use.
  - Human activities should occur at levels that do not adversely affect the harbour porpoise community at the site.

3299. For the purposes of the assessment, the potential effects have been considered in relation to the Rockabill to Dalkey Island SAC conservation objectives outlined in **Table 9.6**.

*Table 9.6 Potential effects in relation to the conservation objectives for the Rockabill to Dalkey Island SAC for harbour porpoise*

| Conservation objective | Potential effect |
|---|---|
| Species range within the site should not be restricted by artificial barriers to site use. | Harbour porpoise within the SAC would not be restricted by any barrier effects from underwater noise associated with the Project or the physical presence of the windfarm.<br>However, significant disturbance or displacement to harbour porpoise out with the SAC as a result of increased underwater noise levels has the potential to affect harbour porpoise from the Rockabill to Dalkey Island SAC and has been assessed. |
| Human activities should occur at levels that do not adversely affect the harbour porpoise community at the site. | Physical and permanent auditory injury from piling would be mitigated and therefore there is no potential for LSE, however this has been assessed in detail in line with the latest advice. |
| | Significant disturbance and displacement as a result of increased underwater noise levels (e.g. from piling) has the potential to affect harbour porpoise from the Rockabill to Dalkey Island SAC and has been assessed. |
| | Any potential increased collision risk with vessels could cause a potential LSE which has been assessed. |

## 9.4.1.5 Bristol Channel Approaches SAC

### Description of designation

3300. The Bristol Channel Approaches SAC extends across the western approaches to the Bristol Channel, from Carmarthen Bay in South Wales to the north coast of Devon and Cornwall. The site covers an area of 5,851km$^2$.

3301. The Bristol Channel Approaches SAC is 234km from the windfarm site, measured as a straight line distance, and 310km measured as a coastline distance.

3302. The Bristol Channel Approaches SAC has been designated because of its importance to harbour porpoise in the winter months (October to March) (JNCC *et al*., 2019b).

### Harbour porpoise population and density

3303. It has been estimated (based on the SCANS-II survey which took place in July 2005) that the site supported approximately 2,147 individuals (95% CI: 810 – 5,693) for at least part of the year, as seasonal differences were likely to occur, and represented approximately 4.7% of the population within the UK part of the CIS MU. Revised “population in the site” estimates based on the 2016 survey (Hammond *et al.,* 2021) were a minimum of 278 (lower 95% CI) and maximum of 1,713 (higher 95% CI). All these estimates were derived from one-month summer surveys and should not be considered as specific population sizes for the site. As such, the widest population is still listed as the supporting population of the site based on the 2005 survey.

3304. As per JNCC *et al.,* (2019b) advice, the reference population for harbour porpoise used in the assessments was the CIS MU.

3305. The average summer density estimate of 1.621 harbour porpoise per km$^2$ from the two year Project site-specific surveys has been used in the assessments.

### Conservation status

3306. Unknown (see North Anglesey Marine SAC **Section 9.4.1.1**).

### Conservation objectives

3307. The conservation objectives for the Bristol Channel Approaches SAC are the same as those for the North Anglesey Marine SAC (see **Section 9.4.1.1**) and have not been repeated here.

## 9.4.2 Project-alone assessment

### 9.4.2.1 Underwater noise

3308. The assessment below refers to the ES assessment, as the population and density estimates used in the EIA were the same as for this assessment. A full underwater noise assessment has been undertaken in Section 11.6.3 of **Chapter 11 Marine Mammals** of the ES, and relevant information from that chapter is summarised in the sections below.

#### Permanent auditory injury from underwater noise during piling

3309. The effect would be relevant to the construction phase only, with effects occurring outside any SAC.

3310. Underwater noise modelling was carried out (**Appendix 11.1** of the ES) to predict the noise levels likely to arise during impact piling and other activities. The modelled impact ranges were used to determine the potential effects on marine mammals. A detailed explanation of the modelling, inputs and assumptions has been provided in Section 11.6.3.1 of **Chapter 11 Marine Mammals** of the ES.

3311. Several scenarios were modelled to determine the worst-case for PTS effects for monopiles and pin-piles of a single strike ($SPL_{peak}$), the total received noise over the whole piling operation, and cumulative effects of sequential piling of four pin-piles or three monopiles.

3312. The maximum predicted impact range for PTS for harbour porpoise was up to 8.2km from $SEL_{cum}$ during sequential monopile installation with maximum hammer energy (6,600kJ), including soft-start and ramp up procedures (**Table 9.7**). Given the distance of the Project windfarm site from the closest SAC (49km), there was therefore no pathway for effects upon harbour porpoise within any SAC considered in this assessment.

3313. An assessment of the maximum number of individuals and percentage of the reference population affected (outside the SAC) under each of the scenarios was undertaken using the assumed worst-case densities and CIS MU.

3314. For PTS the maximum impact was up to 243 harbour porpoise, which represented up to 0.4% of the CIS MU (**Table 9.8**). Given the embedded mitigation, it is concluded that there would be no LSE on the reference population (and no AEoI on any SAC) from PTS. The final approved piling MMMP would reduce the risk of PTS still further. The final MMMP for piling would be based on the Draft MMMP (Document Reference 6.5) which has been included with the DCO Application.

*Table 9.7 Predicted PTS impact ranges (and areas) for harbour porpoise at the Project from a single strike and from cumulative exposure for maximum hammer energy (taken from Table 11.21 of the ES)*

| Impact | Criteria and threshold (Southall *et al*., 2019) | Monopile Maximum impact range (km) and area ($km^2$) *Maximum hammer energy (6,600kJ)* | Pin-pile Maximum impact range (km) and area ($km^2$) *Maximum hammer energy (2,500kJ)* | Monopile (sequential piling) Maximum impact range (km) and area ($km^2$) *Maximum hammer energy (6,600kJ)* | Pin-pile (sequential piling) Maximum impact range (km) and area ($km^2$) *Maximum hammer energy (2,500kJ)* |
|---|---|---|---|---|---|
| PTS from single strike (without mitigation) | $SPL_{peak}$ Unweighted (202 dB re 1µPa) Impulsive | 0.69km ($1.5km^2$) | 0.54km ($0.9km^2$) | N/A | N/A |
| PTS from cumulative SEL (including soft-start and ramp-up) | $SEL_{cum}$ Weighted (155 dB re $1µPa^2s$) Impulsive | 8.1km ($150km^2$) | 5.1km ($60km^2$) | 8.2km ($150km^2$) | 5.2km ($61km^2$) |

*Table 9.8 Maximum number of harbour porpoise (and % of reference population) that could be at risk of PTS from single strike and from cumulative exposure ($SEL_{cum}$) during installation of three sequential monopiles or four pin-piles (taken from Table 11.23 and 11.24, of the ES)*

| **Impact** | **Criteria and threshold (Southall *et al*., 2019)** | **Monopile with maximum hammer energy of 6,600kJ**<br><br>**Maximum number of individuals (% of reference population)** | **Pin-pile with maximum hammer energy of 2,500kJ**<br><br>**Maximum number of individuals (% of reference population)** |
|---|---|---|---|
| Single strike at maximum hammer energy | $SPL_{peak}$ Unweighted<br>(202 dB re 1µPa) Impulsive | 2.4<br>(0.004% of CIS MU) | 1.5<br>(0.002% of CIS MU) |
| Cumulative exposure ($SEL_{cum}$) during sequential piling of four pin-piles or three monopiles | $SEL_{cum}$ Weighted (155 dB re 1µPa$^2$s)<br>Impulsive | 243<br>(0.4% of CIS MU) | 98.9<br>(0.2% of CIS MU) |

3315. The MMMP for piling (see **Section 9.3.1**) would reduce the risk of PTS from the first strike of the soft-start, single strike of the maximum hammer energy and risk of PTS from cumulative exposure. The MMMP for piling would be developed post-consent in consultation with the MMO and other relevant organisations and would be based on the latest information, scientific understanding and guidance, and detailed project design. The final MMMP for piling would be based on the Draft MMMP (Document Reference 6.5) submitted with the DCO Application.

3316. Given the embedded mitigation, it has been concluded that there would be no LSE on the reference population (and no AEoI on any SAC).

3317. The piling MMMP would reduce the risk of PTS still further. The MMMP is likely to include establishing a monitoring zone and ADD activation prior to the soft-start commencing.

3318. With the application of this mitigation the risk of PTS would be further reduced.

### Disturbance impacts from underwater noise during piling

3319. The effect is relevant to the construction phase only with effects occurring outside any SAC.

3320. Disturbance from underwater noise from piling has been assessed in detail in Section 11.6.3.2 of **Chapter 11 Marine Mammals** of the ES.

3321. For harbour porpoise several methods were used to assess the effect of disturbance:

- EDR approach
- Dose Response Curve
- iPCoD
- Disturbance during ADD activation

3322. The most recent SNCB guidance recommends that a potential disturbance range or EDR of 26km (approximate area of 2,124km$^2$) around monopile locations (without noise abatement) and 15km (approximate area of 707km$^2$) for pin-piles with and without noise abatement should be used to assess harbour porpoise disturbance for SACs in England, Wales and Northern Ireland (JNCC *et al*., 2020).

3323. As outlined in **Section 9.4.1**, the nearest SAC is 49km from the windfarm site. As such there is no potential overlap with the SACs in this assessment and no direct effects on harbour porpoise within the SACs.

3324. An assessment of the maximum number of individuals and percentage of the reference population affected (outside the SAC) has been undertaken. The worst-case, based on 26km EDR for a monopile was that up to 3,443 harbour porpoise (5.5% of the CIS MU population) could be disturbed (see Table 11.28 of **Chapter 11 Marine Mammals** of the ES). Based on the dose-response approach, up to 1,858 harbour porpoise (3.02% of the CIS MU) could be disturbed.

3325. For 15km EDR for pin-piles, up to 1,146 (1.8% of CIS MU) could be disturbed (see Table 11.28 of **Chapter 11 Marine Mammals** of the ES).

3326. The maximum duration (considering the worst-case high strike rate scenario) of effect for active piling assuming two OSPs and all WTGs using monopiles and 26km EDR was 140.6 hours. The maximum duration of effect for active piling assuming two OSPs and all WTGs using pin-piles and 15km EDR was 513.6 hours.

3327. The total duration of the installation campaign for WTGs and OSPs foundations is expected to be between 9 - 12 months. The duration of piling has been based on a worst-case scenario and a very precautionary approach, and, as it has been shown at other OWFs, the duration used in the assessment can be overestimated. For example, at the Beatrice Offshore Wind Farm, it was estimated that each pin-pile would require five hours of active piling time. However, during construction, the total duration of piling ranged from 19 minutes to two hours and 45 minutes, with an average duration of one hour and 15 minutes per pile (Beatrice Offshore Wind Farm Ltd, 2018).

3328. The duration of any potential displacement effect would differ depending on the distance of the individual from the piling activity and the noise level the animal was exposed to.

3329. Section 11.6.3.2 of **Chapter 11 Marine Mammals** of the ES reviewed studies undertaken on the effects of disturbance which concluded that although there was potential for adverse short-term effects of construction on harbour porpoise, there was no indication of negative effects of windfarm construction at the population level (Brandt *et al*., 2016, Booth *et al*., 2017 and Nabe-Nielsen *et al*., 2018).

3330. As the 26km EDR assessment has indicated over 5% of the reference population may be disturbed, assessments from iPCoD modelling have been considered further.

3331. Assuming a worst-case of 3,443 harbour porpoise that could be disturbed on every piling day (assuming piling over 37 days), the iPCoD model estimated there to be only the slightest discernible impact to the harbour porpoise population (**Table 9.9** and **Plate 9.1**). It should be noted that the numbers of disturbed harbour porpoise were precautionary as they have been based on the high site-specific density which has been applied across the entire 26km EDR range.

3332. The median population size was predicted to be 100% of the un-impacted population size at the end of 2028 (one year after the piling has completed). By the end of 2029 (two years after piling ends) the median population size for the impacted population was predicted to be 99.89% of the un-impacted population size. Beyond 2029, the impacted population was expected to maintain the same stable trajectory as the un-impacted population (as far as 2052 which was the end point of the modelling).

3333. For harbour porpoise, the modelling indicated there was no potential for a significant impact of disturbance due to there being less than a 1% population level impact over both the first six years and 25 year modelled periods. **Therefore, it has been concluded that there would be no LSE on the harbour porpoise CIS MU population (and no AEoI on any SAC) from the effects of disturbance impacts from underwater noise during piling.**

*Table 9.9 Results of the iPCoD modelling for the Project, giving the mean population size of the harbour porpoise population (CIS MU) for years up to 2052 for both impacted and un-impacted populations in addition to the mean and median ratio between their population sizes*

| Year | Un-impacted population mean | Impacted population mean | Median impacted as % of un-impacted |
|---|---|---|---|
| Start | 62,516 | 62,516 | 100.00% |
| End 2028 | 62,451 | 62,451 | 100.00% |
| End 2029 | 62,424 | 62,268 | 99.89% |
| End 2032 | 62,524 | 62,403 | 99.89% |
| End 2037 | 62,307 | 62,180 | 99.89% |
| End 2047 | 62,036 | 61,908 | 99.89% |
| End 2052 | 61,876 | 61,750 | 99.89% |

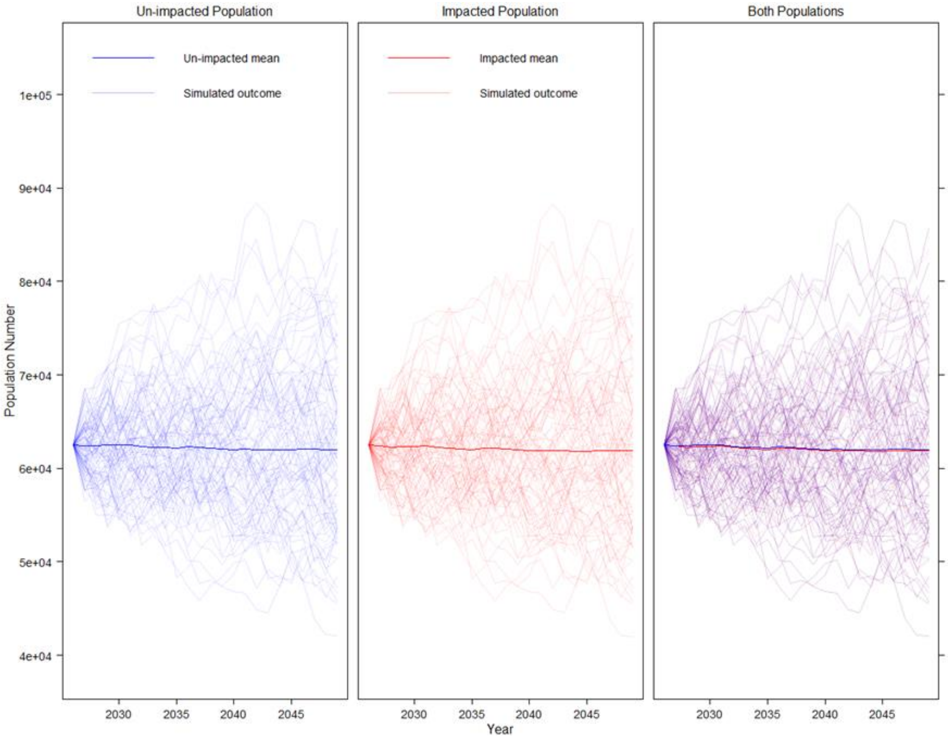


*Plate 9.1 Simulated worst-case harbour porpoise population sizes for both the un-impacted and the impacted populations for the Project (scientific notation used in these charts, e.g. 4e+04 = 40,000).*

## Underwater noise and disturbance from other sources

### Construction

3334. Section 11.6.3.3 of **Chapter 11 Marine Mammals** of the ES details the effects of disturbance impacts from underwater noise from seabed preparation, dredging, trenching, cable installation and rock placement. Section 11.6.3.4 of **Chapter 11 Marine Mammals** of the ES details the effects of underwater noise from the presence of vessels.

3335. A review of various studies was used to determine the maximum potential disturbance range for other construction activities and vessels. Studies undertaken during the construction of two Scottish windfarms (Beatrice OWF and Moray East OWF) (Benhemma-Le Gall *et al*., 2021) found that there was a reduction in porpoise presence detected at up to 12km from pile driving, and up to 4km from construction activities. The 4km radius has been used as the disturbance range for other construction activities, including vessels. For the 37 construction vessels that could be in the Project site at any one time in addition to the 4km buffer for each vessel, the total impact area of 1859.8km$^2$ was an unrealistic worst-case. This scenario did not take into account the overlap in the 4km disturbance range between vessels and the area was approximately 21 times the size than the Project site alone (87km$^2$). In the **Chapter 11 Marine Mammals** of the ES, Plate 11.8 presents such a scenario (for illustrative purpose only), where 37 vessels were on site and within a 4km buffer demonstrating the use of this area (285.4km$^2$) was considered to be sufficient to assess the impacts of vessels during construction.

3336. Taking into account the distance of the Project windfarm site from the closest SAC (49km), there was no pathway for disturbance effects directly upon harbour porpoise within any SAC considered in this assessment.

3337. The assessments took into account the number of construction activities, other than piling, that could be undertaken at the same time, and the maximum number of vessels that could be on site at any one time; these have been summarised in **Table 9.10** and **Table 9.11**.

3338. As a precautionary approach, the potential disturbance from two activities (such as cable laying, dredging, trenching or rock placement) occurring at the same time, including vessels used in this assessment, has been based on a potential impact area of 100.54km$^2$. The maximum number of harbour porpoise that could be disturbed was up to 163 (0.3% of the CIS MU).

3339. The maximum number of harbour porpoise that could be disturbed from up to 37 vessels on site at the same time for the Project windfarm site and 4km buffer was up to 462.6 harbour porpoise (0.74% of CIS MU).

3340. There would be no potential for additive effects (i.e. the disturbance from construction activities plus vessel activities), as the disturbance range of 4km for construction activities included the vessel undertaking the work. In addition, the 26km or 15km EDR for piling would encompass any effects from other construction activities and vessels on site during piling.

3341. The worst-case assessments indicated that for underwater noise and disturbance from other sources, less than 1% (up to 0.74%) of the harbour porpoise CIS MU population could be temporarily disturbed.

3342. Given that this effect was lower than for disturbance impacts from underwater noise during piling and the effect would occur outside of any SAC**, it has been concluded that there would be no LSE on the harbour porpoise CIS MU population (and no AEoI on any SAC) from the effects of disturbance impacts from underwater noise from other sources during construction.**

*Table 9.10 Maximum number of harbour porpoise (and % of CIS MU) that could be disturbed as a result of underwater noise associated with other (non-piling) construction activities, including vessels undertaking the work (taken from Table 11.52 of the ES)*

| Potential impact | Maximum number of individuals (% of reference population) that could be disturbed for one activity (50.27km$^2$) | Maximum number of individuals (% of reference population) that could be disturbed for two activities (100.54km$^2$) |
|---|---|---|
| Disturbance based on 4km disturbance range | 81.5 (0.13% of CIS MU) | 163.0 (0.3% of CIS MU) |

*Table 9.11 Maximum number of harbour porpoise (and % of CIS MU) that could be disturbed as a result of underwater noise associated with construction vessels (taken from Table 11.52 of the ES)*

| Potential impact | Maximum number of individuals (% of reference population) for one vessel (50.27km$^2$) | Maximum number of individuals (% of reference population) for revised site and 4km buffer (285.4km$^2$) |
|---|---|---|
| Vessel disturbance based on 4km disturbance range | 81.5 (0.13% of CIS MU) | 462.6 (0.74% of CIS MU) |

**Operation and maintenance**

3343. Underwater noise and disturbance during operation could result from operational noise from WTGs, maintenance work (such as rock placement or cable repairs) and vessels. Each of these sources has been considered separately in detail in Sections 11.6.4.1, 11.6.4.2 and 11.6.4.3 of **Chapter 11 Marine Mammals** of the ES.

3344. A review of most recent research has been used to determine the potential disturbance of harbour porpoise from underwater operational noise from WTGs (see Section 11.6.4.1 of **Chapter 11 Marine Mammals** of the ES). The studies indicated that any disturbance would be in the immediate area of the operational turbine, depending on ambient noise levels. There was no evidence of any lasting disturbance or exclusion of harbour porpoise around OWFs during operation, with reports of harbour porpoise moving through and foraging within operational OWFs.

3345. Therefore, there was no indication that there would be a LSE on the harbour porpoise CIS MU population from the effects of disturbance impacts from underwater noise of operational WTGs.

3346. As a precautionary approach, a 4km impact range has also been used as a potential disturbance range for maintenance activities, including vessels undertaking the work, based on construction activities (see Section 11.6.3.3

of **Chapter 11 Marine Mammals** of the ES). The potential disturbance from cable repairs and rock placement occurring at the same time has been assessed based on maximum impact area of 100.53km$^2$ (**Table 9.12**). The maximum number of harbour porpoise that could be disturbed was up to 163 (0.3% of CIS MU).

3347. The impact area that has been assessed for 37 vessels (285.4km$^2$) during construction presented the worst-case also for operation and maintenance activities and was therefore not assessed again (see Section 11.6.4.3 in **Chapter 11 Marine Mammals** of the ES).

3348. Based on a standard year of maintenance, it was expected that up to three vessels could be on site at any given time. As such an assessment of the number of animals potentially disturbed by three vessels (150.81km$^2$) The maximum number of harbour porpoise that could be disturbed was up to 244.5 (0.39% of CIS MU).

*Table 9.12 Maximum number of harbour porpoise (and % of CIS MU) that could be disturbed as a result of underwater noise associated with maintenance activities, including vessel undertaking the work (taken from Table 11.68 of the ES)*

| Potential impact | Maximum number of individuals (% of reference population) that could be disturbed for two activities (100.53km$^2$) |
|---|---|
| Disturbance based on 4km disturbance range | 163 (0.3% of CIS MU) |

*Table 9.13 Maximum number of harbour porpoise (and % of CIS MU) that could be disturbed as a result of underwater noise associated vessels during operation and maintenance (taken from Table 11.72 of the ES)*

| Potential Impact | Maximum number of individuals (% of reference population) for up to 3 vessels (150.81km$^2$) |
|---|---|
| Disturbance based on three vessels | 244.5 (0.39% of CIS MU) |

3349. Given that this effect would be lower than for disturbance impacts from underwater noise during piling and the effect would occur outside of any SAC, **it has been concluded that there would be no LSE on the harbour porpoise CIS MU population (and no AEoI on any SAC) from disturbance impacts from underwater noise from maintenance activities and vessels during operation and maintenance.**

### Decommissioning

3350. Potential effects on harbour porpoise associated with underwater noise during decommissioning have not been assessed in detail. This was because further assessments would be carried out ahead of any decommissioning works being undertaken. These assessments would take account of known information at that time, including relevant guidelines and requirements. The detailed Decommissioning Programme would provide details of the techniques to be employed and any relevant mitigation measures required.

3351. It is not possible to provide details of the methods that could be used during decommissioning at this time. However, it is expected that the activity levels would be comparable to construction (with the exception of pile driving noise which would not occur).

3352. During decommissioning, the potential effects on harbour porpoise were anticipated to be similar or less than the worst-case for the construction phase (depending on the methods used). The effects would, therefore, be comparable to those described in construction.

3353. Given that this effect would be lower than for disturbance impacts from underwater noise during piling and the effect would occur outside of any SAC, **it has been concluded that there would be no LSE on the harbour porpoise CIS MU population (and no AEoI on any SAC) from the effects of disturbance impacts from underwater noise during decommissioning.**

#### 9.4.2.2 Barrier effects as a result of underwater noise

##### Construction

3354. Underwater noise during construction could have the potential to create a barrier effect, preventing movement of harbour porpoise between important feeding and/or breeding areas, or potentially increase swimming distances if harbour porpoise avoid the area and go around it.

3355. As outlined in Section 11.6.3.5 of **Chapter 11 Marine Mammals** of the ES, the Project windfarm site itself was not considered to be of particular importance to harbour porpoise as reflected in several modelling studies (such as by Heinänen and Skov, 2015) which did not predict areas of high harbour porpoise density in or around the windfarm site. The site-specific surveys however (**Appendix 11.2**) showed otherwise, as high numbers of harbour porpoises were recorded utilising the area throughout the year.

3356. Any temporary barrier effects as a result of underwater noise at the Project windfarm site would be unlikely to restrict harbour porpoise accessing foraging areas. The two-year monthly aerial surveys reported an increased number of harbour porpoise at the site. However, it is important to note that these animals exhibit a broad range of prey preferences and extensive foraging ranges. Consequently, the higher observed numbers at the Project site should not be interpreted as inferring an exclusive or restrictive feeding ground, as harbour porpoise have been known to maintain flexibility in utilizing various foraging areas beyond the Project site.

3357. The Project windfarm site was not located on any known migration routes for harbour porpoise. They may migrate outside the wider project area, with potential routes from northern UK to destinations like Iceland (Andersen, 2003, Figure 1). It was noteworthy that porpoise from the CIS sub-population displayed seasonal movements towards the northwest of Scotland, as documented by Gaskin (1984).

3358. The Project windfarm site would be located approximately 30km from the nearest point on the coast. The maximum potential impact range during piling at the Project windfarm site would be from disturbance effects (26km using the EDR approach), there would therefore be no potential for any barrier effects between the Project windfarm site and the coast as a result of underwater noise during piling.

3359. Underwater noise from piling would be for a maximum of approximately 37 days (assuming 24hr days) in total over the construction period of up to two and a half years (with foundation installation expected over 9 – 12 months). Other construction activities and vessels that could result in barrier effects would be temporary, not consistent throughout the offshore construction period, and would be limited to only part of the overall construction period and area at any one time. If there were potential barrier effects across the entire Project windfarm site (87km$^2$) this would be a small area in relation to the movements and foraging ranges of harbour porpoise in and around the IS.

3360. There is unlikely to be any significant long-term impact from any temporary barrier effects due to underwater noise, any areas affected would be relatively small in comparison to the range of marine mammals and any effects would not be continuous throughout the offshore construction period. The effect would occur outside of any SAC. **It has therefore been concluded that there would be no LSE on the harbour porpoise CIS MU population (and no AEoI on any SAC) from barrier effects during construction.**

### Operation and maintenance and decommissioning

3361. No barrier effects as a result of underwater noise during operation and maintenance were anticipated. As outlined above, no significant disturbance effects from underwater noise would be anticipated during operation and maintenance.

3362. Any behavioural responses or disturbance would be limited to the close vicinity of the operational WTG. The minimum spacing between WTGs means there would be no potential for underwater noise around individual WTGs to overlap. Taking into account the relatively small impact areas for underwater noise around operational WTGs, there was unlikely to be the potential for barrier effects to marine mammals as a result of operational noise.

3363. During decommissioning, the potential effects on harbour porpoise were anticipated to be similar or less than the worst-case for the construction phase (depending on the methods used). The effects would therefore be comparable to those described in construction.

3364. This effect would be lower than for disturbance impacts from underwater noise during construction, the effect would occur outside of any SAC and the Project windfarm site would not be located on any known migration routes for harbour porpoise. **It has therefore been concluded that there would be no LSE on the harbour porpoise CIS MU population (and no AEoI on any SAC) from disturbance effects of underwater noise during operation and maintenance or decommissioning.**

#### 9.4.2.3 Vessel interactions

#### Construction

3365. During the construction phase, there would be an increase in the number of vessels in the windfarm site. The maximum number of vessels that may be on the Project windfarm site at any one time was estimated to be up to a total of 37 vessels. The number, type and size of vessels would vary depending on the activities taking place at any one time. This effect has been considered in detail in Section 11.6.3.6 of **Chapter 11 Marine Mammals** of the ES, the assessment below summarises the information presented there.

3366. It was estimated that approximately eight individuals (0.012% of the CIS MU) could be at increased risk of collision during construction per year (see Table 11.56 of **Chapter 11 Marine Mammals** of the ES). This would be a permanent effect and in the worst-case assumed to be lethal for the individuals.

3367. It was considered that the quantified assessment was highly precautionary. Marine mammals are able to detect and avoid vessels. However, vessel strikes have been known to occur. This was possibly due to distraction whilst foraging and socially interacting, or due to the marine mammals' inquisitive nature (Wilson *et al*., 2007). Therefore, increased vessel movements, especially those outwith recognised vessel routes, can pose an increased risk of vessel collision to marine mammals. Harbour porpoise are small and highly mobile, and, given their responses to vessel noise (e.g. Thomsen *et al*., 2006; Polacheck and Thorpe, 1990), would be expected to largely avoid vessel collisions. Modelling by Heinänen and Skov (2015) indicated a negative relationship between the number of ships and the distribution of harbour porpoise in the Irish and Celtic Seas during summer, suggesting that the species could exhibit avoidance behaviour which reduced the risk of collision risk with vessels.

3368. It was therefore considered unlikely that up to eight harbour porpoise could be at increased collision risk with vessels during construction, considering the existing number of vessel movements in the area, and that vessels within the Project windfarm site would be stationary for much of the time or very slow moving. In addition, taking into account the disturbance effect from vessels, the actual risk was likely to be very low.

3369. As outlined in **Section 9.3.1** the commitment to mitigation measures would further reduce the potential risk of collision. Where possible, vessels would follow set routes and hence areas where marine mammals were accustomed to vessels, in order to reduce any increased collision risk. Predictability of vessel movement by marine mammals has been known to be a key aspect in minimising the potential risks imposed by vessel traffic (Nowacek *et al.,* 2001, Lusseau, 2003, 2006). Vessels travelling at high speeds were considered to be more likely to collide with marine mammals, and those travelling at speeds below 10 knots would rarely cause any serious injury (Laist *et al*., 2001). All vessel movements would be kept to the minimum that was required to develop the Project, to reduce any potential collision risk. Additionally, vessel operators would use good practice (as suggested in the Outline PEMP (Document Reference 6.2)) to reduce any risk of collisions with marine mammals.

3370. The mitigation measures to manage collision risk would be agreed with the relevant stakeholders and would be detailed within the PEMP.

3371. Given the relatively low actual risk to harbour porpoise and the commitment to mitigation measures to reduce that risk further, **it is concluded that there would be no LSE on the harbour porpoise CIS MU population (and no AEoI on any SAC) from the effects of vessel interactions during construction.**

### Operation and maintenance and decommissioning

3372. The increased risk of collision with vessels during operation and maintenance would be less than assessed for the construction period. During the operation and maintenance phase, the maximum number of vessels that could be on the windfarm site at any one time has been estimated at up to a total of ten vessels, Project whereas a standard year would only have three (**Table 9.4**). The number, type and size of vessels would vary depending on the activities taking place at any one time. The vessels in the Project windfarm site during operation and maintenance would be slow moving or stationary.

3373. Section 11.6.4.6 of **Chapter 11 Marine Mammals** of the ES assessed the potential for increased collision with vessels during operation and maintenance and concluded that approximately two individuals (0.004% of the CIS MU) could be at increased risk per year (see Table 11.74 of **Chapter 11 Marine Mammals** of the ES).

3374. During decommissioning, the potential effects on harbour porpoise are anticipated to be similar or less than the worst-case for the construction phase. The effects would therefore be comparable to those described in construction.

3375. Given that this effect is lower than for construction and the commitment to mitigation measures to reduce that risk further, **it is concluded that there would be no LSE on the CIS MU harbour porpoise population (and no AEoI on any SAC) from the effects of vessel interactions during operation or decommissioning.**

3376. Assessments were made on a standard maintenance year, but, given the low values, **it was anticipated that there would also be no LSE on the reference population (and no AEoI on the SAC) during a heavy maintenance year.**

#### 9.4.2.4 Changes to prey resources

##### Construction

3377. The potential effects on prey species during construction can result from physical disturbance and loss of habitat; increased SSC and sediment deposition; and underwater noise. **Chapter 10 Fish and Shellfish Ecology** of the ES, provides an assessment of these impact pathways on the relevant fish and shellfish species and concluded impacts of negligible to minor adverse significance in EIA terms. **Chapter 11 Marine Mammals** of the ES considered these effects in terms of potential indirect effects on harbour porpoise (see Section 11.6.3.7 Changes to Prey Resources).

3378. The diet of the harbour porpoise consists of a wide variety of prey species and varies geographically and seasonally, reflecting changes in available food resources. Harbour porpoise have relatively high daily energy demands and need to capture enough prey to meet their daily energy requirements. It has been estimated that, depending on the conditions, harbour porpoise can rely on stored energy (primarily blubber) for three to five days, depending on body condition (Kastelein *et al*., 1997).

3379. However, any reductions in prey availability would be small scale, localised and temporary (intermittent effects over the 2.5 year construction period). It was considered highly unlikely therefore that potential reductions in prey availability as a result of construction activities would result in detectable changes to harbour porpoise population.

3380. It is also important to note that there is unlikely to be any additional displacement of harbour porpoise as a result of any changes in prey availability during piling as they would already be disturbed from the area.

3381. Given that this effect would be limited and would occur outside of any area considered important for harbour porpoise foraging (i.e. outside of SACs), **it has been concluded that there would be no LSE on the harbour porpoise CIS MU population (and no AEoI on any SAC) from the effects of changes to prey resources during construction.**

### Operation and maintenance and decommissioning

3382. Changes to prey resource during operation and maintenance have been assessed in Section 11.6.4.7 of **Chapter 11 Marine Mammals** of the ES. As for construction, this assessment has been based upon the conclusions of **Chapter 10 Fish and Shellfish Ecology** of the ES and considered a range of potential impacts, including permanent habitat loss, introduction of hard substrate and EMF as well as the impacts considered for construction. Although new impacts have been considered for operation and maintenance, some effects such as physical disturbance; increased SSC and sediment deposition; and underwater noise would be reduced when compared to construction. Therefore, it is considered highly unlikely that potential reductions in prey availability as a result of operational activities would result in detectable changes to harbour porpoise population.

3383. During decommissioning, the potential effects on harbour porpoise are anticipated to be similar or less than the worst-case for the construction phase. The effects would therefore be comparable to those described in construction.

3384. Given that this effect would be limited and would occur outside of any area considered important for harbour porpoise foraging (i.e. outside of SACs), **it has been concluded that there would be no LSE on the harbour porpoise CIS MU population (and no AEoI on any SAC) from the effects of changes to prey resources during operation and maintenance or decommissioning.**

## 9.4.2.5 Changes to water quality

### Construction

3385. The disturbance of seabed sediments has the potential to increase SSCs and release any sediment-bound contaminants (such as heavy metals and hydrocarbons that may be present within them) into the water column. The accidental release of contaminants (e.g. through spillage) also has the potential to affect water quality. Section 11.6.3.8 of **Chapter 11 Marine Mammals** of the ES considers these effects in detail.

3386. Throughout the construction phase, best practice techniques and due diligence regarding the potential for pollution would be followed throughout all construction activities. Any risk of accidental release of contaminants (e.g. through spillage) would be mitigated in line with the PEMP and any changes to water quality as a result of any accidental release of contaminants (e.g. through spillage or vessel collision) would be negligible. Therefore, the potential for pollutants to be released into the environment has not been considered further in this assessment.

3387. Section 11.6.3.8 of **Chapter 11 Marine Mammals** of the ES considers increases in SSCs and remobilisation of existing contaminated sediments. With regard to increases in suspended sediment, harbour porpoise often inhabit turbid environments and utilise sonar to sense the environment around them and there was little evidence that turbidity affects harbour porpoise directly (Todd *et al*., 2014). As such, any increases in SSC would be unlikely to have a direct effect on harbour porpoise.

3388. As outlined in **Chapter 8 Marine Sediment and Water Quality** of the ES, site specific data indicated that for all potential contaminants tested for within the sediments of the windfarm site, concentrations were negligible. There would be therefore no potential for any direct or indirect effects on marine mammals from remobilisation of contaminated sediments.

3389. Given the distance of the Project windfarm site from the closest SAC (49km), there is no pathway for water quality effects directly upon harbour porpoise within any SAC considered in this assessment.

3390. Given that water quality effects would be negligible, **it has been concluded that there would be no LSE on the harbour porpoise CIS MU population (and no AEoI on any SAC) during construction.**

### Operation and maintenance and decommissioning

3391. During the operation and maintenance phase, there would be potential for increases in SSCs and release of any sediment-bound contaminants. The scale of these impacts would be small, infrequent, of short-term duration, and of a lower magnitude than during the construction phase.

