11. MARINE ORNITHOLOGY

11.1 Introduction

This chapter of the EIA Report presents an assessment of the potential impacts upon offshore ornithology and assesses the likely significant effects arising from the construction, operation and maintenance, and decommissioning of the Project, as detailed in Chapter 5: Project Description. This chapter has been prepared on behalf of FST by Cork Ecology with additional input from Hi-Def Aerial Surveying Ltd and Dr Tom Gittings.

Additional supporting information on offshore and intertidal ornithology is presented in the Offshore Ornithology Baseline Technical Report, hereafter referred to as the Baseline Ornithology Report (EIAR Appendix 11-1).

This chapter also summarises information contained within other chapters and technical reports:

- Chapter 8: Marine Water and Sediment Quality: to be referenced for an overview on the suspended sediment concentrations expected during construction, operation and maintenance, and decommissioning phases, which can have direct impacts on foraging seabirds (e.g. impairment of visibility and therefore foraging ability which might be expected to reduce foraging success), as well as indirect impacts on their prey;
- Chapter 9: Benthic Ecology: to be referenced for an overview of the potential impacts to benthic species, which could indirectly impact seabirds;
- Chapter 10: Fish and Shellfish Ecology: to be referenced for an overview of the potential impacts to fish species, which could indirectly impact seabirds;
- Chapter 11: Appendix 11.2 - Seabird Displacement Matrices Technical Report: to be referenced for a description of the approach and predicted displacement and mortality outputs.
- Chapter 11: Appendix 11-3 - Seabird Collision Risk Modelling Technical Report: to be referenced for a description of the approach and results;
- Chapter 11: Appendix 11-4 – Migratory non-seabird Collision Risk Modelling Technical Report: to be referenced for a description of the approach and results;
- Chapter 11: Appendix 5 – Offshore Ornithology Connectivity and Apportioning Technical Report, hereafter referred to as the Offshore Ornithology Apportioning Report: to be referenced for a description of the connectivity of qualifying interest seabird species at Special Protection Areas (SPAs) to the Project.
- Chapter 11: Appendix 6 - Sceirde Rocks Offshore Wind Farm Population modelling report, hereafter referred to as the PVA Assessment: to be referenced for a description of the approach and results; and
- Chapter 11: Appendix 7 - Digital video aerial surveys of seabirds and marine megafauna at Fuinneamh Sceirde Teoranta: 2-Year Report October 2021 to September 2023, hereafter referred to as the Aerial Survey Two Year Report: to be referenced for a detailed description of the two years of baseline aerial surveys over the wider survey area out to 10 km.

11.1.1 Statement of Authority

Colin Barton of Cork Ecology is the lead author of the Offshore Ornithology EIAR chapter. Colin graduated from the University of Aberdeen in 1992, with a BSc. Honours degree in Biology (Ecology). Colin has worked as an independent consultant for offshore wind projects since 2001, specialising in all aspects of ornithology. He has provided ornithological support and advice for several offshore wind projects in Irish and UK waters, with key inputs including survey design, ESAS training and advice, data input and validation, database management and analysis, the writing of baseline and impact assessment chapters on birds, input into HRA/NIS documents on birds and post-construction monitoring.

Monthly digital aerial surveys were conducted by HiDef Aerial Surveying Ltd. In addition, HiDef undertook analysis of the digital survey data and prepared Baseline Ornithology Report as well as the Offshore Ornithology Apportioning Report.

Ben Cockshull and Dr Kelly Mcleod were the principal staff within Hi-Def who were involved in the preparation of the technical reports upon which the assessments in this chapter were based. Ben is a Project Manager at HiDef and studied at the University of Exeter for a BSc in Biological Sciences including marine biology, microbial ecology and operations management. He has over four years project management experience, having previously worked as a Scientific Consultant in the maritime sector working on a wide range of projects across Europe, the Middle East and West Africa. At HiDef, Ben has been involved with projects for Offshore Wind Farms and conservation. Kelly, as Associate Director (Science), is a marine mammal scientist with more than 20 years' experience working for the NCC and the Sea Mammal Research Unit. She provides scientific oversight to all HiDef projects and has direct responsibility for the delivery of technical reports. Kelly continues to author numerous scientific papers and regularly undertakes peer-review for journal manuscripts. In addition, the following staff at HiDef also had input to the preparation of technical reports and digital survey data for this chapter:

- Catherine Irwin: Catherine completed a Geography BSc at the University of Glasgow with a background in physical geography and spatial analysis. As Head of Projects, leads the Projects team to ensure the smooth day-to-day running of our DAS program. She has over 10 years' experience in the DAS sector and is responsible for the delivery of many high-profile projects and has also contributed to peer-reviewed studies and papers.
- David Thain: David is a Senior Project Manager responsible for the planning and oversight of a variety of projects. He works closely with the operations team to observe that deliverables are met on time and to scope, whilst administering change controls that effect's the project plan. David has over 10 years' experience in Project Management, initially from an Architectural management aspect. More recently he has moved to projects that focus on digital modelling of spatial GIS data.
- Diane Pavat: Diane is an Ecological Consultant and studied for a BSc in Biology from the Université Grenoble Alpes and her MSc in Marine Conservation from the University of Aberdeen. She has over three years' experience in using R-studio and ArcGIS to interrogate and analyse offshore wind farm DAS data. Recently, she presented a poster at the international Conference for Wind energy and Wildlife impact 2023 on her research on black-legged kittiwake flight height trends in the UK and Ireland.
- Polly Brown: Polly joined HiDef in 2024 as an Ecological Consultant after completing her BSc in Zoology at the University of Glasgow and MRes in Biodiversity and Conservation from the University of Leeds. During her MRes she studied foraging ranges of seabirds to assess the impact of offshore wind farms on colonies in southeast Asia. Polly prepares technical reports to support HiDef's clients, including offshore wind developers, alongside Government departments and agencies.
- Rory Thomson: Rory Thomson is a Marine Data Scientist at HiDef, who has a background in quantitative methods and statistics. He has over two years of experience in environmental consenting and studied for an MSc in Quantitative Methods in Biodiversity, Conservation and Epidemiology at the University of Glasgow. With his strong background in R-studio, Rory assists in analysing and modelling DAS data.

Dr Tom Gittings conducted the Collision Risk Modelling (CRM) for this EIAR chapter. Tom is an ecologist with 28 years' experience in professional consultancy work and research. Tom specialises in ecological surveying, monitoring and evaluation, ecological impact assessment, habitat management, and avian, invertebrate, wetland and woodland ecology. He is currently working as an independent ecological consultant. His previous experience includes working for the RPS Group (a multi-disciplinary environmental consultancy) and carrying out research into forest and wetland biodiversity in the Department of Zoology Ecology and Plant Science at University College Cork. He has a BSc (Hons) and a PhD in Ecology and is a member of the Chartered Institute of Ecology and Environmental Management. His recent consultancy work includes assessments for planning applications (including Appropriate Assessments, Environmental Impact Statements, and expert witness work at oral hearings), large-scale habitat surveys, preparation of management plans, contributions to multidisciplinary conservation plans, and specialist ecological survey and research.

Tom has specific expertise in ornithological assessments for wind farm projects. He has been involved in numerous wind farm projects. His input to these projects has variously included field surveys, collision risk modelling, population modelling, writing the ornithological sections of EIS/EIAR and NIS reports, expert witness services at oral hearings, and provision of scoping advice and peer review services. He also has a wide range of other ornithological expertise, with a particular focus on waterbird ecology. Tom has also lectured on Appropriate Assessment, Environmental Impact Assessment, Habitat Survey, Woodland Management, and Invertebrate Ecology to a number of courses in University College Cork and University College Dublin. Tom was the recipient of the Distinguished Recorder Award 2014 from the National Biodiversity Data Centre in recognition of his contribution to invertebrate recording in Ireland.

This EIAR chapter was reviewed by Padraig Cregg of MKO. Padraig is a Principal Ornithologist with MKO and has over eleven years of experience working in environmental consultancies. Padraig is a Principal Ornithologist with MKO and has over eleven years of experience working in environmental consultancies. Padraig's key strengths and areas of expertise are in ornithology and ecology surveying and in writing Natura Impact Statements (NIS) and the Biodiversity chapters of Environmental Impact Assessment Reports (EIAR) to accompany development permission applications. Since joining MKO Padraig has been involved in survey design, execution, project management and the impact assessment of over 40 proposed wind farm developments. He has played a key role in project managing these planning applications through the statutory planning system, with more projects in the pipeline.

11.2 Legislation, Policy and Guidelines

This section outlines the legislation, policy and guidance that is relevant to the assessment of the potential impacts on offshore and intertidal ornithology associated with the construction, operation and maintenance, and maintenance and decommissioning of the Offshore Site, in addition to those listed in Section 1.1.2 of Chapter 1: Introduction of this EIAR.

11.2.1 Legislation, Policy and Guidelines

Over and above the legislation presented in Chapter 1: Introduction and Chapter 2: Background and Policy, the legislation, policy and guidance relevant to the assessment of potential effects from the Project on Offshore Ornithology are outlined below.

Birds Directive 2009/147/EC

The Directive 2009/147/EC of the European Parliament and of the Council of 30 November 2009 on the conservation of wild birds ("Birds Directive") seeks to conserve all wild birds in the EU by setting out rules for their protection, management and control. The Directive covers birds, their eggs, nests and habitats. EU countries must take action to maintain or restore the populations of endangered species to a level, which is in line with ecological, scientific and cultural requirements, while taking into account economic and recreational needs.

The Birds Directive aims to protect all naturally occurring wild bird species present in the EU and their most important habitats. In addition to halting the decline or disappearance of bird species, the

Directive aims to allow bird species to recover and thrive over the long-term. To achieve these aims, EU countries are required to take any necessary measures to maintain or restore bird populations.

Bird species listed on Annex I and migratory species are subject to special conservation measures to protect their habitat, through the establishment of Special Protection Areas (SPAs), under Directive 2009/147/EC on the Conservation of Wild Birds (the Wild Birds Directive 1979). These SPAs must have conditions favourable to these species survival and be situated in the birds' natural area of distribution (i.e. where they naturally occur). Particular attention is paid to wetlands. These SPAs form part of the Natura 2000 network of protected ecological sites.

Council Directive 92/43/EEC of 21 May 1992 on the conservation of natural habitats and of wild fauna and flora ("Habitats Directive") is transposed into Irish national law by the European Communities (Birds and Natural Habitats) Regulations 2011 ("Habitats Regulations") and Part XAB of the Planning and Development Acts 2000 (as amended). Species listed in Annex I and II of the Habitats Directive (and afforded protection under the Habitats Regulations);

Article 12 of the Habitats Directive requires that the measures for the strict protection of species listed in Annex IV (a), prohibiting all forms of deliberate capture or killing, deliberate disturbance particularly during breeding, rearing, hibernation and migration, and deliberate deterioration and destruction of breeding sites or resting places; In particular, EU member states must prohibit:

- All forms of deliberate capture or killing in the wild;
- Deliberate significant disturbance, particularly during breeding and rearing;
- The destruction of, or damage to, nests or eggs, or removal of nests;
- The use of any method for large-scale and non-selective capture or killing such as with nets, cages and glue; and
- The keeping, transport and sale of specimens taken from the wild.

Wildlife Act 1976 to 2021

The Wildlife Act 1976 to 2021 (the "Wildlife Act") is the principal national legislation providing for the protection of wildlife and the control of some activities that may adversely affect wildlife in Ireland. The Wildlife Act came into operation on 1 June 1977. It was the only major legislation concerned with wildlife that was passed in the previous 45 years. It replaced the Game Preservation Act, 1930, and the Wild Birds (Protection) Act, 1930.

The aims of the Wildlife Act are to provide for the protection and conservation of wild fauna and flora, to conserve a representative sample of important ecosystems, to provide for the development and protection of game resources and to regulate their exploitation, and to provide the services necessary to accomplish such aims.

The main objectives of the Wildlife Amendment Act (2000) S.I. No. 176/2023 include aims to improve some existing measures, and introduce new ones, to enhance the conservation of wildlife species and their habitats; enhance a number of existing controls in respect of hunting, which are designed to serve the interests of wildlife conservation; ensure or strengthen compliance with international agreements; strengthen the protective regime for Special Areas of Conservation (SACs) and to give specific statutory recognition to the Minister's responsibilities in regard to promoting the conservation of biological diversity, in light of Ireland's commitment to the UN Convention on Biological Diversity.

Bonn Convention

The UN Convention on Migratory Species (CMS), also known as the Bonn Convention, is an environmental treaty of the United Nations that provides a global platform for the conservation and sustainable use of terrestrial, aquatic and avian migratory animals and their habitats.

The Convention requires signatories to conserve migratory species and their habitats by providing strict protection for endangered migratory species (Appendix I of the Convention) and lists migratory species which would benefit from multilateral Agreements for conservation and management (Appendix II of the Convention). Ireland is a party member of the Bonn Convention. The main pieces of legislation to ensure that the provisions of the Bonn convention are applied include the Birds Directive and the Habitats Directive.

Bern Convention

The European Community is a contracting party to the Convention on the Conservation of European Wildlife and Natural Habitats adopted at Bern on 19 September 1979. The aim of the Bern Convention is to ensure the conservation of European wildlife and natural habitats by means of cooperation between member States. The Bern Convention co-ordinates the action of European States in adopting common standards and policies for the sustainable use of biological diversity, thus contributing to the improvement of the quality of life of Europeans and the promotion of sustainable development.

11.2.2 Policy

The following policy documents have been considered in the preparation of this chapter:

- The Offshore Renewable Energy Development Plan (OREDP) (Ireland) (DCCAE, 2014);
- National Marine Planning Framework (DHLGH, 2021);
- Marine Planning Policy Statement (Ireland) (DHLGH, 2019);

11.2.3 Guidance

This chapter has been drafted considering the following guidance and associated supporting publications:

- Guidance on EIS and NIS Preparation for Offshore Renewable Energy Projects, (DCCAE, 2017);
- Guidance on Marine Baseline Ecological Assessments & Monitoring Activities: Offshore Renewable Energy Projects Parts 1 and 2, (DCCAE, 2018a&b);
- Guidelines on the information to be contained in Environmental Impact Assessment Reports, (EPA, 2022);
- Guidelines for Ecological Impact Assessment in the UK and Ireland: Terrestrial, Freshwater, Coastal and Marine version 1.2, (CIEEM, 2022);
- Using a collision risk model to assess bird collision risks for offshore windfarms, (Band, 2012);
- Attributing seabirds at sea to appropriate breeding colonies and populations, (Butler et al., 2020);
- JNCC Review of data used to calculate avoidance rates for collision risk modelling of seabirds, (Ozsanlav-Harris et al., 2023);
- Guidance on ornithological cumulative impact assessment for offshore wind developers, (King et al., 2009);
- Assessment methodologies for offshore wind farms, (Maclean et al., 2009);
- A stochastic collision risk model for seabirds in flight. Marine Scotland commissioned report (McGregor et al., 2018);
- A Population Viability Analysis Modelling Tool for Seabird Species: Guide for using the Population Viability Analysis (PVA) tool (v2.0) user interface, (Mobbs et al., 2020);
- Natural England Offshore Wind Marine Environmental Assessments: Best Practice Advice for Evidence and Data Standards. Phase II: Expectations for pre-application

engagement and best practice advice for the evidence plan process, (Parker et al., 2022b);
- Natural England Offshore Wind Marine Environmental Assessments: Best Practice Advice for Evidence and Data Standards. Phase III: Expectations for data analysis and presentation at examination for offshore wind applications (Parker et al., 2022c);
- Interim Guidance on Apportioning Impacts from Marine Renewable Developments to Breeding Seabird Populations in Special Protection Areas, (NatureScot, 2018);
- NatureScot Seasonal Periods for Birds in the Scottish Marine Environment, (NatureScot, 2020);
- NatureScot Guidance to support Offshore Wind Applications: Guidance Notes 1 – 11, (NatureScot, 2023);
- Desk-based revision of seabird foraging ranges used for HRA screening, (Woodward *et al.,* 2019);
- Statutory Nature Conservation Bodies (SNCBs) Interim Displacement Advice Note, (SNCBs, 2022a);
- SNCB Interim Advice On The Treatment Of Displacement For Red-Throated Diver, (SNCBs, 2022b); and
- Joint advice note from the SNCBs regarding bird collision risk modelling for offshore wind developments, (SNCBs, 2024).
- Joint advice note from the Statutory Nature Conservation Bodies (SNCBs) regarding bird collision risk modelling for offshore wind developments. (JNCC, *et al.,* 2024).

11.3 Scoping and Consultation

Stakeholder consultation has been ongoing throughout the EIA process and has played an important part in ensuring the scope of the baseline characterisation and impact assessment are appropriate with respect to the Project and the requirements of the regulators and their advisors.

The Scoping Report was distributed to key stakeholders in September 2023. The scoping responses received relevant to offshore ornithology are provided in Table 11-1 below, which provides a high-level response on how these comments have been addressed within the EIAR.

Further consultation has been undertaken throughout the pre-application stage. The list below summarises the consultation activities carried out relevant to offshore ornithology:

- Meeting with An Bord Pleanála (ABP) on 19 September 2023 – some discussion took place regarding the use of other monitoring methods and data sets in addition to digital aerial surveys (DAS). Xodus advised ABP that the EIAR would review and incorporate all possible data sets and where certain methods were not used, a justification of why this was appropriate. The datasets and information sources for this chapter are outlined in Section 11.8.7.

Table 11-1 Summary of consultation relating to offshore ornithology

Consultee	Comment	Where the comment has been addressed in the EIAR
Birdwatch Ireland	No response	N/A
Marine Institute of Ireland	No response	N/A

Consultee	Comment	Where the comment has been addressed in the EIAR
NPWS	E-mail sent 23/1/2024 re Seabirds Count book (Burnell *et al.,* 2023) and Colhoun *et al.,* 2023 South Connemara tern survey report. Email also included an unpublished count of 32,836 pairs of Manx shearwater breeding on Cruagh Island (D. Tierney, pers. comm.).	Reference to Burnell *et al.,* 2023 and the Seabirds Count data is made throughout Offshore Ornithology chapter - relevant regional counts are summarised in Table 11-11. South Connemara tern survey report (Colhoun *et al.,* 2023). Relevant results presented in Table *11-7*. Manx shearwater count presented in Table *11-7* and Table 11-11.

11.4 Survey Methodology

11.4.1 Study Areas

11.4.1.1 Offshore Ornithology Regional Study Area

The Offshore Ornithology Regional Study Area was determined by the area within which potential impacts to breeding seabirds could occur and was based on the foraging ranges of breeding seabirds. Many seabirds have large foraging ranges which in some cases extend several hundred kilometres from their breeding colonies. Birds may therefore overlap (i.e. have connectivity) with the Offshore Site, even when the colonies they originate from are a significant distance away. The Offshore Ornithology Regional Study Area therefore also encompasses the Special Protection Area (SPA) breeding colonies with potential connectivity to the Offshore Site during the breeding season (Figure 11-1).

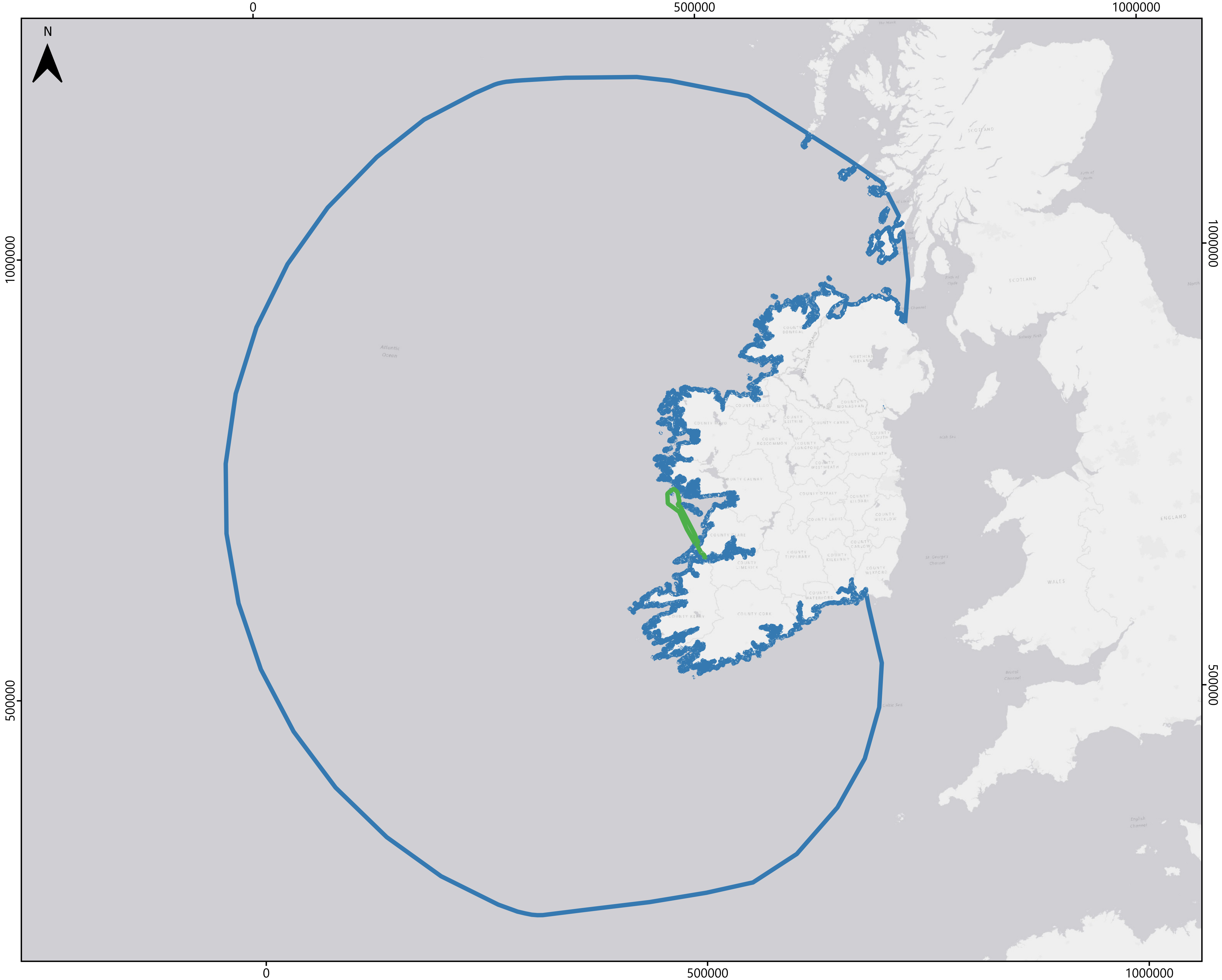
Published mean-maximum foraging ranges (plus one standard deviation (+1 S.D.)) in Woodward et al. (2019) were used to define the Offshore Ornithology Regional Study Area. Gannet has the largest foraging range (315.2 km ± 194.2 km) of the key species considered in the ornithology assessment. The Offshore Ornithology Regional Study Area therefore extends 509.4 km from the Offshore Array Area (OAA) (Figure 11-2). Foraging areas from SPA and non-SPA breeding colonies for other key species in this assessment will fall within the mean-maximum foraging range of gannet. Therefore, this approach is appropriate to define the maximum extent of the Offshore Ornithological regional study area. This approach has been used in recent EIAR assessments for the Irish east coast Phase 1 projects, for example Arklow Bank II (SSE Renewables, 2024), NISA (Ove Arup & Partners, 2024) and Oriel Wind Farm (RPS, 2024).

Individuals from a breeding colony that may potentially be affected by the Offshore Site could also be affected by potential impacts from other OWF developments within the foraging range of breeding seabirds from that colony. The cumulative study area for each species will therefore be defined by implementing a search area equivalent to the species-specific mean-maximum foraging range (+ 1 S.D.) along a marine pathway, from those potentially affected breeding colonies of that species.

In the non-breeding season, seabirds are not constrained by colony location and, depending on individual species, range widely within Irish waters and beyond. For this assessment, the Cumulative Study Area for seabird species in the non-breeding season (where an assessment is deemed to be required) was based on Furness (2015) which presents Biologically Defined Minimum Population Scales (BDMPS). This is outlined further in Section 11.7.3. This approach was agreed between the east coast Phase 1 developers and to maintain consistency in approach has been adopted here.

11.4.1.2 Offshore Ornithology Study Area

For the purposes of this EIAR chapter, the Offshore Ornithology Study Area is defined as the OAA and a surrounding four km buffer, which equates to the area covered by the baseline monthly digital aerial surveys (Figure 11-2). Irish guidance (DCCAE, 2018) suggests that for sites larger than 10 km^2, a buffer of 4 km around the site is adequate. A buffer of 4 km around a potential offshore wind farm site was also recommended in a review of assessment methodologies for offshore wind farms for COWRIE in the UK (MacLean et al., 2009).



SCEIRDE ROCKS WINDFARM
Feirm Ghaoithe Charraigeacha Sceirde

LEGEND

- EIAR Site Boundary
- Regional Study Area

0 100 200 km
Kilometres

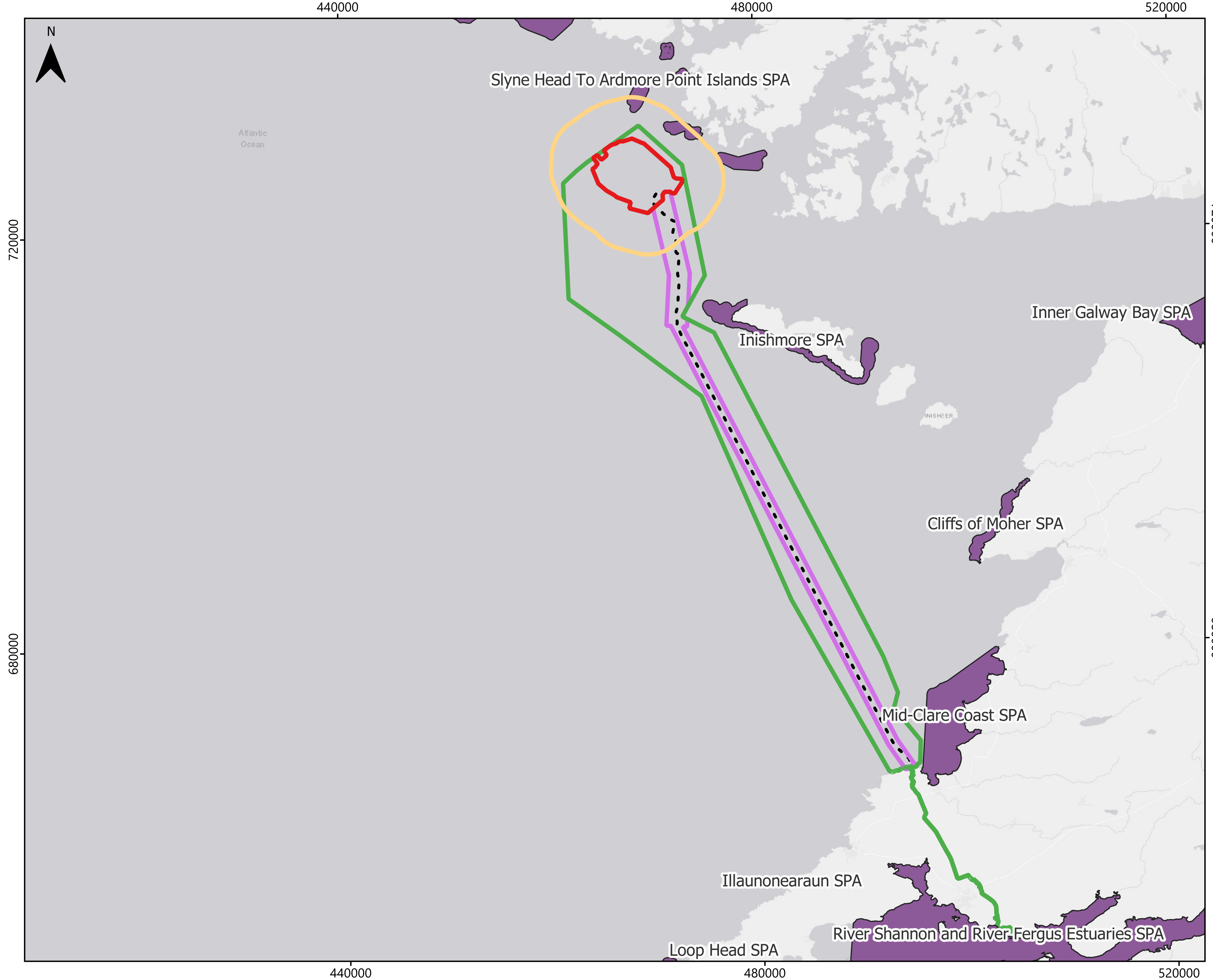
PROJECT TITLE
Sceirde Rocks

MAP TITLE
Offshore Ornithology Regional Study Area

VER	REMARKS	DATE	Drawn	Approved
V1		2024-12-16	CF	ÓM

DRAWING NO
Figure 11-1

SCALE	PAPER SIZE	DATUM	PROJECTION
1:5,160,000	A4	WGS 84	UTM 29N



SCEIRDE ROCKS WINDFARM
Feirm Ghaoithe Charraigeacha Sceirde

LEGEND

- EIAR Site Boundary
- Offshore Array Area (OAA)
- Offshore Export Cable
- Offshore Export Cable Corridor
- Ornithology Study Area (4km buffer from OAA)
- Coastal Special Protection Area (SPA's)

0 10 20 km
Kilometres

Data Source

PROJECT TITLE
Sceirde Rocks

MAP TITLE
Offshore Ornithology Study Area

VER	REMARKS	DATE	Drawn	Approved
V1		2025-01-15	CF	ÓM

DRAWING NO
Figure 11-2

SCALE	PAPER SIZE	DATUM	PROJECTION
1:440,000	A4	WGS 84	UTM 29N

11.4.2 Baseline Data Sources

The characterisation of the receiving environment has been informed based on information from a series of site-specific monthly digital aerial surveys conducted between October 2021 and September 2023. As highlighted in the DCCAE guidance, marine environmental data is derived from a wide range of existing available data sources (DCCAE, 2017), therefore a thorough desk-based study of published datasets has also been conducted. Full details of the data sources considered in the development of the Offshore Ornithology baseline are presented in Table 11-2.