3392. During decommissioning, the potential water quality effects are anticipated to be similar or less than the worst-case for the construction phase. The effects would therefore be comparable to those described in construction.

3393. Given that water quality effects would be negligible, **it has been concluded that there would be no LSE on the harbour porpoise CIS MU population (and no AEoI on any SAC) during operation, maintenance or decommissioning.**

## 9.4.2.6 Potential interactions of Project effects

3394. The effects identified and assessed in this section have the potential to interact with each other. The effects of the Project were:

- PTS from underwater noise
- Disturbance from underwater noise
- Barrier effects
- Vessel interactions
- Changes to prey resources
- Changes to water quality

3395. There would be no interactions between the effects of vessel interaction (i.e collision risk), changes to prey resources, water quality and barrier effects. However, the potential combined effects of disturbance from piling, other construction activities and vessels at the Project may cause an additive disturbance pathway. This has been further discussed in Section 11.10 in **Chapter 11 Marine Mammals** of the ES). Additional mitigation measures as outlined in the PEMP would also reduce the potential for collision risk (see **Table 9.3**).

3396. The anticipated effects on marine mammal receptors were not expected to interact in a way that would lead to a combined effect of greater significance than the assessments presented for each individual phase. It should also be noted that a high level of precautionary measures were implemented in the assessment process, further contributing to the overall understanding and mitigation of potential impacts.

#### 9.4.2.7 Summary of Project-alone conclusions

3397. There would be no overlap of permanent and temporary noise impact ranges within any SAC.

3398. Due to embedded mitigation and commitment to securing mitigation measures (i.e PTS mitigation through the MMMP and to manage the residual low collision risk through best practice vessel practices secured in the PEMP) it is considered that permanent effects upon harbour porpoise would be avoided during construction, operation and maintenance or decommissioning.

3399. Disturbance of harbour porpoise outside any SAC potentially caused by underwater noise and vessel interactions would affect less than 5% of the population.

3400. **It has been concluded that there would be no LSE on the harbour porpoise CIS MU population during construction, operation and maintenance or decommissioning. In addition, any effects would occur outside any SAC boundary.**

3401. Indirect effects (i.e. on water quality or prey resources) would occur outside any SAC boundary, and were considered to be not significant.

3402. None of the assessed effects were within an SAC or were considered to have a LSE on the harbour porpoise CIS MU population during construction, operation and maintenance or decommissioning. As such**, it is concluded that there would be no adverse effect on integrity of the North Anglesey Marine SAC, North Channel SAC, West Wales Marine SAC, Rockabill to Dalkey Island SAC or the Bristol Channel Approaches SAC in relation to the conservation objectives 'The species is a viable component of the site' or 'There is no significant disturbance of the species'.**

3403. Indirect effects (i.e. on water quality or prey resources) were considered to be insignificant and would occur outside any SAC boundary. **As such, it is concluded that there would be no adverse effect on integrity of the North Anglesey Marine SAC, North Channel SAC, West Wales Marine SAC, Rockabill to Dalkey Island SAC or the Bristol Channel Approaches SAC in relation to the conservation objective 'The condition of supporting habitats and processes, and the availability of prey is maintained'.**

3404. The confidence in the assessment for all impacts was considered high considering the baseline information and site-specific data.

### 9.4.3 Potential in-combination effects of the Project with Transmission Assets

3405. A 'combined' assessment has been made with the Transmission Assets[27], for the purpose of an in-combination assessment considering its functional link with the Project.

3406. Due to the ZOI the North Anglesey Marine SAC, North Channel SAC, and Bristol Channel Approaches SAC were screened in for both the Project and the Transmission Assets and West Wales Marine SAC and Rockabill to Dalkey Island SAC were screened in for the Project.

3407. For the Transmission Assets ISAA Project-alone assessment (Morgan Offshore Wind Limited and Morecambe Offshore Windfarm Ltd, 2023b), there would be no adverse effect on the site integrity on any of the screened-in sites, including those listed only for the Project. As for the Project, the distance to the closest SACs was outside the ZoI. A full quantitative assessment has been provided in the assessment of all plans and projects, including the Transmission Assets and is not repeated here. An assessment has been made below of each impact considering the information in **Section 9.4.4.1** and understanding the interactions between the projects.

[27] As the Transmission Assets includes infrastructure associated with both the Project and the Morgan Offshore Wind Project Generation Assets, it should be noted that the combined assessment considers the transmission infrastructure for both the Project and the Morgan Offshore Wind Project Generation Assets.

#### 9.4.3.1 Underwater noise and barrier effects

3408. The key interaction was identified as piling and UXO clearance during construction of the projects.

3409. Given that the Project and Transmission Asset would be outwith any SAC and as potential PTS effects would be mitigated by any consented project, **it is concluded that there would be no LSE on the reference population (and no AEoI on the SAC).**

#### 9.4.3.2 Vessel interactions

3410. During all phases, there would be additional effects due to increased vessel presence from both projects.

3411. Given that the Project and Transmission Assets would be outwith any SAC and both projects would adhere to good practice, **it has been concluded that there would be no LSE on the reference population (and no AEoI on the SAC).**

#### 9.4.3.3 Indirect effects (changes to prey resource and water quality)

3412. During all phases, there would be additional effects due to increased vessel presence from both projects and additional pressure on prey resource.

3413. Given the impacts identified for both projects on prey species and that the Project and Transmission Assets would be outwith any SAC and both projects would adhere to good practice, **it is concluded that there would be no significant in-combination effect on the SAC reference populations (and no AEoI).**

### 9.4.4 Assessment of the potential effects of the Project in-combination with other plans and projects

3414. Section 11.7 of **Chapter 11 Marine Mammals** of the ES details the CEA. This in-combination assessment has been based upon the cumulative assessment and provided a summary of the key information from that assessment without repeating every step of the process. Key information has been taken from **Chapter 11 Marine Mammals** of the ES and carried through with regard to the effect on designated sites.

3415. The effects screened into the in-combination assessment and the identification of the other plans, projects and activities that may result in in-combination effects have been provided in **Appendix 11.4 Marine Mammal CEA Project Screening** (Document Reference 5.2.11.4).

#### 9.4.4.1 Underwater noise

**Permanent auditory injury from underwater noise**

3416. PTS could occur as a result of piling during OWF installation or detonation of underwater explosives (used occasionally during the removal of underwater structures and UXO clearance) (JNCC, 2010a,b[28]). However, if there were the potential for any PTS, from any project, suitable mitigation would be put in place to reduce any risk to marine mammals. Other activities such as dredging, drilling, rock placement, vessel activity, operational windfarms, oil and gas installations or wave and tidal sites would emit broadband noise in lower frequencies and PTS from these activities would be very unlikely.

3417. Given that the Project would be outwith any SAC there was no potential for AEoI from PTS onset in-combination with other projects, as all projects should ensure mitigation is in place to negate the potential for PTS. **Therefore, the potential for PTS in-combination has been screened out and not assessed further.**

**Disturbance from underwater noise during construction**

3418. Section 11.7.3.1 of **Chapter 11 Marine Mammals** of the ES considers disturbance in relation to several sub-effects and then considers them all together: underwater noise impacts from piling at other OWFs; underwater noise impacts from construction activities (other than piling) at other OWFs; and disturbance from other industries and activities (which included geophysical survey, seismic survey and UXO clearance). The combined results from these assessments have been summarised in **Table 9.17**. Where a quantitative assessment has been possible, the potential magnitude of disturbance at other projects has been based on the publicly available project-specific density estimates or numbers of animals impacted. Details can be found in **Appendix 11.4**.

3419. Where there was no project specific information a speculative (or indicative) assessment for a potential activity has been undertaken the results of potential disturbance were only indicative. These assessments were highly conservative and not based on any project specific information such as densities or impact ranges and have been quantified using known disturbance ranges. As such, the assessment for disturbance from underwater noise would be based on the outcome of the population modelling which takes into account projects specific effects and was deemed the most accurate.

[28] DRAFT guidelines for minimising the risk of injury to marine mammals from UXO clearance in the marine environment (JNCC, 2023b) were issued for consultation in 2023. It is anticipated that the publication of the guidelines will occur after submission of this DCO Application.

**Disturbance from piling**

3420. The potential disturbance to harbour porpoise from underwater noise during piling has been assessed based on the 26km EDR for harbour porpoise at each OWF (2,123.7km$^2$), as a worst-case scenario (**Table 9.14**). This assessment considered the effect of all projects at a population level, noting the Project would not overlap with any SAC.

3421. The UK and European OWFs screened in for having a construction period that could potentially be piling at the same time as the Project were (see **Appendix 11.4**):

- AyM OWF (PINS Tier 1)
- Mona OWF (PINS Tier 2)
- Morgan OWF (PINS Tier 2)
- Morgan and Morecambe Offshore Wind Farms Transmission Assets (PINS Tier 2)
- Erebus OWF (PINS Tier 1)
- White Cross (PINS Tier 1)

3422. This short list of OWF projects that could be piling at the same time as the Project could change as projects develop, but this was the best available information at the time of writing, and was considered to reflect the limitations and constraints to project delivery.

3423. The following caveats should be noted in terms of this worst-case assessment:

- The potential areas of disturbance assumed that there would be no overlap in the areas of disturbance between different projects.
- It was assumed that all OWF projects would be 100% piled, if piled foundations were an option.
- The approach was based on the potential for single piling at each OWF at the same time as single piling at the windfarm site. This approach allowed for some of the OWFs not to be piling at the same time, while others could be simultaneously piling. This was considered to be the most realistic worst-case scenario, as it was highly unlikely that all other OWFs would be simultaneously piling at exactly the same time as piling at the Project, especially given the limited active piling time.
- The actual duration for active piling time for the Project (a maximum of 619 hours and 36 minutes hours including soft-start, ramp-up and ADD activation (using pin-piles for OSP and WTG)), which could disturb marine mammals is only a very small proportion of the potential construction period, and this would be the case for other OWFs. This means that there would be a limited window for temporal overlap and any in-combination effect to occur.

- In practice, the potential temporary effects would be less than those predicted in this assessment as there is likely to be a great deal of variation in timing, duration (noting this has been typically overestimated in assessments) and hammer energies used throughout the various OWF construction periods. In addition, not all individuals would be displaced over the entire potential disturbance range (26km) used within the assessments. For example, a study of harbour porpoise at Horns Rev II (Brandt *et al.,* 2011), indicated that at closer distances (2.5 to 4.8km) there was 100% avoidance, however, this proportion decreased significantly moving away from the pile driving activity and at distances of 10km to 18km avoidance was 32% to 49%. At 21km, the abundance was reduced by just 2%

3424. It is also important to note that the harbour porpoise density used in the assessment for the Project (1.621 per $km^2$) was from a two year site-specific survey which has been skewed by two months of very high numbers. The resulting density used was much higher than would be expected from the use of Gilles *et al.,* (2023) (0.5153 per $km^2$).

3425. If all projects that overlap with the piling window of the Project were to apply the use of a 26km EDR, it would present an unrealistic and overly precautious assessment as it would assume that piling at all projects would happen simultaneously and would not take into consideration a possible overlap of disturbance areas for some of the projects.

3426. Instead, the total number of harbour porpoise disturbed within the 26km EDR informed the more realistic population modelling, using the iPCoD model. This model took into account the worst-case disturbed numbers of animals from each project and was deemed the most accurate. The results were based on assessments in Section 11.7.3.2 in **Chapter 11 Marine Mammals** of the ES (detailed information to iPCoD see **Appendix 11.3 Marine Mammal UXO Assessment** (Document Reference 5.2.11.3)).

3427. The median population size was predicted to be 100% of the un-impacted population size at the end of 2028 (one year after the piling was expected to commence) (**Table 9.14** and **Plate 9.2**). By the end of 2028 (the year that piling was expected to end), the median population size for the impacted population was predicted to be 99.78% of the un-impacted population size. Beyond 2028, the impacted population was expected to maintain the same stable trajectory as the un-impacted population (as far as 2052 which was the end point of the modelling, at which point the median impacted to un-impacted ratio was 99.26%).

3428. For harbour porpoise, the analysis showed no significant risk to populations due to there being less than a 1% population level effect on average per year over both the first six years and 25 year modelled periods.

*Table 9.14 Quantified in-combination assessment for the potential disturbance of harbour porpoise during piling at OWFs including the Project (taken from Table 11.86 of the ES)*

| Year | Un-impacted population mean | Impacted population mean | Median impacted as % of un-impacted |
|---|---|---|---|
| Start | 62,516 | 62,516 | 100.00% |
| End 2028 | 62,574 | 62,569 | 100.00% |
| End 2029 | 62,509 | 62,278 | 99.78% |
| End 2032 | 62,389 | 61,703 | 99.22% |
| End 2037 | 62,482 | 61,818 | 99.26% |
| End 2047 | 62,436 | 61,770 | 99.27% |
| End 2052 | 62,564 | 61,897 | 99.26% |

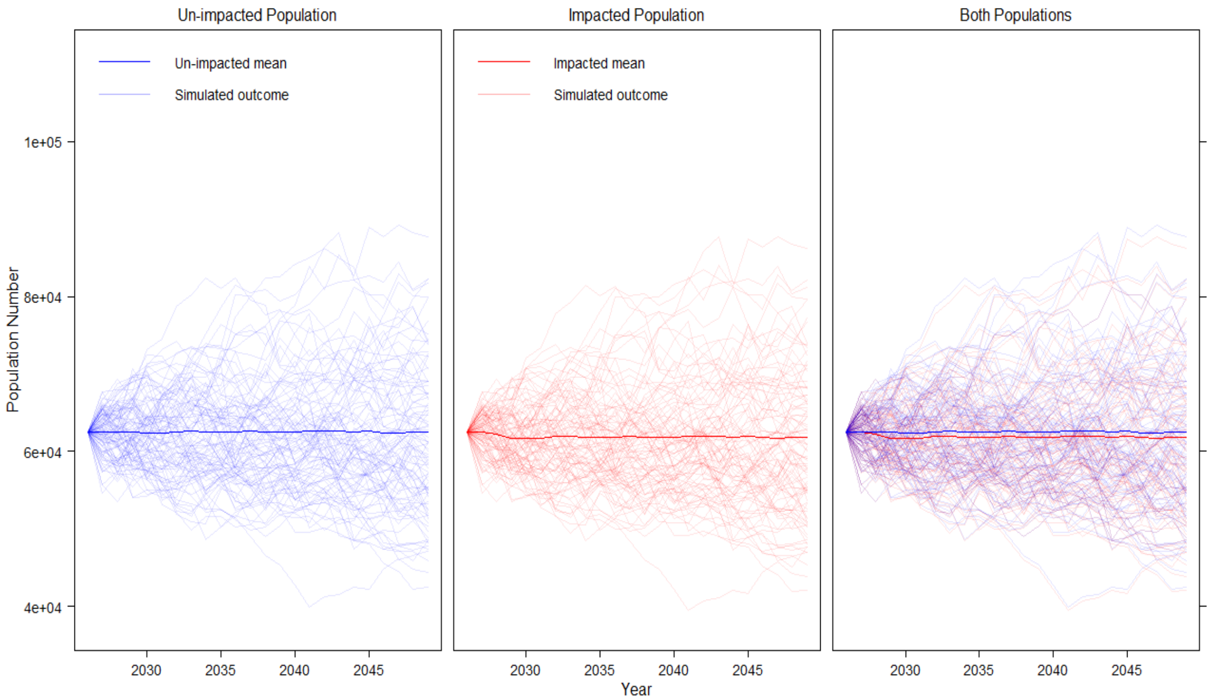


*Plate 9.2 Simulated worst-case harbour porpoise population sizes for both the un-impacted and the impacted populations for the in-combination for the potential disturbance of harbour porpoise during piling at OWFs including the Project (scientific notation used in these charts, e.g. 4e+04 = 40,000).*

3429. It should be noted that the Project has already committed to no concurrent Project piling as embedded mitigation as the assessment suggested that no LSE was expected, no additional mitigation has been proposed.

**Underwater noise impacts from construction activities (other than piling)**

3430. OWFs screened in for other construction activities that could have potential in-combination effects with other construction activities at the Project (see **Appendix 11.4** of the ES) were:

- Codling Wind Park (PINS Tier 2)
- Dublin Array (PINS Tier 2)
- North IS Array (PINS Tier 2)
- Sceirde Rocks (PINS Tier 2)

3431. During the construction of the Project, there would be the potential for overlap with impacts from the non-piling construction activities at other offshore wind farms. Although, it is noted that these were all Tier 2 projects and the certainty on scheduling, and thus temporal overlap, was low. Noise sources that could cause potential disturbance impacts during OWF construction activities (other than pile driving) can include vessels, mooring installation, seabed preparation, cable installation works and rock placement.

3432. The potential impact area for harbour porpoise has been based on the worst-case disturbance range of 4km (50.27km$^2$), which included construction activity and vessels.

3433. The in-combination disturbance effect of other construction activities for all OWFs including the Project was up to 3,589.2 individuals, which represented 5.7% of the CIS MU (**Table 9.15**).

*Table 9.15 Quantified in-combination assessment for the potential disturbance of harbour porpoise during construction activities, including vessels, other than piling OWFs including the Project*

| OWF | Harbour porpoise density (/km$^2$) | Impact area (km$^2$) | Maximum number of individuals potentially disturbed |
|---|---|---|---|
| Codling | 0.942 | 50.27 | 47.4 |
| Dublin Array | 0.942 | 50.27 | 47.4 |
| North IS Array | 0.942 | 50.27 | 47.4 |
| Sceirde Rocks | 0.092 | 50.27 | 4.6 |
| Total number of harbour porpoise | | | *146.7* |
| Percentage of CIS MU | | | *0.2%* |

**Disturbance from other industries and activities**

3434. Section 11.7.3 of **Chapter 11 Marine Mammals** of the ES considers the effects from geophysical surveys; aggregate extraction and dredging; seismic surveys and UXO clearance. This section quantifies the potential impacts from other industries and activities, with the assessment of effects alongside the Project given in **Table 9.16**.

3435. For geophysical survey, as a worst-case it has been assumed there would be the potential for disturbance from two geophysical surveys, based on 5km EDR (JNCC *et al*., 2020). A review of seismic surveys within the UK indicated that surveys were being undertaken for approximately 52% of the time (Business Energy and Industrial Strategy (BEIS[29]), 2020). This data has been applied to geophysical surveys due to their similarity in approach. Taking this into account, up to 103.5km of surveys could be undertaken in one day.

3436. This could result in the disturbance of up to 614 harbour porpoise (0.99% of the CIS MU), based on total area of 1192.1km$^2$ for the two geophysical surveys, including turning area.

3437. For aggregate extraction and dredging, two aggregate/ dredging projects have been screened in that could have potential cumulative disturbance impacts with piling at the Project:

- North Bristol Deep 1601
- North Bristol Deep 1602

3438. Studies have indicated that harbour porpoise may be displaced by dredging operations within 600m of the activities (Diederichs *et al*., 2010; Todd *et al*., 2014). Therefore, disturbance would be up to 2.26km$^2$ for two aggregate projects. This could result in approximately 0.04 harbour porpoise (0.0001% of the CIS MU) being disturbed.

3439. For seismic surveys, the worst-case assumed disturbance from a single survey, with a potential impact area based on 12km EDR, following the most recent SNCB guidance (JNCC *et al*., 2020). Seismic surveys are a moving source, travelling up to 199km in one day, of which 52% (103.5km) is actual survey time. The total impact area for harbour porpoise used for the assessment was 1694.39km$^2$ for one seismic survey. This could result in approximately 872.6 harbour porpoise (1.4% of the CIS MU) being disturbed. However, it was noted that there were no known licence or licence applications for seismic surveys at the time of assessment, that could overlap with

[29] As of February 2023, BEIS is known as the DESNZ

construction of the Project, and this has been included for information purposes at this stage.

3440. Mitigation measures required for UXO clearance include the use of low-order clearance techniques, which could include a small donor charge, rather than full high-order detonation. It was therefore highly unlikely that more than one UXO high-order detonation would occur at exactly the same time or as another UXO high-order detonation, even if they had overlapping UXO clearance operation durations. The assessment was therefore based on potential for disturbance from one high-order and one low-order UXO clearance without mitigation (worst-case).

3441. For harbour porpoise, the potential impact area of 2,123.7km$^2$ was based on a 26km EDR for UXO high order detonation, and 78.5km$^2$ for low-order detonation, following the current SNCB guidance for the assessment of impact to harbour porpoise in the Southern North Sea SAC following the current JNCC (2023b) guidance. This could result in approximately 1,134.2 harbour porpoise (1.8% of the CIS MU) being disturbed.

3442. As outlined in BEIS[30] (2020), due to the nature of the sound arising from the detonation of UXO (i.e. each blast lasting for a very short duration), marine mammals, including harbour porpoise, were not predicted to be significantly displaced from an area. Any changes in behaviour, if they occur, would be an instantaneous response and short-term. Existing guidance suggested that disturbance behaviour was not predicted to occur from UXO clearance if undertaken over a short period of time (JNCC, 2010b).

*Table 9.16 Quantified in-combination assessment for potential disturbance of harbour porpoise from other industries and activities (out with the Project)*

| Activity | Harbour porpoise density (/km$^2$) | Impact area (km$^2$) | Maximum number of individuals potentially disturbed | % CIS MU |
|---|---|---|---|---|
| Geophysical surveys x2 | 0.515 | 1,192.1 | 614 | 0.99 |
| Aggregate extraction and dredging x2 | 0.0157 | 2.26 | 0.04 | 0.0001 |
| Seismic x1 | 0.515 | 1,694.39 | 872.6 | 1.4 |

[30] As of February 2023, BEIS is known as the DESNZ

| Activity | Harbour porpoise density (/km$^2$) | Impact area (km$^2$) | Maximum number of individuals potentially disturbed | % CIS MU |
|---|---|---|---|---|
| UXO (two clearance events) | 0.515 | 2,202.2 | 1,134.2 | 1.8 |
| **Total** | | **3,045.75** | **2,620.84** | **4.19%** |

**Summary of disturbance effects during construction**

3443. For harbour porpoise, there were no effects predicted within any SAC. Considering a precautionary number of noise sources with the potential for in-combination disturbance effects, up to 3.69% of the harbour porpoise CIS MU population would be at risk of disturbance at any one time (see **Table 9.17**). This figure is the total of the in-combination effects (including the indicative assessment) that would result in disturbance (outside any SAC) and would affect less than 5% of the reference population. Modelling by Booth *et al.,* (2017) from the North Sea, and the presented modelling results above suggested that this level of effect would not lead to LSE upon the harbour porpoise population.

3444. There would be no predicted adverse effects on site integrity as:

- The effect of the Project has been based on the worst-case harbour porpoise density which was not representative of the wider area or the whole CIS MU. The resulting density used was much higher than would be expected from use of Gilles *et al.,* (2023) or neighbouring OWF projects.
- The assessment assumed the six in-combination projects (largely Tier 2 projects where scheduling was uncertain) would all be piling at exactly the same time as the Project and using monopiles. Experience from the build-out of projects in the Southern North Sea suggests that this is unrealistic. Piling durations are typically overestimations, further limiting potential for temporal overlap.
- There would be no spatial overlap of effects with any SAC (of any of the projects).
- Not all individuals would be displaced over the entire potential disturbance range used within the assessments.
- Behavioural effects from UXO clearance, if they occur, would be an instantaneous response and short-term. Guidance suggested that disturbance behaviour would not be predicted to occur from UXO clearance if undertaken over a short period of time (JNCC, 2010b)

*Table 9.17 Quantified in-combination assessment for the potential disturbance of harbour porpoise from all underwater noise sources during construction (Taken from Table 11.107 of the ES) (Grey rows are projects and activities that may take place and were therefore indicative assessments)*

| Impact | Maximum number of individuals potentially disturbed |
|---|---|
| Piling at other OWF including the worst-case disturbance from the Project | Based on iPCoD modelling, <1% of the population disturbed over the first six years and 25 year period modelled. |
| Construction activities at other OWF other than piling | 146.7 (0.2%) |
| Other activities[31] | 2,620.84 (4.19%) |
| **Total for all projects that were currently (or expected to be) in the planning process (realistic worst-case scenario)** | |
| **Total** | **<1% of the CIS MU and the SAC population** |

3445. Based on the current worst-case total, the in-combination assessment for underwater noise for all projects that were currently (or expected to be) undertaken at the same time as the Project was less than 5% of the reference and SAC population. **As such, this would suggest there would be no potential for LSE on the harbour porpoise CIS MU population (and no AEoI on any SAC) from disturbance during construction at the Project.**

### Disturbance from underwater noise during operation and maintenance

3446. Underwater noise and disturbance during operation could come from several sources, including operational noise from WTGs, noise from maintenance work (such as cable repairs) and from vessels as well as other OWFs, activities and industries.

3447. A review of relevant studies available at the time of assessment was used to determine the potential disturbance of harbour porpoise from underwater operational noise from WTGs (see Section 11.6.4 of **Chapter 11 Marine Mammals** of the ES). The studies indicated that any disturbance would be in the immediate area of the operational turbine, depending on ambient noise levels. There was no evidence of any lasting disturbance or exclusion of harbour porpoise around windfarm sites during operation, with reports of harbour porpoise moving through and foraging within operational windfarm sites. This was consistent with BEIS[32] (2020) that concluded, due to the low

[31] As listed in **Table 9.16**

[32] As of February 2023, BEIS is known as the DESNZ

noise levels associated with operational OWFs, there would be no potential for significant impact from the operation of OWFs.

3448. Therefore, there is no indication that there would be a LSE on the harbour porpoise CIS MU population from the effects of disturbance impacts from underwater noise of operational turbines that could contribute to in-combination effects.

3449. Effects from maintenance activities at OWFs, such as additional rock placement or cable re-burial, would be very localised, short in duration and temporary. The potential for in-combination effects from maintenance activities, including vessels at OWFs would be less than the in-combination effects assessed for construction activities other than piling.

3450. **Therefore, operational noise from OWF WTGs and maintenance activities, including vessels, was screened out from further consideration within the CEA screening and has not been considered further in this in-combination assessment.**

### Underwater noise from the decommissioning

3451. The potential for in-combination impacts during the decommissioning of the Project is currently unknown. It is not possible to provide details of the methods that would be used during decommissioning at this time. However, it is expected that the activity levels would be comparable or less to construction (with the exception of pile driving noise which would not occur which would be the main contributor to underwater noise effects).

3452. During decommissioning, the potential effects on harbour porpoise have been anticipated to be similar or less than the worst-case for the construction phase (depending on the methods used). Crucially, any in-combination effect would be dependent upon simultaneous decommissioning and it is not possible to provide a realistic estimate of which projects may be decommissioned and when.

3453. **The potential impacts from decommissioning have been screened out from further consideration within the CEA screening and have not been considered further in this in-combination assessment.**

### 9.4.4.2 Barrier effects

3454. Due to the low noise levels associated with operational OWFs, BEIS[33] (2020) concluded that there would be no potential for significant impact to harbour porpoise from the operation of OWFs. Effects from maintenance activities at

[33] As of February 2023, BEIS is known as the DESNZ

OWFs, such as additional rock placement or cable re-burial, would be very localised, short in duration and temporary.

3455. The assessment in **Section 9.4.2**, indicates that there would be no potential barrier effects from underwater noise or disturbance from the Project during construction, operation and maintenance, and decommissioning. If there were no LSE from disturbance, this would support the conclusion that underwater noise would likewise not create a barrier effect. Therefore, there would be no contribution to any in-combination effects as a result of underwater noise.

3456. The physical presence of a windfarm could be perceived as having the potential to create a physical barrier, preventing movement or migration of marine mammals between important feeding and/or breeding areas, or potentially increasing swimming distances if marine mammals avoid the site and go round it.

3457. As outlined in Section 11.6.4.5 of **Chapter 11 Marine Mammals** of the ES, the spacing between WTGs (**Table 9.4**) would allow marine mammals to move between WTGs and through the operational windfarm site. As outlined in Section 11.6.4.1 of **Chapter 11 Marine Mammals** of the ES, information from operational windfarms showed no evidence of exclusion of harbour porpoise.

3458. Based on the review of marine mammal presence within operational OWFs, the potential for any barrier effect due to the physical presence of the windfarm would be negligible. Therefore, there would be no contribution to any in-combination effects.

3459. During construction, barrier effects arising through loud noise would be intermittent and limited to just a fraction of the 2.5 year construction window. During operation, barrier effects arising through loud noise would be limited to the Project boundaries. The long distances between the Project and the SACs would allow for animals to move freely and not encounter barriers through physical turbine presence, nor from the generated noise.

3460. **Given the limited spatial effect, it was considered that there would be no potential for an in-combination barrier effect from underwater noise or physical presence and no LSE on the harbour porpoise CIS MU population (and no AEoI on any SAC) from the activities.**

#### 9.4.4.3 Vessel interactions

3461. Given the low risk of collision to harbour porpoise and the commitment to best practice measures to manage residual risk, it was concluded in **Chapter 11 Marine Mammals**, that there would be no LSE from the Project on the harbour porpoise CIS MU population from vessel interactions during construction, operation and maintenance or decommissioning. It was considered that any consented OWF project would require similar mitigation which would reduce collision risks.

3462. Vessels associated with aggregate extraction and dredging are large and typically slow moving, using established transit routes to and from ports. Therefore, the potential in-combination collision risk with vessels was considered to be extremely low or negligible.

3463. **Given the low risk to harbour porpoise and mitigation measures across OWF projects, it has been concluded that there would be no LSE on the harbour porpoise CIS MU population (and no AEoI on any SAC) from the effects of vessel interactions during construction, operation and maintenance or decommissioning.**

#### 9.4.4.4 Changes to prey resources

3464. No significant impacts with regard to changes to prey resources were expected as a result of the Project (see **Section 9.4.2**).

3465. For any potential changes to prey resources, it has been assumed that any potential effects on marine mammal prey species from underwater noise, including piling, would be the same or less than those for marine mammals. Therefore, there would be no additional in-combination effects above those assessed for harbour porpoise, i.e. if prey were disturbed from an area as a result of underwater noise, harbour porpoise would be disturbed from the same or greater area. As a result, any changes to prey resources would not affect harbour porpoise as they would already be disturbed from the area.

3466. Any effects to prey species were likely to be intermittent, temporary and highly localised, with potential for recovery following cessation of the disturbance activity. Any permanent loss or changes of prey habitat would typically represent a small percentage of the potential habitat for prey species in the surrounding area.

3467. Taking into account the assessment for the Project-alone, there would be no potential for a LSE on the harbour porpoise CIS MU population during construction, operation or decommissioning. This assessment has been made on the basis that there would be a wide range of prey species taken by harbour porpoise over the extent of their foraging ranges. Furthermore, much of the effect would occur outside of any area considered important for foraging (i.e. outside of SACs).

3468. **It has been concluded that there would be no LSE on the harbour porpoise CIS MU population (and no AEoI on any SAC) from the in-combination effect with other plans and projects on changes in prey species.**

#### 9.4.4.5 Changes to water quality

3469. No significant impacts with regard to water quality were expected as a result of the Project (see **Section 9.4.2**). Therefore, there would be no contribution to any in-combination effects.

3470. Aggregate and dredging projects have the potential for increased sediment suspension (and therefore, impacts to marine mammal species), however any changes to water quality as a result of aggregate extraction and dredging would be very localised and temporary. Other OWFs or other construction projects were also considered to have highly localised and temporary effects and would be spatially separated so there would be no potential for significant additive effects.

3471. **It has been concluded that considering the Project in-combination with other plans and projects there would be no LSE on the harbour porpoise CIS MU population (and no AEoI on any SAC).**

#### 9.4.4.6 Summary of in-combination assessment

3472. Effects from the Project would not overlap any SAC and the plans, projects and activities included within the in-combination assessment have been screened in on the basis that they were in the same MU, rather than overlap or proximity with any SAC. It has been noted in the plan level HRA (NIRAS, 2021) that only projects within 26km of an SAC were considered to contribute to disturbance effects and therefore no adverse effect on integrity was predicted for any harbour porpoise SAC at the plan level. Results are therefore highly precautionary, particularly considering the low likelihood of temporal overlap.

3473. Due to the mitigation outlined for all projects, it is considered that permanent effects upon harbour porpoise would be avoided.

3474. Modelling from the North Sea showed no significant risk to harbour porpoise populations considering a large number of developments (Booth *et al.,* 2017). Modelling for the Project showed that disturbance effects would only occur outside of SACs and impact less than 5% of the harbour porpoise CIS MU population during construction. The assessment is also considered over precautionary, particularly given the unlikely event of all activities having temporal overlap.

3475. It is considered that there would be no adverse effect on integrity of the North Anglesey Marine SAC, North Channel SAC, West Wales Marine SAC, Rockabill to Dalkey Island SAC or the Bristol Channel Approaches SAC in relation to the conservation objective ‘The species is a viable component of the site’.

3476. Indirect effects (i.e. on water quality or prey resources) were considered to be insignificant and would occur outside any SAC boundary. As such, it is considered that there would be no adverse effect on integrity of the North Anglesey Marine SAC, North Channel SAC, West Wales Marine SAC, Rockabill to Dalkey Island SAC or the Bristol Channel Approaches SAC in relation to the conservation objective 'The condition of supporting habitats and processes, and the availability of prey is maintained'.

3477. **The confidence in the assessment for all impacts is considered medium, yet highly precautionary, particularly given the consideration of a large number of plans or projects and the unlikelihood of temporal overlap of all these activities.**

## 9.5 Bottlenose dolphin

### 9.5.1 Relevant sites

#### 9.5.1.1 Pen Llŷn a`r Sarnau SAC

##### Description of designation

3478. The Pen Llŷn a'r Sarnau SAC encompasses areas of sea, coast and estuary and supports a significant presence of grey seal and bottlenose dolphin. The SAC is 1,460km$^2$ and covers areas including coastal lagoons, shallow inlets and bays, estuaries, reefs and sandbanks (NRW, 2018b).

3479. Bottlenose dolphins are considered of significant importance within Pen Llŷn a'r Sarnau SAC, even though they do not appear to form a semi-resident group within the sea area encompassed by this site. Bottlenose dolphins do not form a discrete site-based population within Pen Llŷn a'r Sarnau SAC but instead should be seen as part of a wider population, including the Cardigan Bay SAC (Sinclair *et al*., 2023).

3480. The SAC is 97km from the windfarm site, measured as a straight line distance, and 128km, measured as a coastline distance.

##### Bottlenose dolphin population and density

3481. In UK waters, bottlenose dolphin have frequently been reported off the east and south-west coast of Scotland, in the IS, and in the Western English Channel, with limited interchange between these inshore groups (BEIS[34], 2022; Cheney *et al*., 2013; IAMMWG, 2023; Robinson *et al.,* 2012).

[34] As of February 2023, BEIS is known as the DESNZ

3482. As outlined in **Appendix 11.2**, there would be potential for individuals from the East and West Scotland, Wales and Galicia to be present in the Project windfarm site, but there was no evidence of connectivity with any other coastal population of bottlenose dolphin in the UK, Ireland, and northern continental Europe (Nykänen *et al*., 2019).

3483. During both years of Project site-specific surveys, from March 2021 to February 2023, within the Project windfarm site and buffer area (see the **Appendix 11.2**), only two bottlenose dolphin were recorded, and these were in February 2023. On five occasions during the geotechnical surveys, five animals were observed.

3484. The SCANS-III survey also recorded no bottlenose dolphin in survey block F, in which the Project would be located (Hammond *et al*., 2017; see **Appendix 11.2**).

3485. The distribution maps by Evans and Waggitt (2023) also indicated very low bottlenose dolphin densities in and around the Project with a low average annual density of 0.0007 individuals per km$^2$ estimated over the area of the SCANS block.

3486. Few bottlenose dolphins were recorded during SCANS-IV (Gilles *et al.,* 2023), resulting in an estimated density of at 0.0104 animals/km$^2$ (CV = 0.700); with an abundance of 127 (95% Confidence Limit (CL)) = 3 – 353) individuals.