Table 11-2 Data sources considered in the development of the Offshore Ornithology Baseline

Data Source	Type of Data	Temporal and Spatial Coverage
Site-specific survey data		
2021-2023 Survey data	Project-specific monthly digital aerial survey data at 1 km transect spacing covering OAA and 4 km buffer	24 surveys conducted between October 2021 and September 2023. Used to inform the EIAR Assessment.
2021-2023 Survey data	Project-specific monthly digital aerial survey data at 2 km transect spacing covering wider area out to 10 km	24 surveys conducted between October 2021 and September 2023. Used to provide context in the EIAR Assessment.
Published survey data from the wider region		
JNCC Report No. 267 (Pollock *et al.* 1997)	Published Report	ESAS survey data collected between 1980 and 1997 in Irish waters, including a period of intensive surveys between 1994 and 1997, which targeted areas around Ireland with poor survey coverage. Used to provide historic context for the wider west coast area.
ObSERVE 2016 visual aerial surveys (Rogan *et al.* 2018)	Published Report	Visual aerial surveys conducted in summer and winter 2016 to assess the occurrence and distribution of seabird species off the west coast of Ireland. Used to provide recent context for the wider west coast area.
ObSERVE 2021-2023 visual aerial surveys (Giralt Paradell *et al.*, 2024)	Published Report	Visual aerial surveys conducted between 2021 and 2023 to assess the occurrence and distribution of seabird species in Irish waters. Used to provide recent context for the wider west coast area.
Bird Atlas 2007-2011 (Balmer *et al.*, 2013)	Published book	Breeding and winter bird distribution atlas for the UK and Ireland. Used to provide recent context for the wider west coast area.

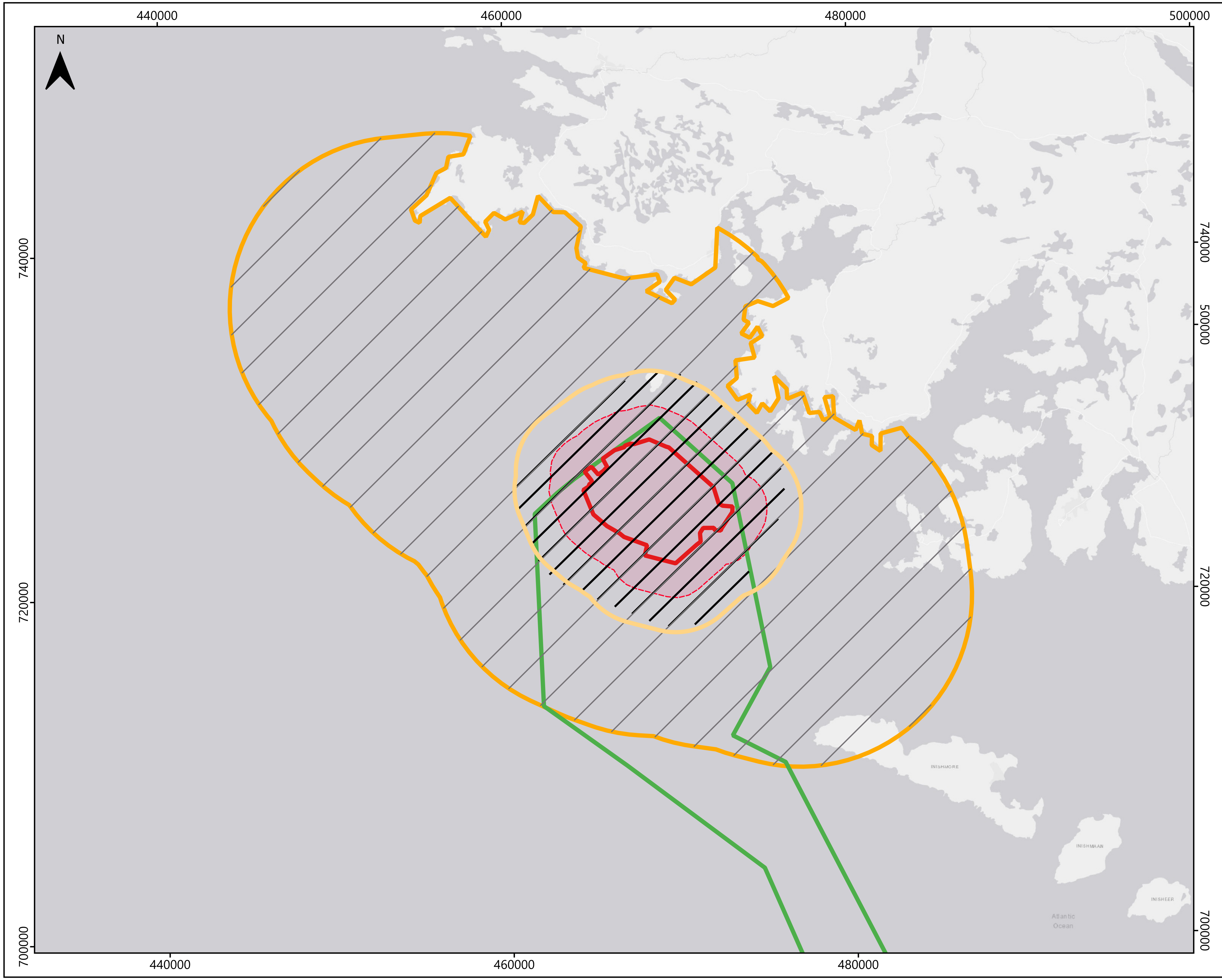
Data Source	Type of Data	Temporal and Spatial Coverage
Seabird colony data from the wider region		
Burnell *et al.,* 2023	Seabirds Count national colony census data	Published data from a census of breeding seabirds in Britain and Ireland between 2015 and 2021. Used to provide SPA reference populations for the EIAR.
Seabird Monitoring Programme	Online Colony Counts	Online database of seabird colony counts in Ireland and UK – most recent data from Seabirds Count national census 2015-2021 and some more recent data. Used to provide SPA reference populations for the EIAR.
Cummins *et al.*, 2019	NPWS Published Report	The Status of Ireland's Breeding Seabirds: Birds Directive Article 12 Reporting 2013 – 2018. Used to provide SPA reference populations for the EIAR.
Colhoun *et al.,* 2023	Report to Science Advisory & Research Directorate, NPWS	2023 survey, results and conservation assessment of breeding terns in South Connemara, Galway. Relevant results presented in Table *11-7.*

11.4.3 Digital Aerial Surveys

In October 2021, HiDef Aerial Surveying Limited (hereafter 'HiDef') were commissioned to undertake a programme of high-resolution digital video aerial surveys (DAS) for seabirds (and other marine fauna) in support of the development proposal, flying a series of strip-transects across the Offshore Site and surrounding buffer area.

The transect-based technique implemented for the baseline surveys has been demonstrated to be highly effective at detecting birds and marine mammals (Thompson *et al.*, 2012; Williamson, 2016; Mendel *et al.*, 2018). The most important aspect of offshore wildlife surveys is ensuring accurate detection rates; if individuals are not being recorded as being present, then population estimates, density and distribution will be incorrect, which may present a risk to the project.

The DAS survey design consisted of 32 strip transects over the original development area and a 10 km surrounding buffer, extending roughly northeast to southwest, perpendicular to the depth contours along the coast to ensure each transect sampled a similar range of habitats (primarily relating to water depth) to reduce the variation in seabird abundance estimates between transects. The survey design consisted of 12 1 km-spaced transects across the OAA (37.28km^2) and a surrounding 2 km buffer, creating an overall area of 100.30 km^2 and achieving approximately 25% coverage. In addition, a series of 2 km-spaced transects were flown over the entire 4 km and 10k m buffers, achieving approximately 15% and 12.5% coverage, for the 4 km and 10 km buffers respectively (Figure 11-3).



SCEIRDE ROCKS WINDFARM
Feirm Ghaoithe Charraigeacha Sceirde

LEGEND

- EIAR Site Boundary
- Offshore Array Area (OAA)
- 1km-spaced flown transects
- 2km-spaced flown transects
- OAA plus 2km buffer
- Offshore Ornithology Study Area (4km OAA buffer)
- Extent of Digital Aerial Survey

0 5 10 km
Kilometres

PROJECT TITLE

Sceirde Rocks

MAP TITLE

Extent of Offshore Digital Aerial Survey

VER	REMARKS	DATE	Drawn	Approved
V1		2024-12-11	CF	ÓM

DRAWING NO **Figure 11-3**

SCALE	PAPER SIZE	DATUM	PROJECTION
1:260,000	A4	WGS 84	UTM 29N

Surveys were flown using an aircraft equipped with four HiDef Gen II digital video cameras with sensors set to a resolution of 2cm Ground Sample Distance. Each camera sampled a strip of 125m width, separated from the next camera by approximately 25m, providing a combined sampled width of 500m within a 575m overall strip. Typically for such surveys, data from two of the four cameras is processed, with the remaining unprocessed data from the other two cameras archived. This unprocessed data can be utilised in situations where seabird densities are very low, or if there is a problem with data from the two processed cameras.

For offshore wind developments a minimum of 10% coverage of the development site and buffer area is typically targeted. For the Sceirde Rocks project, 14.5% site coverage was targeted, with data from two out of the four cameras being processed. This ensured a survey with sufficient coverage and number of transects for precise abundance estimation, with the remaining unprocessed data archived. Monthly survey coverage over the two years of baseline surveys is presented in the Aerial Survey Two Year Report.

The aerial surveys were flown along the transect pattern shown in Figure 11-3 at a height of approximately 500 – 550m (approximately 1,650 – 1,800ft) above sea level (ASL). Flying at this height ensures that there is no risk of flushing species that are easily disturbed by aircraft noise. Thaxter *et al.* (2016) recommended a minimum flight altitude of 460 – 500m ASL for marine bird surveys in order to avoid disturbance to sensitive species such as common scoter and red-throated diver.

Monthly surveys were carried out under the following weather conditions:

- Cloud base above survey altitude (500-550 m);
- No rain;
- Wind speed of less than 30 mph at sea level;
- Sea state of less than 6;
- Surveys commence 1.5 hours after sunrise and finish 1.5 hours before sunset to avoid glare issues.

Digital aerial surveys are one of the recommended survey methods in the DCCAE Guidance, as they can cover a large area over a short period (DCCAE, 2018a&b), and have been demonstrated to be highly effective at detecting birds and marine mammals (Thompson *et al.*, 2012; Williamson, 2016; Mendel et al., 2018). The duration of 24 months of baseline surveys commencing in October and the use of a 4 km buffer around the OAA complies with both the DCCAE guidance (DCCAE, 2018a&b) and current NatureScot guidance (NatureScot, 2023). In addition, the HiDef survey technology and technique has been approved and used by the UK statutory nature conservation bodies (HiDef, 2025). Based on the above, as well as professional judgement and experience gained on other projects, it is considered that the survey methodology used to collect the 24 months of baseline seabird survey data was fit for purpose and meets current industry standards.

Further details of the site-specific ornithology surveys undertaken for the Offshore Site as well as the data analysis undertaken on the survey data are presented in the Baseline Ornithology Report.

As the baseline site characterisation for this Offshore EIA Report has been based on 24 months of recent digital aerial survey data, it is considered to be representative of the OAA and surrounding buffer area for the purpose of impact assessment.

11.5 Assessment Methodology

This assessment considers the potential impacts associated with the construction, operation and maintenance, and decommissioning of the Offshore Site and the potential effects on offshore ornithology. The impact assessment process and methodology follow the principles and approach outlined in Chapter 4: EIA Methodology. The methodology and parameters assessed have also taken into account issues identified through consultation with stakeholders as detailed in Section 11.3 and the understanding of baseline conditions informed by the data sources referenced in Section 11.4.2.

The Project Description (Chapter 5) and the project activities for all stages of the Project life cycle (construction, operation and maintenance, and decommissioning) have been assessed against the ornithology baseline to identify any potential direct and indirect effects and interactions between the Offshore Site and the environment, as well as any potential cumulative effects with other projects. These potential impacts are then assessed to determine a level of significance of effect upon the receiving environment.

11.5.1 Assessment Criteria

The offshore ornithology impact assessment has followed the methodology set out in Chapter 4: EIAR Methodology of the EIAR, with some adaptations to make it applicable to ornithological receptors.

The process for determining the significance of effects is a two-stage process that involves defining the sensitivity of the receptors and the magnitude of the potential impacts. This section describes the criteria applied in this chapter to assign values to the sensitivity of the key bird species and the magnitude of potential impacts.

11.5.1.1 Sensitivity of Receptor Criteria

For offshore ornithology, one of the core components of the assessment of potential impacts and their effects on birds is the sensitivity of a species.

In addition, there is a need to consider the conservation importance of a species when determining the overall sensitivity to any potential impact or effect. This needs to be taken on a species-by-species basis, as a species with a high conservation importance may not be sensitive to a specific effect, while a species with a low conservation importance might be very sensitive to the effect. For example, kittiwake is a species listed as a qualifying feature for some SPAs in Ireland and has a conservation concern listing of 'Red' in Ireland because of recent population declines (Gilbert *et al*, 2021). However, kittiwakes are not considered to be particularly sensitive to human disturbance as there are several examples of the species nesting on buildings or structures such as oil rigs or bridges. Red-throated diver is also a species listed as a qualifying feature for some SPAs in Ireland and is currently 'Amber-listed' in the most recent Birds of Conservation Concern in Ireland (BOCCI) rankings (Gilbert *et al*, 2021). However, red-throated diver is considerably more sensitive to human-related disturbance than kittiwake.

The conservation importance of a species is based on the status of the population from which individuals are predicted to originate from. For this assessment, conservation importance is primarily related to the degree of connectivity of receptor species to SPAs in the region. Criteria for defining the sensitivity and conservation importance in this chapter are outlined in Table 11-3.

Table 11-3 Defining criteria of conservation importance

Importance	Defining Criteria
International	Internationally designated sites within mean maximum foraging range +1 S.D. of the OAA in the breeding season (after Woodward *et al.,* 2019). Regularly occurring species protected under international law (i.e., Annex I species listed as qualifying interests of SPAs within mean maximum foraging range +1 S.D. of the OAA for breeding species, or nearby non-breeding season SPA).
National	Nationally designated sites within mean maximum foraging range +1 S.D. of the OAA. Species protected under national law. Regularly occurring Annex I or Birds Directive Migratory species which are not listed as qualifying interests of SPAs within mean maximum foraging range +1 S.D. of the OAA. BoCCI 'Red' list (Gilbert *et al.,* 2021) species that have nationally important populations within the Offshore Ornithology study area.
Regional	BoCCI 'Red' list (Gilbert *et al.,* 2021) species that have regionally important populations within the Offshore Ornithology study area (i.e., are locally widespread and/or abundant).
Local	The species is common throughout Irish waters but forms a key component of the bird assemblages in the Offshore Ornithology study area.

Previous reviews of post-construction studies of seabirds at OWFs have ranked individual seabird species for their sensitivity to potential impacts such as collision, disturbance and displacement (e.g. Furness and Wade, 2012, Furness *et al.*, 2013, Bradbury *et al.,* 2014, Dierschke *et al.*, 2016). Conclusions from these reviews have been used to inform definitions of sensitivity for seabird species. a summary of conservation importance has also been included (Table 11-4).

Additional consideration has also been given to the current Birds of Conservation Concern in Ireland (BoCCI4) national conservation status for particular species, where appropriate (Gilbert et al., 2021). This is summarised in Table 11-7.

Table 11-4 Sensitivity and conservation importance of seabird species

Receptor sensitivity	Definition
High	Species has low tolerance of sources of disturbance such as noise, light, vessel movements, offshore structures and human activity or high vulnerability to collision impacts. The receptor is of international importance and/or there is clear connectivity to a particular SPA.
Medium	Species has moderate tolerance of sources of disturbance such as noise, light, vessel movements, offshore structures and human activity or moderate vulnerability to collision impacts.

Receptor sensitivity	Definition
	The receptor is of national or international importance and/or individuals at risk are probably drawn from a particular SPA, although other colonies (inc. non-SPAs) may also contribute to the population at risk.
Low	Species has high tolerance of sources of disturbance such as noise, light, vessel movements, offshore structures and human activity or low vulnerability to collision impacts. The receptor is of national importance and/or it is not possible to determine connectivity to any SPAs with any certainty, or no SPAs designated for this species.
Negligible	Species has very high tolerance of sources of disturbance such as noise, light, vessel movements, offshore structures and human activity or very low vulnerability to collision impacts. The receptor is of local importance and/or no SPAs are designated for this species.

11.5.1.2 Magnitude of Impact Criteria

The criteria for defining magnitude levels for key bird species in this chapter are outlined in Table 11-5. This set of criteria has been determined on the basis of changes to bird populations. As a guide, it has been based on summing predicted adult mortality in the breeding season and mortality of all age classes (adults and immature birds) in the non-breeding season and presenting this figure as an overall percentage increase in the baseline mortality in terms of the regional population for each key species. This approach is based on guidance for OWF assessments from NatureScot (NatureScot, 2023). For comparison, mortality has also been calculated based on summing predicted mortality (all ages) in the breeding and non-breeding seasons and presenting this figure as an overall percentage increase in the baseline mortality in terms of the regional population for each key species. This approach is based on guidance for OWF assessments from Natural England (Parker *et al.*, 2022c).

A guide percentage has been included for each of the categories of impact magnitude in

Table 11-5. These guide percentages were agreed with the Irish East Coast Phase 1 developers as an approach for assessing impact magnitude in order to improve the consistency in approach between the Phase 1 project EIARs. Timescales are based on definitions provided in EPA guidance (EPA, 2022).

Table 11-5 Magnitude of the Impact

Magnitude	Definition
High	A change in the size or extent of distribution of the relevant regional population or the population that is the interest feature of a specific protected site that is predicted to irreversibly alter the population in the short-to-long term and to alter the long-term viability of the population and/or the integrity of the protected site. Recovery from that change predicted to be achieved in the long-term or irreversible following cessation of the project activity. Guide: Predicted increase to baseline mortality rate is above 5%.

Magnitude	Definition
Medium	A change in the size or extent of distribution of the relevant regional population or the population that is the interest feature of a specific protected site that occurs in the short and long-term, but which is not predicted to alter the long-term viability of the population and/or the integrity of the protected site. Recovery from that change predicted to be achieved in the medium-term (i.e. in seven to 15 years) following cessation of the project activity. Guide: Predicted increase to baseline mortality rate is between 2% and 5%.
Low	A change in the size or extent of distribution of the relevant regional population or the population that is the interest feature of a specific protected site that is sufficiently small-scale or of short duration to cause no long-term harm to the feature/population. Recovery from that change predicted to be achieved in the short-term (i.e. in one to seven years) following cessation of the project activity. Guide: Predicted increase to baseline mortality rate is between 1% and 2%.
Negligible	Very slight change from the size or extent of distribution of the relevant regional population or the population that is the interest feature of a specific protected site. Recovery from that change predicted to be rapid (i.e. no more than 12 months) following cessation of the project related activity. Guide: Predicted increase to baseline mortality rate is less than 1%.

In the assessments, the predicted magnitude was also sense-checked against relevant Population Viability Analysis PVA outputs (where available) for the species under consideration. As a result, some magnitude ratings have been revised, depending on the PVA predictions. Further details are provided in the assessment sections.

11.5.1.3 Determining Significance of Effects

Assessment of the significance of the potential effects of Sceirde Rocks Offshore Wind Farmon offshore ornithology was determined by correlating the magnitude of the impact and the sensitivity of the receptor in a matrix table (Table 11-6). This matrix table is based on terminology describing the degrees of effect significance presented in the EPA EIAR guidance (EPA, 2022). In addition, in the assessment, the conservation importance of the receptor was also considered using expert judgement to sense-check the matrix outcome.

Table 11-6 Significance of Potential Effects

Magnitude criteria	Definition	Significance
Imperceptible	An effect capable of measurement but without significant consequences.	Not Significant.
Not Significant	An effect which causes noticeable changes in the character of the environment but without significant consequences.	
Slight Effects	An effect which causes noticeable changes in the character of the environment without affecting its sensitivities.	
Moderate Effects	An effect that alters the character of the environment in a manner that is consistent with existing and emerging baseline trends.	Significant; tolerable.

Significant Effects	An effect which, by its character, magnitude, duration or intensity, alters a sensitive aspect of the environment.	Significant; not tolerable. Mitigation measures must be in place to prevent, reduce, or avoid the impact, and if not possible then compensatory measures are proposed.
Very Significant	An effect which, by its character, magnitude, duration or intensity, significantly alters most of a sensitive aspect of the environment.	
Profound Effects	An effect which obliterates sensitive characteristics.	

Effects above moderate significance are therefore considered important in the decision-making process, whilst effects of moderate significance or less warrant little, if any, weight in the decision-making process. However, it should be noted that while impacts of slight significance are not significant in their own right, it is important to distinguish these from other non-significant impacts as they may contribute to significant impacts cumulatively or through interactions.

11.6 Consideration of Data Sources and Quality

The data sources used in this chapter are detailed in Table 11-2 with additional relevant information from the Baseline Ornithology Report. The desktop data used are the most up to date publicly available information obtained from the applicable data sources as cited.

There is a high degree of variability in the marine environment, both spatially and temporally. However, as the baseline site characterisation for the Ornithology Study Area has been based on 24 months of recent digital aerial survey data, it is considered to be representative of the OAA and surrounding buffer area for the purpose of impact assessment.

There was no survey undertaken in February 2022 due to unsuitable weather conditions however, two surveys were flown on 1st and 19th March 2022. Data from the 1st March survey was used as a proxy for the missed February 2022 survey. The use of additional survey data from subsequent months as proxy data for missed months due to weather has been applied in previous EIAR submissions e.g. the Berwick Bank OWF project, Scotland (RPS, 2022) and does not affect the integrity of the dataset used for this assessment.

Overall, it is considered that the digital aerial survey data are representative of the species present in the OAA and surrounding 4 km buffer area throughout the year and that the dataset is both robust and comprehensive and is therefore suitable for the purpose of the impact assessment.

11.7 Baseline Characterisation

11.7.1 Offshore Ornithology Study Area

For the purposes of this Offshore Ornithology EIAR chapter, the Offshore Ornithology Study Area is defined as the OAA and a 4 km buffer around this (Figure 11-2).

A technical report has been prepared to provide a detailed characterisation of the receiving offshore ornithology baseline, hereafter the Baseline Ornithology Report. Data to inform this characterisation of the receiving environment has been collated from a series of site-specific surveys supplemented with a thorough desk-based study of published data. Data was drawn from 24 months of site-specific digital aerial surveys and existing published datasets (Table 11-2).

This section is intended to be a summary of the key findings presented in the Baseline Ornithology Report for the OAA and 4 km buffer and also the wider area out to 10 km. Detail from the Baseline

Ornithology Report has not been repeated within this chapter in order to present a clear and concise impact assessment.

A summary of the baseline environment for offshore ornithology including the Offshore Export Cable (OEC) route is provided in the following sections. Full details of the analysis undertaken on the baseline digital aerial survey data is provided in the Baseline Ornithology Report, which includes information on survey design and methods, as well as the analysis techniques implemented to characterise the baseline.

Between October 2021 and September 2023, 17 seabird species were regularly recorded (more than 10 birds, raw numbers) on digital aerial baseline surveys in the Offshore Ornithology Study Area. A further eight species were recorded occasionally on baseline surveys. Although not recorded in the Offshore Ornithology Study Area on baseline surveys, red-throated diver is also included in Table *11-7*, as there were some sightings of this sensitive species recorded beyond the 4 km buffer. A summary of these species and their conservation status is presented in Table *11-7*. Information for the wider surrounding area has also been included for some species, where relevant. Population estimates presented include apportioning of unidentified birds to species level, as detailed in the Baseline Ornithology Report.

Table 11-7 Summary of Results of Baseline Surveys in Offshore Ornithology Study Area

Species and Conservation Status	Summary of Baseline Results
Red-throated Diver *Gavia stellata* BoCCI[1] Amber listed, Birds Directive Migratory Species, Birds Directive Annex 1	Not recorded in the OAA and 4 km buffer on baseline surveys. Within the wider 10km buffer around the OAA, a total of 91 red-throated divers (raw numbers) were recorded on baseline surveys in Year 1, with four birds recorded in Year 2. All birds were recorded outside the 4 km buffer around the OAA, with the majority of sightings in inshore coastal waters between the coastline and the 4 km buffer around the OAA. Most observations were made between October and April in both years, during the non-breeding season.
Great Northern Diver *Gavia immer* BoCCI Amber-listed, Birds Directive Migratory Species	Within the OAA, birds were recorded between October and May in Year 1, with peak estimated numbers recorded in April (10 birds). In Year 2, birds were recorded between December and May, with peak estimated numbers recorded in December and April (12 birds). Recorded in the OAA and 4 km buffer between October and May in Year 1, with an estimated peak of 52 birds in April 2022. In Year 2, great northern divers were recorded between December and May, with a peak estimate of 54 birds in April 2023.
Fulmar *Fulmarus glacialis* BoCCI Amber listed, Birds Directive Migratory Species	Recorded in the OAA and 4 km buffer in low numbers, primarily in the breeding season. Peak estimated numbers in Year 1 were 57 birds in December 2021 and 22 birds in September 2022. In Year 2, peak estimated numbers were recorded in June 2023 (31 birds) and August 2023 (34 birds).
Manx Shearwater *Puffinus puffinus* BoCCI Amber listed, Birds Directive Migratory Species	Within the OAA, birds were recorded between March and July in Year 1, with peak estimated numbers recorded in May (485 birds). In Year 2, birds were recorded between April and August, with peak estimated numbers recorded in June (388 birds).

Species and Conservation Status	Summary of Baseline Results
	In the OAA and 4 km buffer, birds were recorded between March and August of Year 1, with peak estimated numbers recorded in May (28,093 birds). In Year 2, birds were recorded between April and September, with peak estimated numbers recorded in May (3,359 birds). Unpublished count of 32,836 pairs of Manx shearwater breeding on Cruagh Island received from NPWS (D. Tierney, pers. comm.).
Cory's Shearwater *Calonectris borealis* BoCCI Amber listed, Birds Directive Migratory Species, Birds Directive Annex 1	Within the OAA, birds were only recorded in August of Year 2, with a peak estimate of 285 birds. In the OAA and 4 km buffer, birds were again only recorded in August of Year 2, with a peak estimate of 1,484 birds.
Great Shearwater *Ardenna gravis*	Within the OAA, birds were only recorded in August of Year 2, with a peak estimate of 13 birds. In the OAA and 4 km buffer, birds were again only recorded in August of Year 2, with a peak estimate of 71 birds.
European Storm Petrel *Hydrobates pelagicus* BoCCI Amber listed, Birds Directive Migratory Species, Birds Directive Annex 1	Within the OAA, storm petrels were only recorded in one month (May 2023), with an estimated number of four birds. Recorded in the OAA and 4 km buffer in May and July of Year 1, with a peak estimated number of 25 birds in July. In Year 2, birds were recorded between May and August, with a peak estimate of 17 birds in August.
Gannet *Morus bassanus* BoCCI Amber listed, Birds Directive Migratory Species	Within the OAA, gannets were recorded between April and August in Year 1, with peak estimated numbers recorded in May (29 birds). In Year 2, birds were recorded in December and between April and September, with peak estimate of 13 birds in September). In the OAA and 4 km buffer in Year 1, gannets were recorded in November and December, and between March and September, with an estimated peak of 46 birds in May. In Year 2, gannets were recorded in most months, with peak estimated numbers recorded in August (133 birds).
Cormorant *Phalacrocorax carbo* BoCCI Amber listed, Birds Directive Migratory Species	Within the OAA, cormorants were only recorded in August and September of Year 1, with a peak estimate of five birds in both months. In Year 2, cormorants were recorded in December, January, June and September, with a peak estimate of 12 birds in June. A similar pattern was recorded in the OAA and 4 km buffer, with peak estimates of five birds in August and September of Year 1 and a peak estimate of 17 birds in December and 20 birds in June of Year 2.

Species and Conservation Status	Summary of Baseline Results
Shag *Gulosus aristotelis* BoCCI Amber listed	Within the OAA, shags were recorded in low numbers between December and August in Year 1, with a peak estimate of 29 birds in March. In Year 2, birds were recorded in low numbers in all months except October, with a peak estimate of 20 birds in March. In the OAA and 4 km buffer in Year 1, shags were recorded in all months except September, with an estimated peak of 79 birds in March. In Year 2, shags were recorded in months except October, with peak estimated numbers recorded in December (66 birds).
Eider *Somateria mollissima* BoCCI Red listed, Birds Directive Migratory Species	Within the OAA, eiders were only recorded in March of Year 1, with a peak estimate of 30 birds. In Year 2, peak estimates of 99 birds in March and 28 birds in April were recorded, with no sightings in other months. The same pattern and peak estimates were also recorded in the OAA and 4 km buffer.
Little Gull *Hydrocoloeus minutus* BoCCI Amber listed, Birds Directive Migratory Species	Within the OAA, little gulls were only recorded in July of Year 2, with a peak estimate of 13 birds. In the OAA and 4 km buffer, little gulls were again only recorded in July of Year 2, with a peak estimate of 21 birds.
Black-headed Gull *Chroicocephalus ridibundus* BoCCI Amber listed, Birds Directive Migratory Species	Black-headed gulls were not recorded within the OAA on baseline aerial surveys. In the OAA and 4 km buffer, black-headed gulls were only recorded in July of Year 1, with peak estimate of 10 birds. In Year 2, birds were only recorded in August, with a peak estimate of 12 birds.
Common Gull *Larus canus* BoCCI Amber listed, Birds Directive Migratory Species	Common gulls were not recorded within the OAA in Year 1. In Year 2, common gulls were recorded in December, March and May, with a peak estimate of nine birds in March. In the OAA and 4 km buffer in Year 1, common gulls were recorded in low numbers in October, November, March, July and September, with a peak estimate of 41 birds in September. In Year 2 the pattern was similar, with common gulls recorded in low numbers in October, December, January, and March to July, with a peak estimate of 25 birds in April.
Lesser black-backed Gull *Larus fuscus* BoCCI Amber listed, Birds Directive Migratory Species	Within the OAA, lesser black-backed gulls were recorded in low numbers in March and May to July in Year 1, with a peak estimate of 17 birds in July. In Year 2, birds were recorded in low numbers in April and July, with a peak estimate of 12 birds in April. In the OAA and 4 km buffer, lesser black-backed gulls were recorded in low numbers between March and September in Year 1, with a peak

Species and Conservation Status	Summary of Baseline Results
	estimate of 180 birds in July. In Year 2, birds were recorded in low numbers between April and July, with a peak estimate of 62 birds in April.
Herring Gull *Larus argentatus* BoCCI Amber listed, Birds Directive Migratory Species	Within the OAA, herring gulls were recorded in low numbers in January, May, August and September in Year 1, with a peak estimate of nine birds in September. In Year 2, birds were recorded in low numbers in October, November, January, April and May, with a peak estimate of 33 birds in April. In the OAA and 4 km buffer, herring gulls were recorded in mostly low numbers in all months except December, March, April and June in Year 1, with a peak estimate of 525 birds in July. In Year 2, birds were recorded in low numbers in all months, with a peak estimate of 81 birds in April.
Great black-backed Gull *Larus marinus* BoCCI Green listed, Birds Directive Migratory Species	Within the OAA, great black-backed gulls were recorded in low numbers predominantly in the non-breeding season in Year 1, with a peak estimate of 12 birds in October. In Year 2, birds were recorded in low numbers in all months except October, July and September, with a peak estimate of five birds in January, February, March, June and August. In the OAA and 4 km buffer, great black-backed gulls were recorded in mostly low numbers in all months except March and June in Year 1, with a peak estimate of 134 birds in July. In Year 2, birds were recorded in low numbers in all months except July, with a peak estimate of 194 birds in May.
Kittiwake *Rissa tridactyla* BoCCI Red listed, Birds Directive Migratory Species	Within the OAA, kittiwakes were recorded in low numbers in most months, apart from January, August and September in Year 1, with a peak estimate of 76 birds in March. In Year 2, birds were recorded in low numbers in all months except October, with a peak estimate of 52 birds in June. Kittiwakes were recorded in the OAA and 4 km buffer in all months except September of Year 1, with peak estimates of 167 birds in March and 266 birds in July. In Year 2, kittiwakes were recorded in all months, with peak estimates in February (352 birds) and July (182 birds).
Common Tern *Sterna hirundo* BoCCI Amber listed, Birds Directive Migratory Species, Birds Directive Annex 1	Within the OAA, common terns were only recorded in June and August of Year 1, with a peak estimate of eight birds in August. In Year 2, birds were only recorded in July, with a peak estimate of 22 birds. Common terns were recorded in the OAA and 4 km buffer between May and August of Year 1, with a peak estimate of 82 birds in July. In Year 2, common terns were only recorded in July, with a peak estimate of 194 birds. A survey of breeding terns in South Connemara, Co. Galway in 2023 recorded 120 breeding pairs of Common terns (Colhoun *et al.,* 2023).
Arctic Tern *Sterna paradisaea*	Within the OAA, Arctic terns were only recorded in June of Year 1, with a peak estimate of 11 birds. In Year 2, birds were only recorded in June and July, with a peak estimate of 12 birds in both months.

Species and Conservation Status	Summary of Baseline Results
BoCCI Amber listed, Birds Directive Migratory Species, Birds Directive Annex 1	Arctic terns were recorded in the OAA and 4 km buffer between May and July of Year 1, with a peak estimate of 94 birds in July. In Year 2, Arctic terns were also recorded between May and July, with a peak estimate of 95 birds. A survey of breeding terns in South Connemara, Co. Galway in 2023 recorded 165 breeding pairs of Arctic terns (Colhoun *et al.,* 2023).
Little Tern *Sternula albifrons* BoCCI Amber listed, Birds Directive Migratory Species, Birds Directive Annex 1	Little terns were not recorded within the OAA on baseline aerial surveys. In the OAA and 4 km buffer, little terns were only recorded in July of Year 2, with a peak estimate of eight birds. A survey of breeding terns in South Connemara, Co. Galway in 2023 recorded 82 breeding pairs (Colhoun *et al.,* 2023).
Sandwich Tern *Thalasseus sandvicensis* BoCCI Amber listed, Birds Directive Migratory Species, Birds Directive Annex 1	Sandwich terns were not recorded within the OAA on baseline aerial surveys. In the OAA and 4 km buffer, Sandwich terns were only recorded in May and July of Year 1, with a peak estimate of 65 birds in July. In Year 2, birds were recorded in April and June, with a peak estimate of nine birds in April. A survey of breeding terns in South Connemara, Co. Galway in 2023 recorded 150 breeding pairs of Sandwich terns (Colhoun *et al.,* 2023).
Guillemot *Uria aalge* BoCCI Amber listed, Birds Directive Migratory Species	Within the OAA, guillemots were recorded in all months in Year 1, with a peak estimate of 246 birds in April. In Year 2, birds were again recorded in all months, with a peak estimate of 508 birds in May. In the OAA and 4 km buffer guillemots were recorded in all months, with higher numbers recorded in the breeding season. The peak estimated number was 5,314 birds in July. In Year 2, guillemots were also recorded in all months, with a peak estimate of 7,114 birds in May.
Razorbill *Alca torda* BoCCI Red listed, Birds Directive Migratory Species	Within the OAA, razorbills were recorded in low numbers between November and July in Year 1, with a peak estimate of 28 birds in March. In Year 2, birds were recorded in low numbers in most months, with a peak estimate of 55 birds in November. In the OAA and 4 km buffer razorbills were recorded in all months except October, April, June and September, with peak estimates of 640 birds in November and 707 birds in July. In Year 2, razorbills were recorded in all months except October and March, with peak estimates of 308 birds in November and 268 birds in May.
Black Guillemot *Cepphus grille*	Within the OAA, black guillemots were only recorded in November and January of Year 1, with a peak estimate of five birds in both months. No black guillemots were recorded in the OAA in Year 2.

Species and Conservation Status	Summary of Baseline Results
BoCCI Amber listed, Birds Directive Migratory Species	In the OAA and 4 km buffer, black guillemots were only recorded in November, January and March of Year 1, with a peak estimate of nine birds in March. In Year 2, black guillemots were only recorded in February and March, with a peak estimate of 17 birds in February.
Puffin *Fratercula arctica* BoCCI Red listed, Birds Directive Migratory Species	Within the OAA, puffins were only recorded in April, May and September of Year 1, with a peak estimate of 37 birds in May. In Year 2, puffins were only recorded in May and June, with a peak estimate of 24 birds in June. In the OAA and 4 km buffer puffins were only recorded in October, April, May and September, with a peak estimate of 132 birds in May. In Year 2, puffins were only recorded between April and July, with a peak estimate of 453 birds in July.

Full details of the numbers of all species and species groups recorded on baseline aerial surveys are presented and discussed in more detail within the Baseline Ornithology Report.

11.7.2 Designated Sites

Key designated SPAs identified for offshore ornithology are described in Table 11-8. Typically, these are the closest designated SPAs to the OAA that support important populations of breeding seabirds, or foraging areas for non-seabirds in the non-breeding season. A full list of designated SPAs for seabirds considered for ornithological connectivity with the OAA are detailed in the Offshore Ornithology Apportioning Report.

Table 11-8 Key designated SPAs and relevant species of qualifying interest for offshore ornithology

Designated site	Relevant species of qualifying interest
Slyne Head to Ardmore Point Islands SPA Site Code: 4159 Distance from OAA = 6.7 km	Breeding species: Sandwich tern (*Sterna sandvicensis*), Arctic tern, Little tern (*Sternula albifrons*) Wintering species: Barnacle goose (*Branta leucopsis*)
Inishmore SPA Site Code: 4152 Distance from OAA = 16 km	Breeding species: Kittiwake, Arctic tern, Little tern, Guillemot
Cruagh Island SPA Site Code: 4170 Distance from OAA = 38.6 km	Breeding species: Manx shearwater Wintering species: Barnacle goose
Cliffs of Moher SPA Site Code: 4005	Breeding species: Fulmar, Kittiwake, Guillemot, Razorbill, Puffin, Chough (*Pyrrhocorax pyrrhocorax*)

Designated site	Relevant species of qualifying interest
Distance from OAA = 42.2 km	
Inner Galway Bay SPA Site Code: 4031 Distance from OAA = 56.5 km	Breeding species: Cormorant (*Phalacrocorax carbo*), Sandwich tern, Common tern Wintering species: Black-throated diver (*Gavia arctica*), Great northern diver, Light-bellied brent goose (*Branta bernicla hrota*), Cormorant, Grey heron (*Ardea cinerea*), Wigeon (*Anas penelope*), Teal (*Anas crecca*), Red-breasted merganser (*Mergus serrator*), Ringed plover (*Charadrius hiaticula*), Golden plover (*Pluvialis apricaria*), Lapwing (*Vanellus vanellus*), Dunlin (*Calidris alpina*), Bar-tailed godwit (*Limosa lapponica*), Curlew (*Numenius arquata*), Redshank (*Tringa totanus*), Turnstone (*Arenaria interpres*), Black-headed gull (*Chroicocephalus ridibundus*), Common gull,
High Island, Inishshark and Davillaun SPA Site Code: 4144 Distance from OAA = 51.1 km	Breeding species: Fulmar, Arctic tern Wintering species: Barnacle goose
Mid-Clare Coast SPA Site Code: 4182 Distance from OAA = 60.6 km	Breeding species: Cormorant Wintering species: Barnacle goose, Ringed plover, Sanderling (*Calidris alba*), Purple sandpiper (*Calidris maritima*), Dunlin, Turnstone

11.7.3 Defining the sensitivity of the Baseline

Impacts have been assessed in relation to relevant biological seasons, as defined by Furness (2015), and a summary of these seasons for seabird species is presented in Table 11-9. Where months overlapped between seasons, the breeding season definition was considered to take precedence over non-breeding season definitions. This approach avoids duplicating single monthly estimates which could artificially inflate seasonal abundance estimates and has been used previously in displacement assessments for offshore wind farms in Scotland (e.g. NnGOWL, 2018), therefore is considered appropriate to use here.

Table 11-9 Definitions of breeding and non-breeding seasons for seabirds (based on Furness, 2015 unless otherwise stated)

Species	Season definitions
Red-throated Diver	Breeding Season - March to August Migration Seasons – September to November and February to April Winter Period – December and January
Great Northern Diver	Breeding Season – Not Applicable Non-breeding Season – September to May
Fulmar	Breeding Season - January to August Migration Seasons – September to October and December Winter Period - November

Species	Season definitions
Manx Shearwater	Breeding Season - April to August Migration Seasons – September to early October and March
Storm Petrel[1]	Breeding Season – Mid-May to October Non-breeding Season – November to mid-May (Not present in significant numbers)
Gannet	Breeding Season - March to September Autumn Migration Period – October to November Spring Migration Period – December to February
Cormorant	Breeding Season - April to August Non-breeding Season – September to March
Shag	Breeding Season - February to August Non-breeding Season – September to January
Eider	Breeding Season – Mid-April to August Non-breeding Season – September to mid-April
Common Gull[1]	Breeding Season - April to August Non-breeding Season – September to March
Lesser black-backed Gull	Breeding Season - April to August Autumn Migration Period – September to October Winter Period – November to February Spring Migration Period – March
Herring Gull	Breeding Season - March to August Non-breeding Season – September to February
Great black-backed Gull	Breeding Season - Late March to August Non-breeding Season – September to March
Kittiwake	Breeding Season - March to August Autumn Migration Period – September to December Spring Migration Period – January to February
Common Tern	Breeding Season - May to August Migration Seasons – Late July to early September and April to May
Arctic Tern	Breeding Season - May to early August Migration Seasons – July to early September and late April to May
Guillemot	Breeding Season - March to July Non-breeding Season – August to February
Razorbill	Breeding Season - April to July Migration Seasons – August to October and January to March Winter Period – November and December

Species	Season definitions
Black Guillemot	Breeding Season - April to August Non-breeding Season – September to March
Puffin	Breeding Season - April to early August Non-breeding Season – mid-August to March

1 NatureScot, 2023

For the breeding season, the regional reference population for seabird species was calculated by summing the most recent counts for breeding colonies (e.g. Burnell *et al.,* (2023) within mean-maximum foraging range (+1 S.D.), as defined in Woodward *et al.* (2019), unless otherwise stated (Table 11-10).

Table 11-10 Mean maximum foraging distance + 1 S.D. for seabirds (Woodward et al., 2019)

Species	Mean maximum foraging range + 1 S.D.
Red-throated Diver	9 km
Great Northern Diver	N/A as species does not breed in Ireland
Fulmar[1]	542.3 ± 657.9 km
Manx Shearwater[1]	1,346.8 ± 1,018.7 km[2]
Storm Petrel	336 km
Gannet	315.2 ± 194.2 km
Cormorant	25.6 ±8 .3 km
Shag	13.2 ± 10.5 km
Eider	21.5 km
Common Gull	50 km
Lesser black-backed Gull	127 ± 109 km
Herring Gull	58.8 ± 26.8 km
Great black-backed Gull	73 km
Kittiwake	156.1 ± 144.5 km
Common Tern	18.0 ± 8.9 km
Arctic Tern	25.7 ± 14.8 km
Guillemot	73.2 ± 80.5 km
Razorbill	88.7 ± 75.9 km
Black Guillemot[3]	0.5-7.0 km

Species	Mean maximum foraging range + 1 S.D.
Puffin	137.1 ± 128.3 km

1 For the EIAR assessment breeding season reference population, only colonies within 509.4 km were included.
2 For comparison, the mean foraging range for Manx shearwater is 136.1±88.7 km (Woodward et al., 2019)
3 Based on Birdlife International, (2023)

Regional reference populations

For the purpose of this assessment, impacts are assessed against relevant regional populations. The reference population for assessment is the Biologically Defined Minimum Population Size (BDMPS).

Guidance from Natural England defines the BDMPS for the breeding season as the breeding population within foraging range from the Offshore Site, plus non-breeding and immature birds (Parker *et al.,* 2022c). This is because it is considered that the population in the region is likely to originate from a much wider range of colonies (not just SPA colonies) and may include young immature birds spending the summer in their wintering area as well as immature birds loosely associated with local colonies (Furness, 2015).

However, based on recent EIARs submitted to Marine Scotland (e.g. West of Orkney (Xodus, 2023) and Berwick Bank (RPS, 2022)), for the breeding season, the regional reference population has only included adults from breeding colonies within mean maximum foraging range (plus 1S.D.), with predicted adult mortality from the impact being assessed being compared to this reference population. In this assessment, both approaches have been presented in tables for the relevant impact assessments, to allow a comparison to be made.

For this assessment, for species with SPAs within mean maximum foraging range, colony counts were taken from the Offshore Ornithology Apportioning Report. For species with no SPAs within mean maximum foraging range (cormorant and black guillemot), most recent population counts have been taken from Burnell *et al.,* (2023).

In order to estimate the number of non-breeding or immature birds in the breeding season BDMPS reference population, the number of breeding adults within mean max plus 1 S.D. foraging range was multiplied by the ratio of immature to adult birds, based on Horswill and Robinson, (2015). This figure was then added to the estimated number of breeding adults to calculate the regional reference population (Table 11-11).

Table 11-11 Breeding season regional reference populations for key seabird species

Species	Breeding Season Regional Reference Population (breeding adults)[1]	Immature to adult ratio (number of immatures per adult)	Breeding Season Regional Reference Population (adult and immature birds)[2]
Red-throated Diver	Species does not breed within mean maximum foraging range		
Great Northern Diver	Species does not breed in Ireland or UK		
Fulmar[3]	68,306 adults	1.083	142,281 birds
Manx Shearwater	363,150 adults[3]	1.132	774,236 birds
Storm Petrel	216,846 adults	-	-
Gannet	93,602 adults	0.761	164,833 birds

Species	Breeding Season Regional Reference Population (breeding adults)[1]	Immature to adult ratio (number of immatures per adult)	Breeding Season Regional Reference Population (adult and immature birds)[2]
Cormorant	840 adults	1.451	2,059 birds
Shag	76 adults	0.792	136 birds
Eider	Zero	-	-
Common Gull	532 adults	0.452	772 birds
Lesser black-backed Gull	5,068 adults	0.876	9,508 birds
Herring Gull	3,186 adults	1.370	7,551 birds
Great black-backed Gull	1,410 adults	1.538	3,579 birds
Kittiwake	26,720 adults	0.898	50,715 birds
Common Tern	256 adults	0.701	435 birds
Arctic Tern	504 adults	0.511	762 birds
Guillemot	74,578 adults	0.916	142,891 birds
Razorbill	9,417 adults	0.876	17,666 birds
Black Guillemot	338 adults	1.681	906 birds
Puffin	26,264 adults	0.842	48,378 birds

1 Regional breeding populations within mean maximum foraging range + 1S.D. only, after Woodward et al., (2019)
2 Regional non-breeding populations were derived as outlined in Offshore Ornithology Apportioning Report
3 Includes recent count of 32,836 pairs on Cruagh Island (D.Tierney, pers comm.*)*

For the non-breeding season, the BDMPS approach devised by Furness, (2015) was used as a basis to estimate suitable regional reference populations for use in the EIAR. However, the BDMPS regions defined by Furness (2015) did not include estimates for the west coast of Ireland, therefore revisions to the Furness (2015) approach were required to take account of this.

For each species, BDMPS (Furness, 2015) regional populations incorporated a proportion of the estimated Irish breeding population. This approach has been discussed and aligned with the approach used by the Irish East Coast Phase One OWFs, in order to maintain a consistency of approach between the Phase 1 projects. This component was removed from the BDMPS population estimate and replaced with the most recent breeding population as estimated in Burnell *et al.,* (2023), for north coast and west coast counties between County Donegal and Mizen Head in County Cork. These population estimates were corrected to include non-adult birds using age group proportions from Horswill and Robinson (2015). This figure was then added to the estimated number of breeding adults to estimate the regional reference population of adults and immatures in the breeding season. Further details and refinements for individual species are presented in the Offshore Ornithology Apportioning Report. Regional reference populations for the non-breeding season are shown in Table 11-12.

Table 11-12 Non-breeding season regional reference populations for seabird species

Species	Regional Reference BDMPS Population		
	Autumn migration period	Winter period	Spring migration period
Great Northern Diver	1,219 birds in non-breeding season[1]		
Fulmar	946,463 birds	674,636 birds	946,463 birds
Manx Shearwater	2,139,846 birds	-	2,139,846 birds
Storm Petrel	Not present in Irish waters in significant numbers in the non-breeding season		
Gannet	662,102 birds	-	770,836 birds
Cormorant	17,343 birds in non-breeding season		
Shag	50,135 birds in non-breeding season		
Common Gull	8,914 birds in non-breeding season[1]		
Lesser black-backed Gull	174,257 birds	54,408 birds	174,257 birds
Herring Gull	190,702 birds in non-breeding season		
Great black-backed Gull	42,708 birds in non-breeding season		
Kittiwake	945,743 birds	-	725,683 birds
Common Tern	64,189 birds	-	64,189 birds
Arctic Tern	74,008 birds	-	74,008 birds
Guillemot	1,287,037 birds in non-breeding season		
Razorbill	631,203 birds	365,711 birds	631,203 birds
Puffin	344,797 birds in non-breeding season		

1 Estimated from HiDef population estimates for wider 10 km survey area (HiDef, 2024) plus mean I-WeBS counts from Lewis, et al., 2019

The impact of additional mortality on seabirds due to effects such as displacement or collision, has been assessed in terms of the change in the baseline mortality rate which could result. Species-specific baseline mortality rates were based on age-specific demographic and survival rates and age class proportions from Horswill and Robinson (2015).

For the breeding season assessment based on adult birds only, the increase in baseline mortality was calculated using the estimated adult baseline survival rate from Horswill and Robinson (2015). For example, for gannet the estimated adult baseline survival rate is 0.919, therefore the corresponding rate for adult mortality is 0.081 (Table 11-13).

Table 11-13 Adult survival and mortality rates, age ration of immature to adult birds and average mortality rates used in this assessment (after Horswill and Robinson, 2015)

Species	Adult Survival	Adult Mortality	Percentage age ratio of adults (%)	Average mortality for all age classes
Great Northern Diver	0.870	0.13	51.4	0.161
Manx Shearwater	0.870	0.13	46.9	0.130
Gannet	0.919	0.081	56.8	0.181
Cormorant	0.868	0.132	40.8	0.297
Shag	0.858	0.142	55.8	0.262
Common Gull	0.828	0.172	68.9	0.253
Lesser black-backed Gull	0.885	0.115	53.3	0.123
Herring Gull	0.834	0.166	42.2	0.172
Great black-backed Gull	0.930	0.07	39.4	0.095
Kittiwake	0.854	0.146	52.7	0.156
Common Tern	0.883	0.117	58.8	0.191
Arctic Tern	0.837	0.163	66.2	0.183
Guillemot	0.939	0.061	52.2	0.136
Razorbill	0.895	0.105	53.3	0.129
Black Guillemot	0.870	0.13	37.3	0.158
Puffin	0.906	0.094	54.3	0.177

For the breeding season assessment based on adult and immature birds, and for the non-breeding season assessments, it has been assumed that all age classes are equally at risk of effects, with each age class affected in proportion to its presence in the population. Therefore, a weighted average baseline mortality rate has been calculated which is appropriate for all age classes for use in assessments, calculated for those species screened in for assessment. These were calculated using the different survival rates for each age class and their relative proportions in the population from Horswill and Robinson (2015). Baseline mortality rates used in this assessment are summarised in Table 11-13. Further details are presented in the Baseline Ornithology Report.

11.7.4 Offshore Export Cable (OEC) route

Given the limited scale of works required for the OEC corridor (i.e. a relatively small number of vessel movements over a relatively small area for a short period of time), no specific surveys were commissioned for the OEC route area between the Offshore Ornithology Study Area and the Landfall. Instead the baseline characteristics of the OEC were drawn from published information on seabird abundance and distribution in the wider area (Table 11-2).

It was considered that this level of detail would be sufficient to inform the assessment work for identifying potential impacts on offshore ornithology receptors within the OEC, given the limited potential for disturbance to seabirds arising from the cable installation activities. Overall, it is considered likely that the seabird species that were regularly recorded in the OAA during baseline surveys are also likely to occur regularly within the OEC.

The route of the OEC does not pass through any designated conservation sites for birds (see Figure 11-2).

11.7.5 Baseline characteristics of the OEC route

The baseline description of the OEC route was based on published reports of European Seabirds At Sea (ESAS) surveys in Irish waters between 1980 and 1997 (Pollock *et al.,*1997). Additional information was also used from reports of visual aerial surveys conducted by University College Cork (UCC) in 2016 and between 2021 and 2022 (Rogan *et al.,* 2018, Giralt Paradell *et al.,* 2024).

Great Northern Divers were regularly recorded around the west coast of Ireland in the Winter Atlas, with birds recorded in every 10 km survey square. Red-throated divers were also regularly recorded in the Galway Bay area, but were slightly less widespread. The distribution of black-throated diver was more restricted to bays off the west coast of Ireland in comparison (Balmer *et al.,* 2013). Diver species were not recorded in the vicinity of the OEC on seabird surveys undertaken off the west coast of Ireland (Pollock *et al.,* 1997). Similarly, visual aerial surveys off the west coast of Ireland in winter 2016 or winter 2022 (Rogan *et al.,* 2018, Giralt Paradell *et al.,* 2024). However, inshore coastal waters were not the primary survey target area for these surveys, which explains the lack of sightings.

Highest densities of fulmars and Manx shearwaters were recorded in the summer months off Galway Bay, with lower densities recorded outside the breeding season (Pollock *et al.,* 1997. A similar distribution pattern was recorded on the ObSERVE II surveys (Giralt Paradell *et al.,* 2024).

Highest densities of gannets were recorded in inshore waters of Galway Bay in September and October on ESAS surveys, with lower densities recorded in the winter months (Pollock *et al.,* 1997). Densities of gannets recorded in the wider west coast region on ObSERVE II surveys were generally low (Giralt Paradell *et al.,* 2024).

Highest densities of shags and cormorants were recorded in inshore, coastal waters of Galway and in inner Galway Bay respectively throughout the year on ESAS surveys, with lower densities recorded further offshore (Pollock *et al.,* 1997). Shags and cormorants were also recorded in highest densities in coastal waters of the wider west coast region covered on ObSERVE II surveys, although there were too few sightings in winter 2022 to map the distribution (Giralt Paradell *et al.,* 2024).

Lesser black-backed gulls were widespread in low densities throughout the outer Galway Bay area on ESAS surveys in summer months, with no birds recorded in winter months (Pollock *et al.,* 1997). Herring gulls and great black-backed gulls were typically more abundant offshore outside the breeding season in outer Galway Bay on ESAS surveys (Pollock *et al.,* 1997).

Highest densities of kittiwakes were recorded in inshore waters of Galway Bay in August and September on ESAS surveys, with lower densities recorded in the winter months (Pollock *et al.,* 1997). Densities of kittiwakes recorded in the wider west coast region on ObSERVE II surveys were generally low, but also showed higher concentrations in inshore areas (Giralt Paradell *et al.,* 2024).

Low densities of terns were recorded in inshore waters of Galway Bay in the summer months on ESAS surveys, no birds recorded in the winter months, reflecting their status as summer migrants to Irish waters (Pollock *et al.,* 1997).

Highest densities of guillemots and razorbills were recorded in inshore waters of Galway Bay between July and September on ESAS surveys, while densities of puffins peaked in June and July. Lower densities of auks were recorded in other months (Pollock *et al.,* 1997). A similar pattern was recorded in the wider west coast region on ObSERVE II surveys (Giralt Paradell *et al.,* 2024).

There will be no disturbance to the intertidal area because this area will be avoided through the use of a trenchless, drilled landfall method, i.e. horizontal directional drilling or "direct pipe" installation. This will involve a construction (drilling) compound in agricultural land well above Mean High Water Springs (MHWS) and a drilled conduit to a location ca. 1 km offshore, entirely avoiding the intertidal area. For this reason, bird surveys of the intertidal area were not undertaken during the baseline data collection phase.

11.8 Likely Significant Effects and Associated Mitigation Measures

11.8.1 Scope of the assessment

The following impacts on Offshore Ornithology will be assessed based on known sensitivities of birds to likely activities associated with construction, operation and maintenance, and decommissioning of OWFs. Each of these is a summary of potential impacts as assessed and identified for assessment. This has been considered and assessed in further detail in the following sections.

Construction/Decommissioning Phases

- Impact 1 - Disturbance and displacement on key bird species as a result of increased vessel activity and other construction/decommissioning activity within the OAA;
- Impact 2 - Disturbance and displacement on key bird species as a result of increased vessel activity and other construction/decommissioning activity along the OEC route;
- Impact 3 - Indirect effects on foraging seabirds as a result of habitat loss/displacement of prey species due to increased noise and disturbance to seabed during construction/decommissioning.

Operation and Maintenance Phase

- Impact 4 - Disturbance and displacement on key bird species as a result of increased vessel activity and other maintenance activities within the OAA;
- Impact 5 - Indirect effects as a result of habitat loss/displacement of prey species due to presence of turbines, increased noise and disturbance to seabed;
- Impact 6 - Displacement and barrier effects on key bird species within the OAA and appropriate buffer from offshore infrastructure;
- Impact 7 - Mortality of key bird species as a result of collision with offshore wind turbines;
- Impact 8 - Disturbance from aviation and navigation lighting.

11.8.2 Design Parameters

This section outlines the key project design parameters used for the assessment of potential effects on offshore ornithology during construction (including pre-construction), operation and maintenance, and decommissioning (Table 11-14). The full Offshore Site design is detailed in Chapter 5: Project Description.

Table 11-14 Project design parameters relevant to offshore ornithology

Potential effect	Design Scenario	Requirement
Construction/decommissioning		
Disturbance and displacement on key bird species as a result of increased vessel activity and other construction/ decommissioning activity within the OAA	A total of four years of construction (including pre-construction activities), including the following activities: › Pre-construction activities over four months › Geophysical, geotechnical, and Unexploded Ordnance (UXO) surveys expected to take four months; › Seabed preparation including boulder clearance, ground preparation (e.g. stonebed placement, dredging and controlled flow excavation), and pre-lay grapnel runs. › Construction activities over 18 months › Installation of 31 no. Gravity Base (GBS) foundations expected to take four months; › Construction of 30 no. Wind Turbine Generators (WTGs), expected to take three months; › Construction of 1 no. OSP expected to take 11 months; › Cable installation via surface lay (protection via cast-iron shell or rock/concrete mattress placement) or buried (jet-trenched) of the following: › A network of up to 73 km of inter-array cables within the OAA, expected to take 16 months; › A single Offshore Export Cable (OEC) of maximum total length of 63.5 km expected to take 15 months. 21 installation vessels are expected to operate at the site: › 1 no. vessel for seabed preparation; › 2 no. vessels for OSP Topside installation; › 4 no. vessels for inter-array cable (IAC) installation; › 5 no. vessels for EC installation; › 4 no. vessels for GBS foundation installation; › 3 no. vessels for WTG installation; and › 2 no. vessels for construction ad major maintenance operations. A total of 23 construction support vessels, with a maximum of 11 present within the Offshore Site at any one time.	Duration of construction activities Number of installation vessels present.