3487. The impact assessments have been based on the worst-case SCANS-IV density to present a precautionary approach:

- 0.0104 bottlenose dolphin/km$^2$

3488. The IAMMWG (2023) defined seven MUs for bottlenose dolphin (see **Appendix 11.2** of the ES). The Project would be located in the IS MU. The IS MU for bottlenose dolphin has an abundance estimate of 293 (CV= 0.54; 95% CI = 108 - 793; IAMMWG, 2023).

3489. Even though there was a migration rate of 25.7% between the coastal populations of Wales/West Scotland and East Scotland, the reference population for bottlenose dolphin used in this assessment has been based on the worst-case of IS MU alone, which, based on the latest IAMMWG estimate, was 293 bottlenose dolphin (IAMMWG, 2023).

3490. Photo-identification studies have revealed that the dolphins present in this site travelled between the Pen Llŷn a'r Sarnau/Lleyn Peninsula and the Sarnau SAC and Bae Ceredigion/Cardigan Bay SAC. Both these sites are within Cardigan Bay and their population should be considered together.

3491. Abundance estimates for the wider Cardigan Bay were 289 (CI = 184-453; CV = 0.23) in 2016 (Lohrengel *et al*., 2018). Therefore, the bottlenose dolphin IS

MU of 293 was representative of the Pen Llŷn a'r Sarnau SAC and Cardigan Bay SAC population.

3492. Analysis of photo-identification data for Cardigan Bay SAC based on a closed population model yielded an estimate of 147 (CI = 127-194; CV = 0.29) in 2016 (Lohrengel *et al.,* 2018). Therefore, the assessments have also been put into the context of the 147 bottlenose dolphin for the Cardigan Bay SAC.

## Conservation status

3493. Based on the most recent 2013-2018 reporting by the JNCC (2019), the overall conservation status for bottlenose dolphin in UK waters was 'unknown' at the time of assessment.

## Conservation objectives

3494. The conservation objectives for bottlenose dolphin are that the Llŷn a'r Sarnau/Lleyn Peninsula and the Sarnau SAC will "*continue to provide a productive and supportive marine area for bottlenose dolphin*". Bottlenose dolphin will continue to be widespread within the waters of the SAC and those frequenting the SAC will reflect a healthy population structure including immature and adult male and female dolphins. The bottlenose dolphins in the SAC will form an important component of a larger population of this species present in Cardigan Bay and in the wider sea area around Wales and the north east Atlantic. The animals using the SAC will reflect good physiological health. The bottlenose dolphins will have access to and sufficient availability of prey, and they will have widespread availability and access to good quality essential habitats free from excessive disturbance. The quality and distribution of essential habitats (such as for feeding, calving, resting and travelling) within the site will be maintained or improved through appropriate management" (NRW, 2018b).

3495. To achieve favourable conservation status, the following conservation objectives (subject to natural processes) would need to be fulfilled and maintained in the long-term. If these objectives were not met, restoration measures would be needed to achieve favourable conservation status.

### *Conservation objective 1: Populations*

3496. The bottlenose dolphin population is maintaining itself on a long-term basis as a viable component of its natural habitat. Important elements include:

- Population size
- Structure
- Production
- Condition of the species within the site

3497. As part of this objective, it should be noted that for bottlenose dolphin:

- Contaminant burdens derived from human activity are below levels that may cause physiological damage, or immune or reproductive suppression

*Conservation objective 2: Range*

3498. The bottlenose dolphin population within the site is such that the natural range of the population is not being reduced or likely to be reduced for the foreseeable future:

- Their range within the SAC and adjacent inter-connected areas is not constrained or hindered
- There are appropriate and sufficient food resources within the SAC and beyond
- The sites and amount of supporting habitat used by these species are accessible and their extent and quality is stable or increasing

*Conservation objective 3: Supporting habitats and species*

3499. The presence, abundance, condition and diversity of habitats and species required to support bottlenose dolphin is such that the distribution, abundance and populations dynamics of the species within the site and population beyond the site is stable or increasing. Important considerations include:

- Distribution
- Extent
- Structure
- Function and quality of habitat
- Prey availability and quality

3500. As part of this objective it should be noted that:

- The abundance of prey species subject to existing commercial fisheries needs to be equal to or greater than that required to achieve maximum sustainable yield and secure in the long term
- The management and control of activities or operations likely to adversely affect the species feature is appropriate for maintaining it in favourable condition and is secure in the long term
- Contamination of potential prey species should be below concentrations potentially harmful to their physiological health
- Disturbance by human activity is below levels that suppress reproductive success, physiological health or long-term behaviour

*Conservation objective 4: Restoration and recovery*

3501. For the purposes of the assessment, the potential effects have been considered in relation to the Pen Llŷn a'r Sarnau SAC conservation objectives.

3502. As part of this objective, it should be noted that populations for the bottlenose dolphin should be increasing.

*Table 9.18 Potential effects in relation to the conservation objectives for the Pen Llŷn a'r Sarnau SAC for bottlenose dolphin*

| Conservation objective | Potential effect |
|---|---|
| Populations/Range | Physical and permanent auditory injury from piling would be mitigated. However, this has been considered in detail in line with current advice. |
| Populations/Range | Significant disturbance and displacement as a result of increased underwater noise levels (e.g. piling) would have the potential to affect bottlenose dolphin. |
| Populations/Range | Increased collision risk with vessels would have the potential to affect bottlenose dolphin. |
| Range/Supporting habitats and species | Changes in water quality and prey availability would have the potential to affect bottlenose dolphin. |

#### 9.5.1.2 Cardigan Bay SAC

##### Description of designation

3503. Cardigan Bay is one of the largest bays in the British Isles, measuring over 100km across its westernmost extent from the Lleyn Peninsula to St. David's Head. The SAC covers 960km$^2$ and the population of bottlenose dolphins forms a primary interest and it was for this feature that the Bay was first designated as a SAC (NRW, 2018c,d).

3504. The SAC is 158km from the windfarm site, measured as a straight line distance, and 197km, measured as a coastline distance.

##### Bottlenose dolphin population and density

3505. The dolphins of Cardigan Bay SAC represent a mobile and wide-ranging population of variable individual residence. Their full range was not known but individuals have been recorded regularly along the southern coast of the Bay have also been seen both north and south of the SAC. Species range varies from year to year and this variation is likely to be predominantly as a consequence of natural environmental changes such as prey distribution.

3506. The Cardigan Bay SAC population is considered to be part of a wider population that ranges across waters of the Irish Sea. All effects that have been assessed for the Cardigan Bay SAC would be the same for the Pen Llŷn a'r Sarnau SAC, and thus not repeated:

- As a precautionary approach impact assessments were based on the worst-case SCANS-IV density of 0.0104 bottlenose dolphin/km$^2$ (Gilles *et al*., 2023)
- The reference population was based on the IS MU of 293 bottlenose dolphin (IAMMWG, 2023)
- Assessments have also been put into the context of the 147 bottlenose dolphin for the Cardigan Bay SAC (Lohrengel *et al.,* 2018)

#### Conservation status

3507. Unknown (see Pen Llŷn a'r Sarnau SAC).

#### Conservation objectives

3508. The conservation objectives were identical to those for the Pen Llŷn a'r Sarnau SAC.

## 9.5.2 Project-alone assessment

### 9.5.2.1 Underwater noise

3509. As outlined in **Section 9.3,** the assessment below refers to the assessment in **Chapter 11 Marine Mammals** of the ES, and the population and density estimates used in the EIA were the same as for this assessment. A full underwater noise assessment was undertaken in Section 11.6.3 of **Chapter 11 Marine Mammals** of the ES, and relevant information from that chapter has been summarised in the sections below.

#### Permanent auditory injury from underwater noise during piling

3510. The effect would be relevant to the construction phase only, with effects occurring outside any SAC.

3511. Underwater noise modelling was carried out (**Appendix 11.1**) to predict the noise levels likely to arise during impact piling and other activities. The modelled impact ranges were used to determine the potential effects on marine mammals. A detailed explanation of the modelling, inputs and assumptions has been provided in Section 11.6.3.1 of **Chapter 11 Marine Mammals** of the ES.

3512. Several scenarios were modelled to determine the worst-case for PTS effects for monopiles and pin-piles for both single strike and cumulative exposure for the total received noise over the whole piling operation, and cumulative effects of sequential piling of four pin-piles. The maximum predicted impact range for PTS was up to 0.1km from Cumulative Exposure ($SEL_{cum}$) during single pile installation of monopile with maximum hammer energy (6,600kJ) without any additional mitigation (**Table 9.19**). Given the distance of the Project windfarm site from the closest SAC (the Pen Llŷn a'r Sarnau SAC is 97km as straight line, or 128km as coastline distance), there was no pathway for effects upon bottlenose dolphin within an SAC.

3513. An assessment of the maximum number of individuals and percentage of the reference population affected under each of the scenarios was undertaken using the assumed worst-case densities and IS MU.

3514. For PTS, the maximum impact was approximately 0.001 bottlenose dolphin, which represents 0.0004% of the IS MU population or 0.0007% of Cardigan Bay SAC (**Table 9.20**). It should be noted that this assumed that no mitigation, other than embedded soft-start and ramp-up, would be in place. Given the embedded mitigation, it has been concluded that there would be no LSE from PTS (and no AEoI on any SAC).

3515. The MMMP would reduce the risk of PTS still further. The final MMMP for piling would be based on the Draft MMMP (Document Reference 6.5) which has been included with the DCO Application.

*Table 9.19 Predicted PTS impact ranges (and areas) for bottlenose dolphin at the Project from a single strike and from cumulative exposure for maximum hammer energy (taken from Table 11.21 of the ES)*

| Impact | Criteria and threshold (Southall *et al.*, 2019) | Monopile Maximum impact range (km) and area (km$^2$) | Monopile (sequential piling) Maximum impact range (km) and area (km$^2$) | Pin-pile Maximum impact range (km) and area (km$^2$) | Pin-pile (sequential piling) Maximum impact range (km) and area (km$^2$) |
|---|---|---|---|---|---|
| | | *Maximum hammer energy (6,600kJ)* | | *Maximum hammer energy (2,500kJ)* | |
| PTS from single strike (without mitigation) | $SPL_{peak}$ Unweighted (230 dB re 1µPa) Impulsive | <0.05km (<0.01km$^2$) | N/A | <0.05km (<0.01km$^2$) | N/A |
| PTS from cumulative SEL | $SEL_{cum}$ Weighted | <0.1km (<0.1km$^2$) | <0.1km (<0.1km$^2$) | <0.1km (<0.1km$^2$) | <0.1km (<0.1km$^2$) |

| Impact | Criteria and threshold (Southall *et al.*, 2019) | Monopile Maximum impact range (km) and area ($km^2$) | Monopile (sequential piling) Maximum impact range (km) and area ($km^2$) | Pin-pile Maximum impact range (km) and area ($km^2$) | Pin-pile (sequential piling) Maximum impact range (km) and area ($km^2$) |
|---|---|---|---|---|---|
| | | *Maximum hammer energy (6,600kJ)* | | *Maximum hammer energy (2,500kJ)* | |
| (including soft-start and ramp-up) | (185 dB re 1μPa$^2$s) Impulsive | | | | |

*Table 9.20 Maximum number of bottlenose dolphin (and % of IS MU and Cardigan Bay SAC) that could be at risk of PTS from single strike and from cumulative exposure ($SEL_{cum}$) of monopile or pin-pile and cumulative exposure from piling four pin-piles (taken from Table 11.23 and 11.24 of the ES)*

| Species | Criteria and threshold (Southall *et al.*, 2019) | Monopile with maximum hammer energy of 6,600kJ Maximum number of individuals (% of reference population) | Pin-pile with maximum hammer energy of 2,500kJ Maximum number of individuals (% of reference population) |
|---|---|---|---|
| Single strike at maximum hammer energy | $SPL_{peak}$ Unweighted (230 dB re 1μPa) Impulsive | 0.0001 (0.00004% of IS MU; 0.00007% of the SAC) | 0.0001 (0.00004% of IS MU; 0.00007% of the SAC) |
| Cumulative exposure ($SEL_{cum}$) during installation of a single monopile or pin-pile | $SEL_{cum}$ Weighted (185 dB re 1μPa$^2$s) Impulsive | 0.001 (0.0004% of IS MU; 0.0007% of the SAC) | 0.001 (0.0004% of IS MU; 0.0007% of the SAC) |
| Cumulative exposure ($SEL_{cum}$) during sequential piling of four pin-piles | $SEL_{cum}$ Weighted (185 dB re 1μPa$^2$s) Impulsive | 0.001 (0.0004% of IS MU; 0.0007% of the SAC) | 0.001 (0.0004% of IS MU; 0.0007% of the SAC) |

## Disturbance impacts from underwater noise during piling

3516. The effect would be relevant to the construction phase only with effects occurring outside any SAC.

3517. Disturbance from underwater noise from piling has been assessed in detail in Section 11.6.3.2 of **Chapter 11 Marine Mammals** of the ES.

3518. Based on the literature in the **Appendix 11.2**., there was no agreed disturbance range for dolphin species due to the impact of piling. For bottlenose dolphin due to the limited data and studies on dolphin behavioural response to piling and construction work in general; the application of the harbour porpoise dose-response data has been undertaken as a highly conservative worst-case method to quantify potential disturbance for bottlenose dolphin.

3519. The latest SNCB guidance recommended that a potential disturbance range or EDR of 26km (approximate area of 2,124km$^2$) around monopile locations (without noise abatement) and 15km (approximate area of 707km$^2$) for pin-piles with and without noise abatement should be used to assess harbour porpoise disturbance for SACs in England, Wales and Northern Ireland (JNCC *et al*., 2020).

3520. If this approach were also used for bottlenose dolphin, for 26km EDR up to 22.1 bottlenose dolphin could be temporarily disturbed (7.5% of IS MU; 15% of Cardigan Bay SAC). For 15km EDR, up to 7.4 bottlenose dolphin could be temporarily disturbed (2.5% of IS MU; 5% of Cardigan Bay SAC).

3521. Using the dose response approach, the estimated numbers of bottlenose dolphin disturbed (and percentage of the relevant MU), based on the worst-case foundation and location were estimated. The approach estimated 56.3 bottlenose dolphin (19.2% of the IS MU; 38.3% of Cardigan Bay SAC) could potentially be disturbed.

3522. Taking into account the difference in hearing sensitivity between harbour porpoise (Very-High Frequency (VHF) cetacean) and bottlenose dolphin (High-Frequency (HF) cetaceans (see Table 11.20 in **Chapter 11 Marine Mammals** of the ES; Southall *et al.,* 2019), even the 15km EDR would be a very precautionary worst-case.

3523. It is also important to note that bottlenose dolphin have a predominantly coastal distribution (see **Appendix 11.2**). They are primarily an inshore species, with most sightings within 10km of land. Studies of bottlenose dolphin off the east coast of Scotland found that the majority of sightings and movements were within 2km of the coastline (Quick *et al*., 2014). The Project windfarm site would be located approximately 30km from the nearest point on the coast; therefore, bottlenose dolphin are unlikely to be significantly disturbed.

3524. **Chapter 11 Marine Mammals** of the ES also detailed how population modelling for bottlenose dolphin was also conducted for population level consequences due to disturbance.

3525. The modelling assumed a worst-case of 56.3 bottlenose dolphin disturbed and 0.001 bottlenose dolphin with PTS on every piling day, the iPCoD model estimated there to be no discernible impact to the IS MU (**Table 9.21**) or the Cardigan Bay SAC population (**Table 9.22**). The median population size was predicted to be 100% of the un-impacted population size at the end of 2028 (one year after the piling has completed). By the end of 2029 (two years after piling ends) the median population size for the impacted population was predicted to be 100% of the un-impacted population size. This lack of discernible effect on the impacted population was maintained until 2052, which was the end point of the modelling.

3526. For bottlenose dolphin, the modelling indicated there would be no potential for a significant impact of disturbance due to less than a 1% population level impact of the IS MU over both the first six years and 25 year modelled periods (**Plate 9.3** and **Plate 9.4**).

*Table 9.21 Results of the iPCoD modelling for the Project, giving the mean population size of the bottlenose dolphin population (IS MU) for years up to 2052 for both impacted and un-impacted populations in addition to the median ratio between their population sizes*

| Year | Un-impacted population mean | Impacted population mean | Median impacted as % of un-impacted |
|---|---|---|---|
| Start | 296 | 296 | 100.00% |
| End 2028 | 295 | 295 | 100.00% |
| End 2029 | 293 | 288 | 100.00% |
| End 2032 | 287 | 283 | 100.00% |
| End 2037 | 278 | 275 | 100.00% |
| End 2047 | 262 | 259 | 100.00% |
| End 2052 | 255 | 252 | 100.00% |

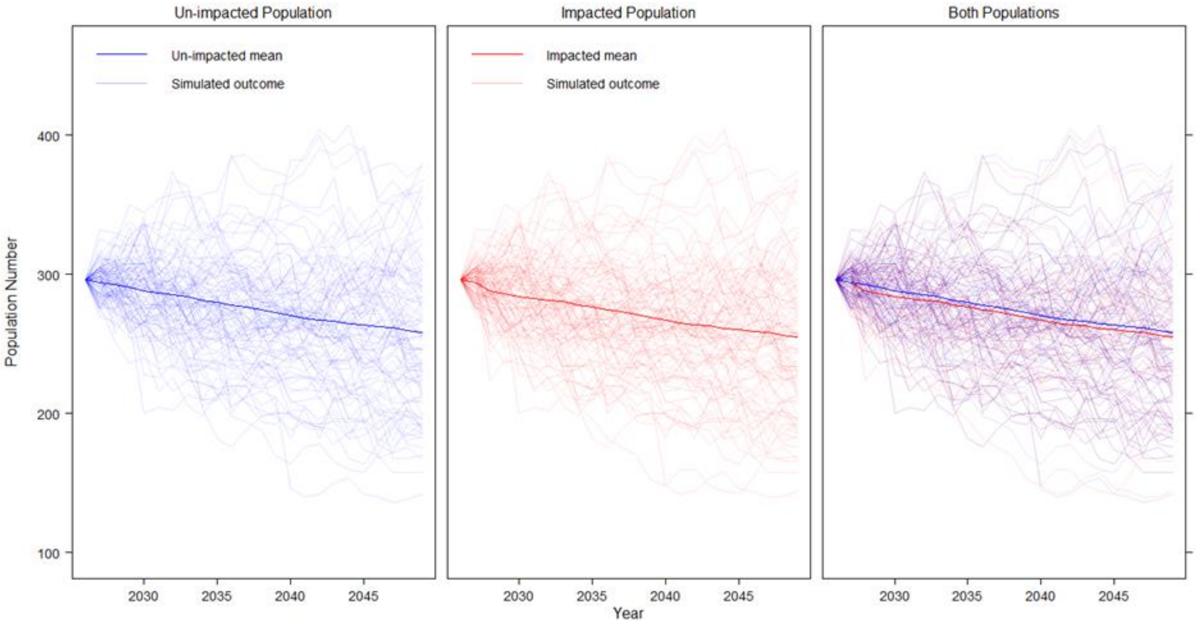


*Plate 9.3 Simulated worst-case bottlenose dolphin population sizes for both the un-impacted and the impacted populations for the Project based on the IS MU*

*Table 9.22 Results of the iPCoD modelling for the Project, giving the mean population size of the Cardigan Bay SAC bottlenose dolphin population for years up to 2052 for both impacted and un-impacted populations in addition to the median ratio between their population sizes*

| Year | Un-impacted population mean | Impacted population mean | Median impacted as % of un-impacted |
|---|---|---|---|
| Start | 148 | 148 | 100.00% |
| End 2028 | 148 | 148 | 100.00% |
| End 2029 | 147 | 143 | 100.00% |
| End 2032 | 145 | 142 | 100.00% |
| End 2037 | 141 | 139 | 100.00% |
| End 2047 | 132 | 130 | 100.00% |
| End 2052 | 128 | 126 | 100.00% |

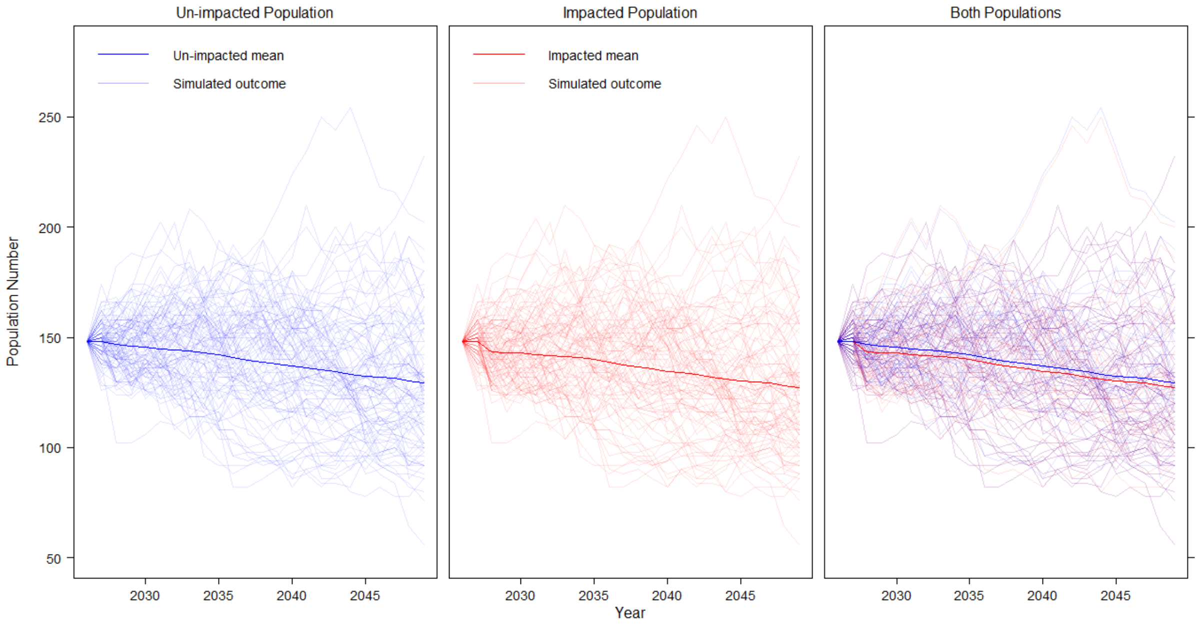


*Plate 9.4 Simulated worst-case Cardigan Bay SAC bottlenose dolphin population sizes for both the un-impacted and the impacted populations for the Project.*

3527. The assessments of the potential disturbance during any ADD activation (assuming a 80 minute activation period) estimated that two individuals (0.58% of IS MU; 1.4% of Cardigan Bay SAC) could be affected (see Table 11.34 of **Chapter 11 Marine Mammals** of the ES).

3528. The maximum duration of effect for active piling (assuming two OSPs and all WTGs using monopiles) would be 190 hours. The maximum duration of effect for active piling (assuming two OSPs and all WTGs using pin-piles) would be 619 hours and 36 minutes (see Table 11.35 of **Chapter 11 Marine Mammals** of the ES). These estimations included soft-start, ramp up and ADD activation times.

3529. The total duration of the installation campaign for WTGs and OSPs foundations is expected to be 9 - 12 months. The duration of piling has been based on a worst-case scenario and a very precautionary approach. In addition, as has been shown at other offshore wind farms, the duration used in the assessment can be overestimated. The duration of any potential displacement effect would differ depending on the distance of the individual from the piling activity and the noise level the animal was exposed to.

3530. The effect would occur outside of any SAC and, based on population modelling, would affect less than 5% of the SAC- and reference population. **It has therefore been concluded that there would be no significant disturbance effect on the bottlenose dolphin IS MU population or the Cardigan Bay SAC population (and no AEoI on any SAC) from underwater noise during piling.**

## Underwater noise and disturbance from other sources

### Construction

3531. Section 11.6.3.3 of **Chapter 11 Marine Mammals** of the ES details the effects of disturbance impacts from underwater noise from seabed preparation, dredging, trenching, cable installation and rock placement. Section 11.6.3.4 of **Chapter 11 Marine Mammals** of the ES details the effects of underwater noise from the presence of vessels.

3532. A review of various studies was used to determine the maximum potential disturbance range for other construction activities and vessels. During the construction of two Scottish windfarms (Beatrice OWF and Moray East OWF), (Benhemma-Le Gall *et al*., 2021), a reduction in harbour porpoise presence was reported up to 4km (50.27km$^2$) distance from construction vessels. This distance has been used as the disturbance range for other construction activities, including associated vessels.

3533. Taking into account the distance of the Project windfarm site from the closest SAC (97km as straight line, or 128km as coastline distance), there is no pathway for disturbance effects directly upon bottlenose dolphin within any SAC considered in this assessment.

3534. The assessments took into account the number of construction activities, other than piling, that could be undertaken at the same time and maximum number of vessels that could be on site at any one time, and have been summarised in **Table 9.23** and **Table 9.24**.

3535. As a precautionary approach, the potential disturbance from two activities (such as cable laying, dredging, trenching and rock placement) occurring at the same time, including vessels undertaking the work, has been assessed based on maximum impact area of 100.54km$^2$ (assuming no overlap in the impact areas between the activities). The maximum number of bottlenose dolphin that could be disturbed was one (0.4% of the IS MU; 0.7% of Cardigan Bay SAC; **Table 9.23**).

3536. The maximum number of bottlenose dolphin that could be disturbed from up to 37 vessels on the revised site (285.4km$^2$) (revised site details in **Chapter 11 Marine Mammals** of the ES in Section 11.6.3.4), with a 4km disturbance range for each vessel was up to three bottlenose dolphin (1.0% of IS MU; 2.0% of Cardigan Bay SAC; **Table 9.24**).

3537. There would be no potential for additive effects (i.e. the disturbance from construction activities plus vessel activities) as the disturbance range of 4km for construction activities includes the vessel undertaking the work.

3538. The worst-case assessments indicate that, for underwater noise and disturbance from other construction activities and vessels, less than 2.7% (up to 1.4% of IS MU) of the bottlenose dolphin population could be temporarily disturbed. For the Cardigan Bay SAC population less than 5% have the potential to be temporarily disturbed (2.7% of Cardigan Bay SAC).

3539. The range of impact and area of disturbance would occur outside of any SAC. **It has been concluded that there would be no LSE on the bottlenose dolphin population (and no AEoI on any SAC) from the effects of disturbance impacts from underwater noise from other sources during construction.**

*Table 9.23 Maximum number of bottlenose dolphin (and % of IS MU and Cardigan Bay SAC) that could be disturbed as a result of underwater noise associated with other (non-piling) construction activities, including vessels undertaking the work (taken from Table 11.48 of the ES)*

| Potential Impact | Maximum number of individuals (% of reference population) that could be disturbed for one activity (50.27km$^2$) | Maximum number of individuals (% of reference population) that could be disturbed for two activities (100.54km$^2$) |
|---|---|---|
| Disturbance based on 4km disturbance range | 0.5 (0.2% of IS MU; 0.3% of SAC) | 1.0 (0.4% of IS MU; 0.7% of SAC) |

*Table 9.24 Maximum number of bottlenose dolphin (and % of IS MU and Cardigan Bay SAC) that could be disturbed as a result of underwater noise associated construction vessels (taken from Table 11.52 of the ES)*

| Potential Impact | Maximum number of individuals (% of reference population) for one vessel (50.27km$^2$) | Maximum number of individuals (% of reference population) for revised site+4km buffer (285.4km$^2$) |
|---|---|---|
| Vessel disturbance based on 4km disturbance range | 0.5 (0.2% of IS; 0.3% of SAC MU) | 3 (1.0% of IS MU; 2.0% of SAC) |

### Operation and maintenance

3540. Underwater noise and disturbance during operation and maintenance could come from operational noise from WTGs, maintenance work (such as rock placement or cable repairs) and vessels. Each of these sources has been considered separately in detail in Sections 11.6.4.1, 11.6.4.2 and 11.6.4.3 of **Chapter 11 Marine Mammals** of the ES.

3541. A review of currently available studies was used to determine the potential disturbance from underwater operational noise from WTGs (see Section 11.6.4.1 of **Chapter 11 Marine Mammals** of the ES). The studies indicated that any disturbance would be in the immediate area of the operational WTG, depending on ambient noise levels. There was no evidence of any lasting disturbance or exclusion around OWFs during operation.

3542. Given the distance of the Project windfarm site from the coast, it is unlikely that there would be a LSE on the bottlenose dolphin population from the effects of disturbance impacts from underwater noise of operational WTGs.

3543. As a precautionary approach, a 4km impact has also been used as a potential disturbance range for maintenance activities including vessels undertaking the work, based on construction activities. The potential disturbance from cable repairs and rock placement occurring at the same time has been assessed based on maximum impact area of 100.53km$^2$ (**Table 9.25**). The potential disturbance from up to three vessels in the Project windfarm site at the same time during operation and maintenance has been assessed based 4km impact range for each vessel, with a maximum impact area of 150.81km$^2$ for three vessels (**Table 9.25**).

*Table 9.25 Maximum number of bottlenose dolphin (and % of IS MU and Cardigan Bay SAC) that could be disturbed as a result of underwater noise associated with maintenance activities at the Project (taken from Tables 11.68 and 11.72 of the ES)*

| Source | Potential Impact | Maximum number of individuals (% of reference population) |
|---|---|---|
| Maintenance activities | Disturbance based on 4km disturbance range | Two activities<br>1.05 (0.4% of IS MU; 0.7% of SAC) |
| Vessels | Disturbance based on 4km disturbance range | Three vessels<br>1.6 (0.53% of IS MU; 1.1% of SAC) |

3544. The worst-case of any of these effects came from the disturbance effects from vessels, assuming that there would be three vessels within the Project windfarm site simultaneously. This would impact approximately 1.6 individuals (0.6% of the IS MU population; 1.1% of the SAC population). Although there is potential for additive effects (i.e. the disturbance from maintenance activities plus vessel activities), it is unlikely that the total area of effect and therefore individuals (% of the IS MU population) would be the sum of the individual effects listed above. These activities would not all be taking place at the same time and any disturbance from rock placement or cable laying would be coincident with and disturb a larger area than the vessel associated with that activity.

3545. Given that this effect would be minimal and would occur outside of any area considered important for foraging, breeding or calving, **it has been concluded that there would be no LSE on the IS MU or the SAC bottlenose dolphin population (and no AEoI on any SAC) from the effects of disturbance impacts from underwater noise during operation.**

**Decommissioning**

3546. Potential effects on bottlenose dolphin associated with underwater noise during decommissioning have not been assessed in detail, as further assessments would be carried out ahead of any decommissioning works to be undertaken taking account of known information at that time, including relevant guidelines and requirements. The detailed Decommissioning Programme would provide details of the techniques to be employed and any relevant mitigation measures required.

3547. It is not possible to provide details of the methods that would be used during decommissioning at this time. However, it is expected that the activity levels would be comparable to construction (with the exception of pile driving noise which would not occur).

3548. During decommissioning, the potential effects on bottlenose dolphin are anticipated to be similar or less than the worst-case for the construction phase (depending on the methods used). The effects would therefore be comparable to those described in construction.

3549. Given that this effect would be lower than for disturbance impacts from underwater noise during piling, and the effect would occur outside of any area considered important for foraging, breeding or calving, **it has been concluded that there would be no LSE on the IS MU or SAC bottlenose dolphin population (and no AEoI on any SAC) from the effects of disturbance impacts from underwater noise during decommissioning.**

### 9.5.2.2 Barrier effects as a result of underwater noise

#### Construction

3550. Underwater noise during construction could have the potential to create a barrier effect, preventing movement of bottlenose dolphin between important feeding and/or breeding areas, or potentially increase swimming distances if bottlenose dolphin avoid the area and go around it. This effect has been considered in detail in Section 11.6.3.5 of **Chapter 11 Marine Mammals** of the ES.

3551. As discussed in Section 11.6.3.5 of **Chapter 11 Marine Mammals** of the ES, there is the potential for bottlenose dolphin to move between areas to the north and south of the windfarm site. However, as reflected in the distribution of bottlenose dolphin in the Irish and Celtic Seas, bottlenose dolphin have a predominantly coastal distribution (see **Appendix 11.2** of the ES). They are primarily an inshore species, with most sightings within 10km of land. Studies of bottlenose dolphin off the east coast of Scotland found that the majority of sightings and movements were within 2km of the coastline, and in waters that were less than 30m deep (Quick *et al.,* 2014). The Project windfarm site would be located approximately 30km from the nearest point on the coast.

3552. Taking into account the movements of bottlenose dolphin along the coast, underwater noise at the Project windfarm site is unlikely to result in any barrier effects to bottlenose dolphin. Any effect would occur outside of any area considered important for foraging, breeding or calving. **It has been therefore concluded that there would be no LSE on the IS MU or SAC bottlenose dolphin population (and no AEoI on any SAC) from barrier effects during construction.**

### Operation and maintenance and decommissioning

3553. As noted above, bottlenose dolphin is primarily an inshore species, with most sightings within 10km of land. Given that any disturbance would be limited to the close vicinity of the operational windfarm site, underwater noise at the Project windfarm site would be unlikely to result in any barrier effects to bottlenose dolphin.

3554. During decommissioning, the potential effects on bottlenose dolphin are anticipated to be similar or less than the worst-case for the construction phase (depending on the methods used). The effects would therefore be comparable to those described in construction.

3555. Given that this effect would be lower than for disturbance impacts from underwater noise during construction, and the effect would occur outside of any area considered important for foraging, breeding or calving, **it has been concluded that there would be no LSE on the IS MU or the SAC bottlenose dolphin population (and no AEoI on any SAC) from barrier effects during operation, maintenance or decommissioning.**

## 9.5.2.3 Vessel interactions

### Construction

3556. During the construction phase there would be an increase in the number of vessels in the windfarm site. The maximum number of vessels that could be on the Project windfarm site at any one time has been estimated to be up to a total of 37 vessels (**Table 9.4**). The number, type and size of vessels would vary depending on the activities taking place at any one time. This effect has been considered in detail in Section 11.6.3.6 of **Chapter 11 Marine Mammals** of the ES, the assessment below summarises the information presented there.

3557. **Chapter 11 Marine Mammals** of the ES estimated that approximately <1 individuals or 0.04% of the IS MU (0.06% of the SAC population) could be at risk of collision during construction per year (see Table 11.56 of **Chapter 11**

**Marine Mammals** of the ES). This would be a permanent effect and in the worst-case lethal for the individuals.

3558. It has been considered that the quantified assessment was highly precautionary. Marine mammals are able to detect and avoid vessels. However, vessel strikes have been known to occur, possibly due to distraction whilst foraging and socially interacting, or due to the marine mammals' inquisitive nature (Wilson *et al.,* 2007).

3559. Feingold and Evans (2013) noted that although collision was a potential risk, there were no records of bottlenose dolphins being killed by boats in Cardigan Bay. Although one case of injury to a female was reported, the injury did not seem to impact on her mobility or reproduction.

3560. It is therefore considered unlikely that bottlenose dolphin would be at increased collision risk with vessels during construction, considering the existing number of vessel movements in the area, and that vessels within the windfarm would be stationary for much of the time or very slow moving. In addition, taking into account the disturbance effects from vessels, the actual risk is likely to be very low.

3561. As outlined in **Section 9.3.1** the commitment to best practice mitigation measures would further reduce the potential risk of collision. Vessel movements, where possible, would follow set vessel routes and hence areas where marine mammals were accustomed to vessels, in order to reduce any increased collision risk. Predictability of vessel movement by marine mammals has been known to be a key aspect in minimising the potential risks caused by vessel traffic (Nowacek *et al*., 2001; Lusseau, 2003; 2006). Vessels travelling at high speeds were considered to be more likely to collide with marine mammals, and those travelling at speeds below 10 knots would rarely cause any serious injury (Laist *et al.,* 2001). All vessel movements would be kept to the minimum number that was required to reduce any potential collision risk. Additionally, vessel operators would use good practice to reduce any risk of collisions with marine mammals.

3562. The mitigation measures to manage collision risk would be agreed with the relevant stakeholders and would be detailed within the PEMP.