Disturbance and displacement on key bird species as a result of increased vessel activity and other construction activity along the OEC route	› Installation of a single OEC of maximum total length of 63.5 km expected to take 15 months › 5 no. vessels for EC installation;	Duration of construction activities Number of installation vessels present.
Indirect effects on foraging seabirds as a result of habitat loss/displacement of prey species due to increased noise and disturbance to seabed during construction	As above	Duration of construction activities Number of installation vessels present.
Operation and maintenance		
Disturbance and displacement on key bird species as a result of increased vessel activity and other maintenance activities within the OAA	Estimated number of maintenance vessels expected for routine inspections, repairs and replacement: › Two CTVs per day with up to four daily return vessel movements; › One SOV per day; › Two annual jack up intervention campaigns (may cover more than two locations); › One repair platform per year; › One drone campaign per year; › Five unscheduled cable repair vessels over the lifetime; › Cable survey vessels required annually for the first 5 years, and one every 5 years thereafter; and › Oil exchange vessels required once every 10 years. › Operational life of 38 years.	Duration and nature of maintenance activities
Indirect effects as a result of long-term habitat loss/displacement of prey species due to presence of turbine foundations	› Operation of 30 no. WTGs and 1 no. OSS; › 31 no. GBS foundations (30 no. for WTGs and 1 no. for OSS); › Minimum spacing of 1,017 m; and › Operational life of 38 years.	Duration and nature of operation. Physical presence of structures (WTG and OSP)
Displacement and barrier effects on key bird species from offshore infrastructure	› Operation of 30 no. WTGs and 1 no. OSS; › Operational life of 38 years. › Combined OAA plus appropriate buffer.	Evidence from existing offshore wind farms indicates that if there is displacement that it will be limited to within 2 km of

		the wind farm boundary for the majority of species. However, for great northern diver, guidance states that a 4 km buffer should be used (SNCBs, 2022), and this has been applied here.
Mortality of key bird species as a result of collision with offshore wind turbines	- Operation of 30 no. WTGs; - Air gap of 32.9m LAT; - Rotor radius = 146m; - Turbine tip height = 324.9m; and - Operational life of 38 years.	Physical presence of structures (WTG and OSS) and underwater sound emissions from WTGs.
Disturbance from aviation and navigation lighting	- Operation of 30 no. WTGs; - Selected peripheral structures will carry marine Aids to Navigation (AtoN) lighting; - Yellow 5s flash; - At least 5nm range; - 360° visibility; - Synchronised; - Located not less than 6m and not more than 30m above Highest Astronomical Tide (HAT); and - Operational life of 38 years.	Potential for nocturnal collisions/ attractions of birds to WTGs.

11.8.3 Mitigation by Design

As described in Chapter 4: EIA methodology, certain measures have been adopted as part of the Offshore Site design to reduce the potential for impacts to the environment. Those relating to offshore ornithology receptors are presented in Table 11-15 below.

Table 11-15 Mitigation measures by design relevant to offshore ornithology

Mitigation measure
Minimum air gap between lower blade tip and sea level was designed to be greater than 30 m LAT in order to minimise collision impacts on flying birds.
Vessels engaged in construction works will typically be travelling at slow (<6 kts) speeds and using consistent routes between ports and the OAA. This will reduce disturbance to offshore ornithology receptors relative to high-speed transiting.
Develop and implement a Project Environmental Management Plan and Monitoring Plan (PEMMP), and Invasive Non-Native Species Management Plan, a Code of Construction Practice (CoCP) and a Marine Pollution Contingency Plan (MPCP). These plans will include a commitment to measures to mitigate against pollution events, biosecurity measures, waste management, measures to avoid the introduction and spread of Invasive Non Native Species, adherence to the BWM Convention and other applicable international regulations, as well as containment procedures.

The use of cable protection will be minimised as far as practicable, and only used where required. Additional external cable protection (e.g. rock placement) will only be used where the minimum target burial depth cannot be achieved, for example in areas of hard ground or at third-party crossings.
Marine pollution prevention under the International Convention for the Prevention of Pollution from Ships (MARPOL) convention requirements will be followed during construction, operation and maintenance and decommissioning.
Development of, and adherence to, a Decommissioning Programme prior to construction and updated throughout the Project lifespan. A Decommissioning Plan has been prepared for the Project (see Chapter 5: Project Description) the details of which will be agreed with the local authority prior to any decommissioning.

11.8.4 Do-Nothing Scenario

Guidance from the EPA (2022), states that the do nothing alternative should be a general description of the evolution of the key environmental factors of the site and environs if the Project did not proceed.

Regarding Offshore Ornithology, if the Project did not proceed, then it is considered likely that breeding seabird populations within foraging range of the project footprint would continue to use the sea area in the breeding season as recorded on baseline surveys. The numbers and species of birds passing through the area in the non-breeding season and on spring or autumn migration would be similar to those recorded on the baseline surveys. These birds would be subject to potential impacts arising from other offshore developments or industries in the vicinity.

However, if the Project did not proceed, then there would be no associated reductions in greenhouse gases and no benefit to reducing the effects of climate change. Current downward pressures on the breeding populations of sensitive seabird species such as kittiwakes, which are considered at risk of the effects of climate change on their prey distribution (RSPB, 2018), would be predicted to continue.

11.8.5 Construction Phase

11.8.5.1 Impact 1: Disturbance and displacement within the OAA during construction

Impact 1 considers disturbance and displacement on key bird species as a result of increased vessel activity and other construction activity within the OAA. Direct temporary disturbance or displacement of birds within the OAA during the construction phase will occur as a result of a range of activities including the use of jack-up vessels during foundation installation/maintenance, installation of inter-array and offshore export cables (including seabed clearance operations prior to cable installation) and anchor placements associated with these activities. Table 11-14 summarises the project design parameters of the Offshore Site considered within this assessment.

Sensitivity of the Receptor

Some seabird species are more susceptible to disturbance than others. There is evidence from studies that demonstrate that species such as divers and scoters may avoid shipping by several kilometres (e.g. Garthe and Hüppop, 2004; Schwemmer et al. 2011), while gulls are not considered susceptible to disturbance, as they are often attracted to fishing boats as a potential food source (e.g. Camphuysen, 1995; Hüppop and Wurm, 2000).

In order to focus this assessment, a screening exercise was undertaken to identify those species likely to be susceptible to disturbance and displacement as a result of increased vessel activity associated with construction (Table 11-16). This was based on previous sensitivity reviews such as Garthe and Hüppop

(2004), who developed a scoring system for such disturbance factors, which is used widely in offshore wind farm EIAs. Similarly, Furness and Wade (2012) developed disturbance ratings for particular species based on Garthe and Hüppop (2004), alongside scores for habitat flexibility and conservation importance in a Scottish context. These were subsequently revised to provide seabird sensitivity scores for species in English territorial waters (Bradbury *et al.,* 2013). In addition, Dierschke *et al.*, (2016) provides a summary of avoidance and attraction evidence from several offshore wind farm projects in Europe.

These sensitivity indices have been used as a basis to inform the likely sensitivity of seabird species recorded within the OAA to disturbance and displacement (Table 11-16). Species with a low sensitivity to disturbance or displacement were screened out of further assessment for this impact. In addition, species that were not recorded or only recorded within the OAA in very small numbers on baseline surveys were also screened out of further assessment for this impact on the basis that there would be no significant effect on any population of these species, due to the very low numbers recorded within the OAA.

Table 11-16 Sensitivity of Species to disturbance and displacement from increased vessel activity in OAA during construction

Species	Sensitivity to Disturbance and Displacement	Screening Result (In/Out)
Red-throated Diver	High	Screened OUT as no red-throated divers were recorded in the OAA and 4 km buffer on baseline surveys and therefore additional disturbance/displacement would be negligible.
Great Northern Diver	High	Screened IN as the species was recorded in in the OAA on baseline surveys and the species is considered to have high sensitivity to disturbance and displacement.
Fulmar	Very low	Screened OUT as the species has a very low sensitivity to disturbance and is not known to avoid vessels.
Manx Shearwater	Very Low	Screened OUT as the species has a very low sensitivity to disturbance and is not known to avoid vessels.
Storm Petrel	Very Low	Screened OUT as the species was recorded in very low numbers in the OAA on baseline surveys and therefore additional disturbance/displacement would be negligible. The species also has a very low sensitivity to disturbance and is not known to avoid vessels.
Gannet	Low	Screened OUT as the species has a low sensitivity to disturbance and displacement from vessels.
Cormorant	Medium	Screened OUT as the species was recorded in very low numbers in the OAA on baseline surveys and therefore additional disturbance/displacement would be negligible.

Species	Sensitivity to Disturbance and Displacement	Screening Result (In/Out)
Shag	Medium	Screened OUT as the species was recorded in very low numbers in the OAA on baseline surveys and therefore additional disturbance/displacement would be negligible.
Eider	Medium	Screened OUT as the species was only occasionally recorded in low numbers in the OAA on baseline surveys and therefore additional disturbance/displacement would be negligible.
Common Gull	Low	Screened OUT as the species has a low sensitivity to disturbance and displacement.
Lesser black-backed Gull	Low	Screened OUT as the species has a low sensitivity to disturbance and displacement.
Herring Gull	Low	Screened OUT as the species has a low sensitivity to disturbance and displacement.
Great black-backed Gull	Low	Screened OUT as the species has a low sensitivity to disturbance and displacement.
Kittiwake	Low	Screened OUT as the species has a low sensitivity to disturbance and displacement.
Common Tern	Low	Screened OUT as the species has a low sensitivity to disturbance and displacement.
Arctic Tern	Low	Screened OUT as the species has a low sensitivity to disturbance and displacement.
Guillemot	Medium	Screened IN due to numbers recorded and classified as medium sensitivity to disturbance and displacement.
Razorbill	Medium	Screened IN due to numbers recorded and classified as medium sensitivity to disturbance and displacement.
Black Guillemot	Medium	Screened OUT as the species was recorded in very low numbers in the OAA on baseline surveys and therefore additional disturbance/displacement would be negligible.
Puffin	Low	Screened OUT as the species has a low sensitivity to disturbance and displacement.

Based on Table 11-16, three species (great northern diver, guillemot and razorbill) were identified as being potentially sensitive to disturbance and displacement from increased vessel activity within the OAA during the construction phase. For each of these species, the magnitude of impact and overall sensitivity to Impact 1 were considered.

For great northern diver, published evidence from reviews indicates that this species has a high sensitivity to disturbance from vessels (e.g. Bradbury *et al.,* 2014). In addition, great northern diver is Amber-listed in the most recent BoCCI review (Gilbert *et al.*, 2021) (Table 11-4). The nearest designated SPA for wintering numbers of this species is Inner Galway Bay SPA, which is approximately 56.5 km from the OAA at its nearest point (Table 11-8). The overall sensitivity of great northern diver to disturbance is therefore considered to be **High.**

For guillemot and razorbill, published evidence from reviews indicates that these species have a medium sensitivity to disturbance from vessels (e.g. Dierschke *et al.*, 2016). These species are not listed on Annex I of the Birds Directive, however there are designated SPAs for breeding guillemots and razorbills within mean maximum foraging range of the OAA, therefore these species can be considered to be of international importance (Table 11-4). It is considered likely that individuals of these species at risk are probably drawn from several SPAs, as well as from other non-SPA colonies, therefore the overall sensitivity of these species to Impact 1 is therefore considered to be **Medium**.

Magnitude of Impact

Construction activities in the OAA could potentially result in the disturbance or displacement of birds as a result of increased vessel activity and noise. This activity will occur intermittently throughout the construction period. The offshore construction works are likely to occur over a period of up to four years (Table 11-14), however, it is considered likely that the majority of offshore construction activities will be conducted between April and September, when weather conditions are typically more favourable. which represents the worst-case for the purposes of this assessment.

The impact is predicted to be of local spatial extent, intermittent, and temporary to short-term duration. The EPA (2022) guidance defines temporary duration as lasting less than one year, while “short-term” duration is defined as between one and seven years duration. However, it is considered that only a small proportion of the total OAA will be affected by construction activities at any one time, and that individual construction activities will typically be completed within a few months. Consequently, only birds in the vicinity of these individual activities will be affected directly.

Baseline surveys show that great northern divers occur in the OAA and surrounding 4 km buffer in the non-breeding season between October and May, with a mean peak of 11 birds in the OAA and 53 birds in the OAA and4 km buffer recorded in April. It is likely that these were individuals congregating prior to migrating to their breeding grounds in Iceland and Greenland (Wernham *et al.,* 2002). No great northern divers were recorded in the OAA and4 km buffer between June and September during baseline surveys (Ornithology Technical Baseline Report).

Based on the baseline survey data, any potential disturbance to great northern divers from vessels will therefore be limited to the non-breeding season, when birds are in the vicinity of the OAA, and there will be no disturbance to great northern divers in the breeding season, therefore reproductive rates will not be affected. .

On this basis, it is considered that any disturbance to great northern diver will be temporary (non-breeding season only), and that the magnitude of any effect will therefore be **Negligible**. Similarly for guillemot and razorbill, the duration of any disturbance will be temporary, and the magnitude of any effect will therefore also be **Negligible** (Table 11-5)**.**

For great northern diver, the magnitude of the impact is deemed to be **Negligible,** and the overall sensitivity of this species is considered to be **High**. Any effects will therefore be **Not Significant** (Table 11-6).

For guillemot and razorbill, the magnitude of the impact is deemed to be **Negligible,** and the overall sensitivity of these species is considered to be **Medium**. There will therefore be a **Slight Negative** effect, which is considered **Not Significant** (Table 11-6).

11.8.5.2 Impact 2 - Disturbance and displacement along the OEC route during construction

Impact 2 considers disturbance and displacement on key bird species as a result of increased vessel activity and other construction activity along the OEC route. The key bird species are considered to be the same species identified as sensitive to Impact 1, due to the similar nature of the activities e.g. increased vessel activity. Direct temporary disturbance or displacement of birds along the OEC route may occur during construction as a result of installation of the offshore export cables. Disturbance arising from these activities has the potential to affect sensitive species directly, for example disturbance of individual seabirds by cable-laying vessels. The route of the OEC does not pass through any designated conservation sites for birds (see Figure 11-2).

Sensitivity of the Receptor

The species scoped in as being sensitive to disturbance and displacement in Table 11-16 will also potentially be affected for Impact 2. Thus, three species (great northern diver, guillemot and razorbill) were identified as being potentially sensitive to disturbance and displacement from increased vessel activity along the OEC route during the construction phase. The sensitivity of great northern diver was considered to be **High**, while the sensitivity of guillemot and razorbill was considered to be **Medium**, for the reasons presented under Impact 1 above.

Magnitude of Impact

Activities resulting in the disturbance or displacement of birds along the OEC route as a result of increased vessel activity and cable-laying activities may occur intermittently throughout the construction period. Installation of the offshore export cables will occur over a period of up to15 months.

Direct disturbance impacts on seabirds are predicted to be of local spatial extent, intermittent, and temporary duration, as the cable-laying operations are predicted to last up to 15 months, (Table 11-14)(although only a small proportion of the total area will be affected at any one time, with individual activities having much shorter durations) and will only affect any birds in the vicinity of these activities directly.

The magnitude for Impact 2 was considered to be the same as for Impact 1 (**Negligible**).

For great northern diver, the magnitude of the impact is deemed to be **Negligible** and the overall sensitivity of this species is considered to be **High**. Any effects will therefore be **Not Significant** (Table 11-6).

For guillemot and razorbill, the magnitude of the impact is deemed to be **Negligible** and the overall sensitivity of these species is considered to be **Medium**. There will therefore be a **Slight Negative** effect, which is considered **Not Significant** (Table 11-6).

11.8.5.3 Impact 3 - Indirect effects on foraging seabirds during construction

Impact 3 considers indirect effects on foraging seabirds as a result of habitat loss/displacement of prey species due to increased noise and disturbance to seabed during construction. Disturbance or displacement to prey species may lead to indirect effects on foraging seabirds during the construction phase. Such indirect effects may be caused by the generation of suspended sediments (e.g. during cable-laying) or underwater noise associated with certain construction activities. An increase in suspended sediment concentration (SSC) may cause fish and mobile invertebrates to temporarily leave the construction area or may smother and hide immobile benthic prey. Suspended sediments also reduce visibility, making it harder for foraging seabirds to see their prey. These activities may lead to a reduction in prey being available within the construction area for foraging seabirds. Such potential effects on benthic invertebrates and fish have been assessed in Chapter 9: Benthic Ecology and Chapter 10: Fish and Shellfish Ecology. The conclusions of those assessments inform this assessment of indirect effects on foraging seabirds in the OAA and along the OEC route.

Sensitivity of the Receptor

Seabird species typically have a variety of target prey species and their foraging ranges would typically cover larger areas than will be affected by construction activities, meaning that they will be able to forage for alternative prey species or to forage in other areas if prey becomes temporarily unavailable during construction activities. The sensitivity of seabirds to indirect effects as a result of habitat loss or displacement of prey species due to increased noise and disturbance during construction is therefore considered to be **low**.

Magnitude of Impact

Construction activities may alter the behaviour or availability of prey species for seabirds, resulting in temporary reduced availability of these prey species to seabirds foraging in the vicinity. The total area of the OAA is 37.3 km^2. The area of temporary habitat disturbance within the OAA will be 0.291 km^2 (Chapter 9), which corresponds to 0.78% of the area of the OAA. Construction of the Offshore Site is currently planned to last for four years, (although only a small proportion of the total area will be affected at any one time, with individual activities having much shorter durations). Therefore, both habitat disturbance to prey species and increases in suspended sediment will be temporary, short-term and localised in extent.

Along the OEC route, the trenching of cables will cause a localised and temporary impact on the habitats within the vicinity. The OEC corridor has a total area of 72.8 km^2. The area of temporary habitat disturbance within the OEC corridor will be 0.999 km^2 (Chapter 9), which corresponds to 1.37% of the OEC corridor. The temporary disturbance associated with OEC installation will not cause a significant reduction in the extent, distribution or quality of habitats that support the prey of foraging seabirds.

The benthic ecology assessment (Chapter 9) concluded that the maximum significance on habitats that could be disturbed as a result of construction activities in the OAA was slight, negative effect. The fish and shellfish ecology assessment (Chapter 10) concluded that the maximum significance on fish species that could be disturbed as a result of construction activities in the OAA was significant (and tolerable) for elasmobranchs and slight, negative effect for other fish species.

The benthic ecology assessment (Chapter 9) concluded that the maximum significance on habitats such as subtidal gravels and muds and subtidal sands and gravels (that are likely to support potential prey species for foraging seabirds) from a temporary increase in SSC as a result of construction activities in the OAA was slight, negative effect. The fish and shellfish ecology assessment (Chapter 10) concluded that the maximum significance on fish species that could be disturbed as a result of a temporary increase in SSC as a result of construction activities in the OAA was slight, negative effect for all receptors.

Based on the assessment presented in the Fish and Shellfish Ecology assessment (Chapter 10), the maximum significance of the impact of underwater noise from unexploded ordnance (UXO) clearance on fish and shellfish species has been assessed as a slight negative effect.

Overall, the Benthic Ecology assessment (Chapter 9) and the Fish and Shellfish Ecology assessment (Chapter 10) concluded no significant effects on potential prey species (benthic organisms, fish or shellfish) or on the habitats that support them from construction activities. The maximum magnitude of any indirect impact on foraging seabirds has therefore been assessed as **Negligible**, with the maximum sensitivity of these receptors being **Medium**. Therefore, the significance of any indirect effect on foraging seabirds during construction activities in the OAA and OEC route is a S**light, Negative** effect, which is **Not Significant** (Table 11-6).

11.8.6 Operation and Maintenance Phase

11.8.6.1 Impact 4: Disturbance from maintenance activities within the OAA

Impact 4 considers disturbance to key bird species as a result of increased vessel activity and other maintenance activities within the OAA during the operation and maintenance. Direct temporary disturbance of birds within the OAA may occur from regular maintenance and service vessels and other maintenance activities.

There is also the potential for major component repairs which would require the use of a jack-up vessel and associated activities. Such events could result in disturbance of bird species and to prey species, although such repairs are considered to be temporary and very occasional. Table 11-14 summarises the project design parameters of the Offshore Site considered within this assessment.

Sensitivity of the Receptor

The scoping exercise undertaken for Impact 1 is also relevant for Impact 4, and the species scoped in as being sensitive to disturbance and displacement in Table 11-16 will also potentially be affected for Impact 4. Thus, three species (great northern diver, guillemot and razorbill) were identified as being potentially sensitive to disturbance from regular maintenance and service vessels and other maintenance activities within the OAA during the operation and maintenance phase. The remaining species scoped out in Table 11-16 are not considered sensitive to Impact 4. The sensitivity of great northern diver was considered to be **High**, while the sensitivity of guillemot and razorbill was considered to be **Medium**, for the reasons presented under Impact 1 above.

Magnitude of Impact

Maintenance activities in the OAA could potentially result in the disturbance of birds as a result of increased vessel activity and noise. This activity will occur regularly throughout the operation and maintenance phase. In addition, there are other potential irregular maintenance activities that may be required occasionally during the operation and maintenance phase.

The impact is predicted to be of local spatial extent, intermittent, and temporary duration. However, it is considered that only a small proportion of the total OAA will be affected by maintenance activities at any one time, and consequently, only birds in the vicinity of these individual activities will be affected directly.

On this basis, it is considered that any disturbance to great northern diver will be intermittent and temporary (non-breeding season only), and that the magnitude of any effect will therefore be **Negligible**. Similarly for guillemot and razorbill, the duration of any disturbance will be intermittent and temporary, therefore the magnitude of any effect will be **Negligible** (Table 11-5).

For great northern diver, the magnitude of the impact is deemed to be **Negligible** and the overall sensitivity of this species is considered to be **High**. Any effects will therefore be **Not Significant** (Table 11-6).

For guillemot and razorbill, the magnitude of the impact is deemed to be **Negligible** and the overall sensitivity of these species is considered to be **Medium**. There will therefore be a **Not Significant** effect (Table 11-6).

11.8.6.2 Impact 5 - Indirect effects on seabirds due to presence of project infrastructure

Impact 5 considers indirect effects on seabirds as a result of habitat loss and displacement of prey species due to the presence of turbines or due to increased noise and disturbance to the seabed. Long term habitat loss will occur directly under all turbine and Offshore Substation (OSS) foundation structures, associated scour protection and cable protection where this is required. The seabed habitats removed by the installation of infrastructure will reduce the amount of suitable habitat and available food resource for fish and shellfish species and communities associated with the baseline substrates/sediments underneath the turbine bases, which could in turn, reduce the availability of these prey fish species for foraging seabirds in the vicinity.

In addition, there is the potential for temporary seabed disturbance resulting from array cable repairs or OEC repairs which may release sediment into the water column, potentially causing fish and mobile invertebrates to temporarily avoid the area. Suspended sediments also reduce visibility, making it harder for foraging seabirds to see their prey. These activities may lead to a reduction in prey being available within the area of maintenance activities for foraging seabirds.

However, turbine foundations may act as “artificial reefs” for fish species (Dannheim *et al.,* 2020), and there is also evidence that some species may target OWFs for food and/or refuge, benefitting from the ecological changes that take place following their installation (Degraer *et al*., 2020). For example, at the Thorntonbank OWF in Belgian waters, herring gulls and great black-backed gulls have been recorded foraging on barnacles in the lower intertidal zone of the turbine jacket foundations (Vanermen *et al.,* 2017).

Sensitivity of the Receptor

Seabird species typically have a variety of target prey species and their foraging ranges would typically cover larger areas than will be affected by maintenance activities, meaning that they will be able to forage for alternative prey species or to forage in other areas if prey becomes temporarily unavailable. The sensitivity of seabirds to indirect effects as a result of habitat loss or displacement of prey species due to increased noise and disturbance during operational maintenance is therefore considered to be **low**.

Magnitude of Impact

Overall, given that the area of the Offshore Site (OAA and OEC corridor) is 110.1 km^2, the area of long-term habitat loss or damage represents 1.52% of the Offshore Site (Chapter 9). Therefore, the habitat loss for prey species will be small in extent.

The benthic ecology assessment (Chapter 9) concluded that with mitigation in design taken into account, the overall ecological function of the benthic habitats remaining within baseline levels, the residual effect of permanent habitat loss as a result of the Project would be a likely, long-term slight negative effect.

The majority of fish species would be able to avoid habitat loss effects due to their mobility and would recover into the areas affected following cessation of construction. Sandeels (and other less mobile prey species) would be affected by long term subtidal habitat loss, although recovery of this species is expected to occur quickly as the sediments recover following installation of infrastructure and adults recolonise and also via larval recolonisation of any sandy sediments. Overall, the fish and shellfish ecology assessment (Chapter 10) concluded that the maximum significance on fish species that could be affected as a result of permanent habitat loss associated with the Project was significant (and tolerable) for elasmobranchs and slight, negative effect for other fish species.

Based on the above, the indirect impact on seabirds as a result of changes in prey availability or distribution associated with permanent habitat loss is predicted to be of local spatial extent, indirect and of medium-term duration, as prey species distribution is considered likely to recover over time. The magnitude is therefore considered to be **Low**.

The magnitude of the impact is deemed to be **Low** and the overall sensitivity of seabird species to Impact 5 is considered to be **Low**. The effect will therefore be of **Slight, Negative, Medium-term Significance**, which is **Not Significant** in EIA terms (Table 11-6).

11.8.6.3 Impact 6 - Displacement and barrier effects within the OAA

Impact 6 considers the potential for displacement and barrier effects on key bird species within the OAA and appropriate buffer from offshore infrastructure. Displacement and/or barrier effects on birds within the OAA and immediate surrounding area during the operation and maintenance phase may occur as a result of the presence of the operational turbines. In this assessment, displacement and barrier effects have been considered together following the approach presented in SNCB guidance (2022a).

Displacement and/or barrier effects resulting from the presence of offshore turbines has the potential to affect individuals of sensitive bird species directly. In effect, this represents habitat loss, which would potentially reduce the area available to forage, rest and/or moult for sensitive seabirds that currently occur within and around the OAA. Displacement may contribute to the overall fitness of individual birds, which could also affect individual breeding success or at an extreme level, could cause mortality of individuals.

The approach for the displacement assessment is presented in the Displacement Assessment Appendix, and follows SNCB guidance (SNCBs, 2022a). Firstly, species sensitivity and habitat specialisation were considered for each species, based on published reviews e.g. Bradbury *et al.,* (2014), as well as numbers of each species likely to be present, based on baseline surveys undertaken between October 2021 and September 2023.

Based on this review, displacement and barrier effects impacts were then assessed using the SNCB approach on 11 species; great northern diver, Manx shearwater, gannet, shag, eider, kittiwake, common tern, Arctic tern, guillemot, razorbill and puffin. Full details of the approach and the seasonal displacement matrices for each of these species are presented in the Displacement Assessment Appendix (Appendix 11-2).

A summary of the predicted displacement mortality within the OAA and appropriate 2 km or 4 km buffer for each of the 11 assessed species is presented in Table 11-17. For species where displacement mortality was predicted to be zero (shag, eider, common tern, Arctic tern), no further assessment was undertaken.

Table 11-17 Predicted displacement mortality for the OAA and appropriate buffer (2 km unless otherwise stated)

Species	Displacement & mortality rates assessed	Predicted displacement	Predicted mortality
Great Northern Diver (4 km buffer)	90-100%; 2%	48-53 birds NBS	One bird per non-breeding season
Manx Shearwater	50%, 1%	3,007 birds BS	18 birds in breeding season; Zero birds in autumn and spring migration periods
Gannet	70%; 1%	51 birds BS 50 birds AUT 4 birds SPR	One bird in breeding season; One bird in autumn migration period; Zero birds in spring migration period
	70%; 3%		Two birds in breeding season; Two birds in autumn migration period; Zero birds in spring migration period
Shag	60%; 1%	19 birds BS 17 birds NBS	Zero birds in breeding and non-breeding seasons
Eider (4 km buffer	60%; 1%	68 birds NBS	Zero birds in non-breeding season
Kittiwake	30%; 1%		Zero birds in breeding season; Zero birds in autumn and spring migration periods
	30%, 3%	28 birds BS 24 birds AUT 43 birds SPR	One bird in breeding season One bird in autumn migration period; One bird in spring migration period
Common Tern	30%, 1%	14 birds BS	Zero birds in breeding season; Zero birds in autumn and spring migration periods
Arctic Tern	30%, 1%	17 birds BS	Zero birds in breeding season; Zero birds in autumn and spring migration periods
Guillemot	50%, 1%	1,608 birds BS 154 birds NBS	16 birds in breeding season; Two birds in non-breeding season
	60%, 1%		Two birds in non-breeding season
	60%, 3%		58 birds in breeding season; Six birds in non-breeding season
	60%, 5%	185 birds NBS 1,930 birds BS	96 birds in breeding season
Razorbill	50%, 1%	110 birds BS 6 birds AUT	One bird in breeding season; Zero birds in autumn and spring

Species	Displacement & mortality rates assessed	Predicted displacement	Predicted mortality
		96 birds WIN 40 birds SPR	migration periods; One bird in winter period
	60%, 1%		Zero birds in autumn and spring migration period; One bird in winter period
	60%, 3%		Four birds in breeding season; Zero birds in autumn migration period; Three birds in winter period One bird in spring migration period
	60%, 5%	132 birds BS 7 birds AUT 115 birds WIN 47 birds SPR	Seven birds in breeding season;
	50%, 1%	38 birds BS 5 birds NBS	Zero birds in breeding and non-breeding seasons
	60%, 1%		Zero birds in non-breeding season
	60%, 3%		One bird in breeding season; Zero birds in non-breeding season
Puffin	60%, 5%	46 birds BS 5 birds NBS	Two birds in breeding season Zero birds in non-breeding season

NBS – Non-breeding season; BS – Breeding Season; AUT – Autumn migration period; WIN – Winter period; SPR – Spring migration period

Displacement mortality was predicted for six species, great northern diver, Manx shearwater, gannet, kittiwake, guillemot, and razorbill, and these species were therefore considered further in the displacement assessment below.