3563. Given the low risk to bottlenose dolphin and the commitment to mitigation measures to manage residual risk, **it has been concluded that there would be no LSE on the IS MU or the SAC bottlenose dolphin population (and no AEoI on any SAC) from the effects of disturbance impacts from vessel interactions during construction.**

### Operation and maintenance and decommissioning

3564. The increased risk of collision with vessels during operation and maintenance would be less than assessed for the construction period. During the operation and maintenance phase, the maximum number of vessels that could be on the Project windfarm site at any one time has been estimated at up to a total of ten vessels in a heavy maintenance year, whereas a standard year would only have three (**Table 9.4**). The number, type and size of vessels would vary depending on the activities taking place at any one time. The vessels in the Project windfarm site during operation and maintenance would be slow moving or stationary.

3565. Section 11.6.4.6 of **Chapter 11 Marine Mammals** of the ES assesses the potential for collision with vessels during operation and maintenance and concluded that approximately <1 individuals or 0.012% of the IS MU (0.02% of the SAC population) could be at risk per year (see Table 11.74 of **Chapter 11 Marine Mammals**).

3566. During decommissioning, the potential effects on bottlenose dolphin are anticipated to be similar or less than the worst-case for the construction phase. The effects would therefore be comparable to those described construction.

3567. Given that this effect would be lower than for potential for collision with vessels during construction and the commitment to mitigation measures to reduce that risk further, **it has been concluded that there would be no LSE on the IS MU bottlenose dolphin population (and no AEoI on any SAC) from the effects of disturbance impacts from vessel interactions during operation or decommissioning.** Assessments were made on a standard maintenance year, but, given the low values, it is anticipated that there would also be no LSE on the reference population (and no AEoI on the SAC) during a heavy maintenance year.

## 9.5.2.4 Changes to prey resources

### Construction

3568. The potential effects on prey species during construction can result from physical disturbance and loss of habitat; increased SSC and sediment deposition; and underwater noise. **Chapter 10 Fish and Shellfish Ecology** of the ES, provided an assessment of these impact pathways on the relevant fish and shellfish species and concluded impacts of negligible to minor adverse significance in EIA terms. **Chapter 11 Marine Mammals** of the ES considers these effects in terms of potential indirect effects on bottlenose dolphin (see **Chapter 11 Marine Mammals** Section 11.6.3.7 Changes to Prey Resource).

3569. Bottlenose dolphin are opportunistic feeders and take a wide variety of fish and invertebrate species. Benthic and pelagic fish (as well as octopus and other cephalopods have all been recorded in the diet of bottlenose dolphin (Santos *et al*., 2001; Santos *et al.,* 2004; Reid *et al.,* 2003) (see **Appendix 11.2**). Food resources appeared to be a primary factor in determining movements and site fidelity in bottlenose dolphins (NRW, 2018c) and clearly the SACs designated for bottlenose dolphin would be the most important areas for feeding. Bottlenose dolphin were not recorded during the first year of site-specific surveys of the Project windfarm site and buffer area, and as previously discussed, the species has, primarily, a coastal distribution.

3570. Any reductions in prey availability would be small scale, localised and temporary and occur in an area that would be suboptimal for bottlenose dolphin feeding. It is considered highly unlikely therefore that potential reductions in prey availability as a result of construction activities would result in detectable changes to the bottlenose dolphin population.

3571. Given that this effect would be limited and would occur outside of any area considered important for bottlenose dolphin foraging (i.e. outside of a SAC and outside of coastal waters), **it is concluded that there would be no LSE on the IS MU or SAC bottlenose dolphin population (and no AEoI on any SAC) from the effects of changes to prey species during construction.**

### Operation and maintenance and decommissioning

3572. Changes to prey resources during operation and maintenance have been assessed in Section 11.6.4.7 of **Chapter 11 Marine Mammals**. As per construction, this assessment was based upon the conclusions of **Chapter 10 Fish and Shellfish Ecology** and considered a suite of impacts including permanent habitat loss, introduction of hard substrate and EMF, as well as the impacts considered for construction. Although new impacts have been considered for operation, some effects such as those of physical disturbance; increased SSC and sediment deposition; and underwater noise would be reduced when compared to construction. It is considered highly unlikely therefore that potential reductions in prey availability as a result of operational activities would result in detectable changes to bottlenose dolphin populations.

3573. During decommissioning, the potential effects on bottlenose dolphin are anticipated to be similar or less than the worst-case for the construction phase. The effects would therefore be comparable to those described construction.

3574. Given that this effect would be limited and would occur outside of any area considered important for bottlenose dolphin foraging (i.e. outside of a SAC and outside of coastal waters), **it has been concluded that there would be no LSE on the IS MU bottlenose or SAC dolphin population (and no AEoI on**

**any SAC) from the effects of changes to prey species during operation or decommissioning.**

### 9.5.2.5 Changes to water quality

#### Construction

3575. The disturbance of seabed sediments has the potential to lead to increases in SSC concentrations and the release of any sediment-bound contaminants (such as heavy metals and hydrocarbons that may be present within them) into the water column. The accidental release of contaminants (e.g. through spillage) also has the potential to affect water quality. During construction there would also be the potential for increased SSCs. Section 11.6.3.8 of **Chapter 11 Marine Mammals** of the ES considered these effects in detail.

3576. Throughout the construction phase, best practice techniques and due diligence regarding the potential for pollution would be followed throughout all construction activities. Any risk of accidental release of contaminants (e.g. through spillage) would be mitigated in line with the PEMP and any changes to water quality as a result of any accidental release of contaminants (e.g. through spillage or vessel collision) would be negligible. Therefore, the potential for pollutants to be released into the environment has not been considered further in this assessment.

3577. Section 11.6.3.8 of **Chapter 11 Marine Mammals** of the ES considers increases in SSC and remobilisation of existing contaminated sediments.

3578. With regard to increases in SSC, bottlenose dolphin have been noted to inhabit turbid environments and utilise sonar to sense the environment around them and there was little evidence that turbidity affected bottlenose dolphin directly (Todd *et al*., 2014). Increased turbidity was unlikely to have a substantial direct effect on bottlenose dolphin that often inhabit naturally turbid or dark environments.

3579. As outlined in **Chapter 8 Marine Sediment and Water Quality** of the ES, site specific data indicated that for all potential contaminants tested for within the sediments of the Project windfarm site concentrations were negligible. There was therefore no potential for any direct or indirect effects on marine mammals from remobilisation of contaminated sediments.

3580. Given that water quality effects would be negligible, **it has been concluded that there would be no LSE on the IS MU or SAC bottlenose dolphin population (and no AEoI on any SAC) during construction.**

#### Operation and maintenance and decommissioning

3581. During the operation and maintenance phase, there would be the potential for increases in SSC and the release of any sediment-bound contaminants. The scale of these impacts would be small, infrequent and of short-term duration and of a lower magnitude than during the construction phase.

3582. During decommissioning, the potential water quality effects are anticipated to be similar or less than the worst-case for the construction phase. The effects would therefore be comparable to those described in construction.

3583. Given that water quality effects would be negligible, **it is concluded that there would be no LSE on the IS MU or SAC bottlenose dolphin population (and no AEoI on any SAC) during operation and maintenance or decommissioning.**

#### 9.5.2.6 Potential interactions of Project effects

3584. The anticipated effects on marine mammal receptors were not expected to interact in a way that would lead to a combined effect of greater significance than the assessments presented for each individual phase. It should also be noted that a high level of precautionary measures were implemented in the assessment process, further contributing to the overall understanding and mitigation of potential impacts.

3585. Interactions of Project effects would be as per those outlined in **Section 9.4.2.6.**

#### 9.5.2.7 Summary of Project-alone effects

3586. There would be no overlap of permanent and temporary noise effects within any SAC.

3587. Due to embedded mitigation and commitment to securing mitigation measures (i.e PTS mitigation through the MMMP and to manage the residual low collision risk through best practice vessel practices secured in the PEMP), it is considered that permanent effects upon bottlenose dolphin would be avoided during construction, operation and maintenance or decommissioning.

3588. Disturbance of bottlenose dolphin potentially caused by underwater noise and vessel interactions would affect less that 5% of the population.

3589. **It is concluded that there would be no LSE on the bottlenose dolphin IS MU population during construction, operation and maintenance or decommissioning. In addition, any effects would occur outside any SAC boundary.**

3590. Indirect effects (i.e. on water quality or prey resources) were considered to be insignificant and would occur outside any SAC boundary.

3591. None of the effects assessed would be within an SAC or were considered to have a significant effect on the IS MU bottlenose dolphin population. **As such, it is considered that there would be no adverse effect on integrity of the**

**Pen Llŷn a'r Sarnau SAC and Cardigan Bay SAC in relation to the conservation objective relating to 'Population' and 'Range'.**

3592. Indirect effects (i.e. on water quality or prey resources) were considered to be insignificant and would occur outside any SAC boundary. **As such, it is considered that there would be no adverse effect on integrity on the Pen Llŷn a'r Sarnau SAC and Cardigan Bay SAC in relation to the conservation objective 'Range' and 'Supporting habitats and species'.**

3593. The confidence in the assessment for all impacts is considered high considering the baseline information and site-specific data.

### 9.5.3 Potential in-combination effects of the Project with Transmission Assets

3594. A 'combined' assessment has been made with the Transmission Assets[35], for the purpose of an in-combination assessment considering its functional link with the Project.

3595. Pen Llŷn a'r Sarnau SAC, Cardigan Bay SAC were screened in for both the Project and the Transmission Assets.

3596. For the Transmission Asset ISAA Project-alone assessment (Morgan Offshore Wind Limited and Morecambe Offshore Windfarm Ltd, 2023b), there was no adverse effect on the site integrity on any of the screened-in sites. As for the Project, the distances to the closest SACs were outside the ZoI. A full quantitative assessment has been provided in the assessment of all plans and projects, including the Transmission Assets and has not been repeated here. An assessment has been made below of each impact, considering the information in **Section 9.5.4.1** and understanding the interactions between the projects.

#### 9.5.3.1 Underwater noise and barrier effects

3597. The key interaction has been identified as piling and UXO during construction for the projects.

3598. Given that the Project and Transmission Assets would be outwith any SAC and potential PTS effects would be mitigated by any consented project, **it is concluded that there would be no LSE on the reference population (and no AEoI on the SAC).**

[35] As the Transmission Assets includes infrastructure associated with both the Project and the Morgan Offshore Wind Project Generation Assets, it should be noted that the combined assessment considers the transmission infrastructure for both the Project and the Morgan Offshore Wind Project Generation Assets.

### 9.5.3.2 Vessel interactions

3599. During all phases, there would be additional effects due to increased vessel presence from both projects.

3600. Given that the Project and Transmission Asset would be outwith any SAC and both projects would adhere to good practice, **it is concluded that there would be no LSE on the reference population (and no AEoI on the SAC).**

### 9.5.3.3 Indirect effects (changes to prey resource and water quality)

3601. During all phases, there would be additional effects due to increased vessel presence from both projects and additional pressure on prey resource.

3602. Given the impacts identified for both projects on prey species and that the Project and Transmission Assets would be outwith any SAC and both projects would adhere to good practice, **it is concluded that there would be no significant in-combination effect on the SAC reference populations (and no AEoI).**

## 9.5.4 Assessment of the potential effects of the Project in-combination with other plans and projects

3603. Section 11.7 of **Chapter 11 Marine Mammals** of the ES details the CEA. This in-combination assessment was based upon the cumulative assessment and provided a summary of the key information from that assessment without repeating every step of the process. Key information has been taken from **Chapter 11 Marine Mammals** of the ES and carried through with regard to the effect on designated sites.

3604. The effects screened into the in-combination assessment and the identification of the other plans, projects and activities that may result in in-combination effects have been provided in **Appendix 11.4**.

### 9.5.4.1 Underwater noise

#### Permanent auditory injury from underwater noise during piling

3605. PTS could occur as a result of piling during OWF installation or detonation of underwater explosives (used occasionally during the removal of underwater structures and UXO clearance) (JNCC, 2010a,b). However, if there was the potential for any PTS, from any project, suitable mitigation would need to be put in place to reduce any risk to marine mammals. Other activities such as dredging, drilling, rock placement, vessel activity, operational windfarms, oil and gas installations or wave and tidal sites would emit broadband noise in lower frequencies and PTS from these activities would be very unlikely.

3606. Given that the Project would be outwith any SAC, there was no potential for AEoI from PTS onset in-combination with other projects, as all projects should ensure mitigation was in place to negate the potential for PTS. Therefore, the potential for PTS in-combination was screened out and not assessed further.

### Disturbance from underwater noise during construction

3607. Section 11.7.3.1 of **Chapter 11 Marine Mammals** of the ES considers disturbance in relation to several sub-effects and then considers them all together: underwater noise impacts from piling at other OWFs; underwater noise impacts from construction activities (other than piling) at other OWFs; and disturbance from other industries and activities (which included geophysical survey, seismic survey and UXO clearance). The combined results from these assessments have been displayed in **Table 9.29** (based upon Table 11.107 of **Chapter 11 Marine Mammals**).

#### Disturbance from piling

3608. The potential disturbance from underwater noise during piling for bottlenose dolphin has been assessed based on the worst-case maximum area modelled for the Project for each species, using TTS/fleeing response as a proxy for disturbance, where no further information of potential disturbance impact ranges were available.

3609. Of the screened UK and European OWFs, five OWFs plus the Morgan and Morecambe Transmission Assets could be piling at the same time as the Project (see **Appendix 11.4**):

- AyM OWF (PINS Tier 1)
- Mona OWF (PINS Tier 2)
- Morgan OWF (PINS Tier 2)
- Morgan and Morecambe Transmission Assets (PINS Tier 2)
- Erebus OWF (PINS Tier 1)
- White Cross OWF (PINS Tier 1)

3610. This short list of OWF projects that could be piling at the same time was precautionary (particularly as half the of the projects were Tier 2 where there is less certainty on schedule) as the Project could change as projects develop, but this was the best available information at the time of writing and was considered to reflect the limitations and constraints to project delivery.

3611. The following caveats should be noted in terms of this worst-case assessment:

- The potential areas of disturbance assume that there would be no overlap in the areas of disturbance between different projects

- It was assumed that all OWF projects would be 100% piled, if piled foundations were an option
- The approach has been based on the potential for single piling at each wind farm at the same time as single piling at the windfarm site. This approach allowed for some of the offshore wind farms not to be piling at the same time, while others could be simultaneously piling. This is considered to be the most realistic worst-case scenario, as it is highly unlikely that all other wind farms would be simultaneously piling at exactly the same time as piling at the Project, especially given the limited active piling time
- The actual duration for active piling time which could disturb marine mammals would be only a very small proportion of the potential construction period, and this would be the case for other OWFs. This means that there would be a limited window temporal overlap and for any in-combination effect to occur
- In practice, the potential temporary effects would be less than those predicted in this assessment as there is likely to be a great deal of variation in timing, duration (noting this has been typically overestimated in assessments), and hammer energies used throughout the various OWF construction periods. In addition, not all individuals would be displaced over the entire potential disturbance range used within the assessments

3612. The disturbance range was assessed using the dose-response curves for harbour porpoise during single pile installation at the Project. This overly precautious number of animals informed the much more realistic population modelling (using iPCoD) for the cumulative effects assessment, taking into account project specific effects and was deemed the most accurate. The results have been based on assessments in Section 11.7.3.2 in **Chapter 11 Marine Mammals** of the ES (detailed information to iPCoD see **Appendix 11.3**). Additional to the population modelling for the IS MU population consequences, modelling has also been conducted using the smaller Cardigan Bay SAC reference population.

3613. For bottlenose dolphin, the potential magnitude of the CEA for disturbance from underwater noise from piling has been assessed as low (in EIA terms) in the ES. This was assessed as a conservative approach due to a 1.39% decrease in one year. Overall the yearly average was less than a 1% population level effect over both the first six years and 25 year modelled periods (**Table 9.26**; **Plate 9.5**).

*Table 9.26 Results of the iPCoD modelling for the cumulative assessment, giving the mean population size of the bottlenose dolphin population (IS MU) for years up to 2052 for both* *impacted and un-impacted populations in addition to the median ratio between their population sizes (taken from Table 11.87 from the ES)*

| Year | Un-impacted population mean | Impacted population mean | Median impacted as % of un-impacted |
|---|---|---|---|
| Start | 296 | 296 | 100.00% |
| End 2028 | 295 | 289 | 100.00% |
| End 2029 | 292 | 281 | 98.61% |
| End 2032 | 286 | 271 | 97.71% |
| End 2037 | 277 | 264 | 97.87% |
| End 2047 | 261 | 249 | 97.80% |
| End 2052 | 254 | 242 | 97.97% |

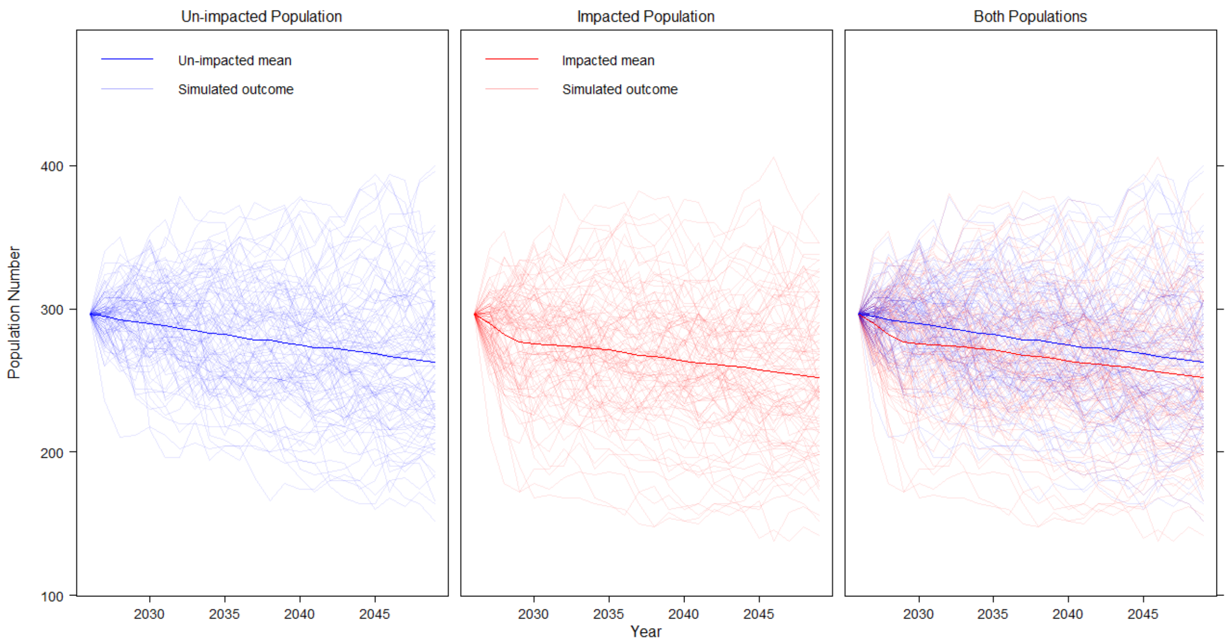


*Plate 9.5 Simulated worst-case IS MU bottlenose dolphin population sizes for both the un-impacted and the impacted populations for in-combination projects (taken from Plate 11.10 from the ES)*

3614. For the Cardigan Bay SAC, the potential for disturbance from underwater noise from piling has been assessed as insignificant based on an average population decrease of 0.9% over the first six years. Overall, in Cardigan Bay, the yearly average was less than a 1% population level effect over both the first six years and 25 year modelled periods (**Table 9.27**; **Plate 9.6**).

*Table 9.27 Results of the iPCoD modelling for the in-combination assessment, giving the mean population size of the Cardigan Bay SAC for years up to 2052 for both impacted and un-impacted populations in addition to the median ratio between their population sizes*

| Year | Un-impacted population mean | Impacted population mean | Median impacted as % of un-impacted |
|---|---|---|---|
| Start | 148 | 148 | 100.00% |
| End 2028 | 147 | 145 | 100.00% |
| End 2029 | 146 | 140 | 97.26% |
| End 2032 | 144 | 133 | 94.44% |
| End 2037 | 140 | 131 | 95.04% |
| End 2047 | 132 | 123 | 94.85% |
| End 2052 | 128 | 119 | 95.10% |

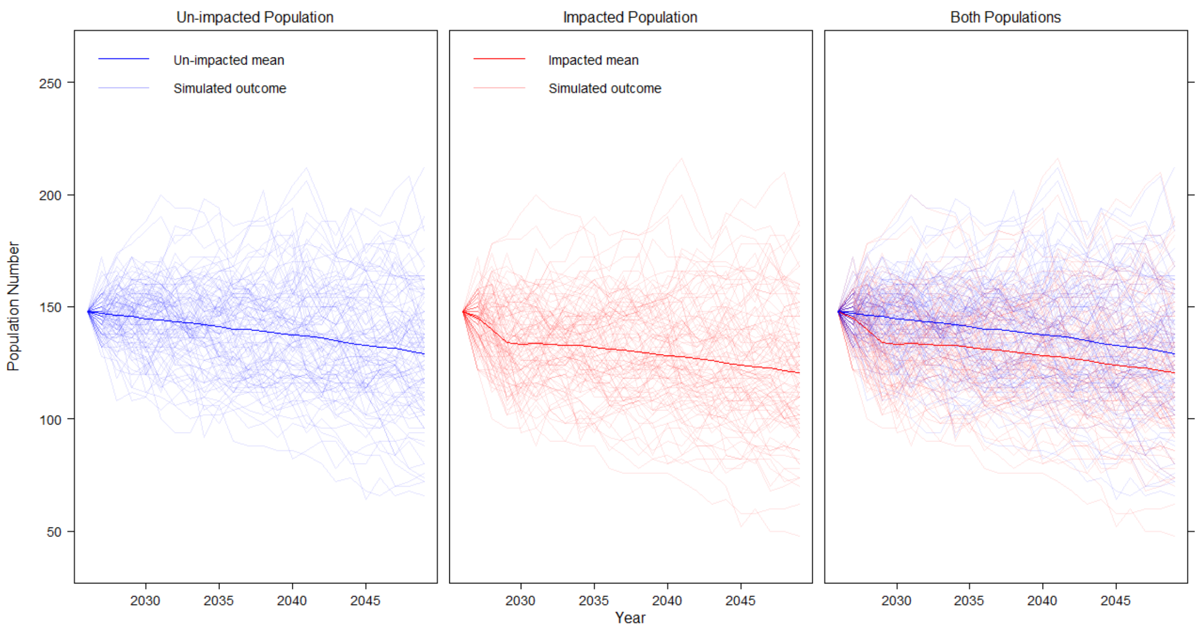


*Plate 9.6 Simulated worst-case Cardigan Bay SAC bottlenose dolphin population sizes for both the un-impacted and the impacted populations for the in-combination projects.*

### Underwater noise impacts from construction activities (other than piling)

3615. The OWF screened in for other construction activities that could have potential in-combination impacts with other construction activities at the Project were:

- Codling (PINS Tier 2)
- Dublin Array (PINS Tier 2)
- North Irish Sea Array (PINS Tier 2)

3616. During the construction of the Project, there would be the potential for overlap with impacts from the non-piling construction activities at other offshore wind farms. Noise sources which could cause potential disturbance impacts during OWF construction activities, other than pile driving, could include vessels, seabed preparation, cable installation works and rock placement (see **Appendix 11.4**).

3617. The potential impact area for bottlenose dolphin has been based on the worst-case disturbance range of 4km (50.27km$^2$), which included construction activity and vessels (see **Section 9.5.2**). As a precautionary approach it has been assumed up to four construction activities (other than piling) could be underway at each OWF, including the Project.

3618. The in-combination disturbance effect of other construction activities for all OWFs would be up to 35.46 individuals, which represented 12.1% of the IS MU or 24% of the Cardigan Bay SAC.

*Table 9.28 Indicative in-combination assessment for the potential disturbance of bottlenose dolphin during construction activities, including vessels, other than piling*

| OWF | Bottlenose dolphin density (/km$^2$) | Impact area (km$^2$) | Maximum number of individuals potentially disturbed | % Population |
|---|---|---|---|---|
| Codling | 0.2352 | 50.27 | 11.82 | 4.0% of IS MU, 8% of SAC |
| Dublin Array | 0.2352 | 50.27 | 11.82 | 4.0% of IS MU, 8% of SAC |
| North Irish Sea Array | 0.2352 | 50.27 | 11.82 | 4.0% of IS MU, 8% of SAC |
| **Total** | | | **35.46** | 12.1% of IS MU; 24% of SAC |

### Disturbance from other industries and activities

3619. Section 11.7.3.1 of **Chapter 11 Marine Mammals** of the ES considers the effects from geophysical surveys; seismic surveys and UXO clearance.

3620. The assessments have been considered together here. No aggregate extraction or dredging was anticipated within the screening area for bottlenose dolphin and therefore this was not considered for in-combination effect (see **Appendix 11.4**)

3621. For the geophysical survey, the worst-case assumed disturbance from two geophysical surveys using an impact area of 707km$^2$. This indicated that approximately 7.4 bottlenose dolphin individuals or 2.5% of the IS MU (5% of the Cardigan Bay SAC) could be disturbed (see Table 11.97 of **Chapter 11 Marine Mammals** of the ES).

3622. For seismic surveys, the worst-case assumed disturbance from a single survey with a potential impact area based on a potential disturbance range of 11km and a survey length of 103.5km, the impact area for one seismic survey was suggested to be 1,518.63km$^2$. This indicated that approximately 15.8 individuals or 5.4% of the IS MU (or 10.7% of the Cardigan Bay SAC) could be disturbed (see Table 11.102 of **Chapter 11 Marine Mammals** of the ES). However, it is noted that there were no known licences or licence applications for seismic surveys at the time of assessment and this has been included for information purposes at this stage.

3623. No aggregate extraction or dredging occurs within the relevant MUs, therefore, this was not considered for in-combination effect (see **Appendix 11.4**).

3624. The worst-case modelled impact ranges for bottlenose dolphin at the Project for TTS/fleeing response (using the impulsive unweighted $SPL_{peak}$) was of 1.1km (3.8km$^2$) for high-order clearance, and 0.13km (0.053km$^2$) for low-order clearance. So, as the worst-case, to represent the wider potential impact area for two low-order UXO clearance events, was based on the 5km disturbance range for harbour porpoise, as a precautionary measure (JNCC, 2023a). The potential effect area during two low-order UXO clearance events was based on a precautionary disturbance range of 5km. This indicated that approximately 1.6 individuals or 0.55% of the IS MU (or 1.1% of the Cardigan Bay SAC) could be disturbed (see Table 11.106 of **Chapter 11 Marine Mammals** of the ES).

3625. As outlined in the BEIS[36] (2020) guidance, due to the nature of the sound arising from the detonation of UXO (i.e. each blast lasting for a very short duration), marine mammals, including bottlenose dolphin, were not predicted to be significantly displaced from an area. Any changes in behaviour, if they occurred, would be an instantaneous response and short-term. The latest guidance suggested that disturbance behaviour was not predicted to occur from UXO clearance if undertaken over a short period of time (JNCC, 2010b).

### Summary of disturbance effects during construction

3626. For bottlenose dolphin, for all noisy activities with the potential for in-combination disturbance effects, up to 23.4% of the IS MU population (41.8% of the SAC population) would be at risk of disturbance at any one time (**Table 9.29**), with the largest proportion of the impact coming from construction activities with other OWFs.

[36] As of February 2023, BEIS is known as the DESNZ

3627. Based on the worst-case in-combination assessment, more than 5% of the reference population would be affected (outside any SAC). As such, this would suggest a potential for significant effects from disturbance during construction. However, this was considered to not be realistic as:

- The assessment assumed the five in-combination projects (mostly Tier 2 projects where scheduling was uncertain) were all piling at exactly the same time as the Project and each using monopiles. Experience from the build-out of projects in the Southern North Sea suggests that this is unrealistic. Piling durations are typically overestimations, further limiting temporal overlap
- The speculative (or indicative) assessments for construction activities at other OWF; geophysical surveys; seismic surveys and UXO clearance assumed all of these activities would occur simultaneously and no mitigation had been applied. Any need for scheduling or further mitigation would be established closer to construction when the timings of projects would be more realistic
- There was no spatial overlap of effects with any SAC.
- Not all individuals would be displaced over the entire potential disturbance range used within the assessments
- Behavioural effects from UXO clearance, if they occur, would be an instantaneous response and short-term. Guidance suggested that disturbance behaviour was not predicted to occur from UXO clearance if undertaken over a short period of time (JNCC, 2010b)
- The Plan Level HRA noted that disturbance would be negligible (and there was no AEoI) "*because the site lies more than 26km from the closest Preferred Project"* (The Crown Estate, 2022)

*Table 9.29 Quantified in-combination assessment for the potential disturbance of bottlenose dolphin from all underwater noise sources during construction (Grey rows are projects and activities that may take place and therefore indicative assessments)*

| Impact | Bottlenose dolphin |
|---|---|
| Piling at other OWFs including the worst-case disturbance from the Project | iPCoD modelling undertaken, <1% population level effect over both the first six years and 25 year modelled periods. |
| Construction activities at other OWF | 12.1% of IS MU; 24% of SAC |
| Geophysical surveys | 2.5% of IS MU; 5% of SAC |
| Seismic surveys | 5.4% of IS MU; 10.7% of SAC |
| UXO clearance | 0.54% of IS MU; 1.1% of SAC |
| **Total for all projects that were currently (or expected to be) in the planning process (realistic worst-case scenario)** | |
| **Total** | **<1% of the IS MU and the SAC population** |

3628. When considering the population modelling undertaken for the Project, less than 5% of the reference and SAC population, it suggested there would be no potential for significant effects from disturbance during construction at the Project (and no AEoI on any SAC).

### Disturbance from underwater noise during operation and maintenance

3629. Underwater noise and disturbance during operation and maintenance could come from multiple sources, from operational noise from WTGs, noise of major maintenance work, rock placement or cable repairs and from the presence of vessels as well as other industries.

3630. Given the distance of the Project windfarm site from the coast, there was no indication that there would be a significant effect on the bottlenose dolphin population, which prefer coastal waters, from the Project.

3631. A review of the most recent studies was used to determine the potential disturbance from underwater operational noise from WTGs (see Section 11.6.4.1 of **Chapter 11 Marine Mammals** of the ES). The studies indicated that any disturbance would be in the immediate area of the operational WTGs, depending on ambient noise levels. There was no evidence of any lasting disturbance or exclusion around OWFs during operation and maintenance and bottlenose dolphin have been frequently observed in and around the Aberdeen Offshore Windfarm (European Offshore Wind Deployment Centre; pers. comm.).

3632. Effects from maintenance activities at OWFs, such as additional rock placement or cable re-burial, would be very localised, short in duration and temporary.

3633. **Operational noise from OWF WTGs and maintenance of windfarm infrastructure was screened out from further consideration within the CEA screening and has not been considered further in this in-combination assessment.**

### Underwater noise from decommissioning

3634. The potential for in-combination impacts during the decommissioning of the Project is unknown. It is not possible to provide details of the methods that would be used during decommissioning at this time. However, it is expected that the activity levels would be comparable to construction (with the exception of pile driving noise which would not occur).

3635. During decommissioning, the potential effects on bottlenose dolphin are anticipated to be similar or less than the worst-case for the construction phase (depending on the methods used). Crucially, any in-combination effect would be dependent upon simultaneous decommissioning and it is not possible to provide a realistic estimate of which projects may be decommissioned and when.

3636. **The potential impacts for the decommissioning were screened out from further consideration within the CEA screening and have not been considered further in this in-combination assessment.**

#### 9.5.4.2 Barrier effects as a result of underwater noise

3637. Bottlenose dolphin have a predominantly coastal distribution (see **Appendix 11.2**), with most sightings within 10km of land. It is therefore considered that underwater noise from the Project would be unlikely to result in any barrier effects to bottlenose dolphin. This would also apply to any non-coastal project and limited the potential for in-combination barrier effects upon bottlenose dolphin.

3638. Given that the Project would be unlikely to materially contribute to any in-combination effect and that most other projects would be likewise located away from the coast, it is considered that there would be no potential for an in-combination barrier effect to occur.

3639. During construction, barrier effects arising through loud noise would be intermittent and limited to just a fraction of the 2.5 year construction window; whilst during operation it would be limited to the Project boundaries. The long distances between the Project and the SACs for bottlenose dolphin would allow for animals to move freely and not encounter barriers through physical turbine presence, nor from the generated noise.

3640. Given the limited spatial effect, **it is considered that there would be no potential for an in-combination barrier effect from underwater noise or physical presence and no LSE on the bottlenose dolphin IS MU population, nor the Cardigan Bay SAC population (and no AEoI on any SAC) from any potential barrier effects.**

#### 9.5.4.3 Vessel interactions

3641. Given the low risk of collision to bottlenose dolphin and the use of best practice mitigation to manage residual risk**, it is concluded that there would be no LSE from the Project on the IS MU and SAC bottlenose dolphin population from vessel interactions during construction, operation and maintenance or decommissioning.** It was considered that any consented OWF project would require similar mitigation which would reduce collision risks.

3642. Vessels associated with aggregate extraction and dredging are large and typically slow moving, using established transit routes to and from ports.

Therefore, the potential increased collision risk with vessels was considered to be extremely low or negligible.

3643. Given the low risk to bottlenose dolphin and the use of mitigation across OWF projects, **it is concluded that there would be no LSE on the IS MU or SAC bottlenose dolphin population (and no AEoI on any SAC) from the effects of vessel interactions during construction, operation and maintenance or decommissioning.**

#### 9.5.4.4 Changes to prey resources

3644. No significant impacts with regard to changes to prey resources are expected as a result of the Project (**Section 9.4.3**).

3645. For any potential changes to prey resources, it has been assumed that any potential effects on marine mammal prey species from underwater noise, including piling, would be the same or less than those for marine mammals. Therefore, there would be no additional in-combination effects above those assessed for marine mammals, i.e. if prey were disturbed from an area as a result of underwater noise, marine mammals would be disturbed from the same or greater area. As a result, any changes to prey resources would not affect marine mammals as they would already be disturbed from the area.

3646. Any effects to prey species were likely to be intermittent, temporary and highly localised, with the potential for recovery following the cessation of the disturbance activity. Any permanent loss or changes of prey habitat would typically represent a small percentage of the potential habitat for prey species in the surrounding area.

3647. **There would be no potential for a significant effect on the IS MU or SAC bottlenose dolphin population (and no AEoI on any SAC) during construction, operation and maintenance or decommissioning.** This conclusion took into account the assessment for the Project-alone, and assumed similar effects for other projects and activities, along with the range of prey species taken by bottlenose dolphin and the extent of their foraging ranges together with the fact that much of the effect would occur outside of any area considered important for foraging (i.e. outside of a marine protected area for the species).

#### 9.5.4.5 Changes to water quality

3648. No significant impacts with regard to water quality are expected as a result of the Project (**Section 9.4.3**).

3649. Other OWFs or other construction projects are also considered to have highly localised and temporary effects and would be spatially separated so there would be no potential for additive effects.

3650. Given that water quality effects would be negligible**, it is concluded that there would be no significant in-combination effect on the IS MU or Cardigan Bay SAC population (and no AEoI on any SAC) during construction, operation and maintenance or decommissioning.**

#### 9.5.4.6 Summary of in-combination assessment

3651. Effects from the Project would not overlap any SAC and the plans, projects and activities included in the in-combination assessment have been screened in on the basis that they were in the same MU, rather than overlap or proximity with any SAC. It should be noted that in the plan level HRA (NIRAS, 2021) only projects within 26km of an SAC were considered to contribute to disturbance effects and therefore, no adverse effect on integrity was predicted for any bottlenose dolphin SAC at the plan level. Results were therefore highly precautionary, particularly considering the low likelihood of temporal overlap.

3652. Due to mitigation outlined for all projects it was considered that permanent effects upon bottlenose dolphin would be avoided.

3653. When assessing the potential in-combination effect during construction, there was the potential for more than 5% of the bottlenose dolphin IS MU and SAC reference population to be disturbed during construction when including speculative activities without mitigation. However, when looking at known projects and impacts using precautionary levels from dose response curve assessments and including them in population modelling, there would be a less than 1% effect at both the IS MU and SAC level.