Great northern diver

Based on the mean seasonal peak of great northern divers in the OAA and 4 km buffer in the non-breeding season, displacement mortality was predicted to be one bird per non-breeding season, based on a displacement rate of 100% and a mortality rate of 2%. (Table 11-17). Further details and the seasonal displacement matrices are presented in the Displacement Assessment Appendix (Appendix 11-2).

The RoI wintering population of great northern divers is estimated to be 2,128 individuals (Lewis, *et al.*, 2019). Based on 5-year mean I-WebS counts between 2011/12 and 2015/16 winters for regularly counted sites between Galway and Donegal, a combined population of 744 birds was estimated for these sites (Table 11-18).

Table 11-18 Recent 5-year mean counts for key wintering sites for Great Northern Diver (Lewis et al., 2019)

Site	Mean count between 2011/12 and 2015/16
Inner Galway Bay	209
Blacksod & Tullahan Bays	196

Site	Mean count between 2011/12 and 2015/16
Donegal Bay	134
Mannin Bay	64
Broadhaven & Sruwadaccon Bays	59
Clew Bay	44
Lough Swilly	27
Lough Foyle	11
Total	744 birds

Baseline surveys in the wider 10 km survey area for the Offshore Site project recorded peak estimates of 382 great northern divers in January 2022 and 567 birds in February 2023 (Aerial Survey Two Year Report). The mean of these two winter counts is 475 birds. If this figure is added to the combined 5-year mean count of 744 birds from the sites listed in Table 11-18, then this would give an estimated non-breeding season population of 1,219 great northern divers. In the absence of a more complete regional population estimate for the non-breeding season, this estimate has been used as the regional reference population for this assessment. It should be noted that this is likely to be a minimum estimate, as birds further offshore would be missed on regular monitoring schemes such as I-WeBS (Crowe, 2005).

For the non-breeding season, estimated seasonal great northern diver displacement mortality was one bird (all ages). As outlined above, the great northern diver regional non-breeding population was estimated to be 1,219 birds. Applying the average mortality rate of 0.161 (Table 11-13), the estimated regional baseline mortality of great northern diver is 196 birds in the non-breeding season (1,219 x 0.161). The additional predicted mortality of one great northern diver in the non-breeding season would increase the baseline mortality rate by 0.51%.

As highlighted by Natural England guidance, where predicted impacts equate to 1% or below of baseline mortality for a population (e.g. colony population) then this level of impact could be considered non-significant (Parker *et al.,* 2022c). Based on this, as the predicted increase in annual baseline mortality for great northern diver was below 1%, PVA was not carried out on the regional population.

Based on the results of the displacement assessment, the magnitude of impact from displacement on the regional great northern diver population was considered to be **Negligible**, as the estimated increase in the annual baseline mortality rate was less than 1% for non-breeding season assessment (Table 11-5).

For this assessment, receptor sensitivity has been based on two reviews of evidence from post-construction studies at offshore wind farms. A review of vulnerability of Scottish seabirds to offshore wind turbines in the context of disturbance and displacement ranked great northern diver as the 3rd most sensitive out of 38 species (Furness *et al.,* 2013). Bradbury *et al.,* (2014), classified the great northern diver population vulnerability to displacement from offshore wind farms in English waters as high.

Great northern divers recorded within the OAA and 4 km buffer in the non-breeding season would qualify as internationally important (Baseline Ornithology Report), as peak estimates exceeded the international threshold of importance (50 birds) (Lewis *et al.,* 2019). On this basis the conservation importance for great northern diver was considered to be medium (Table 11-4).

Overall, based on the conservation importance and likely sensitivity to offshore wind turbines, great northern diver sensitivity to displacement associated with the Offshore Site is likely to be **High**.

For great northern diver, the magnitude of the impact is deemed to be **Negligible,** and the overall sensitivity of this species is considered to be **High**. Any effect is therefore considered **Not Significant** (Table 11-6).

Manx shearwater

Based on the mean seasonal peak of Manx shearwaters in the OAA and 2 km buffer in the breeding season, displacement mortality was predicted to be 18 birds per breeding season, based on a displacement rate of 30% and a mortality rate of 1%. There was zero displacement mortality predicted in the autumn and spring migration periods (Table 11-17). Further details and the seasonal displacement matrices are presented in the Displacement Assessment Appendix (Appendix 11-2).

The predicted displacement mortality of 18 birds includes non-breeding adults and immature birds, as well as breeding adults. Studies have shown that for several seabird species, in addition to breeding birds, colonies are also attended by many immature individuals and a smaller number of non-breeding adults (e.g. Wanless *et al.,* 1998). There is little information on the breakdown of immature and non-breeding adults present at a colony, however, this has been estimated using proportions from Horswill and Robinson (2015) (Ornithology Baseline Report). Based on the proportion of immature Manx shearwaters from the population age ratio (0.532), 53.2% of the population present are immature birds, with a corresponding 46.8% of the population being adult birds. This means that an estimated mortality of 18 Manx shearwaters would involve eight adult birds and 10 immature birds.

In addition, a proportion of adult birds present at colonies in the breeding season will opt not to breed in a particular breeding season. However, it is not known how many adult Manx shearwaters that attend a colony may be non-breeding "sabbatical" birds in any particular breeding season (Baker *et al.,* 2022). Therefore, for this assessment, it was assumed that all adults were breeding birds, which is a precautionary approach.

The breeding season regional reference population for breeding adult Manx shearwaters is 363,150 birds, (Table 11-11). Applying the adult mortality rate of 0.130 (Table 11-13), the estimated regional baseline mortality of Manx shearwater is 47,210 birds in the breeding season (363,150 x 0.130). The additional predicted mortality of eight breeding adult Manx shearwaters in the breeding season would increase the baseline mortality rate by 0.017% (Table 11-19).

Table 11-19 Increase in estimated baseline mortality for Manx shearwater in the OAA plus 2 km buffer as a result of displacement, based on adults only breeding population

Season	Predicted seasonal mortality	Regional baseline popn	Annual regional baseline mortality	Increase in baseline mortality
Breeding (adults only)	8 adults	363,150adults	47,210 adults	0.017%
Autumn	0	-	-	0
Spring	0	-	-	0
Annual total	8 birds	-	-	0.017%

A comparison of estimated Manx shearwater mortality against a regional population consisting of adult and immature birds is shown in Table 11-20. Applying a mortality rate of 1%, the additional mortality due to displacement effects was 18 birds (all ages) in the breeding season. The total Manx shearwater regional breeding population for all ages is estimated to be 774,236 birds (Table 11-11). The average mortality for all age classes is 0.13 (Table 11-13). Applying this mortality rate, the estimated regional baseline mortality of Manx shearwater is 100,651 birds per breeding season (all ages) (774,236 x 0.13). The additional predicted mortality of 18 Manx shearwaters (all ages) in the breeding season would increase the baseline mortality rate by 0.018% (Table 11-20).

As highlighted by Natural England guidance, where predicted impacts equate to 1% or below of baseline mortality for a population (e.g. colony population) then this level of impact could be considered non-significant (Parker *et al.,* 2022c). Based on this, as the predicted increases in annual baseline mortality for Manx shearwater were below 1%, PVA was not carried out on the regional population.

Based on the results of the displacement assessment, the magnitude of impact from displacement on the regional Manx shearwater breeding population was considered to be **Negligible**, as the estimated increases in the annual baseline mortality rate were less than 1% for the breeding season assessment.

Table 11-20 Increase in estimated baseline mortality for Manx shearwater in the OAA plus 2 km buffer as a result of displacement, based on all ages

Season	Predicted seasonal mortality	Regional baseline popn	Annual regional baseline mortality	Increase in baseline mortality
Breeding	18 birds	832,938	108,282	0.018%
Autumn	0	-	-	0
Spring	0	-	-	0
Annual total	18 birds	-	-	0.018%

For this assessment, receptor sensitivity has been based on three reviews of evidence from post-construction studies at offshore wind farms. A review of post-construction studies of seabirds at offshore wind farms in European waters concluded that Manx shearwater was one of the species which weakly avoided offshore wind farms, although evidence for this species was limited and other factors such as flexibility of habitat use and extensive foraging range should also be considered (Dierschke *et al.,* 2016). A review of vulnerability of Scottish seabirds to offshore wind turbines in the context of disturbance and displacement ranked Manx shearwater as the 34th most sensitive out of 38 species (Furness *et al.,* 2013). Bradbury *et al.,* (2014), classified the Manx shearwater population vulnerability to displacement from offshore wind farms in English waters as very low.

Evidence from reviews presented above and from post-construction studies summarised in the Displacement Matrices Technical Report, indicates that Manx shearwater sensitivity to displacement from operational offshore wind farms is likely to be Low.

Manx shearwaters recorded within the OAA would qualify as internationally important in the breeding season Baseline Ornithology Report, with individuals potentially originating from a number of SPAs in the region, as well as non-SPA colonies. On this basis the conservation importance for Manx shearwater was considered to be medium (Table 11-4).

Overall, based on available evidence from published studies, and the origin of birds from SPA and non-SPA colonies in the region, it is considered that Manx shearwater sensitivity to displacement associated with the Offshore Site is likely to be **Low**.

For Manx shearwater, the magnitude of the impact is deemed to be **Negligible,** and the overall sensitivity of this species is considered to be **Low.** Any effect is therefore considered **Not Significant** (Table 11-6).

Gannet

Studies on foraging gannets have shown that they are capable of extending their foraging distances in response to prey distribution, indicating that birds would easily absorb the minor increases in flight distances that a barrier such as an offshore wind farm could cause (Hamer *et al.,* 2007; Hamer *et al*., 2011). In addition, this species was rated as having a low sensitivity to barrier effects by Maclean *et al.* (2009) and Langston (2010). A review by Furness and Wade (2012) concluded that gannets use a wide range of habitats over a large area, usually with a relatively wide range of prey species, and therefore have a high flexibility of habitat use.

Based on the above, it is considered unlikely that there will be any mortality resulting from displacement from the OAA and the 2 km buffer, as displaced birds would be able to forage elsewhere off the west coast of Ireland. However, for the purposes of this assessment, the increase in baseline mortality rate has been considered based on both 1% and 3% of all displaced gannets from the OAA and a 2 km buffer suffering mortality as a consequence of being displaced.

In addition, a recent review of gannet displacement and mortality based on evidence from 25 OWFs recommended that a maximum rate of 1% mortality should be used for assessing potential impacts associated with displacement for gannets from OWFs (APEM, 2022a). Therefore, the use of 3% mortality is considered to be over-precautionary.

Further details about the displacement assessment approach and the seasonal displacement matrices are presented in the Displacement Assessment Appendix (Appendix 11-2).

In the breeding season, based on a displacement rate of 70%, 51 birds were predicted to be displaced. However, this includes non-breeding adults and immature birds, as well as breeding adults. In the breeding season (March to September) age was recorded for 85 gannets on baseline surveys, with 34% of birds aged as adults and 66% of birds aged as immature. Further details are presented in the Ornithology Baseline Report. Based on this breakdown, it has been assumed that an estimated displacement of 51 gannets would involve 17 adult birds and 34 immature birds.

In addition, a proportion of adult birds present at colonies in the breeding season will opt not to breed in a particular breeding season. It has been estimated that 10% of adult gannets may be “sabbatical” birds in any particular breeding season (Xodus, 2023), and this has been applied for this assessment. On this basis, 1.7 displaced adult gannets were considered not to be breeding, however, for this assessment numbers have been rounded to the nearest whole bird for clarity, therefore two adult gannets were considered not to be breeding. The number of displaced gannets was considered to be 15 breeding adults, two non-breeding adults and 34 immature birds. Applying a mortality rate of 1% to 15 breeding adults would result in a mortality of 0.15 adult gannets. Applying a mortality rate of 3% to 15 breeding adults would result in a mortality of 0.45 adult gannets.

Due to the low numbers of birds involved, for the purposes of this assessment, it has been assumed that all predicted mortality involved adult breeding gannets, which is considered to be precautionary. Therefore, it was assumed that based on a displacement mortality rate of 1%, displacement mortality was predicted to involve one adult gannet in the breeding season. Based on a mortality rate of 3%, displacement mortality was predicted to involve two adult gannets in the breeding season.

The breeding season regional reference population for breeding adult gannets is 93,602 breeding birds (Table 11-11). Applying the adult mortality rate of 0.081 (Table 11-13), the estimated regional baseline mortality of gannet is 7,582 birds in the breeding season (93,602 x 0.081). Based on a displacement mortality rate of 1%, the additional predicted mortality of one adult gannet in the breeding season would increase the baseline mortality rate by 0.013%. Based on a displacement mortality rate of 3%, the additional predicted mortality of two adult gannets in the breeding season would increase the baseline mortality rate by 0.026% (Table 11-21).

Table 11-21 Increase in estimated baseline mortality for gannets in the OAA plus 2 km buffer as a result of displacement

Season	Predicted seasonal mortality	Regional baseline popn	Annual regional baseline mortality	Increase in baseline mortality
1% mortality rate				
Breeding	1 bird	93,602	7,582	0.013%
Autumn	1 bird	662,102	119,840	0.001%
Spring	0	-	-	0
Annual total	2 birds	-	-	0.014%
3% mortality rate				
Breeding	2 birds	93,602	7,582	0.026%
Autumn	2 birds	662,102	119,840	0.003%
Spring	0	-	-	0
Annual total	4 birds	-	-	0.03%

In the autumn migration period of the non-breeding season, based on a displacement rate of 70%, 50 gannets (all ages) were predicted to be displaced. Applying a mortality rate of 1% would result in a predicted mortality of one gannet. Applying a mortality rate of 3% would result in a predicted mortality of two gannets.

The regional reference population for the autumn migration period has been estimated as 662,102 birds (all ages) (Table 11-12). Applying the average mortality rate of 0.181 (Table 11-13), the estimated regional baseline mortality of gannet is 119,840 birds in the autumn migration period (662,102 x 0.181). Based on a displacement mortality rate of 1%, the additional predicted mortality of one gannet in the autumn migration period would increase the baseline mortality rate by 0.001%. Based on a displacement mortality rate of 3%, the additional predicted mortality of two gannets in the autumn migration period would increase the baseline mortality rate by 0.003% (Table 11-21).

In the spring migration period of the non-breeding season, based on a displacement rate of 70%, four gannets (all ages) were predicted to be displaced. Applying a mortality rate of 1%, the predicted additional mortality due to displacement effects was zero gannets in the spring migration period. Similarly, applying a mortality rate of 3%, the predicted additional mortality due to displacement effects was zero gannets.

Predicted annual gannet mortality due to displacement effects for adult gannets in the breeding season and all ages in the autumn and spring migration periods, based on a 1% mortality rate involved two gannets, which corresponds to an increase in the annual baseline mortality rate of 0.014%. Based on a 3% mortality rate, predicted gannet mortality involved four birds, which corresponds to an increase in the annual baseline mortality rate of 0.03% (Table 11-21).

A comparison of estimated gannet mortality against a regional population consisting of adult and immature birds gives the same result. Applying a mortality rate of 1%, the predicted additional mortality due to displacement effects was one gannet (all ages) in the breeding season and one gannet in the autumn migration period. Similarly, applying a mortality rate of 3%, the predicted additional mortality due to displacement effects was two gannets (all ages) in the breeding season, and two gannets in the autumn migration period.

The breeding season regional reference population for adult and immature gannets is 164,833 birds (all ages) (Table 11-11). Applying the average mortality rate of 0.181 (Table 11-13), the estimated regional baseline mortality of gannet is 29,835 birds in the breeding season (164,833 x 0.181). Based on a displacement mortality rate of 1%, the additional predicted mortality of one gannet in the breeding season would increase the baseline mortality rate by 0.003%. Based on a displacement mortality rate of 3%, the additional predicted mortality of two gannets (all ages) in the breeding season would increase the baseline mortality rate by 0.007%.

As highlighted by Natural England guidance, where predicted impacts equate to 1% or below of baseline mortality for a population (e.g. colony population) then this level of impact could be considered non-significant (Parker *et al.,* 2022c). Based on this, as the predicted increase in annual baseline mortality for gannet was below 1%, PVA was not carried out on the regional population.

Based on the results of the displacement assessment, the magnitude of impact from displacement on the regional gannet population was considered to be **Negligible**, as the estimated increases in the annual baseline mortality rate were less than 1% (Table 11-5).

For this assessment, receptor sensitivity has been based on three reviews of evidence from post-construction studies at offshore wind farms. A review of post-construction studies of seabirds at offshore wind farms in European waters concluded that gannet was one of the species which strongly or nearly completely avoided offshore wind farms, however, other factors such as flexibility of habitat use and extensive foraging range should also be considered (Dierschke *et al.,* 2016). A review of vulnerability of Scottish seabirds to offshore wind turbines in the context of disturbance and displacement ranked gannet as the 28th most sensitive out of 38 species (Furness *et al.,* 2013). Bradbury *et al.,* (2014), classified the gannet population vulnerability to displacement from offshore wind farms in English waters as very low.

Gannets recorded within the OAA would qualify as internationally important in the breeding season (Offshore Ornithology Baseline Report), with individuals potentially originating from a number of SPAs in the region, as well as non-SPA colonies. On this basis the conservation importance for gannet was considered to be medium (Table 11-4).

Overall, based on available evidence from published studies, and the origin of birds from SPA and non-SPA colonies in the region, it is considered that gannet sensitivity to displacement associated with Sceirde Rocks Offshore Windfarm is likely to be **Medium**.

For gannet, the magnitude of the impact is deemed to be **Negligible,** and the overall sensitivity of this species is considered to be **Medium.** Any effect is therefore considered **Not Significant** (Table 11-6).

Kittiwake

Recent guidance for OWF projects in Scottish waters recommended that a displacement rate of 30% should be used for kittiwakes (NatureScot, 2023), and this displacement rate has been applied for this assessment. NatureScot guidance also recommended that mortality rates of 1% and 3% throughout the year should be used for kittiwake in displacement assessments (NatureScot, 2023). These mortality rates have also been applied for this assessment. Further details of the displacement assessment approach and the seasonal displacement matrices are presented in the Displacement Assessment Appendix (Appendix 11-2).

However, as presented in the Displacement Assessment Appendix (Appendix 11-2), available evidence from post-construction studies and reviews indicates that displacement of kittiwakes by offshore wind turbines is not likely to occur to any significant extent. It is therefore considered that 30% displacement with a mortality rate of 1% is suitably precautionary for an assessment of displacement effects from the Project on kittiwakes.

Due to the low numbers of birds involved, for the purposes of this assessment, it has been assumed that all predicted mortality involved adult breeding kittiwakes, which is considered to be precautionary, as both adult and immature kittiwakes were recorded in the Offshore ornithology Study Area on baseline surveys throughout the year (Baseline Ornithology Report). The number of kittiwake mortalities was therefore compared against the estimated regional population of breeding adult kittiwakes,

In the breeding season, the mean seasonal peak of kittiwakes in the OAA plus 2 km buffer was 93 birds. Based on a displacement rate of 30%, this would mean that an estimated 28 kittiwakes would be displaced from the OAA and 2 km buffer in the breeding season. Applying a 1% mortality rate would therefore involve 0.28 kittiwakes. Applying a 3% mortality rate would involve 0.84 kittiwakes (Table 11-22).

The breeding season regional reference population for kittiwake is 26,720 breeding adults (Table 11-11). Applying the adult mortality rate of 0.146 (Table 11-13), the estimated regional baseline mortality for kittiwake is 3,901 birds in the breeding season (26,720 x 0.146). Based on a displacement mortality rate of 1%, the additional predicted mortality of 0.28 kittiwakes in the breeding season would increase the baseline mortality rate by 0.007%. Based on a displacement mortality rate of 3%, the additional predicted mortality of 0.84 kittiwakes in the breeding season would increase the baseline mortality rate by 0.02% (Table 11-22).

Table 11-22 Increase in estimated baseline mortality for kittiwakes in the OAA plus 2 km buffer as a result of displacement

Season	Predicted seasonal mortality	Regional baseline popn	Annual regional baseline mortality	Increase in baseline mortality
1% mortality rate				
Breeding	0.28 birds	26,720	3,901	0.007%
Autumn	0.24 birds	945,743	147,536	0.0002
Spring	0.43 birds	725,683	113,207	0.0004
Annual total	0.95 birds	-	-	0.008
Annual total rounded to whole birds	1 bird	-	-	0.01
3% mortality rate				
Breeding	0.84 birds	26,720	3,901	0.02%
Autumn	0.72 birds	945,743	147,536	0.0005%
Spring	1.29 birds	725,683	113,207	0.001
Annual total	2.85 birds	-	-	0.02%

Season	Predicted seasonal mortality	Regional baseline popn	Annual regional baseline mortality	Increase in baseline mortality
Annual total	3 birds	-	-	0.02%

In the autumn migration period of the non-breeding season, the mean seasonal peak of kittiwakes in the OAA plus 2 km buffer was 79 birds. Based on a displacement rate of 30%, this would mean that an estimated 24 kittiwakes would be displaced from the OAA and 2 km buffer in the autumn migration period. Applying a 1% mortality rate would therefore involve 0.24 kittiwakes. Applying a 3% mortality rate would involve 0.72 kittiwakes (Table 11-22).

The regional reference population for the autumn migration period has been estimated as 945,743 birds (Table 11-12). Applying the average mortality rate of 0.156 (Table 11-13), the estimated regional baseline mortality of kittiwake is 147,536 birds in the autumn migration period (945,743 x 0.156). Based on a displacement mortality rate of 1%, the additional predicted mortality of 0.24 kittiwakes in the autumn migration period would increase the baseline mortality rate by 0.0002%. Based on a displacement mortality rate of 3%, the additional predicted mortality of 0.72 kittiwakes in the autumn migration period would increase the baseline mortality rate by 0.0005% (Table 11-22).

In the spring migration period of the non-breeding season, the mean seasonal peak of kittiwakes in the OAA plus 2 km buffer was 144 birds. Based on a displacement rate of 30%, this would mean that an estimated 43 kittiwakes would be displaced from the OAA and 2 km buffer in the spring migration period. Applying a 1% mortality rate would therefore involve 0.43 kittiwakes. Applying a 3% mortality rate would involve 1.29 kittiwakes (Table 11-22).

The regional reference population for the spring migration period has been estimated as 725,683 birds (Table 11-12). Applying the average mortality rate of 0.156 (Table 11-13), the estimated regional baseline mortality of kittiwake is 113,207 birds in the spring migration period (725,683 x 0.156). Based on a displacement mortality rate of 1%, the additional predicted mortality of 0.43 kittiwakes in the spring migration period would increase the baseline mortality rate by 0.0004%. Based on a displacement mortality rate of 3%, the additional predicted mortality of 1.29 kittiwakes in the spring migration period would increase the baseline mortality rate by 0.001% (Table 11-22).

Additional annual predicted mortality as a result of displacement was predicted to involve one kittiwake based on a 1% mortality rate, or three kittiwakes based on a 3% mortality rate (Table 11-22). This corresponds to an increase in the baseline mortality rate of 0.01% and 0.02% respectively.

As highlighted by Natural England guidance, where predicted impacts equate to 1% or below of baseline mortality for a population (e.g. colony population) then this level of impact could be considered non-significant (Parker *et al.,* 2022c). Based on this, as the predicted increases in annual baseline mortality for kittiwake were below 1%, PVA was not carried out on the regional population.

Based on the results of the displacement assessment, the magnitude of impact from displacement on the regional kittiwake population in the breeding season and the autumn and spring migration periods of the non-breeding season was considered to be **Negligible**, as the estimated increases in the annual baseline mortality rates were less than 1% (Table 11-5).

For this assessment, receptor sensitivity has been based on three reviews of evidence from post-construction studies at offshore wind farms. A review of post-construction studies of seabirds at offshore wind farms in European waters concluded that kittiwake was one of the species which were hardly affected by OWFs or with attraction and avoidance approximately equal over all studies (Dierschke *et al.,* 2016). A review of vulnerability of Scottish seabirds to offshore wind turbines in the context of disturbance and displacement ranked kittiwake as the 24th most sensitive out of 38 species (Furness *et* *al.,* 2013). Bradbury *et al.,* (2014), classified the kittiwake population vulnerability to displacement from offshore wind farms as very low.

Kittiwakes recorded within the OAA would qualify as internationally important in the breeding season (Offshore Ornithology Baseline Report), with individuals potentially originating from a number of SPA and non-SPA colonies within mean maximum foraging range. On this basis the conservation importance for kittiwake was considered to be medium (Table 11-4).

Overall, based on the conservation importance, with SPAs for breeding kittiwake within mean maximum foraging range of the OAA, together with evidence from reviews and post-construction studies presented above indicates that kittiwake sensitivity to displacement associated with the Sceirde Rocks project is considered to be **Low**.

For kittiwake, the magnitude of the impact is deemed to be **Negligible,** and the overall sensitivity of this species is considered to be **Low.** Any effect is therefore considered **Not Significant** (Table 11-6)

Guillemot

Based on the mean seasonal peak of guillemots in the OAA and 2 km buffer, and a displacement rate of 50% and a mortality rate of 1%, displacement mortality was predicted to be 16 birds in the breeding season. Based on a displacement rate of 60%, and a mortality rate of 3%, displacement mortality was predicted to be 58 birds in the breeding season, increasing to 96 birds, if a mortality rate of 5% is applied. It should be noted that evidence from post-construction monitoring indicates that applying displacement rates greater than 50% and mortality rates of more than 1% is overly precautionary. Results and conclusions from such studies are presented in the Displacement Assessment Appendix (Appendix 11-2).

For the non-breeding season in the OAA and 2 km buffer, and a displacement rate of 50% and a mortality rate of 1%, displacement mortality was predicted to be two birds. Based on a displacement rate of 60%, and a mortality rate of 1%, displacement mortality was predicted to be two birds in the non-breeding season, increasing to six birds, if a mortality rate of 3% is applied (Table 11-17). Further details and the seasonal displacement matrices are presented in the Displacement Assessment Appendix (Appendix 11-2).

However, this estimate includes non-breeding adults and immature birds, as well as breeding adults. Studies have shown that for several seabird species, in addition to breeding birds, colonies are also attended by many immature individuals and a smaller number of non-breeding adults (e.g. Wanless *et al.,* 1998). There is little information on the breakdown of immature and non-breeding adults present at a colony, however, this has been estimated using proportions from Horswill and Robinson (2015) (Offshore Ornithology Baseline Report), summarised in Table 11-13. Based on the proportion of adult guillemots from the population age ratio (0.522), 52.2% of the population present were assumed to be adult birds, with a corresponding 47.8% of the population assumed to be immature birds. This means that between eight and 50 guillemots displaced from the OAA and 2 km buffer during the breeding season would be adult birds, with between eight and 46 immature birds also displaced.

However, a proportion of adult birds present at colonies in the breeding season will opt not to breed in a particular breeding season. It has been estimated that 7% of adult guillemots may be “sabbatical” birds in any particular breeding season (Xodus, 2023), and this has been applied for this assessment. On this basis, between one and four displaced adult guillemots were considered not to be breeding, therefore guillemot mortality was considered to be between seven and 46 breeding adults, one to four non-breeding “sabbatical” adults and between eight and 46 immature birds.

The breeding season regional reference population for guillemot is 74,578 breeding adults (Table 11-11). Applying the adult mortality rate of 0.061 (Table 11-13), the estimated regional baseline mortality of guillemot is 4,549 birds in the breeding season (74,578 x 0.061). The additional predicted mortality of between seven and 46 guillemots in the breeding season would increase the baseline mortality rate by between 0.15% and 1.01% (Table 11-23).

Table 11-23 Increase in estimated baseline mortality for guillemots (adults only in breeding season) in the OAA plus 2 km buffer as a result of displacement

Season	Predicted seasonal mortality	Regional baseline popn	Annual regional baseline mortality	Increase in baseline mortality
50% displacement and 1% mortality rate				
Breeding (adults)	7	74,578	4,549	0.15%
Non-breeding (all ages)	2	1,287,037	175,037	0.001%
Annual total	9	-	-	0.151%
60% displacement and 5% mortality rate				
Breeding (adults)	46	74,578	4,549	1.01%
Non-breeding (all ages)	6	1,287,037	175,037	0.003%
Annual total	52	-	-	1.013%

The non-breeding season regional reference population for guillemot is 1,287,037 birds (Table 11-12). Applying the average mortality rate of 0.136 (Table 11-13), the estimated regional baseline mortality of guillemot is 175,037 birds in the breeding season (1,287,037 x 0.136). The additional predicted mortality of between two and six guillemots in the non-breeding season would increase the baseline mortality rate by between 0.001% and 0.003% (Table 11-23).

Predicted annual guillemot mortality due to displacement effects for adults in the breeding season and all ages in the non-breeding season was between nine and 52 birds, which corresponds to an increase in the annual baseline mortality rate of between 0.151% and 1.019% (Table 11-23).