3654. For operation and maintenance and decommissioning, less than 5% of the bottlenose dolphin reference and SAC population would be disturbed, thus there was no potential for significant effect on the IS MU or SAC population. Furthermore, considering the distances of the projects in the in-combination assessment to any of the SACs considered, none would have disturbance effects which would occur within an SAC.

3655. Indirect effects (i.e. on water quality or prey resources) were considered to be insignificant and would occur outside any SAC boundary.

3656. None of the effects would be within an SAC or were considered to have a significant effect on the IS MU bottlenose dolphin population. **As such, it is considered that there would be no adverse effect on integrity on the Pen Llŷn a'r Sarnau SAC and Cardigan Bay SAC in relation to the conservation objectives relating to 'Population' and 'Range'.**

3657. Indirect effects (i.e. on water quality or prey resources) were considered to be insignificant in-combination and would occur outside any SAC boundary. **As such, it is considered that there would be no adverse effect on integrity on the Pen Llŷn a'r Sarnau SAC and Cardigan Bay SAC in relation to the**

**conservation objectives relating to 'Range' and 'Supporting habitats and species'.**

3658. The confidence in the assessment for all impacts is considered medium, yet highly precautionary, particularly given the consideration of a large number of plans or projects and the unlikelihood of temporal overlap of all these activities.

# 9.6 Grey seal

## 9.6.1 Relevant sites

### 9.6.1.1 Pen Llŷn a`r Sarnau SAC

#### Description of designation

3659. The Pen Llŷn a'r Sarnau SAC encompasses areas of sea, coast and estuary, and supports a significant presence of grey seal. The site covers 1,460km$^2$ including areas of coastal lagoons, shallow inlets and bays, estuaries, reefs and sandbanks.

3660. The SAC is 97km from the Project site, measured as a straight line distance, or 128km as coastline distance.

#### Grey seal population and density

3661. Grey seals present within the SAC at any one time do not form a discrete population but are considered part of the SW England and Wales MU. The breeding colonies at Pen Llŷn and Bardsey Island are the larger sites within North Wales, with a number of important sites located in the north of the SAC.

3662. The grey seal density estimates for the Project have been calculated from the SAC specific seal at-sea usage maps (Carter *et al.,* 2022) based on the grids that overlapped with the Project. The mean at-sea density estimate used in the assessments was:

- 0.00011 individuals per km$^2$ for the Project windfarm site and 4km buffer

3663. As per latest NRW guidance (Sinclair *et al*., 2023) there is no SAC specific grey seal population figure, but they are part of the SW England and Wales MU. The adjusted population estimates for the relevant MUs for grey seal were:

- SW England MU = 1,988 grey seal (SCOS, 2022)
- Wales MU = 3,579 grey seal (SCOS, 2022)

3664. The total reference population for the assessment was 5,567 grey seal.

### Conservation status

3665. The most recent assessment for the conservation status of the grey seal was conducted in 2017 and found the species to be in a Favourable condition (NRW, 2018b).

### Conservation objectives

3666. To achieve favourable conservation status all the following conservation objectives, subject to natural processes, would need to be fulfilled and maintained in the long-term. If these objectives were not met, restoration measures would be needed to achieve favourable conservation status.

*Conservation objective 1: Populations*

3667. The population is maintaining itself on a long-term basis as a viable component of its natural habitat. Important elements include:

- Population size
- Structure, production
- Condition of the species within the site.

3668. As part of this objective it should be noted that for bottlenose dolphin and grey seal:

- Contaminant burdens derived from human activity should be below levels that may cause physiological damage, or immune or reproductive suppression
- For grey seal, populations should not be reduced as a consequence of human activity.

*Conservation objective 2: Range*

3669. The species population within the site is such that the natural range of the population is not being reduced or likely to be reduced for the foreseeable future. As part of this objective it should be noted that for bottlenose dolphin and grey seal:

- Their range within the SAC and adjacent inter-connected areas should not be constrained or hindered
- There should be appropriate and sufficient food resources within the SAC and beyond
- The sites and amount of supporting habitat used by these species should be accessible and their extent and quality should be stable or increasing

*Conservation objective 3: Supporting habitats and species*

3670. The presence, abundance, condition and diversity of habitats and species required to support this species is such that the distribution, abundance and populations dynamics of the species within the site and population beyond the site is stable or increasing. Important considerations include:

- Distribution
- Extent
- Structure
- Function and quality of habitat
- Prey availability and quality.

3671. As part of this objective it should be noted that:

- The abundance of prey species subject to existing commercial fisheries needs to be equal to or greater than that required to achieve maximum sustainable yield and secure in the long term
- The management and control of activities or operations likely to adversely affect the species feature is appropriate for maintaining it in favourable condition and is secure in the long term
- Contamination of potential prey species should be below concentrations potentially harmful to their physiological health
- Disturbance by human activity is below levels that suppress reproductive success, physiological health or long-term behaviour.

3672. For the purposes of the assessment, the potential effects have been considered in relation to the Pen Llŷn a'r Sarnau SAC conservation objectives, as outlined in **Table 9.30**.

*Table 9.30 Potential effects in relation to the conservation objectives for the Pen Llŷn a'r Sarnau SAC for grey seal*

| Conservation objective | Potential effect |
|---|---|
| Populations/Range | Physical and permanent auditory injury from piling would be mitigated, however this was considered in detail in line with current advice. |
| Populations/Range | Significant disturbance and displacement as a result of increased underwater noise levels (e.g. or piling) have the potential to affect grey seal. |
| Populations/Range | Increased collision risk with vessels has the potential to affect grey seal. |
| Range/Supporting habitats and species | Changes in water quality and prey availability have the potential to affect grey seal. |

#### 9.6.1.2 Cardigan Bay SAC

3673. This site supports the same grey seal population due to interconnectivity with the Pen Llŷn a'r Sarnau SAC and has therefore identical conservation status and objectives.

3674. Grey seal is graded as C[37] in this site and, due to the distance from the Project (158km), had a very low density of 0.000006 individuals per km$^2$ for the windfarm site.

3675. It has not been further assessed in the RIAA, as the effects arising from the Project-alone would be less than those assessed for Pen Llŷn a'r Sarnau SAC (**Section 9.6.1.1**) due to the lower density and increased distance.

#### 9.6.1.3 Pembrokeshire Marine SAC

##### Description of designation

3676. Pembrokeshire Marine SAC is one of the largest marine designated sites in the UK. It is recognised to have a variety of habitats, from reefs to subtidal sandbanks, covering an area of 1,380km$^2$. The SAC is a multiple interest site that has been selected for the presence of eight marine habitat features and seven species features, including the grey seal.

3677. The closest point to the Project windfarm site would be approximately 229km from the Pembrokeshire Marine SAC, measured in a straight line, or 254km when measured as a coastline distance.

3678. This SAC could be well within foraging range for grey seals (Carter *et al.*, 2022).

##### Grey seal population and density

3679. Within the Pembrokeshire Marine SAC site selection document, grey seal was a qualifying species and a primary reason for the site selection. Based on pup production estimates, the Welsh 'population' forms around 3.3% of the UK or about 2.7% of the European population. The Pembrokeshire coast contains the main colony in Wales and is the most southerly in Europe of any significant size (Baines *et al*., 1995).

3680. This population itself is not isolated but extends from SW England to SW Scotland and SE Ireland.

[37] Features that are of national importance, but which occur on sites primarily selected for other (A or B grade) features are listed but are not given site accounts for reasons of space. These "secondary" features are graded C when SACs are submitted to the EC.

3681. Grey seal pup production within the Pembrokeshire Marine SAC has increased over the last decade or more (Bull *et al*., 2021) and occurs from August to December, with the peak of the pupping season becoming earlier over the observed period with no indication of reaching carrying capacity (Bull *et al*., 2021). Annual pup production within the site is approximately 980 births, which is approximately 75% of the south-west Wales population, with the largest breeding sites being associated with Ramsey and Skomer islands.

3682. The grey seal density estimates have been calculated from the SAC specific seal at-sea usage maps (Carter *et al.,* 2022) based on the grids that overlapped with the Project. The mean at-sea density estimate used in the assessments was:

- 0.00000009 individuals per km$^2$ for the Project windfarm site and 4km buffer

3683. As per latest NRW guidance (Sinclair *et al*., 2023) there was no SAC specific grey seal population, but they would form part of the SW England and Wales MU. The adjusted population estimates for the relevant MUs for grey seal were:

- SW England MU = 1,988 grey seal (SCOS, 2022)
- Wales MU = 3,579 grey seal (SCOS, 2022)

3684. The total reference population for the assessment was 5,567 grey seals.

### Conservation status

3685. The most recent assessment for the conservation status of the grey seal was conducted in 2017 and found the species to be in a Favourable condition (NRW, 2018e).

### Conservation objectives

3686. The conservation objectives are the same as for other Welsh sites such as the Pen Llŷn a'r Sarnau SAC and have been described in detail in **Section 9.5.1.1**.

## 9.6.2 Project-alone assessment

3687. For the following assessments the SAC specific densities and reference populations (**Table 9.31**) have been used as discussed under **Section 9.6.1**.

*Table 9.31 SAC specific densities and reference populations used in the RIAA*

| SAC | Density | Reference population[38] | Source |
|---|---|---|---|
| Pen Llŷn a`r Sarnau | 0.00011 | 5567<br>(SW England MU + Wales MU | Sinclair *et al.*, (2023) (for SAC reference population) |
| Cardigan Bay | 0.000006 | | |
| Pembrokeshire Marine | 0.00000009 | | |

3688. Although densities have been provided for all three SAC sites, the following assessments have been made on the nearest and worst-case density from the Pen Llŷn a`r Sarnau SAC. All three sites shared the same reference population and thus all effects assessed for Pen Llŷn a`r Sarnau SAC would be the same or less for Cardigan Bay SAC and Pembrokeshire Marine SAC.

### 9.6.2.1 Underwater noise

3689. The assessment below refers to the same impact ranges used in the ES assessment, however the reference population and density estimates used in this assessment have been adapted to be SAC specific.

#### Permanent auditory injury from underwater noise during piling

3690. The effect would be relevant to the construction phase only, with effects occurring outside any SAC.

3691. High exposure levels from underwater noise sources can cause auditory injury or hearing impairment, taking the form of a permanent loss of hearing sensitivity (PTS). The potential for auditory injury is not just related to the level of the underwater sound and its frequency relative to the hearing bandwidth of the animal, but it is also influenced by the duration of exposure.

3692. Underwater noise modelling was carried out (**Appendix 11.1**) to predict the noise levels likely to arise during impact piling and other activities. The modelled impact ranges were used to determine the potential effects on marine mammals. A detailed explanation of the modelling, inputs and assumptions has been provided in Section 11.6.3.1 of **Chapter 11 Marine Mammals** of the ES.

3693. Several scenarios were modelled to determine the worst-case for PTS effects for monopiles and pin-piles including cumulative exposure of sequential piling of three monopiles or four pin-piles. As per modelled impact ranges for

[38] Correction factor applied to those at sea and not available to count (SCOS, 2022)

monopiles, the piling of three sequential piles was nearly the same as that for single pile installation and has been taken forward as the worst-case.

3694. The maximum predicted impact range for PTS was up to 0.95km from cumulative exposure ($SEL_{cum}$) during the monopile installation with maximum hammer energy (6,600kJ) without any additional mitigation (**Table 9.32**) Given the distance of the nearest Project windfarm site from the SAC (97km as straight line, or 128km as coastline distance), there was therefore no pathway for PTS upon grey seal within any of the SACs.

*Table 9.32 Predicted PTS impact ranges (and areas) at the Project from a single strike and from cumulative exposure for maximum hammer energy (taken from Table 11.21 of the ES)*

| Impact | Criteria and threshold (Southall *et al.*, 2019) | Monopile Maximum impact range (km) and area ($km^2$) | Monopile (sequential piling) Maximum impact range (km) and area ($km^2$) | Pin-pile Maximum impact range (km) and area ($km^2$) | Pin-pile (sequential piling) Maximum impact range (km) and area ($km^2$) |
|---|---|---|---|---|---|
| | | Maximum hammer energy (6,600kJ) | | Maximum hammer energy (2,500kJ) | |
| PTS from single strike (without mitigation) | $SPL_{peak}$ Unweighted (218 dB re 1μPa) Impulsive | 0.06km (0.01$km^2$) | *N/A* | <0.05km (0.01$km^2$) | *N/A* |
| PTS from cumulative SEL (including soft-start and ramp-up) | $SEL_{cum}$ Weighted (185 dB re 1μ$Pa^2$s) Impulsive | 0.95km (1.9$km^2$) | 0.98km (2.0$km^2$) | <0.1km (<0.1$km^2$) | <0.1km (<0.1$km^2$) |

3695. An assessment of the maximum number of individuals and percentage of the reference population affected under each of the scenarios was then undertaken using the SAC specific densities and reference population (**Table 9.33**). The worst-case density was from the closest SAC in relation to the Project, Pen Llŷn a`r Sarnau SAC (**Table 9.31**). To assess the effects of permanent auditory injury from underwater sounds, only this density has been taken forward for assessment. It was highly unlikely that there would be any effects on the remaining SACs, located further away and with lower densities associated with the Project (**Table 9.31**).

3696. Given the short impact ranges modelled for grey seal (the maximum being <1km) and the low SAC densities, the number of animals that would be affected was extremely low and insignificant (**Table 9.33**). Given the embedded mitigation, **it has been concluded that there would be no LSE on the reference population (and no AEoI on the SAC) from PTS**

3697. The final approved MMMP would reduce the risk of PTS still further. The final MMMP for piling would be based on the Draft MMMP (Document Reference 6.5) which has been included with the DCO Application.

*Table 9.33 Maximum number of individuals (and % of reference population) from the Pen Llŷn a`r Sarnau SAC (worst-case SAC) that could be at risk of PTS from single strike and from cumulative exposure ($SEL_{cum}$) of monopile or pin-pile and the respective cumulative exposure*

| Criteria and threshold (Southall *et al*., 2019) | Monopile Maximum impact range (km) and area ($km^2$) | Monopile (sequential piling) Maximum impact range (km) and area ($km^2$) | Pin-pile Maximum impact range (km) and area ($km^2$) | Pin-pile (sequential piling) Maximum impact range (km) and area ($km^2$) |
|---|---|---|---|---|
| | Maximum hammer energy (6,600kJ) | | Maximum hammer energy (2,500kJ) | |
| **Single strike at maximum hammer energy** | | | | |
| $SPL_{peak}$ Unweighted (218 dB re 1µPa) Impulsive | 0.0000011 (0.00000002% of the SAC reference pop.) | *N/A* | 0.0000011 (0.00000002% of the SAC reference pop.) | *N/A* |
| **Cumulative exposure ($SEL_{cum}$) during installation** | | | | |
| $SEL_{cum}$ Weighted (185 dB re 1µPa$^2$s) Impulsive | 0.00021 (0.000004% of the SAC reference pop.) | 0.0002 (0.000004%of the SAC reference pop.) | 0.000011 (0.0000002% of the SAC reference pop.) | 0.000011 (0.0000002% of the SAC reference pop.) |

### Disturbance impacts from underwater noise during piling

3698. The impact of underwater noise generated by piling in this area has been evaluated, considering a precautionary disturbance range of 25km around a monopile (Russell, 2016). While there are different methods to assess disturbance (such as dose-response curves, population modelling, *etc*.), the choice to use the 25km range has been informed by the **Chapter 11 Marine Mammals** of the ES. Particularly, the site-specific densities of grey seals were significantly higher than those found in the SACs. Among all the options considered for the disturbance assessment, the 25km range represented the worst-case scenario.

3699. As outlined in **Section 9.6.1**, the nearest SAC, Pen Llŷn a`r Sarnau, is 97km (as straight line, or 128km as coastline distance) from the Project windfarm site and therefore has no potential overlap with it, or other SACs. There would be no direct effects on grey seal within the SACs, but there may be effects on grey seal foraging outside the SAC boundaries.

3700. **Table 9.34** assesses the maximum number of individuals and percentage of the reference population potentially affected.

*Table 9.34 Maximum number of grey seal from the Pen Llŷn a`r Sarnau SAC (worst-case SAC) (and % of SAC reference population) that could be disturbed during piling at the Project based on a disturbance range of 25km*

| SAC | 25km disturbance range (1963.5km$^2$) for monopile<br>Maximum number of individuals (% of reference population) |
|---|---|
| Pen Llŷn a`r Sarnau | 0.2 (0.004% of the SAC reference pop.) |

3701. The disturbance effect would occur outside of any SAC and affect less than 5% of the SAC reference population. **It is concluded that there would be no LSE on the reference population (and no AEoI on any SAC) from disturbance from underwater noise during piling.**

### Underwater noise and disturbance from other sources

#### Construction

3702. Section 11.6.3.3 of **Chapter 11 Marine Mammals** of the ES details the effects of disturbance impacts from underwater noise from seabed preparation, dredging, trenching, cable installation and rock placement. Section 11.6.3.4 of **Chapter 11 Marine Mammals** of the ES details the effects of underwater noise from the presence of vessels.

3703. The insignificant effect of piling on grey seal SAC populations has been highlighted in the preceding assessment on permanent injury from piling. It was deemed unnecessary to repeat an assessment for noise from other sources, which would have even smaller impact ranges. Furthermore, the effect of TTS has been screened out as it would have no permanent effect on the conservation objectives listed in **Section 9.6.1**. This section has therefore examined the effect of disturbance only.

3704. A review of various studies was used to determine the maximum potential disturbance range for other construction activities and vessels. During the construction of two Scottish windfarms (Beatrice OWF and Moray East OWF), (Benhemma-Le Gall *et al*., 2021), a reduction in harbour porpoise presence was reported up to 4km (50.27km$^2$) distance from construction vessels. This distance has been used as the disturbance range for other construction activities, including vessels.

3705. Therefore, as agreed through the EPP and ETGs the assessments for grey seal have been based on the same disturbance impact range of 4km (50.27km$^2$) due to the absence of data on potential disturbance range for other construction activities and vessels.

3706. Given the potential distance of the Project windfarm site from the Pen Llŷn a`r Sarnau SAC (97km as straight line, or 128km as coastline distance), there was no pathway for disturbance effects directly upon grey seal within the site, but may extend to the seals foraging outwith the boundaries of the SAC site.

3707. As a precautionary approach, the potential disturbance from four activities (cable laying, dredging, trenching and rock placement) occurring at the same time, including vessels undertaking the work, has been assessed based on a maximum impact area of 100.54km$^2$ (assuming no overlap in the impact areas between the activities). The maximum number of disturbed grey seal associated with the relevant SAC has been summarised in **Table 9.35**.

3708. The assessment of disturbance from construction vessels has been detailed in Section 11.6.3.4 in **Chapter 11 Marine Mammals** in the ES. A 4km disturbance range has been used for 37 vessels, which would equate to a total impact area of 1859.8km$^2$ and present an unrealistic worst-case. This scenario did not take into account the overlap in the 4km disturbance range between vessels and the area was approximately 21 times the size than the Project site alone (87km$^2$).

3709. Additionally, according to Benhemma-Le Gall *et al.,* (2021) there were several ongoing construction activties and a variety of support vessels present during the porpoise detections in the study. This suggested that an assumption of 4km per vessel was altogether unrealistic. The assessment has been based on an impact area equivalent to the windfarm site, along with a 4km buffer (285.4km$^2$).

3710. The maximum number of disturbed grey seal associated with the relevant SAC are summarised in **Table 9.35**.

*Table 9.35 Maximum number of grey seal (and % of reference population) from the Pen Llŷn a`r Sarnau SAC that could be disturbed as a result of underwater noise associated with other (non-piling) construction activities at the Project.*

| Potential impact | Maximum number of grey seal (% of Pen Llŷn a`r Sarnau SAC reference population) |
|---|---|
| **Disturbance from other activities** | |
| One activity (50.27km$^2$) | 0.0056 (0.0001% of the SAC reference pop.) |
| Two activities (100.54km$^2$) | 0.011 (0.0002% of the SAC reference pop.) |
| **Disturbance from vessels** | |
| One vessel (50.27km$^2$) | 0.0056 (0.0001% of the SAC reference pop.) |
| 37 vessels within the windfarm site + 4km buffer (285.4km$^2$) | 0.032 (0.0006% of the SAC reference pop.) |

3711. Given that this effect would be minimal and would occur outside of any area considered important for foraging or breeding (i.e. a SAC), **it is concluded that there would be no LSE on the reference population (and no AEoI on any SAC) from underwater noise from other sources during construction.**

### Operation and maintenance

3712. Underwater noise and disturbance during operation could come from multiple sources, from operational noise from WTGs, noise of major maintenance work such as rock placement or cable repairs and from the presence of vessels. Each of these sources has been considered separately in detail in the following sections of **Chapter 11 Marine Mammals** of the ES: 11.6.4.1, 11.6.4.2 and 11.6.4.3. The conclusions of these assessments have been brought together for the purpose of this assessment.

3713. A review of the most recent studies was used to determine the potential disturbance of harbour porpoise from underwater operational noise from WTGs (see Section 11.6.4.1 of **Chapter 11 Marine Mammals** of the ES). The studies indicated that any disturbance would be in the immediate area of the operational turbine, depending on ambient noise levels. There was no evidence of any lasting disturbance or exclusion of harbour seal around OWFs during operation, with reports of harbour seal moving through and foraging within operational OWFs. It is therefore considered that the same would apply for grey seal.

3714. There is no LSE arising from the underwater noise and disturbance during piling. Following this, it is not anticipated that operational turbines would cause an effect greater than that of piling noise and therefore this impact has not been assessed further. As outlined previously, TTS would not have a long-lasting effect on the conservation objectives on the site and has been screened out.

3715. However, the disturbance from these maintenance activities, including vessels undertaking the work, would be more likely to affect grey seals when the precautionary range of 4km (Benhemma-Le Gall *et al.,* (2021) was applied.

3716. The potential disturbance from cable repairs and rock placement occurring at the same time has been assessed based on maximum impact area of 100.54km$^2$ (**Table 9.36**).

3717. Based on a standard year of maintenance, it was expected that up to three vessels could be on site at any given time. As such, an assessment of the number of animals from the relevant SACs that could potentially be disturbed by three vessels (150.81km$^2$) has been presented in **Table 9.35**.

*Table 9.36 Maximum number of grey seal (and % of reference population) from the Pen Llŷn a`r Sarnau SAC that could be disturbed as a result of underwater noise associated with other (non-piling) activities at the Project during operation and maintenance.*

| Potential impact | Maximum number of grey seal (% of reference population) |
|---|---|
| **Disturbance from other activities** | |
| One activity (50.27km$^2$) | 0.0056 (0.0001% of the SAC reference pop.) |
| Two activities (100.54km$^2$) | 0.011 (0.0002% of the SAC reference pop.) |
| **Disturbance from vessels** | |
| One vessel (50.27km$^2$) | 0.0056 (0.0001% of the SAC reference pop.) |
| Three vessels (150.81km$^2$) | 0.017 (0.0003% of the SAC reference pop.) |

3718. Given that this effect would be minimal and would occur outside of any area considered important for foraging or breeding (i.e. a SAC), **it is concluded that there would be no LSE on the reference population (and no AEoI on any SAC) the effects of disturbance impacts from underwater noise during operation.**

3719. Assessments were made on a standard maintenance year, but, given the low values, **it is anticipated that there would also be no LSE on the reference population (and no AEoI on the SAC) during a heavy maintenance year**.

**Decommissioning**

3720. Potential effects on grey seal associated with underwater noise during decommissioning have not been assessed in detail, as further assessments would be carried out ahead of any decommissioning works to be undertaken taking account of known information at that time, including relevant guidelines and requirements. The detailed Decommissioning Programme would provide details of the techniques to be employed and any relevant mitigation measures required.

3721. It is not possible to provide details of the methods that would be used during decommissioning at this time. However, it is expected that the activity levels would be comparable to construction (with the exception of pile driving noise which would not occur).

3722. During decommissioning, the potential effects on grey seal are anticipated to be similar or less than the worst-case for the construction phase (depending on the methods used). The effects would therefore be comparable to those described in construction.

3723. Given that this effect would be lower than for disturbance impacts from underwater noise during piling, and the effect would occur outside of any area considered important for foraging or breeding (i.e a SAC), **it is concluded that there would be no LSE on the reference population (and no AEoI on any SAC) from the effects of disturbance impacts from underwater noise during decommissioning.**

### 9.6.2.2 Barrier effects as a result of underwater noise

### Construction

3724. Underwater noise during construction could have the potential to create a barrier effect, preventing movement of grey seal between important feeding and/or breeding areas, or potentially increase swimming distances if grey seal avoid the area and go around it. This effect was considered in detail in Section 11.6.3.5 of **Chapter 11 Marine Mammals** of the ES.

3725. As outlined in **Chapter 11 Marine Mammals** of the ES, the two main grey seal haul-out sites in the NW England MU are at West Hoyle Bank (often referred to as Hilbre Island; approximately 45km from the Project) and at South Walney (approximately 35km from the Project) (SCOS, 2022).

3726. Taking into account the distance of the Project windfarm site from the coast and from grey seal haul-out sites, there would be no potential for underwater noise at the Project windfarm site to result in barrier effects to seals moving to and from haul-out sites.

3727. Underwater noise from piling and ADD use would be for a maximum of approximately 620 hours or 26 days in total (assuming 24-hour days, see Table 11.35 in **Chapter 11 Marine Mammals**) over the construction period of up to two and a half years. Other construction activities and vessels that could result in barrier effects would be temporary, not consistent throughout the offshore construction period, and would be limited to only part of the overall construction period and area at any one time. If there were potential barrier effects across the entire Project windfarm site ($87km^2$) this would be a small area in relation to the movements and foraging ranges of grey seal in and around the area. Although the modelled noise levels were larger than the Project site, a 25km disturbance range has been applied (informed by the **Chapter 11 Marine Mammals** of the ES; see **Section 9.7.2.1**) to assess the worst-case and only had a negligible effect on the Pen Llŷn a`r Sarnau SAC population.

3728. There was unlikely to be any significant long-term impact from any temporary barrier effects due to underwater noise, any areas affected would be relatively small in comparison to the range of marine mammals and any effects would not be continuous throughout the offshore construction period.

3729. The effect would occur outside of any area considered important for foraging or breeding (i.e a SAC). **It is concluded that there would be no LSE on the reference population (and no AEoI on any SAC) from the effects of disturbance impacts from barrier effects during construction.**

### Operation and maintenance and decommissioning

3730. No significant effect from TTS or disturbance impacts from underwater noise would be anticipated during operation and maintenance of the windfarm. Any behavioural responses or disturbance would be limited to the close vicinity of the operational turbines. The minimum spacing commitment between WTGs (**Table 9.4**) means there would be no potential for underwater noise around individual turbines to overlap.

3731. Taking into account the relatively small impact areas for underwater noise around operational WTGs, there is unlikely to be the potential for barrier effects to marine mammals as a result of operational noise.

3732. During decommissioning, the potential effects on grey seal are anticipated to be similar or less than the worst-case for the construction phase (depending on the methods used). The effects would therefore be comparable to those described in construction.

3733. Given that this effect would be lower than for disturbance impacts from underwater noise during construction and the effect would occur outside of any area considered important for foraging or breeding (i.e a SAC), **it has been concluded that there would be no LSE on the reference population (and no AEoI on any SAC) from the effects of disturbance impacts from barrier effects during operation or decommissioning.**

#### 9.6.2.3 Vessel interactions

##### Construction

3734. During the construction phase, there would be an increase in the number of vessels in the windfarm site. The maximum number of vessels that could be on the Project windfarm site at any one time has been estimated as up to a total of 37 vessels (**Table 9.4**). The number, type and size of vessels would vary depending on the activities taking place at any one time.

3735. This effect has been considered in detail in Section 11.6.3.6 of **Chapter 11 Marine Mammals** of the ES where Table 11.56 summarised the most recent available data of the Cetacean Strandings Investigation Programme (CSIP) record strandings of marine mammals in Wales and England for the relevant species and detailed the number of deaths caused by either vessel strike, or physical trauma with an unknown cause (which could be attributed to vessel strike).

3736. The collision risk rate (4.32% for grey seals) has been calculated based on the number of deaths attributed to vessels strike, or other physical trauma, which could have been caused by collision with a vessel, as a proportion of the total known causes of death for grey seal.

3737. As a result, approximately five grey seal from the SAC population (any SAC) could be at risk from vessel collision per year, with just under 0.1% of the population at risk (based on 2,583 annual vessel transits that would be associated with construction (**Table 9.4**). This would have a long-lasting effect that has been, in the worst-case, assumed to be lethal for the individuals. However, it was considered that the quantified assessment was highly precautionary. Marine mammals are able to detect and avoid vessels. However, vessel strikes have been known to occur, possibly due to distraction whilst foraging and socially interacting, or due to the marine mammals' inquisitive nature (Wilson *et al*., 2007).

3738. In 2016, a study was conducted to determine the likelihood of harbour seal injury occurring due to co-presence with large vessels within the Moray Firth (Onoufriou *et al.,* 2016) (see **Section 9.7.2.3**). This study concluded that there was no relationship between areas of high co-occurrence and incidences of injury. Whilst this conclusion was specific to harbour seals, it was considered unlikely that grey seal would be at increased collision risk with vessels during construction.

3739. Considering that the vessel movements would be between larger ports (to be confirmed prior to start of construction) and the Project, there would be no overlap of vessel routes with any SAC for grey seal. Taking into account the existing number of vessel movements in the area with the potential to cause disturbance, and that vessels within the windfarm would be stationary for much of the time or very slow moving, the actual risk was likely to be very low.

3740. As outlined in **Section 9.3.1,** the commitment to best practice mitigation measures would further reduce the potential risk of collision. The mitigation measures would be agreed with the relevant stakeholders and would be detailed within the PEMP.

3741. Given the low SAC densities of grey seals in the Project area, the low risk and the commitment to mitigation to reduce that risk further, **it is concluded that there would be no LSE on the reference population (and no AEoI on any SAC) from vessel interactions during construction.**

### Operation and maintenance and decommissioning

3742. The increased risk of collision with vessels during operation and maintenance would be less than assessed for the construction period. Approximately two grey seals (1.7), or 0.03% of the SAC reference population could be at risk from collision per year.

3743. During the operation and maintenance phase the maximum number of vessels that could be on the Project windfarm site at any one time has been estimated at up to a total of 10 vessels in a heavy maintenance year, with up to 832 vessel transits per year (**Table 9.4**). The number, type and size of vessels would vary depending on the activities taking place at any one time. The vessels in the Project windfarm site during operation and maintenance would be slow moving or stationary.

3744. During decommissioning, the potential effects on grey seal are anticipated to be similar or less than the worst-case for the construction phase. The effects would therefore be comparable to those described in construction.

3745. Given that this effect would be lower than for potential for collision with vessels during construction and the commitment to mitigation measures to reduce that risk further, **it is concluded that there would be no significant likely effects**

**on the reference population (and no AEoI on any SAC) from vessel interactions during operation or decommissioning.**

#### 9.6.2.4 Changes to prey resources

##### Construction

3746. The potential effects on prey species during construction can result from physical disturbance and loss of habitat, increased SSC and sediment deposition and underwater noise. **Chapter 10 Fish and Shellfish Ecology** of the ES, provides an assessment of these impact pathways on the relevant fish and shellfish species and concluded impacts of negligible to minor adverse significance in EIA terms. **Chapter 11 Marine Mammals** of the ES considers these effects in terms of potential indirect effects on grey seal (see Section 11.6.3.7 of **Chapter 11 Marine Mammals**).

3747. Any reductions in prey availability would be small scale, localised and temporary and occur in an area that would not be considered important for grey seal feeding. Therefore, it was considered highly unlikely that potential reductions in prey availability as a result of construction activities would result in detectable changes to the grey seal population.

3748. It is also important to note that there was unlikely to be any additional displacement of marine mammals as a result of any changes in prey availability during piling as marine mammals would already be disturbed from the area.

3749. Given that this effect would be limited and would occur outside of any area considered important for grey seal foraging (i.e a SAC), **it is concluded that there would be no LSE on the reference population (and no AEoI on any SAC) from the effects of changes to prey species during construction.**

##### Operation and maintenance and decommissioning

3750. Changes to prey resource during operation have been assessed in Section 11.6.4.7 of **Chapter 11 Marine Mammals** of the ES. As per construction, this assessment has been based upon the conclusions of **Chapter 10 Fish and Shellfish Ecology** of the ES and considered a range of potential impacts including permanent habitat loss, introduction of hard substrate and EMF, as well as the impacts considered for construction. Although new impacts have been considered for operation, some effects such as those of physical disturbance; increased SSC and sediment deposition; and underwater noise would be reduced when compared to construction. Therefore, it is considered highly unlikely that potential reductions in prey availability as a result of operational activities would result in detectable changes to grey seal populations.

3751. During decommissioning, the potential effects on grey seal are anticipated to be similar or less than the worst-case for the construction phase. Therefore, the effects would be comparable to those described in construction.

3752. Given that this effect would be limited and would occur outside of any area considered important for grey seal foraging (i.e a SAC), **it is concluded that there would be no LSE on the reference population (and no AEoI on any SAC) from the effects of changes to prey species during operation or decommissioning.**

### 9.6.2.5 Changes to water quality

#### Construction

3753. Disturbance of seabed sediments has the potential to lead to increases in SSCs and release of any sediment-bound contaminants (such as heavy metals and hydrocarbons that may be present within them) into the water column. The accidental release of contaminants (e.g. through spillage) also has the potential to affect water quality. During construction, there would be the potential for increased SSC. Section 11.6.3.8 of **Chapter 11 Marine Mammals** of the ES considered these effects in detail.

3754. Throughout the construction phase, best practice techniques and due diligence regarding the potential for pollution would be followed throughout all construction activities. Any risk of accidental release of contaminants (e.g. through spillage) would be mitigated in line with the PEMP and any changes to water quality as a result of any accidental release of contaminants (e.g. through spillage or vessel collision) would be negligible. Therefore, the potential for pollutants to be released into the environment has not been considered further in this assessment.

3755. Section 11.6.3.8 of **Chapter 11 Marine Mammals** of the ES considers increases in SSC and remobilisation of existing contaminated sediments.

3756. Increased SSC would be unlikely to have any direct or indirect effects on marine mammals as they have been known to often inhabit turbid environments. Pinnipeds are likely to use other senses instead of, or in-combination with, vision to sense the environment around them. Studies have shown that vision was not essential to seal survival, or ability to forage (Todd *et al*., 2014).

3757. As outlined in **Chapter 8 Marine Sediment and Water Quality** of the ES, Project site specific data indicated that for all potential contaminants tested for within the sediments of the Project windfarm site concentrations were negligible. Therefore, there would be negligible potential for any direct or indirect effects on marine mammals from remobilisation of contaminated sediments.

3758. Given the distance of the Project windfarm site from the closest SAC (Pen Llŷn a'r Sarnau SAC is 97km as straight line, or 128km as coastline distance), there would be no pathway for water quality effects directly upon grey seal within the site.

3759. Given that water quality effects would be negligible, **it is concluded that there would be no LSE on the reference population (and no AEoI on any SAC) from the effects during construction.**

### Operation and maintenance and decommissioning

3760. During the operation and maintenance phase, there would be the potential for increases in SSC and the release of any sediment-bound contaminants. The scale of these impacts would be small, infrequent and of short-term duration and of a lower magnitude than during the construction phase.

3761. During decommissioning, the potential water quality effects are anticipated to be similar or less than the worst-case for the construction phase. Therefore, the effects would be comparable to those described in construction.