A comparison of estimated guillemot mortality against a regional population consisting of adult and immature birds is shown in Table 11-24. Applying a displacement rate of 50% and a mortality rate of 1%, the additional predicted mortality due to displacement effects was 16 guillemots (all ages) in the breeding season. Based on a displacement rate of 60% and a mortality rate of 5%, the additional predicted mortality due to displacement effects was 96 guillemots. The total guillemot regional breeding population (all ages) is estimated to be 142,891 birds (Table 11-11.). The average mortality for all age classes is 0.136 (Table 11-13). Applying this mortality rate, the estimated regional baseline mortality of guillemots is 19,433 birds per breeding season (all ages) (142,891 x 0.136). The additional predicted mortality of between 16 and 96 guillemots in the breeding season would increase the baseline mortality rate by between 0.08% and 0.49% (Table 11-24).

Table 11-24 Increase in estimated baseline mortality for guillemots (all ages) in the OAA plus 2 km buffer as a result of displacement

Season	Predicted seasonal mortality	Regional baseline popn	Annual regional baseline mortality	Increase in baseline mortality
50% displacement and 1% mortality rate, all ages				
Breeding	16	142,891	19,433	0.08
Non-breeding	2	1,287,037	175,037	0.001%
Annual total	18	-	-	0.081%
60% displacement and 5% mortality rate, all ages				
Breeding	96	142,891	19,433	0.49
Non-breeding	6	1,287,037	175,037	0.003%
Annual total	102	-	-	0.493%

As before, predicted guillemot mortality from displacement effects in the non-breeding season was between two and six birds. Therefore, predicted annual guillemot mortality due to displacement effects for all ages was between 18 and 102 birds, which corresponds to an increase in the annual baseline mortality rate of between 0.081% and 0.493% (Table 11-24).

As highlighted by Natural England guidance, where predicted impacts equate to 1% or below of baseline mortality for a population (e.g. colony population) then this level of impact could be considered non-significant (Parker *et al.,* 2022c). Based on this, as the predicted increases in annual baseline mortality for guillemot were below 1% or very close to 1%, PVA was not carried out on the regional population.

Based on the results of the displacement assessment, the magnitude of impact from displacement on the regional guillemot population in the breeding and non-breeding seasons was considered to be **Negligible to Low**, as estimated increases in the annual baseline mortality rate ranged between less than 1% to 1.013% (Table 11-5).

For this assessment, receptor sensitivity has been based on three reviews of evidence from post-construction studies at offshore wind farms. A review of post-construction studies of seabirds at offshore wind farms in European waters concluded that guillemot was one of the species that weakly avoided offshore wind farms (Dierschke *et al.,* 2016). A review of vulnerability of Scottish seabirds to offshore wind turbines in the context of disturbance and displacement ranked guillemot as the 11th most sensitive out of 38 species (Furness *et al.,* 2013). Bradbury *et al.,* (2014), classified the guillemot population vulnerability to displacement from offshore wind farms in English waters as moderate.

Guillemots recorded within the OAA would qualify as internationally important in the breeding season (Offshore Ornithology Baseline Report), with individuals potentially originating from a number of SPA colonies within mean maximum foraging range, and also some non-SPA colonies. On this basis the conservation importance for guillemot was considered to be medium (Table 11-4).

Overall, based on available evidence from published studies, and the origin of birds from SPA and non-SPA colonies in the region, it is considered that guillemot sensitivity to displacement associated with the Offshore Site is likely to be **Medium**.

For guillemot, the magnitude of the impact is deemed to be **Negligible to Low** and the overall sensitivity of this species is considered to be **Medium.** The significance of any effect on guillemots from displacement and barrier effects associated with the Offshore Site is therefore at worst a **Slight Negative** effect, which is not significant (Table 11-6).

Razorbill

Based on the mean seasonal peak of razorbills in the OAA and 2 km buffer, and a displacement rate of 50% and a mortality rate of 1%, displacement mortality was predicted to be one bird in the breeding season. Based on a displacement rate of 60%, and a mortality rate of 3%, displacement mortality was predicted to be four birds in the breeding season, increasing to seven birds, if a mortality rate of 5% was applied. For the autumn migration period, razorbill displacement mortality was predicted to be zero birds for both 50% and 60% displacement rates and 1% and 3% mortality rates. For the winter period, based on a displacement rate of 50% and a mortality rate of 1%, displacement mortality was predicted to be one bird. Based on a displacement rate of 60%, and a mortality rate of 1%, displacement mortality was also predicted to be one bird, increasing to three birds, if a mortality rate of 3% was applied. For the spring migration period, based on a displacement rate of 50% and a mortality rate of 1%, displacement mortality was predicted to be zero birds. Based on a displacement rate of 60%, and a mortality rate of 1%, displacement mortality was also predicted to be zero birds, increasing to one bird, if a mortality rate of 3% was applied (Table 11-25). Further details and the seasonal displacement matrices are presented in the Displacement Assessment Appendix (Appendix 11-2).

However, this estimate includes non-breeding adults and immature birds, as well as breeding adults. Studies have shown that for several seabird species, in addition to breeding birds, colonies are also attended by many immature individuals and a smaller number of non-breeding adults (e.g. Wanless *et al.,* 1998). There is little information on the breakdown of immature and non-breeding adults present at a colony, however, this has been estimated using proportions from Horswill and Robinson (2015) (Offshore Ornithology Baseline Report), summarised in Table 11-13. Based on the proportion of adult razorbills from the population age ratio (0.533), 53.3% of the population present are adult birds, with a corresponding 46.7% of the population being immature birds. This means that between one and four razorbills displaced from the OAA and 2 km buffer during the breeding season would be adult birds, with between zero and three immature birds also displaced.

However, a proportion of adult birds present at colonies in the breeding season will opt not to breed in a particular breeding season. It has been estimated that 7% of adult razorbills may be "sabbatical" birds in any particular breeding season (Xodus, 2023), and this has been applied for this assessment. However, applying this to the small number of adult razorbills predicted to suffer mortality from displacement does not change the predicted number of breeding birds, therefore razorbill mortality was considered to involve between one and four breeding adults, zero non-breeding "sabbatical" adults and zero to three immature birds.

The breeding season regional reference population for razorbill is 9,417 breeding adults (Table 11-11). Applying the adult mortality rate of 0.105 (Table 11-13), the estimated regional baseline mortality of adult razorbills is 989 birds in the breeding season (9,417 x 0.105). The additional predicted mortality of between one and four adult razorbills in the breeding season would increase the baseline mortality rate by between 0.10% and 0.40% (Table 11-25).

For the autumn migration period of the non-breeding season, estimated seasonal razorbill mortality was predicted to be zero birds based on both 50% and 60% displacement rates and 1% and 3% mortality rates (Table 11-25).

For the winter period of the non-breeding season, estimated seasonal razorbill mortality was predicted to be between one and three birds (Table 11-25). The total razorbill regional population for the winter period is estimated to be 365,711 birds (Table 11-12). The increase in baseline mortality was calculated based on an estimated average mortality rate of 0.129 (Table 11-13). Applying this mortality rate, the estimated regional baseline mortality of razorbills is 47,177 birds in the winter period (365,711 x 0.129). The additional predicted mortality of between one and three razorbills in the winter period would increase the baseline mortality rate by between 0.002% and 0.006% (Table 11-25).

For the spring period of the non-breeding season, estimated seasonal razorbill mortality was predicted to be between zero and one bird (Table 11-25). The total razorbill regional population for the spring migration period is estimated to be 631,203 birds (Table 11-12). The increase in baseline mortality was calculated based on an estimated average mortality rate of 0.129 (Table 11-13). Applying this mortality rate, the estimated regional baseline mortality of razorbills is 81,425 birds in the spring migration period (631,203 x 0.129). The additional predicted mortality of between zero and one razorbills in the spring migration period would increase the baseline mortality rate by between 0% and 0.001% (Table 11-25).

Table 11-25 Increase in estimated baseline mortality for razorbills (adults only in breeding season) in the OAA plus 2 km buffer as a result of displacement

Season	Predicted seasonal mortality	Regional baseline popn	Annual regional baseline mortality	Increase in baseline mortality
50% displacement and 1% mortality rate				
Breeding (adults)	1	9,417	989	0.10%
Autumn migration (all ages)	0	-	-	0%
Winter period (all ages)	1	365,711	47,177	0.002%
Spring migration (all ages)	0	-	-	0%
Annual total	2	-	-	0.102%
60% displacement and 5% mortality rate				
Breeding (adults)	4	9,417	989	0.40%
Autumn migration (all ages)	0	-	-	0%
Winter period (all ages)	3	365,711	47,177	0.006%
Spring migration (all ages)	1	631,203	81,425	0.001%
Annual total	8	-	-	0.41%

Predicted annual razorbill mortality due to displacement effects for adults in the breeding season and all ages in the autumn migration, winter and spring migration periods of the non-breeding season was between two and eight birds, which corresponds to an increase in the annual baseline mortality rate of between 0.102% and 0.41% (Table 11-25).

A comparison of estimated razorbill mortality against a regional population consisting of adult and immature birds is shown in Table 11-26. Applying a displacement rate of 50% and a mortality rate of 1%, the additional predicted mortality due to displacement effects was one razorbill (all ages) in the breeding season. Based on a displacement rate of 60% and a mortality rate of 5%, the additional predicted mortality due to displacement effects was seven razorbills. The total razorbill regional breeding population (all ages) is estimated to be 17,666 birds (Table 11-11). The average mortality for all age classes is 0.129 (Table 11-13). Applying this mortality rate, the estimated regional baseline mortality of razorbills is 2,279 birds per breeding season (all ages) (17,666 x 0.129). The additional predicted mortality of between one and seven razorbills in the breeding season would increase the baseline mortality rate by between 0.04% and 0.31% (Table 11-26).

Table 11-26 Increase in estimated baseline mortality for razorbills (all ages) in the OAA plus 2 km buffer as a result of displacement

Season	Predicted seasonal mortality	Regional baseline popn	Annual regional baseline mortality	Increase in baseline mortality
50% displacement and 1% mortality rate, all ages				
Breeding	1	17,666	2,279	0.04%
Autumn migration	0	-	-	0%
Winter period	1	365,711	47,177	0.002%
Spring migration	0	-	-	0%
Annual total	2	-	-	0.042%
60% displacement and 5% mortality rate, all ages				
Breeding	7	17,666	2,279	0.31%
Autumn migration	0	-	-	0%
Winter period	3	365,711	47,177	0.006%
Spring migration	1	631,203	81,425	0.001%
Annual total	11	-	-	0.323%

As before, predicted razorbill mortality from displacement effects in the autumn migration, winter and spring migration periods of the non-breeding season was between one and four birds. Therefore, predicted annual razorbill mortality due to displacement effects for all ages was between two and 11 birds, which corresponds to an increase in the annual baseline mortality rate of between 0.042% and 0.323% (Table 11-26).

As highlighted by Natural England guidance, where predicted impacts equate to 1% or below of baseline mortality for a population (e.g. colony population) then this level of impact could be considered non-significant (Parker *et al.,* 2022c). Based on this, as the predicted increases in annual baseline mortality for razorbill were below 1%, PVA was not carried out on the regional population.

Based on the results of the displacement assessment, the magnitude of impact from displacement on the regional razorbill population in the breeding and non-breeding seasons was considered to be **Negligible**, as the predicted increases in the annual baseline mortality rate were less than 1% (Table 11-5).

For this assessment, receptor sensitivity has been based on three reviews of evidence from post-construction studies at offshore wind farms. A review of post-construction studies of seabirds at offshore wind farms in European waters concluded that razorbill was one of the species that weakly avoided offshore wind farms (Dierschke *et al.,* 2016). A review of vulnerability of Scottish seabirds to offshore wind turbines in the context of disturbance and displacement ranked razorbill as the 12th most sensitive out of 38 species (Furness *et al.,* 2013). Bradbury *et al.,* (2014), classified the razorbill population vulnerability to displacement from offshore wind farms in English waters as moderate.

Razorbills recorded within the OAA would qualify as internationally important in the breeding season (Offshore Ornithology Baseline Report), with individuals potentially originating from a number of SPA colonies within mean maximum foraging range, and also some non-SPA colonies. On this basis the conservation importance for razorbill was considered to be medium (Table 11-4).

Overall, based on the conservation importance, with SPAs for breeding razorbills within mean maximum foraging range of the OAA, together with evidence from reviews and post-construction studies presented above indicates that razorbill sensitivity to displacement associated with the Offshore Site is likely to be **Medium**.

For razorbill, the magnitude of the impact is deemed to be **Negligible** and the overall sensitivity of this species is considered to be **Medium.** Any effect is therefore considered **Not Significant** (Table 11-6).

Puffin

Based on the mean seasonal peak of puffins in the OAA and 2 km buffer, and a displacement rate of 50% and a mortality rate of 1%, displacement mortality was predicted to be zero birds in the breeding season. Based on a displacement rate of 60%, and a mortality rate of 3%, displacement mortality was predicted to be one puffin in the breeding season, increasing to two puffins, if a mortality rate of 5% was applied.

For the non-breeding season in the OAA and 2 km buffer, and a displacement rate of 50% and a mortality rate of 1%, displacement mortality was predicted to be zero birds. Based on a displacement rate of 60%, and mortality rates of 1% and 3%, displacement mortality was also predicted to be zero birds (Table 11-27). Further details and the seasonal displacement matrices are presented in the Displacement Assessment Appendix (Appendix 11-2).

The breeding season regional reference population for puffin is 26,264 breeding adults (Table 11-6). Applying the adult mortality rate of 0.094 (Table 11-13), the estimated regional baseline mortality of puffin is 2,469 birds in the breeding season (26,264 x 0.094). Assuming that all affected puffins were adult birds, then the additional predicted mortality of between one and two puffins in the breeding season would increase the baseline mortality rate by between 0.04% and 0.081% (Table 11-27).

Table 11-27 Increase in estimated baseline mortality for puffins (adults only in breeding season) in the OAA plus 2 km buffer as a result of displacement

Season	Predicted seasonal mortality	Regional baseline popn	Annual regional baseline mortality	Increase in baseline mortality
50% displacement and 1% mortality rate				
Breeding (adults)	0	-	-	0%
Non-breeding (all ages)	0	-	-	0%
Annual total	0	-	-	0%
60% displacement and 3% mortality rate				
Breeding (adults)	1	26,264	2,469	0.04
Non-breeding (all ages)	0	-	-	0%
Annual total	1	-	-	0.04%
60% displacement and 5% mortality rate				
Breeding (adults)	2	26,264	2,469	0.081
Non-breeding (all ages)	0	-	-	0%
Annual total	2	-	-	0.081%

For the non-breeding season, estimated seasonal puffin mortality was predicted to be zero birds, therefore predicted annual puffin mortality due to displacement effects for adults in the breeding season and all ages in the non-breeding season was between one and two birds, which corresponds to an increase in the annual baseline mortality rate of between 0.04% and 0.081% (Table 11-27).

As highlighted by Natural England guidance, where predicted impacts equate to 1% or below of baseline mortality for a population (e.g. colony population) then this level of impact could be considered non-significant (Parker *et al.,* 2022c). Based on this, as the predicted increases in annual baseline mortality for puffin were below 1%, PVA was not carried out on the regional population.

Based on the results of the displacement assessment, the magnitude of impact from displacement on the regional puffin population in the breeding and non-breeding seasons was considered to be **Negligible**, as the predicted increases in the annual baseline mortality rate were less than 1% (Table 11-5).

For this assessment, receptor sensitivity has been based on reviews of evidence from post-construction studies at offshore wind farms. A review of vulnerability of Scottish seabirds to offshore wind turbines in the context of disturbance and displacement ranked puffin as the 17th most sensitive out of 38 species (Furness *et al.,* 2013). Bradbury *et al.,* (2014), classified the puffin population vulnerability to displacement from offshore wind farms in English waters as low.

Puffins recorded within the OAA would qualify as internationally important in the breeding season (Offshore Ornithology Baseline Report), with individuals potentially originating from a number of SPA colonies within mean maximum foraging range, and also some non-SPA colonies. On this basis the conservation importance for puffin was considered to be medium (Table 11-4).

Overall, based on the conservation importance, with SPAs for breeding puffins within mean maximum foraging range of the OAA, together with evidence from reviews and post-construction studies presented above indicates that puffin sensitivity to displacement associated with the Offshore Site is likely to be **Medium**.

For puffin, the magnitude of the impact is deemed to be **Negligible** and the overall sensitivity of this species is considered to be **Medium.** Any effect is therefore considered **Not Significant** (Table 11-6).

11.8.6.4 Impact 7 – Collision Mortality within the OAA

Impact 7 considers the potential for mortality of key bird species as a result of collision with offshore wind turbines. There is potential risk to birds arising from collision with operating turbines resulting in injury or fatality. This may occur when birds fly through an OWF whilst foraging for food, commuting between breeding colonies and foraging areas, or during migration.

CRM has been undertaken for the Offshore Site. The approach used for CRM for this assessment was based on the NatureScot guidance (NatureScot, 2023a), with reference to the East Coast Phase 1 Projects method statement (GoBe, 2022) and the review of that method statement (ABPmer, 2023). The Avian Stochastic CRM (Caneco, 2022) was used for the CRM, following the guidance in Donovan (2017). Full detailed methods and results are presented in the Seabird CRM Technical Report.

The project design elements outlined in Table 11-14, describe the turbine scenario considered within this assessment. Further details are presented in the Seabird CRM Technical Report.

An initial screening of all seabird species recorded on baseline surveys was conducted, based on the species sensitivity to collision effects, the densities of flying birds recorded on baseline surveys and whether the species are qualifying features of nearby seabird colonies (Table 11-28). Species that were ranked Very High or High sensitivity were scoped into the CRM assessment. In addition, common and Arctic tern were also included in the CRM assessment, as there are SPA breeding colonies in the vicinity for these species.

Table 11-28 Initial screening of seabirds for CRM assessment

Species	Sensitivity to collision impacts[1]	Summary of density of birds in flight in OAA	Screened IN or OUT
Red-throated Diver	Moderate	Zero	OUT
Great Northern Diver	Moderate	Low	OUT
Fulmar	Very Low	Low	OUT
Manx Shearwater	Very Low	High	OUT
Storm Petrel	Low	Low	OUT
Gannet	High	Low	IN
Cormorant	Low	Low	OUT
Shag	Moderate	Low	OUT
Eider	Low	Low	OUT

Species	Sensitivity to collision impacts[1]	Summary of density of birds in flight in OAA	Screened IN or OUT
Common Gull[1]	High	Low	IN
Lesser black-backed Gull	Very High	Low	IN
Herring Gull	Very High	Low	IN
Great black-backed Gull	Very High	Low	IN
Kittiwake	High	Moderate	IN
Common Tern	Moderate	Low	IN
Arctic Tern	Moderate	Low	IN
Guillemot	Very Low	High	OUT
Razorbill	Very Low	Moderate	OUT
Black Guillemot	Very Low	Low	OUT
Puffin	Very Low	Low	OUT

1 Based on Furness *et al.,* 2013 and Bradbury *et al.,* 2014

Following the initial screening exercise, CRM was undertaken on the following eight seabird species:

- Gannet
- Kittiwake
- Common gull
- Great black-backed gull
- Herring gull
- Lesser black-backed gull
- Common tern, and
- Arctic tern

Details of all turbine parameters and species parameters used in the CRM are presented in the Seabird CRM Technical Report (Appendix 11-3).

A key parameter in the CRM is the species-specific avoidance rate, which accounts for the fact that birds will take action to avoid colliding with the rotors (at a range of scales, from the whole wind farm to individual turbine blades). This adjustment is required in the model since baseline survey data are collected before turbines are present and hence do not contain any avoidance behaviour. The avoidance rates used in this assessment for each species have been derived from NatureScot (NatureScot, 2023).

Annual collision estimates for the proposed 30 turbines and Band Option 2 for the key species considered in the CRM assessment are summarised in Table 11-29. Band Option 2 uses generic flight height distributions, rather than site-specific data to calculate the proportion of flight activity at collision risk height in the calculation of predicted transits, and is the recommended flight height option by the UK SNCBs (e.g. NatureScot, 2023, JNCC, *et al.,* 2024). Further details are presented in the Seabird CRM Technical Report.

Table 11-29 Estimated annual number of collisions in the OAA

Species	Estimated annual collisions (plus lower and upper 95% confidence limits)
Gannet	0.8 (0.1 – 1.9)
Kittiwake	8.2 (4.1 – 14.0)
Common gull	0.3 (0.0 – 1.6)
Great black-backed gull	6.1 (1.5 – 13.0)
Herring gull	4.5 (0.0 – 13.6)
Lesser black-backed gull	3.1 (0.0 – 7.8)
Common tern	0.4 (0.0 – 1.3)
Arctic tern	0.2 (0.0 – 1.3)

The CRM assessments are presented for each species below.

11.8.6.4.1 Gannet

Estimated number of gannet collisions are presented in Table 11-30. Figures are presented for the breeding season and the autumn and spring migration periods of the non-breeding season, based on the proposed Wind Turbine Generator (WTG) layout. Predicted collision numbers were very low for the breeding and non-breeding seasons, with a total of less than one collision annually.

Table 11-30 Estimated numbers of gannet collisions by season in the OAA

Season	Mean number of collisions	Lower 95% confidence limit	Upper 95% confidence limit
Breeding (Mar-Sep)	0.7	0.1	1.9
Autumn migration (Oct-Nov)	0	0	0
Spring migration (Dec-Feb)	0.03	0.0	0.18
Annual collisions	0.8	0.1	1.09
Total (to nearest whole bird)	1	0	2

Annual total taken from Table 11-29. Note totals based on seasonal breakdown may differ slightly due to rounding.

In the breeding season (March to September), the total mean estimated number of gannet collisions was 0.7 bird (Table 11-30). As this number is very small, it was considered that there was no requirement to take account of non-breeding adults and immature birds therefore, breeding season gannet collision mortality was considered to involve 0.7 breeding, adult birds.

The total gannet regional breeding population is estimated to be 93,602 adult birds (Table 11-6). For the breeding season assessment based on adult birds only, the increase in baseline mortality was calculated based on an estimated adult gannet baseline survival rate of 0.919, therefore the corresponding rate for adult gannet mortality is 0.081 (Table 11-13). Applying this mortality rate, the estimated regional baseline mortality of gannets is 7,582 adult birds per breeding season (93,602 x 0.081). The additional predicted mortality of 0.7 breeding adult gannets in the breeding season would increase the baseline mortality rate by 0.01% (Table 11-31).

For the autumn and spring migration periods, estimated seasonal gannet mortality from collision was 0.03 birds, which was rounded to zero birds (Table 11-30). Overall, predicted annual gannet mortality due to collision effects involved 0.8 gannets, which corresponds to an increase in the annual baseline mortality rate of 0.01% (Table 11-31).

Table 11-31 Increase in estimated baseline mortality for adult gannets in the OAA as a result of collision

Season	Regional baseline population	Annual Regional Baseline Mortality	Increase in baseline mortality (%)
Breeding (Mar-Sep) (Adults only)	93,602	7,582	0.01
Autumn migration (Oct-Nov) (All ages)	662,102	-	0
Spring migration (Dec-Feb) (All ages)	770,836	-	0
Total	-	-	0.01

As highlighted by Natural England guidance, where predicted impacts equate to 1% or below of baseline mortality for a population (e.g. colony population) then this level of impact could be considered non-significant (Parker *et al.,* 2022c). Based on this, as the predicted increase in annual baseline mortality for gannet was below 1%, PVA was not carried out on the regional population.

Based on the results of the collision assessment, the magnitude of impact from collision effects on the regional gannet population was considered to be **Negligible**, as the estimated increase in the annual baseline mortality rate was below 1% (Table 11-5).

For gannet, there is evidence that gannets show a high degree of avoidance of offshore wind farms. A detailed study (Krijgsveld *et al.,* 2011) using radar and visual observations to monitor the post-construction effects of the Windpark Egmond aan Zee OWEZ established that 64% of gannets avoided entering the wind farm (macro-avoidance) and a similar result (80% macro avoidance) was also observed during a study at the Thanet wind farm (Skov *et al.,* 2018). Leopold *et al.,* (2013) reported that most gannets avoided Dutch offshore wind farms and did not forage within them.

In addition, post-construction studies for the Beatrice offshore wind farm in the Moray Firth, Scotland reported that gannet showed a marked difference in distribution within the wind farm on post-construction surveys than on pre-construction surveys, with only two birds recorded within the wind farm boundary across all post-construction surveys undertaken in Year 1. Spatial modelling indicated a significant decrease centred on the wind farm and extending towards the coast with no areas of significant increase. Beyond the region of decrease, the density in the remainder of the survey area was almost identical when comparing pre- and post-construction data (MacArthur Green, 2021). The Beatrice Year 2 post-construction analysis concluded that virtually no gannets were recorded within the wind farm, and that the current collision avoidance rate used in assessments may well be an underestimate of the level of avoidance this species exhibits (MacArthur Green, 2023).

Gannets recorded within the OAA on baseline surveys would qualify as internationally important in the breeding season (Offshore Ornithology Baseline Report), with individuals potentially originating from a number of SPAs in the region, as well as non-SPA colonies. On this basis the conservation importance for gannet was considered to be medium (Table 11-4).

Overall, based on available evidence from published studies indicating high levels of wind farm avoidance, and the origin of birds from SPA and non-SPA colonies in the region, it is considered that gannet sensitivity to collision effects associated with the Offshore Site is likely to be **Medium**.

For gannet, the magnitude of the impact is deemed to be **Negligible,** and the overall sensitivity of this species is considered to be **Medium**. Any effect on gannets from collision effects associated with the Offshore Site is therefore considered **Not Significant** (Table 11-6).

Gannet displacement and collision effects combined

The SNCBs guidance on displacement assessments (SNCBs, 2022a) states that collision and displacement impacts should be combined for species that are considered likely to be affected by both displacement and collision effects. The guidance does acknowledge that this approach includes a degree of precaution, as there is the potential for double-counting. As highlighted by NatureScot in the NnG OWF Scoping Opinion (Marine Scotland, 2017), collision risk and displacement are considered to be mutually exclusive impacts, and therefore combining mortality estimates for displacement and collision should be considered extremely precautionary.

Results from the collision and displacement assessments for gannet were combined, using the annual predicted mortality totals (Table 11-32).

Table 11-32 Combined annual estimated mortality for gannet (all ages) as a result of displacement and collisions

	Combined estimated mortality	Increase in baseline mortality (%) (breeding adults)	Increase in baseline mortality (all ages)
Annual displacement	2	0.0068	0.0068
Annual collisions	0.8	0.01	0.01
Combined total	2.8	0.017	0.017

Combined estimated annual gannet mortality due to collision and displacement effects showed a maximum increase in the annual baseline mortality rate of 0.017%.

The magnitude of impact from combined displacement and collision effects on the regional gannet population was considered to be **Negligible**, as the estimated increases in the annual baseline mortality rate were below 1% (Table 11-5).

For gannet, the magnitude of the impact is deemed to be **Negligible,** and the overall sensitivity of this species is considered to be **Medium**, with individuals potentially originating from a number of SPAs in the region, as well as non-SPA colonies. Any effect on gannets from combined displacement and collision effects associated with the Offshore Site is therefore considered **Not Significant** (Table 11-6).

11.8.6.4.2 Kittiwake

Estimated number of kittiwake collisions are presented in Table 11-33. Figures are presented for the breeding season and the autumn and spring migration periods of the non-breeding season, based on the proposed WTG layout. Highest numbers of collisions were predicted for the breeding season, with lower collisions predicted for the autumn and spring migration periods of the non-breeding season.

Table 11-33 Estimated numbers of kittiwake collisions by season in the OAA

Season	Mean number of collisions	Lower 95% confidence limit	Upper 95% confidence limit
Breeding (Mar-Aug)	4.4 birds (3.7 breeding adults)	2.0	7.9
Autumn migration (Sep-Dec)	2.8	0.8	6.1
Spring migration (Jan-Feb)	1.0	0.2	2.3
Annual collisions	7.5	4.1	14.0
Total (to nearest whole bird)	8	3	16

Annual total taken from Table 11-29. Note totals based on seasonal breakdown may differ slightly due to rounding.

In the breeding season (March to August), the total mean estimated number of kittiwake collisions was 4.44 birds (Table 11-33). However, this includes non-breeding adults and immature birds, as well as breeding adults. Based on the proportion of immature kittiwakes recorded on baseline surveys in the breeding season (see Offshore Ornithology Baseline Report), it was assumed that 91.95% of the population present are adult birds. This would mean that an estimated 4.1 kittiwakes predicted to collide during the breeding season would be adult birds.

Similarly, a proportion of adult birds present at colonies in the breeding season will opt not to breed in a particular breeding season. It has been estimated that 10% of adult kittiwakes may be “sabbatical” birds in any particular breeding season (Xodus, 2023), and this has been applied for this assessment. On this basis, 0.41 adult kittiwakes predicted to collide were considered not to be breeding, with 3.7 birds considered to be breeding adult kittiwakes.

The total kittiwake regional breeding population is estimated to be 26,720 adult birds (Table 11-6). For the breeding season assessment based on breeding adult birds only, the increase in baseline mortality was calculated based on an estimated adult kittiwake baseline survival rate of 0.854, therefore the corresponding rate for adult mortality is 0.146 (Table 11-13). Applying this mortality rate, the estimated regional baseline mortality of kittiwakes is 3,901 adult birds per breeding season (26,720 x 0.146). The additional predicted mortality of 3.7 breeding adult kittiwakes in the breeding season would increase the baseline mortality rate by 0.095% (Table 11-34).

For the autumn migration period, estimated seasonal kittiwake collision mortality was 2.8 birds, (Table 11-33). The kittiwake regional population for the autumn migration period is estimated to be 945,743 birds (Table 11-12). The increase in baseline mortality was calculated based on an average kittiwake baseline mortality rate of 0.156 (Table 11-13). Applying this mortality rate, the estimated regional baseline mortality of kittiwakes is 147,536 birds for the autumn migration period (945,743 x 0.156). The additional predicted mortality of 2.8 kittiwakes in the autumn migration period of the non-breeding season would increase the baseline mortality rate by 0.002% (Table 11-34).