3762. Given that water quality effects would be negligible, **it is concluded that there would be no LSE on the reference population (and no AEoI on any SAC) from the effects during operation or decommissioning.**

## 9.6.2.6 Disturbance at haul out sites

### Construction

3763. The port(s) used to supply the Project would be confirmed post-consent, at this stage assumed within a 50km range and considered in relation to transit routes with regard to the seal haul out sides in the study area. The two main haul out sites at West Hoyle Bank (Hilbre Island) and at South Walney (SCOS, 2022) were outwith any grey seal SACs, and Project vessels would follow main shipping routes into the entry into any port. Best practice measures would be committed to and would be agreed through the PEMP. **It is concluded that there would be no LSE on the reference population (and no AEoI on any SAC) from disturbance at haul out sites.**

### Operation and maintenance and decommissioning

3764. As for construction, **there would be no LSE on any SAC reference population (and no AEoI) from disturbance at haul-out sites during operation and maintenance and decommissioning.**

## 9.6.2.7 Potential interactions of Project effects

3765. The anticipated effects on marine mammal receptors were not expected to interact in a way that would lead to a combined effect of greater significance than the assessments presented for each individual phase. It should also be noted that a high level of precautionary measures were implemented in the assessment process, further contributing to the overall understanding and mitigation of potential impacts.

3766. Interactions of Project effects as per those outlined in **Section 9.4.2.6**.

#### 9.6.2.8 Summary of Project-alone effects

3767. There would be no overlap of permanent, long-term and temporary noise impact ranges within any SAC.

3768. Due to embedded mitigation and commitment to securing mitigation measures (i.e PTS mitigation through the MMMP and to manage the residual low collision risk through best practice vessel practices secured in the PEMP), it was considered that permanent effects upon grey seal would be avoided during construction, operation and maintenance, and decommissioning.

3769. Given that disturbance to the grey seal reference population was lower than 0.1%, there would be no potential for significant effect on the SAC reference population during construction, operation or decommissioning.

3770. Indirect effects (i.e. on water quality or prey resources) were considered to be insignificant and would occur outside the SAC boundary and be minimal compared to the species range.

3771. None of the effects assessed would be within an SAC or were considered to have a significant effect on the SAC reference population of grey seal. As such, it is considered that there would be no adverse effect on integrity of the three SACs in relation to the conservation objective relating to ‘Population’ and ‘Range’.

3772. Indirect effects (i.e. on water quality or prey resources) were considered to be insignificant and would occur outside any SAC boundary. **As such, it is considered that there would be no adverse effect on integrity of the of the three SACs in relation to the conservation objective ‘Range’ and ‘Supporting habitats and species’.**

3773. The confidence in the assessment for all impacts is considered high considering the baseline information and site-specific data.

### 9.6.3 Potential in-combination effects of the Project with Transmission Assets

3774. A 'combined' assessment has been made with the Transmission Assets[39], for the purpose of an in-combination assessment considering its functional link with the Project.

3775. Pen Llŷn a'r Sarnau SAC and Pembrokeshire Marine SAC were screened in for both the Project and the Transmission Assets and The Maidens SAC, Lundy SAC and Isles of Scilly Complex SAC were screened in for Transmission Assets.

3776. For the Transmission Assets ISAA Project-alone assessment (Morgan Offshore Wind Limited and Morecambe Offshore Windfarm Ltd, 2023b), there would be no adverse effect on the site integrity on any of the screened-in sites, including those listed only for Transmission Assets. As for the Project, the distance to the closest SACs would all be outside the ZoI. A full quantitative assessment has been provided in the assessment of all plans and projects, including the Transmission Assets and has not been repeated here. An assessment has been made below of each impact, considering the information in **Section 9.6.4** and understanding the interactions between the projects.

#### 9.6.3.1 Underwater noise and barrier effects

3777. The key interaction was identified as piling and UXO during construction for the projects.

3778. Given that the Project and Transmission Assets would be outwith any SAC and potential PTS effects would be mitigated by any consented project, **it is concluded that there would be no LSE on the reference population (and no AEoI on the SAC).**

#### 9.6.3.2 Vessel interactions

3779. During all phases, there would be additional effects due to increased vessel presence from both projects.

3780. Given that the Project and Transmission Assets would be outwith any SAC and both projects would adhere to good practice, **it is concluded that there would be no LSE on the SAC reference population (and no AEoI on the SAC).**

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[39] As the Transmission Assets includes infrastructure associated with both the Project and the Morgan Offshore Wind Project Generation Assets, it should be noted that the combined assessment considers the transmission infrastructure for both the Project and the Morgan Offshore Wind Project Generation Assets.

#### 9.6.3.3 Indirect effects (changes to prey resource and water quality)

3781. During all phases, there could be additional effects due to increased vessel presence from both projects and additional pressure on prey resource.

3782. Given the impacts identified for both projects on prey species and that the Project and Transmission Assets would be outwith any SAC and both projects would adhere to good practice, **it is concluded that there would be no significant in-combination effect on the SAC reference populations (and no AEoI).**

#### 9.6.3.4 Disturbance at haul-out sites

3783. There were two grey seal breeding or haul out sites identified in the NW England MU (SCOS, 2022), West Hoyle Bank (often referred to as Hilbre Island) in the Dee estuary, in Cheshire (approximately 45km from the Project) and at South Walney, in Cumbria (approximately 35km from the Project.

3784. Given the distance to the haul-out sites and the SACs that were screened in and adherence to good practice, **it is concluded that there would be no LSE on the reference population (and no AEoI on the SAC) during construction, operation and maintenance or decommissioning.**

### 9.6.4 Assessment of the potential effects of the Project in-combination with other plans and projects

3785. Section 11.7 of **Chapter 11 Marine Mammals** of the ES details the CEA. This in-combination assessment has been based upon the cumulative assessment and provided a summary of the key information from that assessment without repeating every step of the process. Key information has been taken from **Chapter 11 Marine Mammals** of the ES and carried through with regard to the effect on designated sites.

3786. The effects screened into the in-combination assessment and the identification of the other plans, projects and activities that may result in in-combination effects have been provided in **Appendix 11.4**.

#### 9.6.4.1 Underwater noise

##### Permanent auditory injury from underwater noise during piling

3787. PTS could occur as a result of piling during OWF installation and detonation of underwater explosives (used occasionally during the removal of underwater structures and UXO clearance) (JNCC, 2010a,b). However, if there was the potential for any PTS, from any project, suitable mitigation would be put in place to reduce any risk to marine mammals. Other activities such as dredging, drilling, rock placement, vessel activity, operational windfarms, oil and gas installations or wave and tidal sites would emit broadband noise in lower frequencies and PTS from these activities would be very unlikely and would require mitigation for any potential effects.

3788. Therefore, the potential risk of PTS has not been considered further in the in-combination assessment.

3789. The Project would be outwith any SAC and there is no potential for AEoI from PTS onset in-combination with other projects, as all projects should ensure mitigation was in place prior to commencement of relevant works to negate the potential for PTS. **Therefore, the potential for PTS in-combination has been screened out and not assessed further.**

### Disturbance from underwater noise during construction

3790. Section 11.7.3.2 of **Chapter 11 Marine Mammals** of the ES considers disturbance in relation to several sub-effects and then considers all together, as follows:

- Underwater noise impacts from piling at other OWFs
- Underwater noise impacts from construction activities (other than piling) at other OWFs; and
- Disturbance from other industries and activities (which includes geophysical survey, seismic survey and UXO clearance).

3791. The results from the assessments have been displayed in **Table 9.37** and **Table 9.39**.

#### Disturbance from piling

3792. The potential disturbance from underwater noise during piling for grey seal has been assessed based on the disturbance range of 25km for the Project (Russell, 2016).

3793. This assessment considered the effect of all projects at a population level, noting the Project would not overlap with any SAC boundary.

3794. Of the UK and European OWFs screened in for having a construction period that could potentially overlap with the construction of the Project, six OWFs could be piling at the same time as the Project (**Appendix 11.4**):

- AyM OWF (PINS Tier 1)
- Mona OWF (PINS Tier 2)
- Morgan OWF (PINS Tier 2)
- Morgan and Morecambe OWFs Transmission Assets (PINS Tier 2)
- Erebus OWF (PINS Tier 1)

- White Cross OWF (PINS Tier 1)

3795. Of these projects, only Erebus OWF and White Cross OWF have been assessed for grey seals in the Pembrokeshire Marine SAC.

3796. This short list of OWF projects that could be piling at the same time as the Project could change as projects develop (noting that three projects were Tier 1, with the most certainty of development and schedule), but this was the best available information at the time of writing, and was considered to reflect the limitations and constraints to project delivery.

3797. The following caveats should be noted in terms of this worst-case:

- The potential areas of disturbance assume that there would be no overlap in the areas of disturbance between different projects
- It was assumed that all OWF projects would be 100% piled, if piled foundations were an option
- The approach has been based on the potential for single piling at each wind farm at the same time as single piling at the Project windfarm site. This approach allowed for some of the offshore wind farms not to be piling at the same time, while others could be simultaneously piling. This is considered to be the most realistic worst-case scenario, as it is highly unlikely that all other wind farms would be simultaneously piling at exactly the same time as piling at the Project, especially given the limited active piling time
- The actual duration for active piling time (a maximum of 619 hours and 36 minutes for the Project) which could disturb marine mammals would be only a very small proportion of the potential construction period, and this would be the case for other OWFs. This meant that there would be a limited window for any in-combination effect to occur
- In practice, the potential temporary effects would be less than those predicted in this assessment as there is likely to be a great deal of variation in timing, duration (noting this has been typically overestimated in assessments), and hammer energies used throughout the various OWF construction periods. This meant that there would be a limited window for temporal overlap and any in-combination effect to occur. In addition, not all individuals would be displaced over the entire potential disturbance range used within the assessments

*Table 9.37 Quantitative assessment for in-combination disturbance for grey seal from the Pen Llŷn a'r Sarnau SAC during piling at other projects*

| Project | Grey seal density (/km²) (based on Pen Llŷn a'r Sarnau SAC relative densities) | Effect area (km²) based on 25km disturbance range | Maximum number of grey seal potentially disturbed during single piling |
|---|---|---|---|
| The Project | 0.00011 | 1,963.5 | 0.22 |
| AyM | *Dose-response-curve assessment* | | 81 |
| Mona | *Dose-response-curve assessment* | | 45 |
| Morgan | *Dose-response-curve assessment* | | 31 |
| Morgan and Morecambe Transmission Assets | *Dose-response-curve assessment* | | 28 |
| Erebus | *These SACs were not assessed as part of their RIAA.* | | - |
| White Cross | | | - |
| **Total number of grey seal** | | | **185.22** |
| **Percentage of SAC population** | | | **3.33%** |

*Table 9.38 Quantitative assessment for in-combination disturbance for grey seal from Cardigan Bay SAC for during piling at other projects*

| Project | Grey seal density (/km²) (based on Cardigan Bay SAC relative densities) | Effect area (km²) based on 25km disturbance range | Maximum number of grey seal potentially disturbed during single piling |
|---|---|---|---|
| The Project | 0.000006 | 1,963.5 | 0.012 |
| AyM | *Dose-response-curve assessment* | | 81 |
| Mona | *Dose-response-curve assessment* | | 45 |
| Morgan | *Dose-response-curve assessment* | | 31 |
| Morgan and Morecambe Transmission Assets | *Dose-response-curve assessment* | | 28 |
| Erebus | *These SACs were not assessed as part of their RIAA.* | | - |
| White Cross | | | - |
| **Total number of grey seal** | | | **185.0** |
| **Percentage of SAC population** | | | **3.32%** |

*Table 9.39 Quantitative assessment for in-combination disturbance for grey seal from the Pembrokeshire Marine SAC during piling at other projects*

| Project | Grey seal density (/km$^2$) (based on Pembrokeshire Marine SAC relative densities) | Effect area (km$^2$) based on 25km disturbance range | Maximum number of grey seal potentially disturbed during single piling |
|---|---|---|---|
| The Project | 0.00000009 | 1,963.5 | 0.00018 |
| AyM | *Dose-response-curve assessment* | | 81 |
| Mona | *Dose-response-curve assessment* | | 45 |
| Morgan | *Dose-response-curve assessment* | | 31 |
| Morgan and Morecambe Transmission Assets | *Dose-response-curve assessment* | | 28 |
| White Cross | 0.004 | 1,963.5 | 7.85 |
| Erebus | *Dose-response-curve assessment* | | 18 |
| **Total number of grey seal** | | | **210.85** |
| **Percentage of SAC population** | | | **3.78%** |

3798. The effect of piling for all OWFs including the Project was quantified as potentially impacting upon approximately 185 grey seal or 3.3% of the Pen Llŷn a'r Sarnau and Cardigan Bay SAC population and assumed no significant in-combination effect on the feature from piling at the same time as the Project. Piling at the Project would cause no significant additional disturbance effect to grey seals from the SAC.

3799. The same applied for the Pembrokeshire Marine SAC, from which up to 211 grey seal individuals or 3.8% of the SAC population could be disturbed. The majority of disturbance data for other OWFs, however, came from other projects using site specific dose response assessments rather than SAC densities, the limitations of which have been discussed above. Each would need to have their own mitigation measures in place wherever there was a likely overlap with any relevant SAC boundaries.

### Underwater noise impacts from construction activities (other than piling)

3800. OWFs screened in for other construction activities that could have potential in-combination impacts with piling activities at the Project were (see **Appendix 11.4**):

- Codling Wind Park (PINS Tier 2)
- Dublin Array (PINS Tier 2)
- North Irish Sea Array (PINS Tier 2)

- Sceirde Rocks (PINS Tier 2) (would be outside the screening area for grey seal)

3801. During the construction of the Project, there would be the potential for overlap with impacts from the non-piling construction activities at other OWFs. Although, it is noted that these were Tier 2 projects and the certainty on scheduling, and thus temporal overlap, was low. Noise sources which could cause potential disturbance impacts during OWF construction activities (other than pile driving) could include vessels, mooring installation, seabed preparation, cable installation works and rock placement.

3802. The potential impact area for grey seal has been based on the worst-case disturbance range of 4km (50.27km$^2$), which included activities from construction vessels (see **Section 9.6.2.3**). As a precautionary approach it has been assumed that one construction activity (other than piling) could be underway at each OWF, including piling at the Project.

3803. The in-combination disturbance effect on grey seal from the Pen Llŷn a'r Sarnau SAC would be up to two grey seals, representing 0.03% of the SAC reference population (**Table 9.40**). The in-combination disturbance effect on grey seal from the Cardigan Bay SAC would be less than one grey seal, representing 0.003% of the SAC reference population (**Table 9.41**). The disturbance effect on grey seal from the Pembrokeshire Marine SAC would be very low with less than 0.001% of the SAC reference population affected (**Table 9.42**).

*Table 9.40 Indicative in-combination assessment for the potential disturbance for grey seal from the Pen Llŷn a'r Sarnau SAC during construction at other projects*

| Project | Grey seal density (/km$^2$) (based on based on Pen Llŷn a'r Sarnau SAC relative densities) | Effect area (km$^2$) | Maximum number of grey seal potentially disturbed during single piling |
|---|---|---|---|
| The Project | 0.00011 | 1963.5 | 0.22 |
| Codling Wind Park | 0.016 | 50.27 | 0.82 |
| Dublin Array | 0.010 | | 0.51 |
| North Irish Sea Array | 0.000041 | | 0.002 |
| Total number of grey seal | | | **1.6** |
| Percentage of SAC population | | | **0.03%** |

*Table 9.41 Indicative in-combination assessment for the potential disturbance for grey seal from the Cardigan Bay SAC during construction at other projects*

| Project | Grey seal density (/km$^2$) (based on Cardigan Bay SAC relative densities) | Effect area (km$^2$) | Maximum number of grey seal potentially disturbed during single piling |
|---|---|---|---|
| The Project | 0.000006 | 1963.5 | 0.012 |
| Codling Wind Park | 0.0023 | 50.27 | 0.12 |
| Dublin Array | 0.0011 | | 0.055 |
| North Irish Sea Array | 0.0000028 | | 0.00014 |
| Total number of grey seal | | | **0.183** |
| Percentage of SAC population | | | **0.0033%** |

*Table 9.42 Indicative in-combination assessment for the potential disturbance for grey seal from the Pembrokeshire Marine SAC during construction at other projects*

| Project | Grey seal density (/km$^2$) (based on based on Pembrokeshire Marine SAC relative densities) | Effect area (km$^2$) | Maximum number of grey seal potentially disturbed during single piling |
|---|---|---|---|
| The Project | 0.00000009 | 1963.5 | 0.00018 |
| Codling Wind Park | 0.00052 | 50.27 | 0.026 |
| Dublin Array | 0.00025 | | 0.013 |
| North Irish Sea Array | 0.0000016 | | 0.00008 |
| Total number of grey seal | | | **0.039** |
| Percentage of SAC population | | | **0.0007%** |

**Disturbance from other industries and activities**

3804. Section 11.7.3.2 of **Chapter 11 Marine Mammals** considers the effects from geophysical surveys, aggregate extraction and dredging, seismic surveys associated with other projects and UXO clearance.

3805. It should be noted that that there were no known licences or licence applications for seismic surveys (to overlap with Project construction) at the time of assessment and this potential combination has been included for information purposes at this stage.

3806. To establish a worst-case scenario for geophysical surveys, it has been assumed that seals within a 1km radius (equating to a total area of 3.1km²) might experience disturbance from this type of survey (BEIS[40], 2020). This projected disturbance would extend across an area of 420.2km² throughout the entire transit zone encompassing two surveys.

3807. For aggregate extraction and dredging, there were two projects screened in that could have potential in-combination disturbance effects with piling at the Project. For this type of activity, based on porpoise displacement (more details in **Section 9.4.2**), a disturbance range of 600m would result in a potential disturbance area of 1.13km$^2$ for each project, or up to 2.26km$^2$ for two aggregate projects.

3808. The potential impact area during a single UXO clearance event on grey and harbour seals, based on the modelled worst-case impact range at the Project for TTS/fleeing response (using the impulsive weighted SELs) of 16km (804.25km$^2$) for high-order clearance, and 0.8km (2.01km$^2$) for low-order clearance.

3809. **Table 9.43** presents an impact assessment for each of the listed industries and activities on grey seal based on the density from the Pen Llŷn a'r Sarnau SAC.

3810. Using the density based on the Pen Llŷn a'r Sarnau SAC (the worst-case) for the Project, the effects would not present any LSE on the population of the nearest SAC. Less than one grey seal and less than 0.012% of the Pen Llŷn a'r Sarnau SAC population would be disturbed if all activities were to occur at the same time.

*Table 9.43 Indicative in-combination assessment for the potential disturbance for grey seal from the Pen Llŷn a'r Sarnau SAC for other industries and activities at other projects*

| Activity | Grey seal density (/km$^2$) based on Pen Llŷn a'r Sarnau SAC relative densities) | Impact area (km$^2$) | Maximum number of individuals potentially disturbed | % SAC ref pop |
|---|---|---|---|---|
| Geophysical surveys x2 | 0.00011 | 420.2 | 0.05 | *0.0009%* |
| Aggregate extraction and dredging x2 | 0.00011 | 2.26 | 0.0003 | 0.000005% |
| Seismic survey x1 | 0.00011 | 5,334.8 | 0.59 | *0.011%* |

[40] As of February 2023, BEIS is known as the DESNZ

| Activity | Grey seal density (/km$^2$) based on Pen Llŷn a'r Sarnau SAC relative densities) | Impact area (km$^2$) | Maximum number of individuals potentially disturbed | % SAC ref pop |
|---|---|---|---|---|
| UXO (two clearance events) | 0.00011 | 806.3 | 0.089 | *0.0016%* |
| **Total** | | | **0.73** | **0.012%** |

3811. **Table 9.44** provides an assessment of each of the listed industries and activities for grey seal based on the density from the Cardigan Bay SAC.

3812. Using the density based on the Cardigan Bay SAC that was considered to represent the worst-case for the Project, the effects from other industries and activities would not present any LSE on the population of the nearest SAC. Less than one grey seal and less than 1% of the Cardigan Bay SAC population would be disturbed if all activities were to occur at the same time.

*Table 9.44 Indicative in-combination assessment for the potential disturbance for grey seal from the Cardigan Bay SAC for other industries and activities at other projects.*

| Activity | Grey seal density (/km$^2$) (based on Cardigan Bay SAC relative densities) | Impact area (km$^2$) | Maximum number of individuals potentially disturbed | % SAC ref population |
|---|---|---|---|---|
| Geophysical surveys x2 | 0.000006 | 420.2 | 0.0025 | 0.000045% |
| Aggregate extraction and dredging x2 | 0.000006 | 2.26 | 0.000014 | 0.0000002% |
| Seismic survey x1 | 0.000006 | 5,334.8 | 0.032 | 0.00057% |
| UXO (two clearance events) | 0.000006 | 806.3 | 0.0048 | 0.000087% |
| **Total** | | | **0.039** | **0.0007%** |

3813. **Table 9.45** provides an impact assessment for each of the listed industries and activities on grey seal based on the density from the Pembrokeshire Marine SAC.

*Table 9.45 Indicative in-combination assessment for the potential disturbance for grey seal from the Pembrokeshire SAC for other industries and activities at other projects*

| Activity | Grey seal density (/km$^2$) (based on Pembrokeshire Marine SAC relative densities | Impact area (km$^2$) | Maximum number of individuals potentially disturbed | % SAC ref population |
|---|---|---|---|---|
| Geophysical surveys x2 | 0.00000009 | 420.2 | 0.05 | 0.0009% |
| Aggregate extraction and dredging x2 | 0.00000009 | 2.26 | 0.0000002 | 0.000000004% |
| Seismic survey x1 | 0.00000009 | 5,334.8 | 0.0005 | 0.000009% |
| UXO (two clearance events) | 0.00000009 | 806.3 | 0.00007 | 0.0000013% |
| **Total** | | | **0.051** | **0.00091%** |

3814. The effects from other industries and activities would not present any LSE on the Pembrokeshire Marine SAC population as less than one grey seal and less than 0.01% of the relevant population would be disturbed if all activities were to occur at the same time.

3815. Mitigation measures required for UXO clearance include the use of low-order clearance techniques, which could include a small donor charge, rather than full high-order detonation which would only be used as a last resort. JNCC guidance referred to the preference of using low-order deflagration, thus this has been carried forward in the assessment, in-combination with piling at the Project.

3816. It would be highly unlikely that more than one UXO high-order detonation would occur at exactly the same time as another UXO high-order detonation, even if they had overlapping UXO clearance operation durations. So as the worst-case one high-order UXO clearance has been carried forward to the assessment in-combination with piling at the Project.

3817. Due to the short duration of the sound arising from the detonation of UXO, marine mammals were not predicted to be significantly displaced from an area. Any changes in behaviour, if they occurred, would be an instantaneous response and short-term. The most recent SNCB guidance suggested that disturbance behaviour was not predicted to occur from UXO clearance if undertaken over a short period of time (JNCC, 2010b).

**Summary of disturbance effects during construction**

3818. For grey seal, the potential for in-combination disturbance from all noisy activities occurring at the same time as Project piling would be less than 4% of either the nearest (Pen Llŷn a'r Sarnau SAC) or furthest relevant (Pembrokeshire Marine SAC) SAC reference population. (**Table 9.46**). As no SACs would overlap with the Project, only seals outwith SACs where they were a designated feature would be affected.

*Table 9.46 Quantified in-combination assessment for the potential disturbance of grey seal from all underwater noise sources during construction (Grey rows were projects and activities that may take place and therefore indicative assessments have been completed)*

| Impact | Grey seal from Pen Llŷn a'r Sarnau SAC | Grey seal from Cardigan Bay SAC | Grey seal from Pembrokeshire Marine SAC |
|---|---|---|---|
| Worst-case disturbance from the Project (piling at the Project) | 0.2 | 0.012 | 0.00018 |
| Piling at other OWF | 185.22 | 185.0 | 210.85 |
| Construction activities at other OWF | 1.3 | 0.17 | 0.039 |
| Geophysical surveys | 0.27 | 0.0025 | 0.05 |
| Aggregates and dredging | 0.0003 | 0.0000002 | 0.0000002 |
| Seismic surveys | 0.59 | 0.032 | 0.0005 |
| UXO clearance | 0.089 | 0.0048 | 0.00007 |
| **Total number of individuals** | **187.81** | **185.22** | **210.94** |
| **Percentage of reference population** | **3.4%** | **3.3%** | **3.8%** |

3819. Based on the worst-case total, the in-combination assessment of grey seals for underwater noise impacts at all projects that were (or expected to be) undertaken at the same time as the Project was less than 5% of the relevant reference and SAC populations. **As such, it has been concluded that considering the Project in-combination with other plans and projects there would be no LSE on relevant reference populations (and no AEoI on any SAC) from disturbance during construction.**

## Disturbance from underwater noise during operation and maintenance

3820. Underwater noise and disturbance during operation and maintenance could come from multiple sources, including from operational noise from WTGs, noise of major maintenance work, rock placement or cable repairs, the presence of vessels as well as other industrial activities.

3821. A review of the most recent scientific literature was used to determine the potential disturbance of harbour porpoise from underwater operational noise from operational WTGs (see Section 11.6.4.1 of **Chapter 11 Marine Mammals**). The research indicated that any disturbance would be in the immediate area of the operational WTG, depending on ambient noise levels. There was no evidence of any lasting disturbance or exclusion of harbour seals around windfarm sites during operation, with reports of harbour seals moving through and foraging within operational windfarm sites. Similar limited impacts have been assumed for grey seal.

3822. Effects from maintenance activities at OWFs, such as additional rock placement or cable re-burial, would be very localised, short in duration and temporary. The potential for in-combination effects from maintenance activities, including vessel traffic associated with OWFs, would be less than the in-combination effects assessed for construction activities other than piling due to the shorter duration and lower level of activity required.

3823. Therefore, operational noise from OWF WTGs and maintenance of OWFs **is considered unlikely to have any significant effect on any relevant reference population (and no AEoI on any SAC) due to the long distances from the projects.**

### Underwater noise from decommissioning

3824. The potential for in-combination impacts during the decommissioning of the Project is unknown. It is not possible to provide details of the methods that would be used during decommissioning at this time. However, it is expected that the activity levels would be comparable to construction (with the exception of pile driving noise which would not occur).

3825. During decommissioning, the potential effects on grey seal were anticipated to be similar or less than the worst-case for the construction phase (depending on the methods used). Crucially, any in-combination effect would be dependent upon simultaneous decommissioning and it is not possible to provide a realistic prediction as to which projects may be decommissioned and when.

3826. **The potential impacts during decommissioning have been screened out from further consideration within the CEA screening (see Appendix 11.4) and have not been considered further in this in-combination assessment.**

#### 9.6.4.2 Barrier effects as a result of underwater noise

3827. There was limited information available on the seal usage of windfarm sites during the construction period, but if a barrier effect arose from underwater noise, it would be intermittent for a limited time period. Given the presence of several windfarms in the IS, there would be the potential for disturbance effects to overlap, however these have been assessed as having no long-term population level effect (see Section 11.7.3.2 in **Chapter 11 Marine Mammals**). Furthermore, the Project would be unlikely to cause a barrier effect to foraging seals, considering their large foraging ranges and the potential distance between the Project and the nearest relevant SAC.

3828. Due to the low noise levels associated with operational OWFs, BEIS[41] (2020) concluded that there would be no potential for significant impact from the operation of OWFs. Effects from maintenance activities at OWFs, such as such as additional rock placement or cable re-burial, would be very localised, short in duration and temporary and limited to the Project boundaries.

3829. Given these limited temporal and spatial effects and the geographical spread of projects across the Irish Sea, it is considered that there was no potential for a barrier effect from underwater noise to occur. **As such it has been concluded that, considering the Project in-combination with other plans and projects, there would be no LSE on any relevant reference grey seal population (and no AEoI on any SAC).**

#### 9.6.4.3 Vessel interactions

3830. Given the low risk of collision to grey seal and the commitment to mitigation to manage the residual risk, it was concluded that there would be no significant effect from the Project on the wider grey seal reference population from the effects of vessel interactions during construction, operation and maintenance or decommissioning. It was considered that any consented OWF project would require similar mitigation which would reduce collision risks.

3831. Vessels associated with aggregate extraction and dredging are large and typically slow moving, using established transit routes to and from ports. Therefore, the potential increased collision risk with vessels was considered to be extremely low or negligible.

3832. Given the low risk to grey seal and the use of mitigation across OWF projects and other industries, **it is concluded that, considering the Project in-combination with other plans and projects, there would be no LSE on the relevant reference population (and no AEoI on any SAC) from the**

[41] As of February 2023, BEIS is known as the DESNZ

**effects of vessel interactions during construction, operation and maintenance or decommissioning.**

#### 9.6.4.4 Changes to prey resources

3833. No significant impacts with regard to changes to prey resources were expected as a result of the Project, as detailed in the **Chapter 11 Marine Mammals** of the ES (for all Project phases).

3834. For any potential changes to prey resources, it has been assumed that any potential effects on marine mammal prey species from underwater noise, including piling, would be the same or less than those for marine mammals. Therefore, there would be no additional in-combination effects above those assessed for marine mammals, i.e. if prey were disturbed from an area as a result of underwater noise, grey seal would be disturbed from the same or a greater area. As a result, any changes to prey resources would not affect marine mammals as they would already be disturbed from the area.

3835. Any effects to prey species were likely to be intermittent, temporary and highly localised, with potential for recovery following cessation of the disturbance activity. Any permanent loss or changes of prey habitat would typically represent a small percentage of the potential habitat for prey species in the surrounding area.

3836. This has been concluded on the basis of the range of prey species taken by grey seal over the extent of their foraging areas. Furthermore, much of the effect would occur outside of any area considered important for foraging (i.e. outside of a SAC). **It is concluded that, considering the Project in-combination with other plans and projects, there would be no LSE on the relevant reference population (and no AEoI on any SAC).**

#### 9.6.4.5 Changes to water quality

3837. No significant impacts with regard to water quality were expected as a result of the Project, as detailed in **Chapter 11 Marine Mammals** of the ES (for all Project phases).

3838. Aggregate and dredging projects and the clearance of UXOs (**Appendix 11.4**) have the potential for increased SSCs (and therefore, impact marine mammal species), however, any changes to water quality as a result of these activities would be very localised and temporary. Other OWFs or other construction projects were also considered to have highly localised and temporary effects and would be spatially separated so there would be no potential for significant additive effects.

3839. Given that effects from changes to water quality would be negligible, **it is concluded that considering the Project in-combination with other plans**

**and projects there would be no LSE on the relevant reference population (and no AEoI on any SAC).**

#### 9.6.4.6 Summary of in-combination assessment

3840. Due to embedded mitigation and commitment to further mitigation measures, it was considered that permanent effects upon grey seal could be avoided (i.e PTS mitigation through the MMMP) or that the existing low collision risk could be further reduced (i.e. reduction of collision risk through vessel management measures) during construction, operation and maintenance or decommissioning of all consented OWFs.

3841. Given that the disturbance of grey seal would be below 5% of the relevant reference population, there would be no potential for LSE during construction, operation and maintenance or decommissioning. Given the potential distances of the projects within the in-combination assessment from the Pen Llŷn a’r Sarnau SAC, none would have significant disturbance effects likely to occur within a SAC.

3842. Indirect effects (i.e. on water quality or prey resources) were considered to be insignificant and would occur outside any SAC boundary.

3843. None of the effects assessed were considered to have a significant effect on the wider reference population for grey seal. **As such, it is considered that there would be no adverse effect on integrity of the Pen Llŷn a’r Sarnau SAC, nor any other SAC in relation to the conservation objectives for ‘Population’ and ‘Range’.**

3844. Indirect effects (i.e. on water quality or prey resources) are considered to be insignificant and would occur outside any SAC boundary. **As such, it is considered that there would be no adverse effect on the integrity of the Pen Llŷn a’r Sarnau SAC nor any other SAC in relation to the conservation objectives for ‘Range’ and ‘Supporting habitats and species’.**

3845. The confidence in the assessment for all impacts is considered medium, yet highly precautionary, particularly given the consideration of a large number of plans or projects and the unlikelihood of temporal overlap of all these activities.

# 9.7 Harbour seal

## 9.7.1 Relevant sites

### 9.7.1.1 Strangford Lough SAC

#### Description of designation

3846. Strangford Lough is a marine inlet on the east coast of County Down, Northern Ireland. The SAC covered an area of 154.0km$^2$. The Lough is almost land-locked, separated from the Irish Sea by the Ards Peninsula to the east, bounded to the south by the Lecale coast and connected to the open sea by the Strangford Narrows (DAERA, 2020).

3847. The SAC is approximately 130km from the windfarm site, measured as a straight line distance, and 136km, measured as a coastline distance.

#### Harbour seal population and density

3848. A review of harbour seal count data from 1992-2017 indicated an annual decline in harbour seal adults of 2.01% and pups of 1.31% in Strangford Lough SAC (Culloch *et al.,* 2018). It was proposed that the SAC population should be at least 200 adults with at least 25% of the population being pups to meet the conservation objectives (Culloch *et al.,* 2018). It was highlighted that since 2007, the Lough has not been surveyed in its entirety and apparent declines need to be considered carefully as this may be related to survey effort.

3849. Although harbour seal populations in other areas on the Northern Irish coast appeared to be stable, harbour seal in Strangford Lough SAC have experienced a continuous decline since 2002 when complete surveys of the Northern Ireland coast (also in 2011 and 2018) were carried out by the Sea Mammal Research Unit (SMRU). The most recent survey was conducted in 2018, when only 93 seals were counted compared with a count of 403 in 2002 (Morris and Duck, 2018).

3850. The harbour seal density estimates for the Project have been calculated from SAC specific seal at-sea usage maps produced by Carter *et al.,* (2022), based on the 5km x 5km grids that overlapped with the Project area. The mean at-sea density estimate used in the assessment was:

- 0.00000001 individuals per km$^2$ for the Project windfarm site and 4km buffer

3851. The reference population for harbour seal from Strangford Lough SAC was estimated to hold 93 individuals (Morris and Duck, 2018). After applying a correction factor for those individuals not available to count (0.72 derived by

SCOS-BP 21/02 in SCOS, 2022), the reference population of Strangford Lough SAC was estimated to hold 106 harbour seal.

### Conservation status

3852. Based on the most recent 2013-2018 DAERA survey report, the attributes measured against the condition assessment targets for harbour seal (**Table 9.47**) resulted in an overall Conservation Status of unfavourable-declining, in relation to the Natura network (Alvarez Alonso and Foster, 2022).

### Conservation objectives

3853. Conservation objective for this site is to maintain (or restore where appropriate) the harbour seal population to favourable condition (DAERA, 2017).

3854. SAC selection feature objective requirements include:

- Maintain and enhance, as appropriate, the Harbour (Common) Seal population
- Maintain and enhance, as appropriate, physical features used by Harbour (Common) Seals within the site

*Table 9.47 Conservation status attributes and targets. Table adapted from Alvarez Alonso and Foster, 2022 (* =primary attribute; should one fail= unfavourable condition)*

| Attribute | Targets |
|---|---|
| *Number of adults | Maintain a population of at least 200 individuals |
| *Number of Pups | Number of pups to be at least 25% of the population |
| *Mother and pup resident time | Resident time to be at least three weeks |
| Habitat availability | Maintain the number of suitable sites for moulting, haul-out and breeding |

## 9.7.2 Project-alone assessment

3855. For the following assessments, the SAC specific density and reference population (**Table 9.48**) have been used as discussed under **Section 9.7.1.1**.

*Table 9.48 SAC specific density and reference population used in the RIAA*

| SAC | Density | Reference population[42] | Source |
|---|---|---|---|
| Strangford Lough SAC | 0.00000001 | 106 | Morris and Duck, 2018 (for reference population) |

[42] Correction factor applied to those at sea and not available to count (SCOS, 2022)

### 9.7.2.1 Underwater noise

3856. The assessment below refers to the same impact ranges used in the ES assessment. However, the reference population and density estimates used in this assessment have been adapted to be SAC specific.

#### Permanent auditory injury from underwater noise during piling

3857. This impact would be relevant to the construction phase only, with any potential impacts occurring outside any SAC.

3858. High exposure levels from underwater noise sources can cause auditory injury or hearing impairment, collectively described as PTS. The potential for auditory injury was not just related to the level of the underwater sound and its frequency relative to the hearing bandwidth of the animal, but was also influenced by the duration of exposure.

3859. Underwater noise modelling was carried out (see **Appendix 11.1**) to predict the noise levels likely to arise during impact piling and other activities. The modelled impact ranges have been used determine the potential effects on marine mammals. A detailed explanation of the modelling, inputs and assumptions has been provided in Section 11.6.3.1 of **Chapter 11 Marine Mammals** of the ES.

3860. Several scenarios were modelled to determine the worst-case for PTS effects for monopiles and pin piles, including the effects of sequential piling of four pin-piles. The maximum predicted impact range for PTS was up to 0.1km from cumulative exposure ($SEL_{cum}$) during single monopile installation at maximum hammer energy (6,600kJ) without any additional mitigation (**Table 9.49**). Given the potential distance of the Project windfarm site from the SAC (135km), there was no pathway for PTS upon harbour seal within the SAC.

3861. An assessment of the maximum number of individuals and percentage of the reference population affected (outside the SAC) under each of the scenarios was then undertaken using the SAC specific density and reference population.

3862. Given the very low densities and the low impact ranges for PTS, the number of animals which would be affected was extremely low and insignificant (**Table 9.49**). Given the embedded mitigation and commitment to further mitigation measures, **it has been concluded that there would be no LSE on the reference population (and no AEoI on the SAC) from PTS.**

3863. The final approved piling MMMP would reduce the risk of PTS still further. The final MMMP for piling would be based on the Draft MMMP (Document Reference 6.5) which has been included with the DCO Application.

*Table 9.49 Maximum number of individual Harbour seal (and % of reference population) from the Strangford Lough SAC that could be at risk of PTS from single strike and from cumulative exposure ($SEL_{cum}$) of monopile or pin-pile underwater noise and the respective cumulative exposure*

| Criteria and threshold (Southall *et al*., 2019) | Monopile Maximum impact range (km) and area (km$^2$) | Monopile (sequential piling) Maximum impact range (km) and area (km$^2$) | Pin-pile Maximum impact range (km) and area (km$^2$) | Pin-pile (sequential piling) Maximum impact range (km) and area (km$^2$) |
|---|---|---|---|---|
| | Maximum hammer energy (6,600kJ) | | Maximum hammer energy (2,500kJ) | |
| **Single strike at maximum hammer energy** | | | | |
| $SPL_{peak}$ Unweighted (218 dB re 1μPa) Impulsive | 0.0000000001 (0.00000000009% of the SAC reference pop.) | *N/A* | 0.0000000001 (0.00000000009% of the SAC reference pop.) | *N/A* |
| **Cumulative exposure ($SEL_{cum}$) during installation** | | | | |
| $SEL_{cum}$ Weighted (185 dB re 1μPa$^2$s) Impulsive | 0.000000019 (0.000000018 % of the SAC reference pop.) | 0.00000002 (0.000000019% of the SAC reference pop.) | 0.000000001 (0.0000000009 % of the SAC reference pop.) | 0.000000001 (0.0000000009% of the SAC reference pop.) |

## Disturbance impacts from underwater noise during piling

3864. This potential impact would be relevant to the construction phase only with effects occurring outside any relevant SAC.

3865. The impact of underwater noise generated by piling in this area has been evaluated, considering a precautionary disturbance range of 25km around a monopile (Russell, 2016). While there were different methods to assess disturbance (such as dose-response curves, population modelling, *etc*.), the choice to use the 25km range has been used to inform the assessment in **Chapter 11 Marine Mammals** of the ES. Among all the options considered for the disturbance assessment, the 25km range represented the worst-case scenario.

3866. As outlined in **Section 9.7.1.1**, Strangford Lough SAC is 135km from the Project site and therefore would have no potential overlap with underwater noise potentially generated by the Project. There would be no direct effect on harbour seal within the SACs, but there may be impacts on harbour seal foraging outside the SAC boundaries.

3867. The assessment in **Table 9.50** provides information on the maximum number of individuals and percentage of the reference population potentially affected.

*Table 9.50 Maximum number of harbour seal (and % of reference population) that could be disturbed during piling at the Project based on a disturbance range of 25km*

| SAC | 25km disturbance range (1,963.5km$^2$) for monopile<br>Maximum number of individuals (% of reference population) |
|---|---|
| Strangford Lough | 0.00002 (0.000019% of the SAC reference pop.) |

3868. The effect would occur outside of any SAC and would affect less than 5% of the reference population. **It was therefore concluded that there would be no LSE on the reference population (and no AEoI on the SAC) from underwater noise during piling.**

3869. Population modelling has been conducted for harbour seal population associated with the Strangford Lough SAC due to the overall Conservation Status of unfavourable-declining to ensure there is no population level effect. The iPCoD framework (Harwood *et al.,* 2013, King *et al*., 2015) has been used to predict the potential medium and long term population consequences of the predicted amount of disturbance resulting from piling at the Project-alone, and in-combination with other relevant projects on the Strangford Lough SAC population. **Chapter 11 Marine Mammals** of the ES also detailed how population modelling for harbour seal was also conducted for population level consequences due to disturbance.

3870. The latest data on the Strangford Lough SAC harbour seal population suggests that the population may be on a declining trajectory (Culloch *et al.,* 2018). In contrast, the Northern Irish harbour seal population in general appears to be stable (Sinclair *et al*., 2020). For this reason, to test each possible scenario, iPCoD modelling has been conducted using two sets of demographic parameters provided by Sinclair *et al.,* (2020). The Northern Ireland parameters model a stable population representative of the wider MU where the SAC is located, whilst the Orkney & North Coast parameters are used to model a declining population to represent the current condition of the SAC population (**Table 9.51**).

*Table 9.51 The demographic parameters used to model population consequences of piling disturbance on the Strangford Lough SAC harbour seal population (Sinclair et al., 2020)*

| MU/SMA | Age calf/pup becomes independent | Age of first birth | Calf/Pup Survival | Juvenile Survival | Adult Survival | Fertility | Growth Rate |
|---|---|---|---|---|---|---|---|
| Northern Ireland | 1 | 4 | 0.4 | 0.78 | 0.92 | 0.85 | 1.000 |
| Orkney & North Coast | 1 | 4 | 0.24 | 0.86 | 0.80 | 0.90 | 0.8956 |

3871. The modelling assumed a worst-case of 0.00002 harbour seal disturbed and 0.00000002 harbour seal with PTS on every piling day from the Project.

3872. The iPCoD model estimated there to be no discernible impact to the Strangford Lough SAC assuming a stable population (**Table 9.52**) or assuming a declining population (**Table 9.53**). The median population size was predicted to be 100% of the un-impacted population size at the end of 2028 (one year after the piling has completed). By the end of 2029 (two years after piling ends) the median population size for the impacted population was predicted to be 100% of the un-impacted population size. This lack of discernible effect on the impacted population was maintained until 2052, which was the end point of the modelling.

3873. For the Strangford Lough SAC, the modelling indicated there was no potential for a significant impact of disturbance due to less than a 1% population level impact of the population over both the first six years and 25 year modelled periods (**Plate 9.7** and **Plate 9.8**) and shows that there are no likley significant effects on the reference population (and no AEoI on the SAC). Piling at the Project would cause no additional disturbance effect to harbour seals from the SAC.

*Table 9.52 Results of the iPCoD modelling for the Project, assuming a stable population (Northern Irish MU/SMA demographic parameters from Sinclair et al., (2020)), giving the mean population size of the harbour seal population (Strangford Lough SAC) for years up to 2052 for both impacted and un-impacted populations in addition to the median ratio between their population sizes*

| Year | Un-impacted population mean | Impacted population mean | Median impacted as % of un-impacted |
|---|---|---|---|
| Start | 106 | 106 | 100.00% |
| End 2028 | 107 | 107 | 100.00% |
| End 2029 | 107 | 107 | 100.00% |
| End 2032 | 107 | 107 | 100.00% |
| End 2037 | 107 | 107 | 100.00% |
| End 2047 | 108 | 108 | 100.00% |
| End 2052 | 108 | 108 | 100.00% |

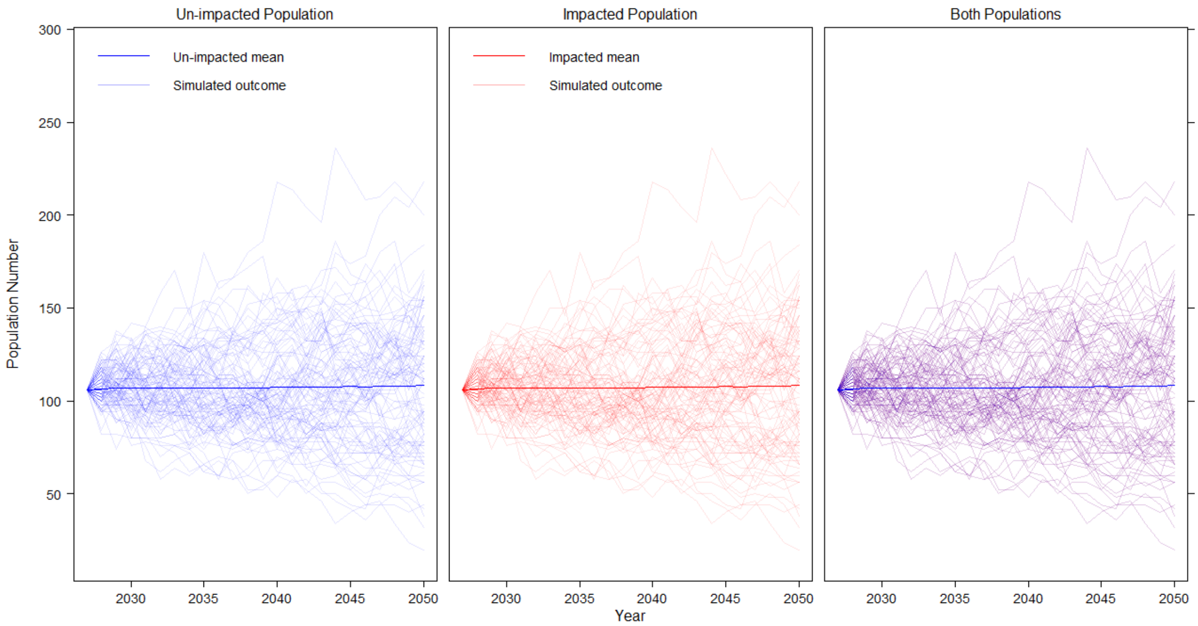


*Plate 9.7 Simulated worst-case harbour seal population sizes for both the un-impacted and the impacted populations for the Strangford Lough SAC, assuming a stable population (Northern Irish MU/SMA demographic parameters from Sinclair et al., (2020))*

*Table 9.53 Results of the iPCoD modelling for the Project, assuming a declining population (Orkney and North Coast MU/SMA demographic parameters from Sinclair et al., (2020)), giving the mean population size of the harbour seal population (Strangford Lough SAC) for years up to 2052 for both impacted and un-impacted populations in addition to the median ratio between their population sizes*

| Year | Un-impacted population mean | Impacted population mean | Median impacted as % of un-impacted |
|---|---|---|---|
| Start | 106 | 106 | 100.00% |
| End 2028 | 96 | 96 | 100.00% |
| End 2029 | 86 | 86 | 100.00% |
| End 2032 | 61 | 61 | 100.00% |
| End 2037 | 35 | 35 | 100.00% |
| End 2047 | 11 | 11 | 100.00% |
| End 2052 | 6 | 6 | 100.00% |

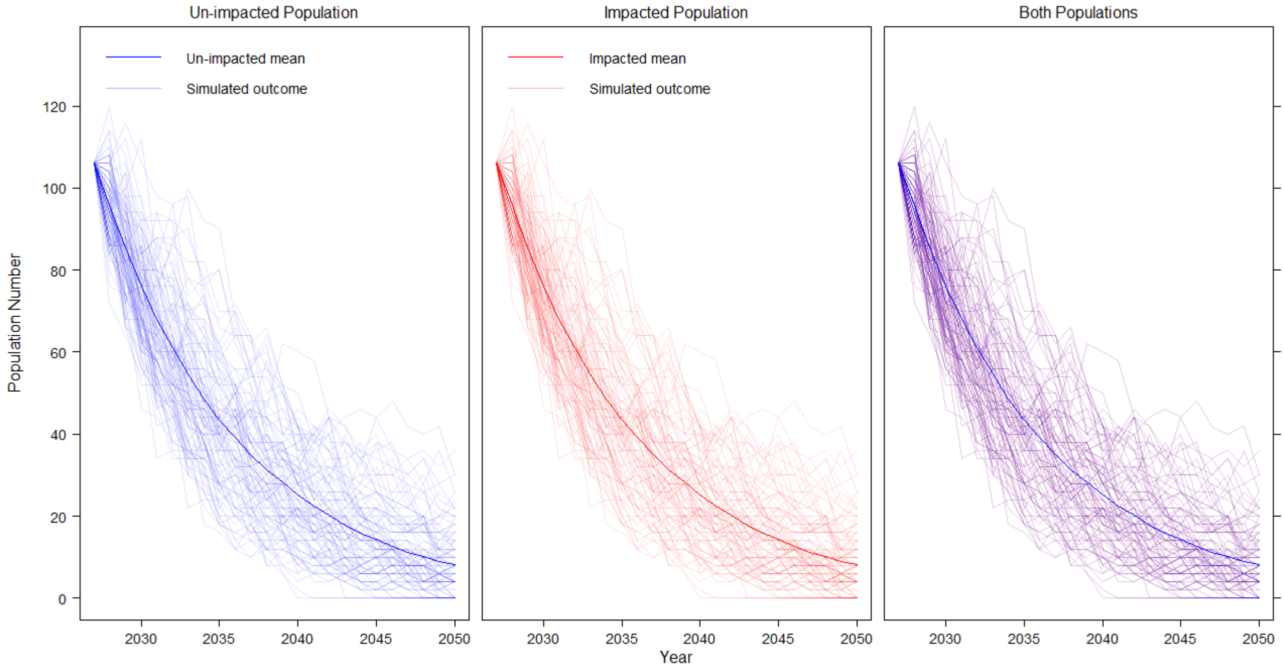


*Plate 9.8 Simulated worst-case harbour seal population sizes for both the un-impacted and the impacted populations of Strangford Lough SAC, assuming a declining population (Orkney and North Coast MU/SMA demographic parameters from Sinclair et al., (2020))*

## Underwater noise and disturbance from other sources

### Construction

3874. Section 11.6.3.3 of **Chapter 11 Marine Mammals** of the ES details the potential impacts of disturbance from underwater noise from seabed preparation, dredging, trenching, cable installation and rock placement activities. Section 11.6.3.4 of **Chapter 11 Marine Mammals** of the ES details the potential impacts of underwater noise from the presence of vessels.

3875. The likely insignificant effect of piling on the harbour seal SAC population has been highlighted in the preceding assessment of potential permanent injury from piling. It was deemed unnecessary to repeat an assessment for noise from other sources, which would have even smaller impact ranges. Furthermore, the effect of TTS has been screened out, as it would have no permanent effect on the conservation objectives listed in **Section 9.6.1**. Therefore, this section has examined the impact of disturbance only.

3876. A review of the most recent scientific literature has been used to determine the maximum potential disturbance range for other construction activities and vessels. During the construction of two Scottish windfarms (Beatrice OWF and Moray East OWF), (Benhemma-Le Gall *et al*., 2021), a reduction in harbour porpoise presence was reported up to 4km (50.27km$^2$) in distance from construction vessels. This distance has been used as the disturbance range for other construction activities, including vessels.

3877. Therefore, assessments for harbour seal have been based on the same disturbance impact range of 4km (50.27km$^2$) due to the absence of data on potential disturbance ranges for other construction activities and vessels.

3878. Taking into account the potential distance of the Project windfarm site from the SAC (135km), there was no pathway for disturbance effects directly upon harbour seal within the site.

3879. As a precautionary approach, the potential disturbance from two activities (such as cable laying, dredging, trenching and rock placement) occurring at the same time, including vessels undertaking the work, has been assessed based on maximum impact area of 100.54km$^2$ (assuming no overlap in the impact areas between the activities). The maximum number of harbour seal that could be disturbed was up to 0.0000010 (0.00000095% of the SAC) (**Table 9.54**).

3880. The assessment of disturbance from construction vessels has been detailed in Section 11.6.3.4 in **Chapter 11 Marine Mammals** in the ES. The approach of the 4km disturbance range has been used for 37 vessels, which would equate to a total impact area of 1859.8km$^2$ and would present an unrealistic worst-case. This scenario would not take into account the overlap in the 4km disturbance range between vessels and the area would be approximately 21 times the size of the Project site alone (87km$^2$).

3881. Additionally, according to Benhemma-Le Gall *et al.,* (2021) there were several ongoing construction activities and a variety of support vessels present during the porpoise detections in their study area. This suggested that an assumption of 4km per vessel would be altogether unrealistic. The assessment has therefore been based on an impact area equivalent to the windfarm site, along with a 4km buffer (285.4km²).

3882. The maximum number of potentially disturbed harbour seal associated with the relevant SAC has been summarised in **Table 9.54**.

*Table 9.54 Maximum number of harbour seal (and % of reference population) from Strangford Lough SAC that could be disturbed as a result of underwater noise associated with other (non-piling) construction activities at the Project*

| Potential impact | Maximum number of harbour seal (% of reference population) |
|---|---|
| **Disturbance from other activities** | |
| One activity (50.27km$^2$) | 0.0000005<br>(0.00000047%% of the SAC reference pop.) |
| Two activities (100.54km$^2$) | 0.000001<br>(0.00000095% of the SAC reference pop.) |
| **Disturbance from vessels** | |
| One vessel (50.27km$^2$) | 0.0000005<br>(0.00000047%% of the SAC reference pop.) |
| 37 vessels within the revised site + 4km buffer (285.4km$^2$) | 0.0000029<br>(0.0000027% of the SAC reference pop.) |

3883. There would be no potential for additive effects (i.e. the disturbance from construction activities plus vessel activities) as the disturbance range of 4km for construction activities included noise from the vessel undertaking the work.

3884. Given that this effect would be minimal and would occur outside of any area considered important for foraging or breeding (i.e a SAC), **it is concluded that there would be no LSE on the reference population (and no AEoI on the SAC) from underwater noise from other sources during construction.**

### Operation and maintenance

3885. Underwater noise and disturbance during operation could come from multiple sources, from operational noise from WTGs, noise of major maintenance work, rock placement or cable repairs, and from the presence of vessels. Each of these sources has been considered separately in detail in the following sections of **Chapter 11 Marine Mammals** of the ES: 11.6.4.1, 11.6.4.2 and 11.6.4.3. The conclusions of these assessments have been brought together for the purposes of this assessment.

3886. A review of the most recent scientific literature was used to determine the potential disturbance of harbour seal from underwater operational noise from WTGs (see Section 11.6.4.1 of **Chapter 11 Marine Mammals** of the ES). The studies indicated that any disturbance would be in the immediate area of the operational turbine, depending on ambient noise levels. There was no evidence of any lasting disturbance or exclusion of harbour seal around OWFs during operation, with reports of harbour seal moving through and foraging within operational OWFs.

3887. There was no LSE arising from the underwater noise and disturbance during construction. Following this, it was not anticipated that operational turbines would cause an effect greater than that of piling noise and therefore this has not been assessed further. As outlined previously, TTS would not have a long-lasting effect on the conservation objectives for the site and has been screened out of further assessment.

3888. However, the disturbance for maintenance activities (including vessels undertaking the work) would be more likely to affect harbour seals if the precautionary range of 4km (Benhemma-Le Gall *et al.,* (2021) were applied.

3889. The potential disturbance from cable repairs and rock placement occurring at the same time has been assessed based on maximum impact area of 100.54km$^2$ (**Table 9.55**).

3890. Based on a standard year of maintenance it was expected that up to three vessels could be on site at any given time. As such, an assessment of the number of animals from the relevant SACs that could potentially be disturbed by three vessels (150.81km$^2$) has been presented in **Table 9.55**.

*Table 9.55 Maximum number of harbour seal (and % of reference population) from the Strangford Lough SAC that could be disturbed as a result of underwater noise associated with maintenance activities at the Project*

| Potential impact | Maximum number of harbour seal (% of reference population) |
|---|---|
| **Disturbance from other activities** | |
| One activity (50.27km$^2$) | 0.0000005<br>(0.00000047%% of the SAC reference pop.) |
| Two activities (100.54km$^2$) | 0.000001<br>(0.00000095% of the SAC reference pop.) |

| Potential impact | Maximum number of harbour seal (% of reference population) |
|---|---|
| **Disturbance from vessels** | |
| One vessel (50.27km$^2$) | 0.0000005<br>(0.00000047%% of the SAC reference pop.) |
| Three vessels (150.81km$^2$) | 0.0000015<br>(0.0000014% of the SAC reference pop.) |

3891. Given that this effect would be minimal (less than 5% of the SAC population) and would occur outside of any area considered important for foraging or breeding (i.e a SAC), **it was concluded that there would be no LSE on the reference population (and no AEoI on the SAC) from disturbance impacts from underwater noise during operation.** Assessments were made on a standard maintenance year, but, given the maximum number of harbour seal (% of reference population), it was anticipated that there would also be no LSE on the reference population (and no AEoI on the SAC) during a heavy maintenance year.

#### Decommissioning

3892. Potential impacts on harbour seal associated with underwater noise during decommissioning have not been assessed in detail, as further assessments would be carried out ahead of any decommissioning works to be undertaken, taking account of known information at that time, including relevant guidelines and requirements. The final Decommissioning Programme would provide details of the methodology to be employed and any relevant mitigation measures required.

3893. It is not possible to provide details of the methods that would be used during decommissioning at this time. However, it is expected that the activity levels would be comparable to construction (with the exception of pile driving noise which would not occur).

3894. During decommissioning, the potential effects on harbour seal are anticipated to be similar or less than the worst-case for the construction phase (depending on the methods used). The effects would therefore be comparable to those described in construction.

3895. Given that this effect would be lower than for disturbance impacts from underwater noise during piling and any impact would occur outside of any area considered important for foraging or breeding (i.e a SAC), **it is concluded that there would be no LSE on the harbour seal reference population (and no AEoI on the SAC) from the impact of disturbance from underwater noise during decommissioning.**

### 9.7.2.2 Barrier effects as a result of underwater noise

#### Construction

3896. Underwater noise during construction could have the potential to create a barrier effect, preventing movement of harbour seal between important feeding and/or breeding areas, or potentially increase swimming distances if harbour seal avoid the area and go around it. This effect has been considered in detail in Section 11.6.3.5 of **Chapter 11 Marine Mammals** of the ES.

3897. As outlined in the ES, there are no significant harbour seal breeding or haul out sites in the NW England MU (SCOS, 2022), there would be therefore no potential for underwater noise at the Project windfarm site to result in barrier effects to seals moving to and from haul-out sites.

3898. Underwater noise from piling and ADD use would be for a maximum of approximately 620 hours or 26 days in total (assuming 24-hour days, see Table 11.35 in **Chapter 11 Marine Mammals**) over the construction period of up to two and a half years (with foundation installation expected over 9 – 12 months). Other construction activities and vessels that could result in barrier effects would be temporary, inconsistent throughout the offshore construction period, and would be limited to only parts of the overall construction period and area at any one time. If there were potential barrier effects across the entire Project windfarm site (87km$^2$), this area would be small in relation to the movements and foraging ranges of harbour seal in and around the IS. Although the modelled noise levels were larger than the Project site, a 25km disturbance range has been applied (informed by the **Chapter 11 Marine Mammals** of the ES; see **Section 9.7.2.1**) to assess the worst-case scenario This resulted in a negligible effect on the Strangford Lough SAC harbour seal population.

3899. There is unlikely to be any significant long-term impact from any temporary barrier effects due to underwater noise, as any areas affected would be relatively small in comparison to the range of harbour seals and any effects would not be continuous throughout the offshore construction period. The effect would occur outside of any SACs, or other areas considered important for foraging or breeding. **It is therefore concluded that there would be no likely significant impact on the reference population (and no AEoI on the SAC) from barrier effects during construction.**

#### Operation and maintenance and decommissioning

3900. No potential likely significant disturbance effects from underwater noise were anticipated during operation. Any behavioural responses or disturbance would be limited to the close vicinity of the operational WTG. The minimum spacing between WTGs (**Table 9.4**) meant there would be no potential for underwater noise around individual WTGs to overlap.

3901. Taking into account the relatively small impact areas for underwater noise around operational WTGs, there was unlikely to be the potential for barrier effects upon marine mammals as a result of operational noise.

3902. During decommissioning, the potential effects on harbour seal are anticipated to be similar or less than the worst-case for the construction phase (depending on the methods used). Therefore, the impacts would be comparable to those described in construction.

3903. Given that this effect would be lower than for disturbance impacts from underwater noise during construction and the effect would occur outside of any area considered important for foraging or breeding (i.e a SAC), **it has therefore been concluded that there would be no likely significant impact on the reference population (and no AEoI on the SAC) from barrier effects during operation or decommissioning.**

#### 9.7.2.3 Vessel interactions

#### Construction

3904. During the construction phase, there would be an increase in the number of vessels in the windfarm site. The maximum number of vessels that could be on the Project windfarm site at any one time has been estimated as up to a total of 37 vessels (**Table 9.4**). The number, type and size of vessels would vary depending on the activities taking place at any one time.

3905. This effect has been considered in detail in Section 11.6.3.6 of **Chapter 11 Marine Mammals** of the ES where Table 11.56 summarises the most recent available data using the CSIP recorded strandings of marine mammals in Wales and England for the relevant species and details the number of deaths caused by either vessel strike or physical trauma with an unknown cause (which could be attributed to vessel strike).

3906. The collision risk rate (3.35% for harbour seal) has been calculated based on the number of deaths attributed to vessels strike, or other physical trauma, which could have been caused by collision with a vessel, as proportion of the total known causes of death for each harbour seal.

3907. As a result, approximately one seal from the Strangford Lough SAC population could be at risk from vessel collision per year, with just over 1% of the population at risk (based on 2,583 annual vessel transits that were associated with construction activities (**Table 9.4**)). This would be a long-lasting effect and, in the worst-case, was assumed to be lethal for the individual. However, it was considered that the quantified assessment was highly precautionary.

Marine mammals would be able to detect and avoid vessels. However, vessel strikes have been known to occur, possibly due to distraction whilst foraging and socially interacting, or due to the marine mammals' inquisitive nature (Wilson *et al*., 2007).

3908. In 2016, SMRU conducted a study to determine the likelihood of harbour seal injury occurring due to co-presence with large vessels within the Moray Firth (Onoufriou *et al.,* 2016). This study used telemetry data of harbour seal within the Moray Firth, alongside vessel Automatic Identification System (AIS) data. The data indicated vessel and seal co-occurrence was high (defined as over less than 42 co-occurrence hours per year) in very localised areas. However, there appeared to be no relationship between areas of high co-occurrence and incidences of injury (Onoufriou *et al*., 2016).

3909. Considering that the vessel movements would be between larger ports and the Project (the Port(s) used to supply the Project would be selected post-consent), there would be no overlap of vessel routes with any SAC designated for harbour seal. Taking into account the existing number of vessel movements in the area, that vessels within the windfarm would be stationary for much of the time or very slow moving, and the disturbance from vessels, the realistic risk, was likely to be very low.

3910. As outlined in **Section 9.3.1,** the commitment to best practice mitigation measures would further reduce the potential risk of collision. The mitigation measures to reduce collision risk would be agreed with the relevant stakeholders and would be detailed within the PEMP and other relevant procedures.

3911. Vessel movements, where possible, would follow set vessel routes and hence areas where marine mammals were accustomed to vessels, in order to reduce any increased collision risk. Predictability of vessel movement by marine mammals has been known to be a key aspect in minimising the potential risks imposed by vessel traffic (Nowacek *et al*., 2001, Lusseau, 2003, 2006). Vessels travelling at high speeds were considered to be more likely to collide with marine mammals, and those travelling at speeds below 10 knots would rarely cause any serious injury (Laist *et al*., 2001). All vessel movements would be kept to the minimum number that was required to reduce any potential collision risk. Additionally, vessel operators would use good practice to reduce any risk of collisions with marine mammals.

3912. Given the relatively low risk to harbour seal and the commitment to mitigation measures to reduce that risk further, **it is concluded that there would be no LSE on the reference harbour seal population (and no AEoI on the SAC) from vessel interactions during construction.**

### Operation and maintenance and decommissioning

3913. The increased risk of collision with vessels during operation and maintenance would be less than assessed for the construction period. Less than one harbour seal (0.4), or 0.33% of the SAC reference population could be at risk from collision, based on the collision risk described above of 3.35% for harbour seal.

3914. During the operation and maintenance phase, the maximum number of vessels that could be on the Project windfarm site at any one time has been estimated at up to a total of ten vessels in a heavy maintenance year, with potentially up to 832 vessel transits per year (**Table 9.4**). The number, type and size of vessels would vary depending on the activities taking place at any one time. The vessels in the Project windfarm site during operation and maintenance would be slow moving or stationary.

3915. During decommissioning, the potential effects on harbour seal are anticipated to be similar or less than the worst-case for the construction phase. The effects would therefore be comparable to those described for construction.

3916. Given that this effect would be lower than for potential collision with vessels during construction and the commitment to mitigation measures to manage residual risk, **it is concluded that there would be no LSE on the reference harbour seal population (and no AEoI on the SAC) from vessel interactions during operation or decommissioning.**

### 9.7.2.4 Changes to prey resources

### Construction

3917. Potential impacts on prey species during construction could result from physical disturbance and loss of habitat, increased SSC and sediment deposition and underwater noise. **Chapter 10 Fish and Shellfish Ecology** of the ES, provides an assessment of these impact pathways on the relevant fish and shellfish species and concluded impacts of negligible to minor adverse significance in EIA terms. **Chapter 11 Marine Mammals** of the ES considers these effects in terms of potential indirect impacts on harbour seal (see **Chapter 11 Marine Mammals** Section 11.6.3.7 Changes to Prey Resources).

3918. Any reductions in prey availability would be small scale, localised, and temporary, and would occur in an area that was not considered important for harbour seal feeding. Therefore, it was considered highly unlikely that potential reductions in prey availability as a result of construction activities would result in detectable changes to the harbour seal population.

3919. It is also important to note that there was unlikely to be any additional displacement of harbour seal as a result of any changes in prey availability during piling as harbour seal would already be disturbed from the area.

3920. Given that this effect would be limited and would occur outside of any area considered important for harbour seal foraging (i.e. outside of SACs), **it is concluded that there would be no LSE on the reference population (and no AEoI on the SAC) from the potential impact of changes to prey species during construction.**

### Operation and maintenance and decommissioning

3921. Changes to prey resource during operation and maintenance have been assessed in Section 11.6.4.7 of **Chapter 11 Marine Mammals** of the ES. As per construction, this assessment has been based upon the conclusions of **Chapter 10 Fish and Shellfish Ecology** of the ES and considered a range of potential impacts including permanent habitat loss, introduction of hard substrate and EMF as well as the impacts considered for construction. Although new impacts have been considered for operation, some effects, such as those from physical disturbance, increased SSC and sediment deposition and underwater noise would be reduced when compared to construction. It was considered highly unlikely therefore that potential reductions in prey availability as a result of operation and maintenance activities would result in detectable changes to harbour seal populations.

3922. During decommissioning, the potential effects on harbour seal are anticipated to be similar or less than the worst-case for the construction phase. The effects would therefore be comparable to those described for construction in **Section 9.6.2**.

3923. Given that this effect would be limited and would occur outside of any area considered important for harbour seal foraging (i.e. outside of SACs), **it is concluded that there would be no LSE on the reference population (and no AEoI on the SAC) from the effects of changes to prey species during operation and maintenance or decommissioning.**

## 9.7.2.5 Changes to water quality

### Construction

3924. The disturbance of seabed sediments has the potential to lead to increases in SSCs and release of any sediment-bound contaminants (such as heavy metals and hydrocarbons that may be present within them) into the water column. The accidental release of contaminants (e.g. through spillage) also has the potential to affect water quality. During construction, there would also be the potential for increased SSCs. Section 11.6.3.8 of **Chapter 11 Marine Mammals** of the ES considered these effects in detail.

3925. Throughout the construction phase, best practice techniques and due diligence regarding the potential for pollution would be followed throughout all construction activities. Any risk of accidental release of contaminants (e.g. through spillage) would be mitigated in line with the PEMP and any changes to water quality as a result of any accidental release of contaminants (e.g. through spillage or vessel collision) would be negligible. Therefore, the potential for pollutants to be released into the environment has not been considered further in this assessment.

3926. Section 11.6.3.8 of **Chapter 11 Marine Mammals** of the ES considers increases in SSCs and remobilisation of existing contaminated sediments.

3927. Increased SSC would be unlikely to have any direct or indirect effects on marine mammals as they are known to often inhabit turbid environments. Pinnipeds are likely to use other senses instead of, or in-combination with, vision to sense the environment around them. Studies have shown that vision was not essential to seal survival, or their ability to forage (Todd *et al*., 2014).

3928. As outlined in **Chapter 8 Marine Sediment and Water Quality** of the ES, site specific data indicated that for all potential contaminants tested for within the sediments of the Project windfarm site concentrations were negligible. There would therefore be no potential for any direct or indirect effects on harbour seal from remobilisation of contaminated sediments.

3929. Given the potential distance of the Project windfarm site from the SAC (135km), there was no pathway for water quality effects directly upon harbour seal within the site.

3930. Given that the effects from changes to water quality would be negligible, **it is concluded that there would be no LSE on the harbour seal reference population (and no AEoI on the SAC) during construction.**

### Operation and maintenance and decommissioning

3931. During the operation and maintenance phase, there would be potential for increases in SSC and the release of any sediment-bound contaminants. The scale of these impacts would be small, infrequent and of short-term duration and of a lower magnitude than during the construction phase.

3932. During decommissioning, the potential water quality effects are anticipated to be similar or less than the worst-case for the construction phase. The effects would therefore be comparable to those described in construction.

3933. Given that the effects from changes to water quality would be negligible, **it is concluded that there would be no LSE on the reference population (and no AEoI on the SAC) during operation and maintenance or decommissioning.**

### 9.7.2.6 Disturbance at haul out sites

3934. As there were no significant harbour seal breeding or haul out sites in the NW England MU (SCOS, 2022), **it is concluded that there could be no LSE on the reference population (and no AEoI on the SAC) during construction, operation and maintenance or decommissioning.**

### 9.7.2.7 Potential interactions of Project effects

3935. The anticipated effects on marine mammal receptors were not expected to interact in a way that would lead to a combined effect of greater significance than the assessments presented for each individual phase. It should also be noted that precautionary measures were implemented in the assessment process, further contributing to the overall understanding and mitigation of potential impacts.

3936. Interactions of Project effects were as per those outlined in **Section 9.4.2.6** and **it is concluded that there could be no LSE on the reference population (and no AEoI on the SAC).**

### 9.7.2.8 Summary of Project-alone conclusions

3937. There would be no overlap of permanent or temporary noise impact ranges within any SAC.

3938. Due to embedded mitigation and the Project's commitment to securing additional mitigation measures (e.g. PTS mitigation through the MMMP and to manage the residual low collision risk through best practice vessel practices secured in the PEMP), it was considered that permanent impacts upon harbour seal would be avoided during construction, operation and maintenance or decommissioning.

3939. Disturbance of harbour seal potentially caused by underwater noise and vessel interactions would affect less than 5% of the relevant reference population.

3940. None of the effects assessed would be within an SAC or were considered to have any LSE on the SAC reference population. **As such, it is considered that there would be no adverse effect on integrity on the Strangford Lough SAC in relation to the conservation objective 'Maintain and enhance, as appropriate, the harbour seal population'.**

3941. Indirect effects (i.e. on water quality or prey resources) were considered to be insignificant and would occur outside any SAC boundary. **As such, it is considered that there would no adverse effect on integrity on the Strangford Lough SAC in relation to the conservation objective 'Maintain**

**and enhance, as appropriate, physical features used by harbour seals within the site'.**

3942. The confidence in the assessment for all impacts is considered high considering the baseline information and site-specific data.

### 9.7.3 Potential in-combination effects of the Project with Transmission Assets

3943. A 'combined' assessment has been made with the Transmission Assets[43], for the purpose of an in-combination assessment considering its functional link with the Project.

3944. Strangford Lough SAC was screened in for both the Project and the Transmission Assets and Murlough Lough SAC was screened in for the Transmission Assets.