For the spring migration period, estimated seasonal kittiwake collision mortality was 1.0 birds (Table 11-33). The kittiwake regional population for the spring migration period is estimated to be 725,683 birds (Table 11-12). The increase in baseline mortality was calculated based on an average kittiwake baseline mortality rate of 0.156 (Table 11-13). Applying this mortality rate, the estimated regional baseline mortality of kittiwakes is 113,207 birds for the spring migration period (725,683 x 0.156). The additional predicted mortality of 1.0 kittiwakes in the spring migration period of the non-breeding season would increase the baseline mortality rate by 0.001% (Table 11-34).

Table 11-34 Increase in estimated baseline mortality for kittiwakes in the OAA as a result of collision

Season	Regional baseline population	Annual Regional Baseline Mortality	Increase in baseline mortality (%)
Breeding (Mar-Aug) (Adults only)	26,720	3,901	0.095
Autumn migration (Sep-Dec) (All ages)	945,743	147,536	0.002
Spring migration (Jan-Feb) (All ages)	725,683	113,207	0.001
Total	-	-	0.098

Predicted annual kittiwake mortality due to collision effects based on adult birds in the breeding season and all ages in the autumn and spring migration periods of the non-breeding season, involved 7.5 birds, which corresponds to an increase in the annual baseline mortality rate of 0.098% (Table 11-34).

A comparison of estimated kittiwake collision mortality against a regional population consisting of adult and immature birds is shown in Table 11-35. The predicted additional mortality due to collision effects was 4.4 kittiwakes (all ages) in the breeding season (Table 11-33). The total kittiwake regional breeding population (all ages) is estimated to be 50,715 birds (Table 11-11). The average mortality for all age classes is 0.156 (Table 11-13). Applying this mortality rate, the estimated regional baseline mortality of kittiwakes is 7,912 birds per breeding season (all ages) (50,715 x 0.156). The additional predicted mortality of 4.4 kittiwakes in the breeding season would increase the baseline mortality rate by 0.056% (Table 11-35).

As above, the additional predicted mortality of 2.8 kittiwakes in the autumn migration period of the non-breeding season would increase the baseline mortality rate by 0.002%. The additional predicted mortality of 1.0 kittiwakes in the spring migration period of the non-breeding season would increase the baseline mortality rate by 0.001% (Table 11-35).

Predicted annual kittiwake mortality due to collision effects based on all ages in the breeding season and the autumn and spring migration periods of the non-breeding season, involved 8.2 birds, which corresponds to an increase in the annual baseline mortality rate of 0.063% (Table 11-35).

Table 11-35 Increase in estimated baseline mortality for kittiwakes in the OAA as a result of collision

Season (All ages)	Regional baseline population	Annual Regional Baseline Mortality	Increase in baseline mortality (%)
Breeding (Mar-Aug)	50,715	7,912	0.056
Autumn migration (Sep-Dec)	945,743	147,536	0.002
Spring migration (Jan-Feb)	725,683	113,207	0.001

Season (All ages)	Regional baseline population	Annual Regional Baseline Mortality	Increase in baseline mortality (%)
Total	-	-	0.059

As highlighted by Natural England guidance, where predicted impacts equate to 1% or below of baseline mortality for a population (e.g. colony population) then this level of impact could be considered non-significant (Parker *et al.,* 2022c). Based on this, as the predicted increases in annual baseline mortality for kittiwake were below 1%, PVA was not carried out on the regional population.

Based on the results of the collision assessment, the magnitude of impact from collision effects on the regional kittiwake population was considered to be **Negligible**, as the estimated increases in the annual baseline mortality rate were below 1% (Table 11-5).

For this assessment, receptor sensitivity has been based on three reviews of evidence from post-construction studies at offshore wind farms. A review of post-construction studies of seabirds at offshore wind farms in European waters concluded that kittiwake was one of the species that was hardly affected by offshore wind farms (Dierschke *et al.,* 2016). A review of vulnerability of Scottish seabirds to offshore wind turbines in the context of collision ranked kittiwake as the seventh most sensitive out of 38 species (Furness *et al.,* 2013). Bradbury *et al.,* (2014) classified the kittiwake population vulnerability to collision mortality from offshore wind farms as high.

Kittiwakes recorded within the OAA would qualify as internationally important in the breeding season (Offshore Ornithology Baseline Report), with individuals potentially originating from a number of SPAs in the region, as well as non-SPA colonies. On this basis the conservation importance for kittiwake was considered to be medium (Table 11-4).

Overall, based on available evidence from published studies indicating a high sensitivity to collision, and the origin of birds from SPA and non-SPA colonies in the region, it is considered that kittiwake sensitivity to collision effects associated with the Offshore Site is likely to be **High**.

For kittiwake, the magnitude of the impact is deemed to be **Negligible,** and the overall sensitivity of this species is considered to be **High**. Any effect on kittiwakes from collision effects associated with the Offshore Site is therefore considered **Not Significant** (Table 11-6).

Kittiwake displacement and collision effects combined

The SNCBs guidance on displacement assessments (SNCBs, 2022a) states that collision and displacement impacts should be combined for species that are considered likely to be affected by both displacement and collision effects. The guidance does acknowledge that this approach includes a degree of precaution, as there is the potential for double-counting. As highlighted by NatureScot in the NnG OWF Scoping Opinion (Marine Scotland, 2017), collision risk and displacement are considered to be mutually exclusive impacts, and therefore combining mortality estimates for displacement and collision should be considered extremely precautionary.

Results from the collision and displacement assessments for kittiwake were combined, using the annual predicted mortality totals (Table 11-36).

Table 11-36 Combined annual estimated mortality for kittiwake as a result of displacement and collisions

	Combined estimated mortality	Increase in baseline mortality (%) (breeding adults)	Increase in baseline mortality (all ages)
Annual displacement	3	0.02	0.02
Annual collisions	8.2	0.098	0.059
Combined total	11.2	0.12	0.08

Combined estimated annual kittiwake mortality due to collision and displacement effects showed a maximum increase in the annual baseline mortality rate of 0.12%.

The magnitude of impact from combined displacement and collision effects on the regional kittiwake population was considered to be **Negligible**, as the estimated increases in the annual baseline mortality rate were below 1% (Table 11-5).

For kittiwake, the magnitude of the impact is deemed to be **Negligible.** The sensitivity of this species to collision is considered to be high, while sensitivity to displacement is considered to be very low (e.g. Btadbury *et al.*, 2014). The conservation importance of kittiwake is considered to be medium, with individuals potentially originating from a number of SPAs in the region, as well as non-SPA colonies. Overall, the sensitivity of kittiwake to combined displacement and collision effects is considered to be medium, bearing in mind that combining mortality estimates for displacement and collision for assessment purposes should be considered extremely precautionary. Any effect on kittiwakes from combined displacement and collision effects associated with the Offshore Site is therefore considered **Not Significant** (Table 11-6).

11.8.6.4.3 Common Gull

Estimated number of common gull collisions are presented in Table 11-37. Figures are presented for the breeding and non-breeding seasons, based on the proposed WTG layout. No common gull collisions were predicted in the breeding season and collision numbers were very low for the non-breeding season, with a total of less than one collision annually.

Table 11-37 Estimated numbers of common gull collisions by season in the OAA

Season	Mean number of collisions	Lower 95% confidence limit	Upper 95% confidence limit
Breeding (Apr-Aug)	0	0	0
Non-breeding (Sep-Mar)	0.3	0	1.6
Annual collisions	0.3	0	1.6
Total (to nearest whole bird)	0	0	2

As the predicted number of common gull collisions was very low, the assessment has been carried out on all ages and it has been assumed that there are no sabbatical (non-breeding) birds involved.

In the breeding season (April to August), the mean estimated number of common gull collisions was zero birds (Table 11-37). Therefore, there was no predicted increase to common gull baseline mortality in the breeding season.

For the non-breeding season, estimated seasonal common gull collision mortality was 0.3 birds (Table 11-37).

The mean RoI wintering population of common gulls is estimated to be 21,438 individuals (Lewis, *et al.,* 2019). Based on 5-year mean I-WebS counts between 2011/12 and 2015/16 winters for regularly counted sites between Galway and Donegal, a combined population of 8,465 common gulls was estimated for these sites. Further details are presented in the Offshore Ornithology Baseline Report.

Baseline surveys in the wider 10 km survey area for the Offshore Site project recorded peak estimates of 686 common gulls in November 2021 and 211 birds in January 2023 (Aerial Survey Two Year Report). The mean of these two winter counts is 449 birds. If this figure is added to the combined 5-year mean I-WeBS count of 8,465 birds, then this would give an estimated non-breeding season population of 8,914 common gulls. In the absence of a more complete regional population estimate for the non-breeding season, this estimate has been used as the regional reference population for this assessment. It should be noted that this is likely to be a minimum estimate, as gulls including common gull are not priority count species for I-WeBS and so are not counted at every site (Crowe, 2005). In addition, data for sites holding fewer than 500 common gulls was not presented in Lewis *et al.,* 2019.

For this assessment, the common gull regional population for the non-breeding season is therefore estimated to be 8,914 birds (Table 11-12). The increase in baseline mortality was calculated based on an average common gull baseline mortality rate of 0.253 (Table 11-13). Applying this mortality rate, the estimated regional baseline mortality of common gull is 2,255 birds for the non-breeding season (8,914 x 0.253). The additional predicted mortality of 0.3 common gulls in the non-breeding season would increase the baseline mortality rate by 0.013% (Table 11-38).

Predicted annual common gull mortality due to collision effects based on all ages in the breeding and non-breeding seasons involved 0.3 birds, which corresponds to an increase in the annual baseline mortality rate of 0.013% (Table 11-38).

Table 11-38 Increase in estimated baseline mortality for common gulls in the OAA as a result of collision

Season (All ages)	Regional baseline population (all ages)	Annual Regional Baseline Mortality	Increase in baseline mortality (%)
Breeding (Apr-Aug)	874	221	0
Non-breeding (Sep-Mar)	8,914	2,255	0.013
Total	-	-	0.013

As highlighted by Natural England guidance, where predicted impacts equate to 1% or below of baseline mortality for a population (e.g. colony population) then this level of impact could be considered non-significant (Parker *et al.,* 2022c). Based on this, as the predicted increase in annual baseline mortality for common gull was below 1%, PVA was not carried out on the regional population.

Based on the results of the collision assessment, the magnitude of impact from collision effects on the regional common gull population was considered to be **Negligible**, as the estimated increase in the annual baseline mortality rate were less than 1% (Table 11-5).

For this assessment, receptor sensitivity has been based on three reviews of evidence from post-construction studies at offshore wind farms. A review of post-construction studies of seabirds at offshore wind farms in European waters concluded that common gull was one of the species that was weakly attracted to offshore wind farms (Dierschke *et al.,* 2016). A review of vulnerability of Scottish seabirds to offshore wind turbines in the context of collision ranked common gull as the sixth most sensitive out of 38 species (Furness *et al.,* 2013). Bradbury *et al.,* (2014), classified the common gull population vulnerability to collision mortality from offshore wind farms as high.

Common gulls recorded within the OAA would qualify as internationally important in the breeding season (Offshore Ornithology Baseline Report), with individuals potentially originating from two SPAs in the region, as well as non-SPA colonies. On this basis the conservation importance for common gull was considered to be medium (Table 11-4).

Overall, based on available evidence from published studies indicating a high sensitivity to collision, and the medium conservation importance in the breeding season, it is considered that common gull sensitivity to collision effects associated with the Offshore Site is likely to be **High**.

For common gull, the magnitude of the impact is deemed to be **Negligible,** and the overall sensitivity of this species is considered to be **High**. Any effect on common gull from collision effects associated with the Offshore Site is therefore considered **Not Significant** (Table 11-6).

11.8.6.4.4 Great black-backed Gull

Estimated number of great black-backed gull collisions are presented in Table 11-39. Figures are presented for the breeding and non-breeding seasons, based on the proposed WTG layout. Predicted numbers of collisions were slightly higher for the non-breeding season, compared to the breeding season.

Table 11-39 Estimated numbers of great black-backed gull collisions by season in the OAA

Season	Mean number of collisions	Lower 95% confidence limit	Upper 95% confidence limit
Breeding (Mar-Aug)	2.4 (1.6 adults)	0	7.7
Non-breeding (Sep-Feb)	3.7	0	10.5
Annual collisions	6.1	1.5	13.0
Total (to nearest whole bird)	6	2	13

Annual total taken from Table 11-29. Note totals based on seasonal breakdown may differ slightly due to rounding.

In the breeding season (March to August), the total mean estimated number of great black-backed gull collisions was 2.4 birds (Table 11-39). However, this includes non-breeding adults and immature birds, as well as breeding adults. As all aged great black-backed gulls recorded on baseline surveys in the breeding season were adults, (see Offshore Ornithology Baseline Report), it was assumed that 100% of the population present are adult birds, therefore breeding season great black-backed gull collision mortality was considered to involve 2.4 adult birds.

A proportion of adult birds present at colonies in the breeding season will opt not to breed in a particular breeding season. It has been estimated that 35% of adult great black-backed gulls may be “sabbatical” birds in any particular breeding season (Xodus, 2023), and this has been applied for this assessment. On this basis, 0.8 adult great black-backed gulls predicted to collide were considered not to be breeding. Therefore, great black-backed gull collision mortality in the breeding season was considered to be 1.6 adult breeding birds.

The total great black-backed gull regional breeding population is estimated to be 1,410 adult birds (Table 11-11). For the breeding season assessment based on breeding adult birds only, the increase in baseline mortality was calculated based on an estimated adult great black-backed gull baseline survival rate of 0.930, therefore the corresponding rate for adult mortality is 0.07 (Table 11-13). Applying this mortality rate, the estimated regional baseline mortality of great black-backed gulls is 99 adult birds per breeding season (1,410 x 0.07). The additional predicted mortality of 1.6 breeding adult great black-backed gulls in the breeding season would increase the baseline mortality rate by 1.62% (Table 11-40).

For the non-breeding season, estimated seasonal great black-backed gull collision mortality was 3.7 birds (Table 11-39). The great black-backed gull regional population for the non-breeding season is estimated to be 42,708 birds (Table 11-12). The increase in baseline mortality was calculated based on an average baseline mortality rate of 0.095 (Table 11-13). Applying this mortality rate, the estimated regional baseline mortality of great black-backed gulls is 4,057 birds for the non-breeding season (42,708 x 0.095). The additional predicted mortality of 3.7 great black-backed gulls in the non-breeding season would increase the baseline mortality rate by 0.09% (Table 11-40).

Table 11-40 Increase in estimated baseline mortality for great black-backed gulls in the OAA as a result of collision

Season	Regional baseline population (adults)	Annual Regional Baseline Mortality	Increase in baseline mortality (%)
Breeding (Mar-Aug) (Adults only)	1,410	99	1.62
Non-breeding (Sep-Feb) (All ages)	42,708	4,057	0.09
Total	-	-	1.7

Predicted annual great black-backed gull mortality due to collision effects based on adult birds in the breeding season and all ages in the non-breeding season, involved 5.3 birds, which corresponds to an increase in the annual baseline mortality rate of 1.7% (Table 11-40).

A comparison of estimated great black-backed gull collision mortality against a regional population consisting of adult and immature birds is shown in Table 11-41. The predicted additional mortality due to collision effects was 2.4 great black-backed gulls (all ages) in the breeding season (Table 11-39). The total great black-backed gull regional breeding population (all ages) is estimated to be 3,579 birds (Table 11-11). The average mortality for all age classes is 0.095 (Table 11-13). Applying this mortality rate, the estimated regional baseline mortality of great black-backed gulls is 340 birds per breeding season (all ages) (3,579 x 0.095). The additional predicted mortality of 2.4 great black-backed gulls in the breeding season would increase the baseline mortality rate by 0.71% (Table 11-41).

As above, the additional predicted mortality of 3.7 great black-backed gulls in the non-breeding season would increase the baseline mortality rate by 0.09%. Predicted annual great black-backed gull mortality due to collision effects based on all ages in the breeding and non-breeding season involved 6.1 birds, which corresponds to an increase in the annual baseline mortality rate of 0.8% (Table 11-41).

Table 11-41 Increase in estimated baseline mortality for great black-backed gulls in the OAA as a result of collision

Season (All ages)	Regional baseline population (adults)	Annual Regional Baseline Mortality	Increase in baseline mortality (%)
Breeding (Mar-Aug)	3,579	340	0.71
Non-breeding (Sep-Feb)	42,708	4,057	0.09
Total	-	-	0.8

As highlighted by Natural England guidance, where predicted impacts equate to 1% or below of baseline mortality for a population (e.g. colony population) then this level of impact could be considered non-significant (Parker *et al.,* 2022c). Based on this, as the predicted increases in annual baseline mortality for great black-backed gull were between 0.81% and 1.7%, PVA was carried out on the regional population.

Details of the approach and results from the PVA are presented in the PVA Technical Appendix (Appendix 11.6), and the results are summarised below. The two ratio metrics recommended by NatureScot (2023) to compare impacted (with Project) and un-impacted (no Project) populations are the counterfactual ratio of the final population sizes, and the counterfactual ratio of the population growth rates.

The PVA for the great black-backed gull regional breeding population produced exponential growth under all considered scenarios, reflecting the absence of density-dependence in the model. These results are not likely to represent realistic trajectories of the future variation in this population, as density-dependent factors are likely to be important in situations where exponential increases occur. The PVA predicted that population growth rates for the great black-backed gull regional population would be reduced by around 0.1% under the mean collision risk scenario, with a predicted counterfactual ratio of 0.999 over the 38-year Project lifetime. After 38 years, the median population size was predicted to be 149,095, compared to the no-Project baseline population of 154,826. The predicted counterfactual ratio for population size was 0.96 (Table 11-42).

NatureScot (2023) state that a counterfactual population growth rate ratio of 0.90, or a counterfactual population size ratio of 0.95, “might be considered to be a small enough effect that the development would not lead to an adverse effect on site integrity”. However, they caution that “there is no standard threshold with respect to what might be considered an “acceptable” level of impact”.

On this basis, it was concluded that the PVA indicated that there would be no adverse effect on the great black-backed gull regional breeding population as a result of the Sceirde Rocks Project. Further details are presented in the PVA Technical Appendix (Appendix 11-6).

Table 11-42 Summary of PVA collision outputs for great black-backed gulls after 37 years)

	Annual Population Growth Rate		Final Population Size	
	Median growth rate	Counterfactual ratio	Median population size	Counterfactual ratio
Baseline (No Project)	1.133	-	154,826	-
Mean collision rate After 38 years with Project	1.132	0.999	149,095	0.96

Based on the results of the collision assessment and the PVA assessment, the magnitude of impact from collision effects on the regional great black-backed gull population was considered to be **Low** as the estimated increases in the annual baseline mortality rate were between 0.82% and 1.7% (Table 11-5), while the PVA outputs did not predict a significant negative effect on the regional breeding population.

For this assessment, receptor sensitivity has been based on three reviews of evidence from post-construction studies at offshore wind farms. A review of post-construction studies of seabirds at offshore wind farms in European waters concluded that great black-backed gull was one of the species that was weakly attracted to offshore wind farms (Dierschke *et al.,* 2016). A review of vulnerability of Scottish seabirds to offshore wind turbines in the context of collision ranked great black-backed gull as the second most sensitive out of 38 species (Furness *et al.,* 2013). Bradbury *et al.,* (2014) classified the great black-backed gull population vulnerability to collision mortality from offshore wind farms as very high.

Great black-backed gull is not listed as a qualifying interest in the breeding season for any SPA within mean maximum foraging distance (NPWS, 2024). In addition, the species is Green-listed in Ireland in terms of its conservation status (Gilbert *et al.,* 2021), indicating that it is not a species of conservation concern. On this basis, it is considered that great black-backed gull is of "local" importance in terms of its conservation value (Table 11-4). Although the species has a very high behavioural sensitivity to collision impacts, it is only of local conservation importance, leading to an overall **Medium** sensitivity to collision risk.

For great black-backed gull, the magnitude of the impact is deemed to be **Low,** and the overall sensitivity of this species is considered to be **Medium**, with local conservation importance in the breeding and non-breeding seasons. The significance of any effect on great black-backed gulls from collision effects associated with the Offshore Site is therefore considered a **Slight Negative** effect which is Not Significant (Table 11-6).

11.8.6.4.5 Herring Gull

Estimated number of herring gull collisions are presented in Table 11-43. Figures are presented for the breeding and non-breeding seasons, based on the proposed WTG layout. Predicted numbers of collisions were higher for the breeding season, compared to the non-breeding season.

Table 11-43 Estimated numbers of herring gull collisions by season in the OAA

Season	Mean number of collisions	Lower 95% confidence limit	Upper 95% confidence limit
Breeding (Mar-Aug)	3.1 (1.4 breeding adults)	0	10.5
Non-breeding (Sep-Feb)	1.4	0	4.5
Annual collisions	4.5	0	13.6
Total (to nearest whole bird)	4	0	15

Annual total taken from Table 11-29. Note totals based on seasonal breakdown may differ slightly due to rounding.

In the breeding season (March to August), the total mean estimated number of herring gull collisions was 3.1 birds (Table 11-43). However, this includes non-breeding adults and immature birds, as well as breeding adults. In the breeding season, 70.59% of all aged herring gulls recorded on baseline surveys were adults, (see Baseline Ornithology Report), therefore it was assumed that 70.59% of the population present are adult birds. For this assessment, herring gull collision mortality was considered to involve 2.2 adult birds in the breeding season.

Similarly, a proportion of adult birds present at colonies in the breeding season will opt not to breed in a particular breeding season. It has been estimated that 35% of adult herring gulls may be "sabbatical" birds in any particular breeding season (RPS, 2022), and this has been applied for this assessment. On this basis, 0.8 adult herring gulls predicted to collide were considered not to be breeding, therefore herring gull collision mortality in the breeding season was considered to involve 1.4 adult breeding birds.

The total herring gull regional breeding population is estimated to be 3,186 adult birds (Table 11-11). For the breeding season assessment based on adult birds only, the increase in baseline mortality was calculated based on an estimated adult herring gull baseline survival rate of 0.834, therefore the corresponding rate for adult mortality is 0.166 (Table 11-13). Applying this mortality rate, the estimated regional baseline mortality of herring gulls is 529 adult birds per breeding season (3,186 x 0.166). The additional predicted mortality of 1.4 breeding adult herring gulls in the breeding season would increase the baseline mortality rate by 0.26% (Table 11-44).

Table 11-44 Increase in estimated baseline mortality for herring gulls in the OAA as a result of collision

Season	Regional baseline population (adults)	Annual Regional Baseline Mortality	Increase in baseline mortality (%)
Breeding (Mar-Aug) (Adults only	3,186	529	0.26
Non-breeding (Sep-Feb) (All ages)	190,702	32,801	0.004
Total	-	-	0.264

For the non-breeding season, estimated seasonal herring gull collision mortality was 1.4 birds (Table 11-43). The herring gull regional population for the non-breeding season is estimated to be 190,702 birds (Table 11-12). The increase in baseline mortality was calculated based on an average baseline mortality rate of 0.172 (Table 11-13). Applying this mortality rate, the estimated regional baseline mortality of herring gulls is 32,801 birds for the non-breeding season (190,702 x 0.172). The additional predicted mortality of 1.4 herring gulls in the non-breeding season would increase the baseline mortality rate by 0.004% (Table 11-44).

Predicted annual herring gull mortality due to collision effects based on adult birds in the breeding season and all ages in the non-breeding season, involved 2.8 birds, which corresponds to an increase in the annual baseline mortality rate of 0.264% (Table 11-44).

A comparison of estimated herring gull collision mortality against a regional population consisting of adult and immature birds is shown in Table 11-45. The predicted additional mortality due to collision effects was 3.1herring gulls (all ages) in the breeding season (Table 11-43). The total herring gull regional breeding population (all ages) is estimated to be 7,551 birds (Table 11-11). The average mortality for all age classes is 0.172 (Table 11-13). Applying this mortality rate, the estimated regional baseline mortality of herring gulls is 1,299 birds per breeding season (all ages) (7,551 x 0.172). The additional predicted mortality of 3.1 herring gulls in the breeding season would increase the baseline mortality rate by 0.24% (Table 11-45).

As above, the additional predicted mortality of 1.4 herring gulls in the non-breeding season would increase the baseline mortality rate by 0.004%. Predicted annual herring gull mortality due to collision effects based on all ages in the breeding and non-breeding season involved 4.5 birds, which corresponds to an increase in the annual baseline mortality rate of 0.244% (Table 11-45).

Table 11-45 Increase in estimated baseline mortality for herring gulls in the OAA as a result of collision

Season (All ages)	Regional baseline population (adults)	Annual Regional Baseline Mortality	Increase in baseline mortality (%)
Breeding (Mar-Aug)	3,560	612	0.24
Non-breeding (Sep-Feb)	190,702	32,801	0.004
Total	-	-	0.244

As highlighted by Natural England guidance, where predicted impacts equate to 1% or below of baseline mortality for a population (e.g. colony population) then this level of impact could be considered non-significant (Parker *et al.,* 2022c). Based on this, as the predicted increases in annual baseline mortality for herring gull were below 1%, PVA was not carried out on the regional population.

Based on the results of the collision assessment, the magnitude of impact from collision effects on the regional herring gull population was considered to be **Negligible** as the estimated increases in the annual baseline mortality rate were below 1% (Table 11-5).

For this assessment, receptor sensitivity has been based on three reviews of evidence from post-construction studies at offshore wind farms. A review of post-construction studies of seabirds at offshore wind farms in European waters concluded that herring gull was one of the species that was weakly attracted to offshore wind farms (Dierschke *et al.,* 2016). A review of vulnerability of Scottish seabirds to offshore wind turbines in the context of collision ranked herring gull as the most sensitive out of 38 species (Furness *et al.,* 2013). Bradbury *et al.,* (2014), classified the herring gull population vulnerability to collision mortality from offshore wind farms as very high.

Herring gull is not listed as a qualifying interest in the breeding season for any SPA within mean maximum foraging distance (NPWS, 2024). The species is Amber-listed in Ireland in terms of its conservation status (Gilbert *et al.,* 2021), indicating that it is not a key species of conservation concern. On this basis, it is considered that herring gull is of medium importance in terms of its conservation value (Table 11-4). Overall, the species has a very high behavioural sensitivity to collision impacts, and a medium conservation importance, leading to an overall **High** sensitivity to collision risk.

For herring gull, the magnitude of the impact is deemed to be **Negligible,** and the overall sensitivity of this species is considered to be **High**, with medium conservation importance in the breeding and non-breeding seasons. Any effect on herring gulls from collision effects associated with the Offshore Site is therefore considered **Not Significant** (Table 11-6).

11.8.6.4.6 Lesser Black-backed Gull

Estimated number of lesser black-backed gull collisions are presented in Table 11-46. Figures are presented for the breeding and non-breeding seasons, based on the proposed WTG layout. Predicted numbers of collisions were higher for the breeding season, compared to the non-breeding season.

Table 11-46 Estimated numbers of lesser black-backed gull collisions by season in the OAA

Season	Mean number of collisions	Lower 95% confidence limit	Upper 95% confidence limit
Breeding (Apr-Aug)	2.8 (1.8 adults)	0	6.7
Autumn migration (Sep-Oct)	0	0	0
Winter period (Nov-Feb)	0	0	0
Spring migration (March)	0.4	0	2.2
Annual collisions	3.1	0	7.8
Total (to nearest whole bird)	3	0	8

Annual total taken from Table 11-29. Note totals based on seasonal breakdown may differ slightly due to rounding.

In the breeding season (April to August), the total mean estimated number of lesser black-backed gull collisions was 2.8 birds (Table 11-46). However, this includes non-breeding adults and immature birds, as well as breeding adults. As all aged lesser black-backed gulls recorded on baseline surveys in the breeding season were adults, (see Offshore Ornithology Baseline Report), it was assumed that 100% of the population present are adult birds, therefore breeding season lesser black-backed gull collision mortality was considered to involve 2.8 adult birds.

A proportion of adult birds present at colonies in the breeding season will opt not to breed in a particular breeding season. It has been estimated that 35% of adult lesser black-backed gulls may be "sabbatical" birds in any particular breeding season (RPS, 2022), and this has been applied for this assessment. On this basis, 0.98 adult lesser black-backed gulls predicted to collide were considered not to be breeding, therefore lesser black-backed gull collision mortality in the breeding season was considered to be 1.8 adult breeding birds.

The total lesser black-backed gull regional breeding population is estimated to be 5,068 adult birds (Table 11-11). For the breeding season assessment based on adult birds only, the increase in baseline mortality was calculated based on an estimated adult lesser black-backed gull baseline survival rate of 0.885, therefore the corresponding rate for adult mortality is 0.115 (Table 11-13). Applying this mortality rate, the estimated regional baseline mortality of lesser black-backed gulls is 583 adult birds per breeding season (5,068 x 0.115). The additional predicted mortality of 1.8 breeding adult lesser black-backed gulls in the breeding season would increase the baseline mortality rate by 0.31% (Table 11-47).

For the non-breeding season, estimated seasonal lesser black-backed gull collision mortality was 0.4 birds in the spring migration period (Table 11-46). The total lesser black-backed gull regional population in the spring migration period is estimated to be 174,257 birds (Table 11-12). The increase in baseline mortality was calculated based on an average baseline mortality rate of 0.123 (Table 11-13). Applying this mortality rate, the estimated regional baseline mortality of lesser black-backed gulls is 21,434 birds for the non-breeding season (174,257 x 0.123). The additional predicted mortality of 0.4 lesser black-backed gulls in the spring migration period of the non-breeding season would increase the baseline mortality rate by 0.002% (Table 11-47).

Predicted annual lesser black-backed gull mortality due to collision effects based on adult birds in the breeding season and all ages in the non-breeding season, involved 2.2 birds, which corresponds to an increase in the annual baseline mortality rate of 0.312% (Table 11-47).