3945. For the Transmission Assets ISAA Project-alone assessment (Morgan Offshore Wind Limited and Morecambe Offshore Windfarm Ltd, 2023b), there would be no adverse effect on site integrity on any of the screened-in sites. As for the Project, the distances to the closest SACs were outside the ZoI. A full quantitative assessment has been provided in the assessment of all plans and projects and has not been repeated here, but an assessment has been made below of each impact considering the information in **Section 9.7.2** and analysing potential interactions between the projects.

#### 9.7.3.1 Underwater noise and barrier effects

3946. The key interaction was identified as piling and UXO during construction for the projects.

3947. Given that the Project and Transmission Assets would be outwith any SAC and potential PTS effects would be mitigated by any consented project, **it is concluded that there would be no LSE on the reference population (and no AEoI on the SAC).**

#### 9.7.3.2 Vessel interactions

3948. During all phases, there would be additional effects due to increased vessel presence from both projects.

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[43] As the Transmission Assets includes infrastructure associated with both the Project and the Morgan Offshore Wind Project Generation Assets, it should be noted that the combined assessment considers the transmission infrastructure for both the Project and the Morgan Offshore Wind Project Generation Assets.

3949. Given that the Project and Transmission Asset would be outwith any SAC and both projects would adhere to good practice, **it is concluded that there would be no LSE on the reference population (and no AEoI on the SAC).**

#### 9.7.3.3 Indirect effects (changes to prey resource and water quality)

3950. During all phases there would be additional effects due to increased vessel presence from both projects and additional pressure on prey resources.

3951. Given the impacts identified for both projects on prey species and that the Project and Transmission Asset would be outwith any SAC and both projects would adhere to good practice, **it is concluded that there would be no likely significant in-combination effect on the SAC reference populations (and no AEoI).**

#### 9.7.3.4 Disturbance at haul out sites

3952. As there are no significant harbour seal breeding or haul out sites in the NW England MU (SCOS, 2022), **it is concluded that there can be no LSE on the reference population (and no AEoI on the SAC) during construction, operation and maintenance or decommissioning.**

### 9.7.4 Assessment of the potential effects of the Project in-combination with other plans and projects

3953. Section 11.7 of **Chapter 11 Marine Mammals** of the ES details the CEA. This in-combination assessment has been based upon the cumulative assessment and provides a summary of the key information from that assessment without repeating every step of the process. Key information has been taken from **Chapter 11 Marine Mammals** of the ES and carried through with regard to the effect on designated sites.

3954. The effects screened into the in-combination assessment and the identification of the other plans, projects and activities that may result in in-combination effects have been provided in **Appendix 11.4** of the ES.

#### 9.7.4.1 Underwater noise

#### Permanent auditory injury from underwater noise during piling

3955. PTS could occur as a result of piling during offshore wind farm installation or through detonation of underwater explosives (used occasionally during the removal of underwater structures and UXO clearance) (JNCC, 2010a, b). However, if there were the potential for any PTS from any project, suitable mitigation would need to be put in place to reduce any risk to marine mammals. Other noise sources such as dredging, drilling, rock placement, vessel activity, operational WTGs, oil and gas installations or wave and tidal sites would emit broadband noise in lower frequencies and PTS from these activities would be very unlikely.

3956. Therefore, the potential risk of PTS has not been considered further in the in-combination assessment.

3957. The Project would be outwith any SAC and that there was no potential for AEoI from PTS onset in-combination with other projects, as all projects should ensure mitigation was in place to negate the potential for PTS. Therefore, the potential for PTS in-combination has been screened out and not assessed further.

## Disturbance from underwater noise during construction

3958. Section 11.7.3.1 of **Chapter 11 Marine Mammals** of the ES considers disturbance in relation to several sub-effects and then considers all together; Underwater noise impacts from piling at other OWFs, underwater noise impacts from construction activities (other than piling) at other OWFs and disturbance from other industries and activities (which included geophysical survey, seismic survey and UXO clearance).

### Disturbance from piling

3959. The potential disturbance from underwater noise during piling for harbour seal has been assessed based on a disturbance range of 25km (Russell, 2016) for the Project.

3960. This assessment considered the effect of all projects at a population level, noting the Project would not overlap with any SAC.

3961. Of the UK and European OWFs screened in for a construction period that could potentially overlap with the construction of the Project, six OWFs could be piling at the same time as the Project (see **Appendix 11.4** of the ES):

- AyM OWF (PINS Tier 1)
- Mona OWF (PINS Tier 2)
- Morgan OWF (PINS Tier 2)
- Morgan and Morecambe OWFs Transmission Assets (PINS Tier 2)
- Erebus OWF (PINS Tier 1)
- White Cross OWF (PINS Tier 1)

3962. This short list could change as projects develop (noting that three projects were Tier 1 with the most certainty of development and schedule), but this was the best available information at the time of writing, and was considered to reflect the limitations and constraints to project delivery.

3963. The following caveats should be noted in terms of this worst-case:

- The potential areas of disturbance assume that there would be no overlap in the areas of disturbance between different projects
- It was assumed that all OWF projects would be 100% piled, if piled foundations were an option
- The approach was based on the potential for single piling at each wind farm at the same time as single piling at the Project windfarm site. This approach allowed for some of the offshore wind farms not to be piling at the same time, while others could be simultaneously piling. This is considered to be the most realistic worst-case scenario, as it is highly unlikely that all other wind farms would be simultaneously piling at exactly the same time as piling at the Project, especially given the limited active piling time
- The actual duration for active piling time for the Project (a maximum of 619 hours and 36 minutes hours including soft-start, ramp-up and ADD activation (using pin-piles for OSP and WTG)) which could disturb marine mammals would be only a very small proportion of the potential construction period, and this would be the case for other OWFs. This meant that there would be a limited window for any in-combination effect to occur
- In practice, the potential temporary effects would be less than those predicted in this assessment as there is likely to be a great deal of variation in timing, duration (noting this has been typically overestimated in assessments), and hammer energies used throughout the various OWF construction periods. This meant that there would be a limited window for temporal overlap and any in-combination effect to occur. In addition, not all individuals would be displaced over the entire potential disturbance range used within the assessments

3964. The effects of piling for all OWFs including the Project were quantified as less than three harbour seal or up to 2.8% of the Strangford Lough SAC population (**Table 9.56**). This assessment was based on the specific dose-response curves from other projects which were rounded to <1, and thus presented an over-estimated number of harbour seals from the Strangford Lough SAC that could be disturbed. If the SAC- specific densities by Carter *et al.,* (2022) were to be applied to Morgan Offshore Wind Project, Mona Offshore Wind Project, and the Transmission Assets (as it has been done for the Project), the resulting densities and the final number of disturbed harbour seals from the SAC would be much lower and present a more realistic, assessment (as shown in **Table 9.56**).

3965. **There would be no LSE on the reference population (and no AEoI on the SAC). Piling at the Project would cause no additional disturbance effect to harbour seals from the SAC.**

*Table 9.56 Quantitative assessment for in-combination disturbance for harbour seal from the Strangford Lough SAC during piling at other projects*

| Project | Harbour seal density (/$km^2$) (based on Strangford Lough SAC relative densities) | Effect area ($km^2$) based on 25km disturbance range | Maximum number of harbour seal potentially disturbed during single piling |
|---|---|---|---|
| The Project | 0.00000001 | 1,963.5 | 0.000020 |
| Mona | *Dose-response-curve assessment* | | <1 |
| Morgan | *Dose-response-curve assessment* | | <1 |
| Morgan and Morecambe Transmission Assets | *Dose-response-curve assessment* | | <1 |
| AyM | *This SAC were not assessed as part of their RIAA.* | | - |
| Erebus | | | - |
| White Cross | | | - |
| **Total number of harbour seal** | | | **<3.00*** |
| **Percentage of SAC population** | | | **<2.8%**** |

** Given the rounding for the other projects, SAC- specific densities by Carter et al., (2022) were applied to Morgan (0.00000071 seals/$km^2$), Mona (0.00000018 seals/$km^2$), and Transmission Assets (0.000000019 seals/$km^2$). ** The total number (including the Project) of disturbed harbour seals from the SAC was 0.0018, or 0.002% of the Strangford Lough SAC reference population.*

3966. For the Strangford Lough SAC, due to the overall Conservation Status the potential for disturbance from underwater noise from piling has been assessed for the effects of piling for all OWFs including the Project. The iPCoD model estimated there to be no discernible impact to the Strangford Lough SAC assuming a stable population (**Table 9.57**), or assuming a declining population (**Table 9.58**), with the median population size was predicted to be 100% of the un-impacted population size over all. For the harbour seal population of the Strangford Lough SAC, the yearly average was less than a 1% population level effect over both the first six years and 25 year modelled periods for both metrics (**Plate 9.9**; **Plate 9.10**) and **it was concluded that there could be no LSE on the reference population (and no AEoI on the SAC).**

Table 9.57 Results of the iPCoD modelling for the Project, in-combination with other plans and projects, assuming a stable population (Northern Irish MU/SMA demographic parameters from Sinclair et al., (2020)), giving the mean population size of the harbour seal population (Strangford Lough SAC) for years up to 2052 for both impacted and un-impacted populations in addition to the median ratio between their population sizes

| Year | Un-impacted population mean | Impacted population mean | Median impacted as % of un-impacted |
|---|---|---|---|
| Start | 106 | 106 | 100.00% |
| End 2027 | 106 | 106 | 100.00% |
| End 2028 | 107 | 107 | 100.00% |
| End 2031 | 107 | 107 | 100.00% |
| End 2036 | 108 | 108 | 100.00% |
| End 2046 | 108 | 108 | 100.00% |
| End 2051 | 109 | 109 | 100.00% |

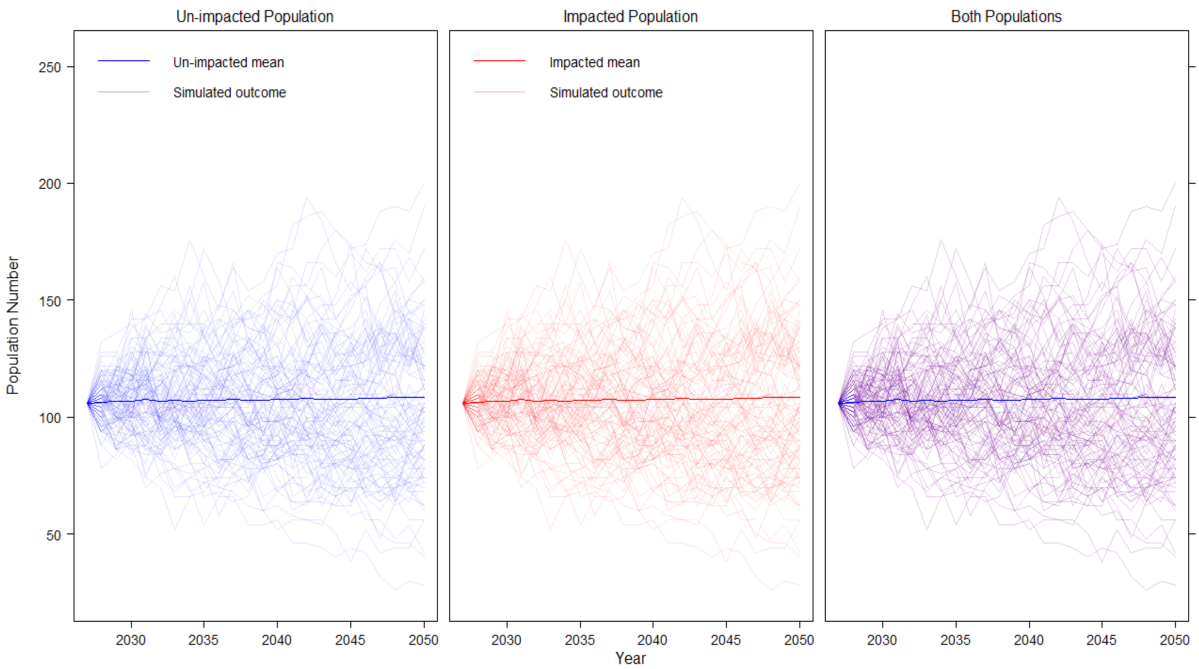


Plate 9.9 Simulated worst-case harbour seal population sizes for both the un-impacted and the impacted populations of Strangford Lough SAC due to piling from the Project in-combination with other plans and projects, assuming a stable population (Northern Irish MU/SMA demographic parameters from Sinclair et al., (2020))

*Table 9.58 Results of the iPCoD modelling for the Project, in-combination with other plans and projects, assuming a declining population (Orkney and North Coast MU/SMA demographic parameters from Sinclair et al., (2020)), giving the mean population size of the harbour seal population (Strangford Lough SAC) for years up to 2052 for both impacted and un-impacted populations in addition to the median ratio between their population sizes*

| Year | Un-impacted population mean | Impacted population mean | Median impacted as % of un-impacted |
|---|---|---|---|
| Start | 106 | 106 | 100.00% |
| End 2027 | 95 | 95 | 100.00% |
| End 2028 | 85 | 85 | 100.00% |
| End 2031 | 62 | 62 | 100.00% |
| End 2036 | 36 | 36 | 100.00% |
| End 2046 | 12 | 12 | 100.00% |
| End 2051 | 7 | 7 | 100.00% |

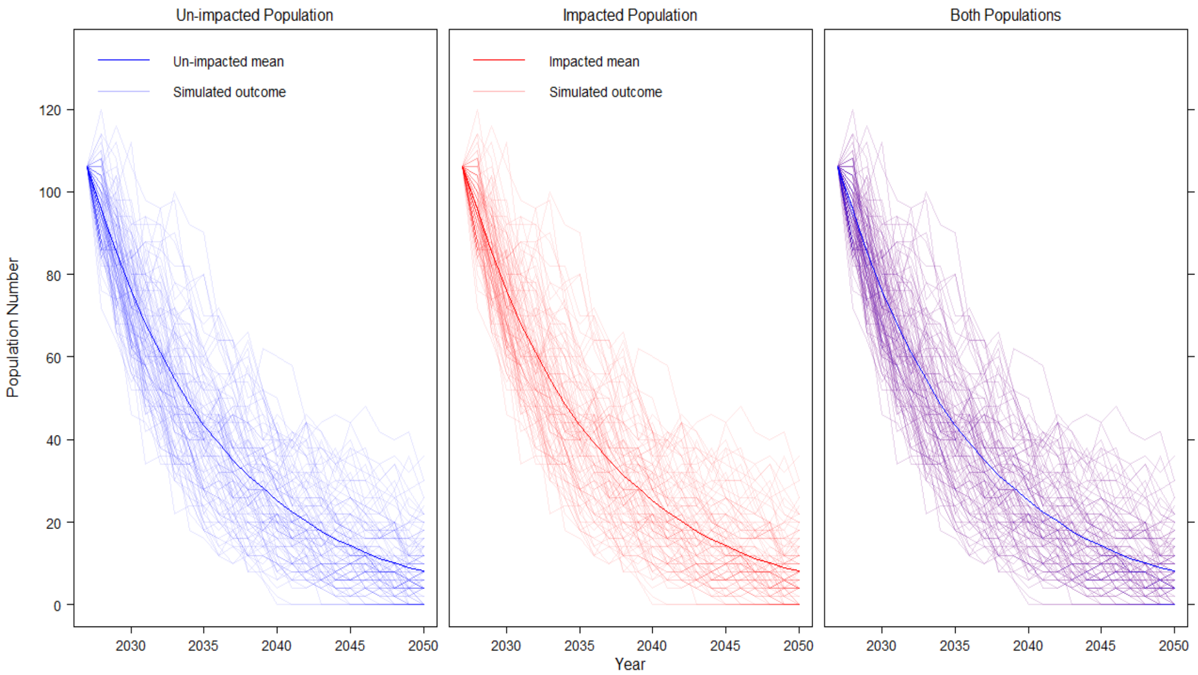


*Plate 9.10 Simulated worst-case harbour seal population sizes for both the un-impacted and the impacted populations of Strangford Lough SAC due to piling from the Project in-combination with other plans and projects, assuming a declining population (Orkney and North Coast MU/SMA demographic parameters from Sinclair et al., (2020))*

### Underwater noise impacts from construction activities (other than piling)

3967. OWFs screened in for other construction activities that could have potential in-combination impacts with piling activities at the Project were (see **Appendix 11.4**):

- Codling Wind Park (PINS Tier 2)
- Dublin Array (PINS Tier 2)
- North Irish Sea Array (PINS Tier 2)
- Sceirde Rocks (PINS Tier 2)

3968. It should be noted that none of these projects were within the harbour seal CEA screening area considered in the ES and effects have therefore not been assessed further. However, it was unlikely that any significant impact would arise from this, given the assessment for impacts during piling.

### Disturbance from other industries and activities

3969. Section 11.7.3.1 of **Chapter 11 Marine Mammals** of the ES considered the effects from geophysical surveys, seismic surveys, aggregation and dredging associated with other projects and UXO clearance.

3970. It should be noted that there were no known licences or licence applications for seismic surveys at the time of assessment (planned to overlap with Project construction) and this has been included for information purposes at this stage as an indicative disturbance assessment.

3971. To establish a worst-case scenario for geophysical surveys, it has been assumed that seals within a 1km radius (equating to a total area of 3.1km²) might experience disturbance from each survey (BEIS[44], 2020). This projected disturbance would extend across an area of 420.2km² throughout the entire transit zone encompassing two surveys.

3972. For aggregate extraction and dredging, there were two projects screened in that could have potential in-combination disturbance effects with piling at the Project. For this type of activity, based on harbour porpoise displacement (which would be used as a proxy; more details in **Section 9.4.4.1**), a disturbance range of 600m would result in a potential disturbance area of 1.13km$^2$ for each project, or up to 2.26km$^2$ for two.

3973. The potential impact area during a single UXO clearance event on grey and harbour seals, based on the modelled worst-case impact range at the Project for TTS/fleeing response (using the impulsive weighted SELs) of 16km (804.25km$^2$) for high-order clearance, and 0.8km (2.01km$^2$) for low-order

[44] As of February 2023, BEIS is known as the DESNZ

clearance. Given that the preferred clearance technique would be of low-order deflagration and the uncertainty regarding UXO potential clearance requirements, one high-order clearance and one low order clearance event have been included to present a more realistic assessment.

3974. **Table 9.59** provides an assessment for each of the listed industries and activities for harbour seal based on the density from the Strangford Lough SAC.

*Table 9.59 Indicative in-combination assessment for the potential disturbance of harbour seal from the Strangford Lough SAC for other industries and activities at other projects*

| Activity | Harbour seal density (/km$^2$) (based on Strangford Lough SAC relative densities | Impact area (km$^2$) | Maximum number of individuals potentially disturbed | % SAC reference population |
|---|---|---|---|---|
| Geophysical surveys x2 | 0.00000001 | 420.2 | 0.0000042 | 0.000004% |
| Aggregate extraction and dredging x2 | *The nearest aggregate extraction sites, North Bristol Deep 1601 & 1602 were outside of the harbour seal screening area* | | | - |
| Seismic survey x1 | 0.00000001 | 5,334.8 | 0.000053 | 0.00005% |
| UXO (two clearance events) | 0.00000001 | 806.26 | 0.0000081 | 0.0000076% |
| **Total** | | | **0.000066** | **0.000062%** |

3975. The in-combination impacts from other industries and activities would not present any likely effect on the Strangford Lough SAC harbour seal population, with less than one harbour seal and less than 1% of the SAC population likely to be affected. There would therefore be no LSE on the reference population (and no AEoI on the SAC).

**Summary of disturbance effects during construction**

3976. For harbour seal, the potential for in-combination disturbance from all noisy activities occurring at the same time as Project piling would have an effect on three harbour seal, or less than 1% of the SAC reference population (**Table 9.60**). As the SAC would not overlap with the Project, only seals foraging or transiting outside the SAC would be affected.

*Table 9.60 Quantified in-combination assessment for the potential disturbance of harbour seal from all underwater noise sources during construction (Grey rows are projects and activities that may take place and therefore indicative assessments)*

| Impact | Harbour seal |
|---|---|
| Piling at other OWFs including the worst-case disturbance from the Project | iPCoD modelling undertaken, <1% population level effect over both the first six years and 25 year modelled periods. |
| Construction activities at other OWF | *Not included as outside screening area* |
| Geophysical surveys | 0.0000042 |
| Aggregates and dredging | *Not included as outside screening area* |
| Seismic surveys | 0.000053 |
| UXO clearance | 0.0000081 |
| **Total for all projects that were currently (or expected to be) in the planning process (realistic worst-case scenario)** | |
| **Percentage of SAC** | **<1% of the Strangford Lough SAC population** |

3977. Based on the current worst-case total, the in-combination assessment for harbour seals for underwater noise for all projects that could be (or were expected to be) undertaken at the same time as the Project was less than 1% of the reference and SAC population. **It is concluded that there is no potential for LSE on the reference population (and no AEoI on the SAC) to occur from disturbance during construction at the Project.**

### Disturbance from underwater noise during operation and maintenance

3978. Underwater noise and disturbance during operation could come from multiple sources, including from operational noise from WTGs, noise of major maintenance work, rock placement or cable repairs, and from the presence of vessels as well as other industrial activities.

3979. A review of the most recent scientific literature was used to determine the potential disturbance of marine mammals from underwater operational noise from WTGs (see Section 11.6.4.1 of **Chapter 11 Marine Mammals** of the ES). The studies indicated that any disturbance would be in the immediate area of the operational turbine, depending on ambient noise levels. There was no evidence of any lasting disturbance or exclusion of harbour seals around windfarm sites during operation (Diederichs *et al*., 2008; Lindeboom *et al*., 2011; Marine Scotland, 2012; McConnell *et al*., 2012; Russell, 2014; Scheidat *et al*., 2011; Teilmann *et al*., 2006; Tougaard *et al*., 2005, 2009), with reports of harbour seals moving through and foraging within operational windfarm sites.

3980. Effects from maintenance activities at OWFs such as additional rock placement or cable re-burial, would be very localised, short in duration and temporary. The potential for in-combination effects from maintenance activities, including vessels at OWFs would be less than the in-combination effects assessed for construction activities other than piling.

3981. **Therefore, operational noise from OWF WTGs and maintenance of OWFs was unlikely to have any significant effect on the reference population (and no AEoI on any SAC) due to the long distances to the projects.**

### Underwater noise from decommissioning

3982. The potential for in-combination impacts during the decommissioning of the Project was unknown at the time of assessment. It is not possible to provide details of the methods that would be used during decommissioning at this time. However, it is expected that the activity levels would be comparable to construction (with the exception of pile driving noise which would not occur).

3983. During decommissioning, the potential effects on harbour seal are anticipated to be similar or less than the worst-case for the construction phase (depending on the methods used). Crucially, any in-combination effect would be dependent upon simultaneous decommissioning and it is not possible to provide a realistic prediction as to which projects may be decommissioned and when.

3984. The potential impacts for the decommissioning have been screened out from further consideration within the CEA (**Appendix 11.4**) and have not been considered further in this in-combination assessment.

### 9.7.4.2 Barrier effects as a result of underwater noise

3985. There was limited information available on the usage of windfarm sites during the construction period, but if a barrier effect arises from underwater noise, it would be intermittent for a limited time period. Given the presence of several windfarms in the IS there would be the potential for disturbance effects to overlap but these have been assessed as having no potential long-term population level effect (see Section 11.7.3.2 in **Chapter 11 Marine Mammals** of the ES). Furthermore, a barrier effect for harbour seal as a result of Project activities would be unlikely given the long distance between the proposed site and the nearest SAC boundary, considering their large foraging ranges.

3986. There was no evidence of any lasting disturbance or exclusion of harbour seals around windfarm sites during operation. Effects from maintenance activities at OWFs, such as additional rock placement or cable re-burial, would be very localised, short in duration, and temporary.

3987. Given these limited temporal and spatial effects and the geographical spread of projects across the IS, it is considered that there would be no potential for a likely significant in-combination barrier effect from underwater noise to occur. **It is concluded there would be no potential for LSE on the reference population (and no AEoI on the SAC) from disturbance during all phases at the Project.**

#### 9.7.4.3 Vessel interactions

3988. Given the low risk of collision to harbour seal and the commitment to mitigation to manage the residual risk, it was concluded in the ES that there would be no LSE from the Project on the wider harbour seal reference population from the impacts of vessel interactions during construction, operation and maintenance or decommissioning. It was considered that any consented OWF project would require similar mitigation which would reduce collision risks.

3989. Vessels associated with aggregate extraction and dredging are large and typically slow moving, using established transit routes to and from ports. Therefore, the potential increased collision risk with vessels was considered to be extremely low or negligible.

3990. Given the low risk to harbour seal and the use of mitigation across OWF projects and other industries, it is concluded that there would be no LSE on the wider harbour seal reference population from the effects of vessel interactions during construction, operation and maintenance or decommissioning. **It is concluded there would be no potential for LSE on the reference population (and no AEoI on the SAC) from disturbance during construction at the Project.**

#### 9.7.4.4 Changes to prey resources

3991. No significant impacts with regard to changes to prey resources would be expected as a result of the Project, as identified in the ES (for all Project phases).

3992. For any potential changes to prey resources, it has been assumed that any potential effects on marine mammal prey species from underwater noise, including piling, would be the same or less than those for marine mammals. Therefore, there would be no additional in-combination effects above those assessed for marine mammals, i.e. if prey were disturbed from an area as a result of underwater noise, marine mammals would be disturbed from the same or greater area. As a result, any changes to prey resources would not affect marine mammals as they would already be disturbed from the area.

3993. Any effects to prey species were likely to be intermittent, temporary and highly localised, with potential for recovery following cessation of the disturbance activity. Any permanent loss or changes of prey habitat would typically represent a small percentage of the potential habitat for prey species in the surrounding area.

3994. Taking into account the assessment for the Project alone, there would be no potential for a LSE on the wider harbour seal reference population during construction, operation and maintenance or decommissioning. This was on the basis that there would be a range of prey species taken by harbour seal over the extent of their foraging areas. Further, much of the effect would occur outside of any area considered important for foraging (i.e. outside of a SAC). **It is concluded there would be no potential for LSE on the reference population (and no AEoI on the SAC) from disturbance during construction at the Project.**

#### 9.7.4.5 Changes to water quality

3995. No significant impacts with regard to water quality were expected as a result of the Project, as detailed in the ES (for all Project phases).

3996. Other OWFs or other construction projects were also considered to have highly localised and temporary effects and would be spatially separated, so there would be no potential for likely significant additive effects.

3997. Given that water quality impacts would be negligible, it is concluded that there would be no likely significant in-combination effect on the wider harbour seal reference population during construction, operation and maintenance or decommissioning. **It is concluded there would be no potential for LSE on the reference population (and no AEoI on the SAC) from disturbance during construction at the Project.**

#### 9.7.4.6 Summary of in-combination assessment

3998. Effects from the Project would not overlap any SAC (Strangford Lough would be 135km from the Project) and the plans, projects and activities included in the in-combination assessment have been screened in on the basis they were within the NW England MU or Northern Ireland MU (as per **Appendix 11.4**).

3999. Due to embedded mitigation and the Project's commitment to further additional mitigation measures it is considered that LSE upon harbour seal would be avoided (e.g. PTS mitigation through the MMMP) and the existing low vessel collision risk could be managed via best practices agreed through the PEMP.

4000. Disturbance of harbour seals outside the SAC would be much less than 5% of the reference population and other indirect effects would be minimal in the scale of the species range.

4001. Given the distances from the SAC and other activities and projects, and limited connectivity, there would be no likely adverse effect on the integrity of the

Strangford Lough SAC in relation to the conservation objective 'Maintain and enhance, as appropriate, the harbour seal population'.

4002. The confidence in the assessment for all impacts is considered medium, yet highly precautionary, particularly given the consideration of a large number of plans or projects and the unlikelihood of temporal overlap of all these activities.

## 9.8 Summary of marine mammal conclusions

4003. A summary of the marine mammal assessments is shown in **Table 9.61**. While a precautionary approach has been taken, when taking into account the application of mitigation and the potential distance between the Project windfarm site and the SACs, it is considered that there would be no adverse effect on integrity.

*Table 9.61 Summary of potential effects on sites designated for marine mammals*

| Site | Qualifying feature | Assessment of Project-alone effects | Assessment of in-combination effects |
|---|---|---|---|
| North Anglesey Marine SAC | Harbour porpoise | No adverse effect on site integrity. | No adverse effect on site integrity. |
| North Channel SAC | | | |
| West Wales Marine SAC | | | |
| Rockabill to Dalkey Island SAC | | | |
| Bristol Channel Approaches SAC | | | |
| Pen Llŷn a'r Sarnau SAC | Bottlenose dolphin | No adverse effect on site integrity. | No adverse effect on site integrity. |
| Cardigan Bay SAC | | | |
| Pen Llŷn a'r Sarnau SAC | Grey seal | No adverse effect on site integrity. | No adverse effect on site integrity. |
| Cardigan Bay SAC | | | |
| Pembrokeshire Marine SAC | | | |
| Strangford Lough SAC | Harbour seal | No adverse effect on site integrity. | No adverse effect on site integrity. |

# 9.9 Marine wildlife application

4004. It is important to note that all cetaceans including harbour porpoise and bottlenose dolphin are listed as European Protected Species (EPS) under Annex IV of the Habitats Directive, and are therefore protected from the deliberate killing (or injury), capture and disturbance throughout their range. Within the UK, The Habitats Directive has been enacted through The Conservation of Habitats and Species Regulations 2017 and the Conservation of Offshore Marine Habitats and Species Regulations 2017. Under these Regulations, it was made an offence to:

- Deliberately capture, injure or kill any cetacean species
- Deliberately disturb them
- Damage or destroy a breeding site or resting place

4005. If required, a Marine Wildlife Licence application would be submitted post-consent. At that point in time, the PDE would have been further refined through detailed design and procurement activities and further detail would be available on the methodology and materials selected for construction, as well as any mitigation measures that would need to be in place as approved in the final MMMPs for piling and UXO clearance.

# 10 Summary

4006. The screening report identified the potential for the Project to interact with European sites with benthic ecology, migratory fish, offshore ornithology and marine mammal qualifying features. This report has provided an Appropriate Assessment and assessed whether any Project activity may have an adverse effect on the integrity of any European sites, either alone or in-combination.

4007. This report has determined that there will no effect on the integrity of any European sites from the Project-alone or in-combination with any other projects or plans. A summary of the findings is provided in **Table 10.1**.

*Table 10.1 Summary of potential effects*

| Site | Qualifying feature | Potential effects | Potential for adverse effect on site integrity, alone and in-combination |
|---|---|---|---|
| **Benthic Ecology** | | | |
| Shell Flat and Lune Deep SAC | ▪ Sandbanks which are slightly covered by seawater all the time (Shell Flat)<br>▪ Reefs (Lune Deep). | ▪ Increased SSCs and deposition<br>▪ Remobilisation of contaminated sediments (all phases<br>▪ Introduction and spread of INNS<br>▪ Risk of deterioration of water quality due to spillages/leakages | No adverse effect on site integrity. |
| **Fish** | | | |
| Dee Estuary/ Aber Dyfrdwy SAC | ▪ River lamprey<br>▪ Sea lamprey | ▪ Temporary and permanent habitat loss/disturbance<br>▪ Increased SSCs and sediment re-deposition<br>▪ Remobilisation of contaminated sediments<br>▪ Underwater noise and vibration<br>▪ EMF<br>▪ Barrier effects<br>▪ Introduction/removal of hard substrate | No adverse effect on site integrity. |
| River Dee and Bala Lake/ Afon Dyfrdwy a Llyn Tegid | ▪ River lamprey<br>▪ Sea lamprey<br>▪ Atlantic salmon | | No adverse effect on site integrity. |
| Afon Gwyrfai a Llyn Cwellyn SAC | ▪ Atlantic salmon | | No adverse effect on site integrity. |
| Afon Eden – Cors Goch Trawsfynydd SAC. | Atlantic salmon | | No adverse effect on site integrity. |
| **Offshore Ornithology (key receptors; refer also to Table 8.199)** | | | |
| Liverpool Bay SPA | Red-throated diver | Disturbance/displacement/barrier effects (construction and decommissioning, operation and maintenance) | No adverse effect on site integrity. |

| Site | Qualifying feature | Potential effects | Potential for adverse effect on site integrity, alone and in-combination |
|---|---|---|---|
| | Common Scoter | Disturbance/displacement/barrier effects (construction and decommissioning, operation and maintenance) | No adverse effect on site integrity. |
| | Little gull | Collision risk (operation and maintenance phase) | No adverse effect on site integrity. |
| | Common tern | Although initially screened in, it was concluded that common terns present at the windfarm site are very unlikely to be associated with the Liverpool Bay SPA population | No adverse effect on site integrity. |
| Morecambe Bay and Duddon Estuary SPA and Ramsar site | Lesser black-backed gull | Collision risk (operation and maintenance phase) | No adverse effect on site integrity. |
| | Herring gull | Collision risk (operation and maintenance phase) | No adverse effect on site integrity. |
| | Sandwich tern | Although initially screened in, it was concluded that Sandwich terns present at the windfarm site are very unlikely to be associated with the Morecambe Bay and Duddon Estuary SPA and Ramsar site population | No adverse effect on site integrity. |
| Ribble and Alt Estuaries SPA and Ramsar site | Lesser black-backed gull | Collision risk (operation and maintenance phase) | No adverse effect on site integrity. |
| Anglesey Terns/Morwenoliaid Ynys Môn SPA | Sandwich tern | Although initially screened in, it was concluded that Sandwich terns present at the windfarm site are very unlikely to be associated with the Anglesey Terns/Morwenoliaid Ynys Môn SPA population | No adverse effect on site integrity. |

| Site | Qualifying feature | Potential effects | Potential for adverse effect on site integrity, alone and in-combination |
|---|---|---|---|
| Glannau Aberdaron ac Ynys Enlli/Aberdaron Coast and Bardsey Island SPA | Manx shearwater | Disturbance/displacement/barrier effects (operation and maintenance) | No adverse effect on site integrity. |
| Copeland Islands SPA | Manx shearwater | Disturbance/displacement/barrier effects (operation and maintenance) | No adverse effect on site integrity. |
| Ailsa Craig SPA | Gannet | Disturbance/displacement/barrier effects and collision risk (operation and maintenance) | No adverse effect on site integrity. |
| Skomer, Skokholm and the Seas off Pembrokeshire/Sgomer, Sgogwm a Moroedd Penfro SPA | Manx shearwater | Disturbance/displacement/barrier effects (operation and maintenance) | No adverse effect on site integrity. |
| Grassholm SPA | Gannet | Disturbance/displacement/barrier effects and collision risk (operation and maintenance) | No adverse effect on site integrity. |
| **Marine mammals** | | | |
| North Anglesey Marine SAC | Harbour porpoise | ▪ Permanent and temporary auditory injury during piling<br>▪ Disturbance impacts from underwater noise<br>▪ Underwater noise and disturbance from other sources<br>▪ Increased SSCs<br>▪ Barrier effects<br>▪ Vessel interactions | No adverse effect on site integrity. |
| North Channel SAC | Harbour porpoise | | No adverse effect on site integrity. |
| West Wales Marine SAC | Harbour porpoise | | No adverse effect on site integrity. |
| Rockabill to Dalkey Island SAC | Harbour porpoise | | No adverse effect on site integrity. |

| Site | Qualifying feature | Potential effects | Potential for adverse effect on site integrity, alone and in-combination |
|---|---|---|---|
| Bristol Channel Approaches SAC | Harbour porpoise | ▪ Changes to prey resources<br>▪ Changes to water quality<br><br>Additionally for seal species:<br>▪ Disturbance at seal haul out sites | No adverse effect on site integrity. |
| Pen Llŷn a`r Sarnau SAC | Bottlenose dolphin | | No adverse effect on site integrity. |
| | Grey seal | | No adverse effect on site integrity. |
| Cardigan Bay SAC | Bottlenose dolphin | | No adverse effect on site integrity |
| | Grey seal | | No adverse effect on site integrity. |
| Pembrokeshire Marine SAC | Grey seal | | No adverse effect on site integrity. |
| Strangford Lough | Harbour seal | | No adverse effect on site integrity. |

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