Table 11-47 Increase in estimated baseline mortality for lesser black-backed gulls in the OAA as a result of collision

Season	Regional baseline population (adults)	Annual Regional Baseline Mortality	Increase in baseline mortality (%)
Breeding (Apr-Aug) (Adults only)	5,068	583	0.31
Autumn migration (Sep-Oct) (All ages)	174,257	21,434	0
Winter period (Nov-Feb) (All ages)	54,408	6,692	0
Spring migration (March) (All ages)	174,257	21,434	0.002
Total	-	-	0.312

A comparison of estimated lesser black-backed gull collision mortality against a regional population consisting of adult and immature birds is shown in Table 11-48. The predicted additional mortality due to collision effects was 2.8 lesser black-backed gulls (all ages) in the breeding season (Table 11-46). The total lesser black-backed gull regional breeding population (all ages) is estimated to be 9,508 birds (Table 11-11). The average mortality for all age classes is 0.123 (Table 11-13). Applying this mortality rate, the estimated regional baseline mortality of lesser black-backed gulls is 1,169 birds per breeding season (all ages) (9,508 x 0.123). The additional predicted mortality of 2.8 lesser black-backed gulls in the breeding season would increase the baseline mortality rate by 0.24% (Table 11-48).

As above, the additional predicted mortality of 0.4 lesser black-backed gulls in the non-breeding season would increase the baseline mortality rate by 0.002%. (Table 11-46). Predicted annual lesser black-backed gull mortality due to collision effects based on all ages in the breeding and non-breeding season involved 3.2 birds, which corresponds to an increase in the annual baseline mortality rate of 0.242% (Table 11-48).

Table 11-48 Increase in estimated baseline mortality for lesser black-backed gulls in the OAA as a result of collision

Season (All ages)	Regional baseline population (adults)	Annual Regional Baseline Mortality	Increase in baseline mortality (%)
Breeding (Apr-Aug)	9,508	1,169	0.24
Autumn migration (Sep-Oct)	174,257	21,434	0
Winter period (Nov-Feb)	54,408	6,692	0
Spring migration (March)	174,257	21,434	0.002
Total	-	-	0.242

As highlighted by Natural England guidance, where predicted impacts equate to 1% or below of baseline mortality for a population (e.g. colony population) then this level of impact could be considered non-significant (Parker *et al.,* 2022c). Based on this, as the predicted increases in annual baseline mortality for lesser black-backed gull were below 1%, PVA was not carried out on the regional population.

Based on the results of the collision assessment, the magnitude of impact from collision effects on the regional lesser black-backed gull population was considered to be **Negligible** as the estimated increases in the annual baseline mortality rate were below 1% (Table 11-5)

For this assessment, receptor sensitivity has been based on three reviews of evidence from post-construction studies at offshore wind farms. A review of post-construction studies of seabirds at offshore wind farms in European waters concluded that lesser black-backed gull was one of the species that was weakly attracted to offshore wind farms (Dierschke *et al.,* 2016). A review of vulnerability of Scottish seabirds to offshore wind turbines in the context of collision ranked lesser black-backed gull as the third most sensitive out of 38 species (Furness *et al.,* 2013). Bradbury *et al.,* (2014), classified the lesser black-backed gull population vulnerability to collision mortality from offshore wind farms as very high.

Lesser black-backed gull is listed as a qualifying interest in the breeding season for one SPA within mean maximum foraging distance (NPWS, 2024). In addition, the species is Amber-listed in Ireland in terms of its conservation status (Gilbert *et al.,* 2021). On this basis, it is considered that lesser black-backed gull is of international importance in terms of its conservation value (Table 11-4).

Estimated numbers of lesser black-backed gulls recorded within the OAA would qualify as internationally important in the breeding season (Offshore Ornithology Baseline Report), with individuals potentially originating from an SPA in the region, as well as non-SPA colonies. On this basis the conservation importance for lesser black-backed gull was considered to be medium (Table 11-4).

This species has a very high behavioural sensitivity to collision impacts, with birds within the site in the breeding season considered likely to be from SPAs and non-SPAs the overall sensitivity to collision risk is considered to be **High**.

For lesser black-backed gull, the magnitude of the impact is deemed to be **Negligible,** and the overall sensitivity of this species is considered to be **High**, with birds within the site in the breeding season considered likely to be from SPAs and non-SPAs. Any effect on lesser black-backed gulls from collision effects associated with the Offshore Site is therefore considered **Not Significant** (Table 11-6).

11.8.6.4.7 Common Tern

Estimated number of common tern collisions are presented in Table 11-49. Predicted collision numbers were very low, based on the proposed WTG layout.

Table 11-49 Estimated numbers of common tern collisions by season in the OAA

Season	Mean number of collisions	Lower 95% confidence limit	Upper 95% confidence limit
Migration-free breeding (June & July)	0.3	0	1.3
Autumn migration (Aug & Sep)	0.1	0	0.5
Spring migration (Apr & May)	0	0	0
Annual collisions	0.4	0	1.3
Total (to nearest whole bird)	0	0	2

As defined by Furness (2015), there is considerable overlap between the breeding season for common tern (May to August) and the autumn migration period (late July to early September). To separate out collision impacts on breeding adult common terns from migrating common terns, the migration-free breeding season, i.e. the breeding season between the spring and autumn migration periods where the majority of adult birds are present at colonies, has been used. Furness (2015) defined the migration-free breeding season as June to mid-July, however for this assessment, the whole of July was considered as breeding season, as this was considered more precautionary. Similarly, Furness (2015) defined the autumn period of the non-breeding season as late July to early September, however for this assessment August and September were considered the autumn migration period. The spring migration period was defined by Furness (2015) as April and May, and this has been used for this assessment.

In the migration-free breeding season (June and July), the total mean estimated number of common tern collisions was 0.3 birds (Table 11-49). For the breeding season assessment, it was assumed that all birds involved were breeding adults.

The total common tern regional breeding population (adults only) is estimated to be 256 birds (Table 11-11). The adult mortality rate is 0.117 (Table 11-13). Applying this mortality rate, the estimated regional baseline mortality of common tern is 30 birds per breeding season (all ages) (256 x 0.117). The additional predicted mortality of 0.3 common terns in the breeding season would increase the baseline mortality rate by 1.0% (Table 11-50).

For the autumn migration period (August and September), estimated seasonal common tern collision mortality was 0.1 birds (Table 11-49). The total common tern regional population in the autumn migration period is estimated to be 64,189 birds (Table 11-12). The increase in baseline mortality was calculated based on an average baseline mortality rate of 0.191 (Table 11-13). Applying this mortality rate, the estimated regional baseline mortality of common terns is 12,260 birds for the autumn migration period (64,189 x 0.191). The additional predicted mortality of 0.1 common terns in the autumn migration period would increase the baseline mortality rate by 0.0008% (Table 11-50).

For the spring migration period of the non-breeding season, estimated seasonal common tern collision mortality was zero birds (Table 11-49). Consequently, there would be no additional predicted mortality of common terns in the spring migration period.

Predicted annual common tern mortality due to collision effects based on adults only in the migration-free breeding season and all ages in the autumn and spring migration periods, involved 0.4 birds, which corresponds to an increase in the annual baseline mortality rate of 1.0% (Table 11-50).

Table 11-50 Increase in estimated baseline mortality for common terns (in the OAA as a result of collision

Season	Regional baseline population (adults)	Annual Regional Baseline Mortality	Increase in baseline mortality (%)
Migration-free breeding (May-Aug) (Adults only)	256	30	1.0
Autumn migration (Aug & Sep) (All ages)	64,189	12,260	0.0008
Spring migration (Apr & May) (All ages)	64,189	12,260	0
Total	-	-	1.0

A comparison of estimated common tern collision mortality against a regional population consisting of adult and immature birds is shown in Table 11-51. The predicted additional mortality due to collision effects was 0.3 common terns (all ages) in the breeding season (Table 11-49). The common tern regional breeding population (all ages) is estimated to be 435 birds (Table 11-11). The average mortality for all age classes is 0.191 (Table 11-13). Applying this mortality rate, the estimated regional baseline mortality of common terns is 83 birds per breeding season (all ages) (435 x 0.191). The additional predicted mortality of 0.3 common terns in the migration-free breeding season would increase the baseline mortality rate by 0.36% (Table 11-51).

For the autumn migration period, the additional predicted mortality of 0.1 common terns would increase the baseline mortality rate by 0.0008%. Predicted annual common tern mortality due to collision effects based on all ages in the breeding and non-breeding seasons, involved 0.4 birds, which corresponds to an increase in the annual baseline mortality rate of 0.36% (Table 11-51).

Table 11-51 Increase in estimated baseline mortality for common terns (in the OAA as a result of collision

Season (All ages)	Regional baseline population (adults)	Annual Regional Baseline Mortality	Increase in baseline mortality (%)
Migration-free breeding (May-Aug)	435	83	0.36
Autumn migration (Aug & Sep)	64,189	12,260	0.0008

Season (All ages)	Regional baseline population (adults)	Annual Regional Baseline Mortality	Increase in baseline mortality (%)
Spring migration (Apr & May)	64,189	12,260	0
Total	-	-	0.36

As highlighted by Natural England guidance, where predicted impacts equate to 1% or below of baseline mortality for a population (e.g. colony population) then this level of impact could be considered non-significant (Parker *et al.,* 2022c). Based on this, as the predicted increases in annual baseline mortality for common tern were between 0.36% and 1.0%, PVA was carried out on the regional population.

Details of the approach and results from the PVA are presented in the PVA Technical Appendix (Appendix 11.6), and the results are summarised below. The two ratio metrics recommended by NatureScot (2023) to compare impacted (with Project) and un-impacted (no Project) populations are the counterfactual ratio of the final population sizes, and the counterfactual ratio of the population growth rates.

The PVA for the common tern regional breeding population predicted modest growth in the median population size over 50 years under all considered scenarios, although the lower limits of the confidence interval for the 97.5 percentile collision risk scenario showed a slight decrease.

The PVA predicted that population growth rate for the common tern regional population would be reduced by around 0.002% under the mean collision risk scenario, with a predicted counterfactual ratio of 0.998 over the 38-year Project lifetime. After 38 years, the median population size was predicted to be 727 birds, compared to the no-Project baseline end population of 778. The predicted counterfactual ratio for population size was 0.93 (Table 11-52).

NatureScot (2023) state that a counterfactual population growth rate ratio of 0.90, or a counterfactual population size ratio of 0.95, "might be considered to be a small enough effect that the development would not lead to an adverse effect on site integrity". However, they caution that "there is no standard threshold with respect to what might be considered an "acceptable" level of impact".

On this basis, it was concluded that the PVA indicated that there would be no adverse effect on the common tern regional breeding population as a result of the Sceirde Rocks Project. Further details are presented in the PVA Technical Appendix (Appendix 11-6).

Table 11-52 Summary of PVA collision outputs for common tern after 37 years)

	Annual Population Growth Rate		Final Population Size	
	Median growth rate	Counterfactual ratio	Median population size	Counterfactual ratio
Baseline (No Project)	1.011	1	778	1
Mean collision rate After 38 years with Project	1.009	0.998	727	0.93

Based on the results of the collision assessment and the PVA assessment, the magnitude of impact from collision effects on the regional common tern population was considered to be Low as the estimated increases in the annual baseline mortality rate were between 0.36% and 1.0% (Table 11-5), while the PVA outputs did not predict a significant negative effect.

For this assessment, receptor sensitivity has been based on reviews of evidence from post-construction studies at offshore wind farms. A review of post-construction studies of seabirds at offshore wind farms in European waters concluded that common tern was one of the species that was hardly affected by offshore wind farms (Dierschke *et al.,* 2016). A review of vulnerability of Scottish seabirds to offshore wind turbines in the context of collision ranked common tern as the 14th most sensitive out of 38 species (Furness *et al.,* 2013). Bradbury *et al.,* (2014), classified the common tern population vulnerability to collision mortality from offshore wind farms as moderate.

Common terns recorded within the OAA would qualify as internationally important in the breeding season (Offshore Ornithology Baseline Report), as the species is listed on Annex I of the EU Birds Directive, and there are SPAs within mean maximum foraging range (+1S.D.). In addition, the species is Amber-listed in Ireland in terms of its conservation status (Gilbert *et al.,* 2021). On this basis the conservation importance for common tern was considered to be medium (Table 11-4). This species has a moderate behavioural sensitivity to collision impacts, with birds within the site in the breeding season considered likely to be from SPAs and non-SPAs, therefore the overall sensitivity to collision risk is considered to be **Medium**.

For common tern, the magnitude of the impact is deemed to be **Low,** and the overall sensitivity of this species is considered to be **Medium**, with birds within the site in the breeding season considered likely to be from SPAs and non-SPAs. The significance of any effect on common terns from collision effects associated with the Offshore Site is therefore considered a **Slight Negative** effect which is **Not Significant** (Table 11-6).

11.8.6.4.8 Arctic Tern

Estimated number of Arctic tern collisions are presented in Table 11-53. Predicted collision numbers were very low, based on the proposed WTG layout.

Table 11-53 Estimated numbers of Arctic tern collisions by season in the OAA

Season	Mean number of collisions	Lower 95% confidence limit	Upper 95% confidence limit
Migration-free breeding (June)	0.2	0	0.9
Autumn migration (July to Sep)	0.1	0	0.6
Spring migration (Apr & May)	0	0	0
Annual collisions	0.2	0	1.3
Total (to nearest whole bird)	0	0	1

As defined by Furness (2015), there is considerable overlap between the breeding season for Arctic tern (May to August) and the autumn migration period (July to early September). To separate out collision impacts on breeding adult Arctic terns from migrating Arctic terns, the migration-free breeding season, i.e. the breeding season between the spring and autumn migration periods where the majority of adult birds are present at colonies, has been used. Furness (2015) defined the migration-free breeding season as June, and this has been used for this assessment. Similarly, Furness (2015) defined the autumn period of the non-breeding season as July to early September, however for this assessment the whole of September was considered part of the autumn migration period. The spring migration period was defined by Furness (2015) as April and May, and this has been used for this assessment.

In the migration-free breeding season (June), the total mean estimated number of Arctic tern collisions was 0.2 birds (all ages) (Table 11-53). For the breeding assessment, it was assumed that all collisions involved breeding adult birds.

The total Arctic tern regional breeding population (adults only) is estimated to be 504 birds (Table 11-11). The average adult mortality is 0.163 (Table 11-13). Applying this mortality rate, the estimated regional baseline mortality of adult Arctic tern is 82 birds per breeding season (504 x 0.163). The additional predicted mortality of 0.2 Arctic terns in the migration-free breeding season would increase the baseline mortality rate by 0.24% (Table 11-54).

For the autumn migration period (July to September), estimated seasonal Arctic tern collision mortality was 0.1 birds (Table 11-53). The total Arctic tern regional population in the autumn migration period is estimated to be 74,008 birds (Table 11-12). The increase in baseline mortality was calculated based on an average baseline mortality rate of 0.183 (Table 11-13). Applying this mortality rate, the estimated regional baseline mortality of Arctic terns is 13,543 birds for the autumn migration period (74,008 x 0.183). The additional predicted mortality of 0.1 Arctic terns in the autumn migration period would increase the baseline mortality rate by 0.0007% (Table 11-54).

For the spring migration period of the non-breeding season, estimated Arctic tern collision mortality was zero birds (Table 11-53). Predicted annual Arctic tern mortality due to collision effects based on all ages in the breeding and non-breeding season involved 0.3 birds, which corresponds to an increase in the annual baseline mortality rate of 0.24% (Table 11-54).

Table 11-54 Increase in estimated baseline mortality for Arctic terns in the OAA as a result of collision

Season	Regional baseline population (adults)	Annual Regional Baseline Mortality	Increase in baseline mortality (%)
Migration-free breeding (June) (Adults only)	504	82	0.240
Autumn migration (July to Sep)(All ages)	74,008	13,543	0.0007
Spring migration (Apr & May)(All ages)	74,008	13,543	0
Total	-	-	0.24

A comparison of estimated Arctic tern collision mortality against a regional population consisting of adult and immature birds is shown in Table 11-55. The predicted additional mortality due to collision effects was 0.2 Arctic terns (all ages) in the breeding season (Table 11-53). The Arctic tern regional breeding population (all ages) is estimated to be 762 birds (Table 11-11). The average mortality for all age classes is 0.183 (Table 11-13). Applying this mortality rate, the estimated regional baseline mortality of Arctic terns is 139 birds per breeding season (all ages) (762 x 0.183). The additional predicted mortality of 0.2 Arctic terns in the migration-free breeding season would increase the baseline mortality rate by 0.14% (Table 11-55).

For the autumn migration period, the additional predicted mortality of 0.1 Arctic terns would increase the baseline mortality rate by 0.0007%. Predicted annual Arctic tern mortality due to collision effects based on all ages in the breeding and non-breeding seasons, involved 0.24 birds, which corresponds to an increase in the annual baseline mortality rate of 0.121% (Table 11-55).

Table 11-55 Increase in estimated baseline mortality for Arctic terns (in the OAA as a result of collision

Season (All ages)	Regional baseline population (adults)	Annual Regional Baseline Mortality	Increase in baseline mortality (%)
Migration-free breeding (June)	762	139	0.14
Autumn migration (July to Sep)	74,008	13,543	0.0007
Spring migration (Apr & May)	74,008	13,543	0
Total	-	-	0.14

As highlighted by Natural England guidance, where predicted impacts equate to 1% or below of baseline mortality for a population (e.g. colony population) then this level of impact could be considered non-significant (Parker *et al.,* 2022c). Based on this, as the predicted increases in annual baseline mortality for Arctic tern were below 1%, PVA was not carried out on the regional population.

Based on the results of the collision assessment, the magnitude of impact from collision effects on the regional Arctic tern population was considered to be **Negligible** as the estimated increases in the annual baseline mortality rate were below 1% (Table 11-5).

For this assessment, receptor sensitivity has been based on reviews of evidence from post-construction studies at offshore wind farms. A review of post-construction studies of seabirds at offshore wind farms in European waters concluded that Arctic tern was one of the species that was hardly affected by offshore wind farms (Dierschke *et al.,* 2016). A review of vulnerability of Scottish seabirds to offshore wind turbines in the context of collision ranked Arctic tern as the 17th most sensitive out of 38 species (Furness *et al.,* 2013). Bradbury *et al.,* (2014), classified the Arctic tern population vulnerability to collision mortality from offshore wind farms as low.

Estimated numbers of Arctic terns recorded within the OAA would qualify as internationally important in the breeding season (Offshore Ornithology Baseline Report), as the species is listed on Annex I of the EU Birds Directive, and there are SPAs within mean maximum foraging range (+1S.D.). In addition, the species is Amber-listed in Ireland in terms of its conservation status (Gilbert *et al.,* 2021). On this basis the conservation importance for Arctic tern was considered to be medium (

Table 11-4). This species has a moderate behavioural sensitivity to collision impacts, with birds within the site in the breeding season considered likely to be from SPAs and non-SPAs, therefore the overall sensitivity to collision risk is considered to be **Medium**.

For Arctic tern, the magnitude of the impact is deemed to be **Negligible,** and the overall sensitivity of this species is considered to be **Medium**, with birds within the site in the breeding season considered likely to be from SPAs and non-SPAs. Any effect on Arctic terns from collision effects associated with the Offshore Site is therefore considered **Not Significant** (Table 11-6).

Migratory non-seabird species

There is the potential risk to migratory birds flying through the OAA to collide with the wind turbines and associated infrastructure. Migratory species are at risk when passing through the area on seasonal migration on spring and autumn passage. Migratory species are not typically recorded during monthly site-specific surveys due to the brief nature of migration timings as well as movement outside of routine survey windows i.e. at night or during poor weather (Woodward *et al.,* 2023). The potential collision risk to each species can be estimated throughout the year by using collision risk modelling (CRM).

Migratory species travel to, from and through western Ireland when moving from high latitude breeding grounds to more southerly wintering areas and vice versa. Some migratory bird species are present within the area year-round and while some groups overwinter in the area, others are present during autumn (post-breeding dispersal) or spring (return) migrations. Many seabirds migrate through western Irish waters while non-seabird species such as waders, waterfowl, passerines and non-passerines also migrate through the area.

The potential collision risk to migratory non-seabird species from the Project has been assessed using the Marine Scotland Avian Migration Collision Risk Model Shiny Application (hereafter 'mCRM app'. Since the tool was originally designed for UK migratory species, it had to be adapted so that it was compatible for use with Irish sites. In the code, the default migration corridors were changed to include birds with migration pathways that overlapped with the OAA and Iceland. The populations used in the tool were also revised to either the Ireland or SPA-specific populations where applicable. The key details of the approach and the results are summarised below, with further information presented in Appendix 11.4.

For the mCRM assessment, those SPAs hosting migratory species not previously considered were determined by level of connectedness of straight-line migration pathways that passed through the OAA. Relevant SPAs were screened in using the percentage of migration pathways which may intersect with the boundary of the OAA of the Project, taking the geometric centre of the SPA as the SPA location. This was undertaken using R code where the country boundaries for Europe (Iceland) and North America (Greenland) obtained from the NPWS website (NPWS, 2024) were combined with the Irish boundary so the migratory paths could be drawn between the sampled points along the coastline of each country. The sampled points were selected approximately 1 km apart along the coastlines of both Iceland and Greenland and points that intersected with land were removed to ensure that only valid landfall locations remained.

From this complete range, 10,000 points were randomly selected from both the Ireland (or the SPA polygon for those species linked to an SPA) and Europe/North America polygons and then converted to polyline objects so that they could be the start and end points for the migratory paths. These polylines were saved and used to establish the percentage of intersection to determine which colonies/sites should be included in the mCRM tool. Furthermore, the boundary of the outermost lines was taken and converted to a polygon to represent the migration corridor for each site. It was assumed that the migration corridor was the same for each species. In the final modelling scenarios, there was a different migration corridor for each individual SPA and for Ireland as a whole.

The proportion of potential migration lines that intersected with the OAA from each SPA was calculated. All SPAs in Ireland were included in initial screening with those that had no lines intersecting with the OAA screened out at this point.

An arbitrary 10% threshold for the percentage of migration paths intersecting with the OAA was set as per other recent assessments such as North Irish Sea Array (NISA) Windfarm Ltd (GoBe, 2023); only those SPAs with migratory features with at least 10% of lines intersecting with the OAA were carried forward. Where there were fewer than 10% of straight-line migration paths crossing the OAA between origin (or destination) and the SPA centroid, that SPA was screened out as it was expected that only negligible numbers of birds would be passing through the site, and any associated collisions would be minimal.

SPAs in which no migratory non-seabird species are designated features were screened out. Once the list of SPAs was complete, the designated migratory species for each site were screened in based on where at least 1% of the Irish population is expected to pass through the OAA each year as per other project assessments (e.g. NISA Windfarm Ltd, GoBe, 2023). Species where less than 1% of the Irish population is likely to pass through the area were screened out; these were coot, curlew, hen harrier and chough. In addition, as merlin, chough and hen harrier are terrestrial qualifying interest species during the breeding season for which the potential migratory numbers are low, these species were also screened out.

Oystercatcher (*Haematopus ostralegus*) and whimbrel (*Numenius phaeopus)* were not listed as a qualifying interest of any of the selected SPAs however, both species have a substantial Icelandic breeding population that passes into or through west Ireland on migration (BirdLife International, 2024a). Autumn migration for whimbrel passes at sea west of Ireland and so only the spring migration is considered (BirdLife International, 2024b). As birds are not concentrated in SPAs, the assessment undertaken looked at the whole migration path from Ireland to Iceland. All migratory species screened into the assessment are presented in Table 11-56.

Table 11-56 Short-list of migratory non-seabird species screened into mCRM assessment

Species		
Barnacle goose	Black-tailed godwit	Dunlin
Greenland white-fronted goose	Mallard	Oystercatcher
Shelduck	Teal	Whimbrel
Wigeon		

Turbine parameters and predicted mean monthly wind availability were as presented in Appendix 11.4. The species-specific avoidance rates and biometric data used in the assessment are default within the mCRM tool and were determined by the British Trust for Ornithology (BTO). Further details are presented in Appendix 11.4. The results of the mCRM for each screened in migratory non-seabird species are presented in Table 11-57.

Table 11-57 Summary of seasonal and annual collision estimates per SPA of screened in migratory non-seabird species within the OAA

Species	Pre-breeding	Post-breeding	Other	Totals
Clonakilty Bay SPA				
Black-tailed godwit	0.007 ± 0.001	0.007 ± 0.001	0.000 ± 0.000	0.014 ± 0.001
Dunlin	0.004 ± 0.000	0.004 ± 0.000	0.000 ± 0.000	0.008 ± 0.000
Shelduck	0.009 ± 0.001	0.009 ± 0.001	0.009 ± 0.001	0.027 ± 0.002
Eirk Bog SPA				
Greenland white-fronted goose	0.000 ± 0.000	0.000 ± 0.000	0.000 ± 0.000	0.000 ± 0.000
Illaunonearaun SPA				
Barnacle goose	0.001 ± 0.000	0.001 ± 0.000	0.000 ± 0.000	0.002 ± 0.000
Killarney National Park SPA				
Greenland white-fronted goose	0.000 ± 0.000	0.000 ± 0.000	0.000 ± 0.000	0.000 ± 0.000
The Gearagh SPA				
Mallard	0.057 ± 0.004	0.057 ± 0.004	0.056 ± 0.004	0.170 ± 0.007
Teal	0.016 ± 0.001	0.016 ± 0.001	0.000 ± 0.000	0.032 ± 0.001
Wigeon	0.022 ± 0.002	0.022 ± 0.002	0.000 ± 0.000	0.044 ± 0.003
Ireland				
Oystercatcher	0.039 ± 0.006	0.040 ± 0.006	0.000 ± 0.000	0.079 ± 0.008
Whimbrel	0.030 ± 0.005	0.000 ± 0.000	0.000 ± 0.000	0.030 ± 0.005

The analysis of migration collisions for these SPA-qualifying migratory non-seabird species show that in all cases, considerably less than a single collision is expected annually. The proportion of birds using Ireland as a staging post or wintering area that are at risk of collision with the Project is extremely small.

Based on the results of the mCRM assessment, the magnitude of impact from collision effects on the migratory non-seabird species assessed was considered to be **Negligible** as the estimated numbers of collisions were all well below one collision per year, which is considered to result in a very slight change from the size or extent of distribution of the relevant regional population or the population that is the interest feature of a specific protected site (Table 11-5).

Langston and Pullan (2003) concluded that species of wildfowl were particularly sensitive to potential collision impacts while species of waders such as black-tailed godwit and curlew were not sensitive to collision impacts. On this basis, sensitivity of migratory non-seabird species was considered to be **High** at worst.

The magnitude of the potential collision impact on migratory non-seabird species is deemed to be **Negligible,** and the overall sensitivity of these species is considered to be **High** at worst. Any effect on migratory non-seabird species from collision effects associated with the Offshore Site is therefore considered **Not Significant** (Table 11-6).

11.8.6.5 Impact 8- Disturbance from turbine lighting

Impact 8 considers the potential for disturbance to seabirds from aviation and navigation lighting on turbines. There is the potential that aviation and navigation lighting on wind turbines could attract or repel birds moving through the OAA at night. There is some evidence that nocturnal lighting may cause changes in bird behaviour and habitat selection (Drewitt and Langston, 2008). However much of this evidence is based on oil and gas platforms, and as offshore wind farms are typically less intensively lit than these installations, any impacts are likely to be less extreme. It is currently planned that selected peripheral turbines will be illuminated with Aids to Navigation (AtoN) lighting. All other turbines will be unlit apart from small white lamps above turbine access doors. Further details are presented in the Lighting and Marking Plan included in Appendix 5-9 of the EIAR.

Sensitivity of the receptor

A significant impact could potentially occur if large numbers of migrant birds fly through an offshore wind farm in a single event, leading to mass disorientation or collisions. However, there is no evidence from existing offshore wind farm studies to suggest mass collision events occur as a result of aviation and navigation lighting that is typically used for these projects. Evidence from Kerlinger *et al.,* (2010) and Welcker *et al.,* (2017) found that nocturnal migrants do not have a higher risk of collision with wind farms than species that migrate during daylight, while mortality rates are not higher at offshore wind farms with lighting compared to those without. Furthermore, studies have shown that nocturnal flight is altered to counteract the risk of collision at offshore wind farms (Dirksen *et al.,* 1998 and Desholm and Kahlert, 2005).

Gannet and kittiwake are considered to be most at risk of collisions with turbines, however both species are unlikely to be active at night, as they either return to their colonies or roost on the sea surface during darkness (Wade *et al.,* 2016). A tracking study by Furness *et al.,* (2018) reported that gannet flight and diving activity was minimal during darkness. Kotzerka *et al.,* (2010) reported that kittiwake foraging trips mainly occurred during daylight hours and that birds were largely inactive during the hours of darkness and therefore risks of interactions with turbines were lower than in daylight hours.

Gulls are known to have low to moderate levels of nocturnal activity but can sometimes be attracted to the lights of fishing vessels and well-lit oil and gas platforms that attract fish to the surface waters (Burke *et al.,* 2012). However, as offshore wind farms are typically less intensively lit than these installations, the degree of nocturnal attraction for gull species is considered likely to be lower than oil platforms or fishing vessels.

While species such as Manx shearwater and storm petrel could be considered at potential risk of attraction to turbine lighting at night, the potential for impacts is still considered low. Although there is some evidence of foraging occurring at night in Scotland (Kane, 2020), Manx shearwater foraging occurs almost exclusively during daylight hours. The majority of nocturnal behaviour would typically be associated with birds rafting close to colonies in the evening and then returning to their burrows after dusk. As there are no Manx shearwater colonies in the immediate vicinity of the Project, and as foraging activity is likely to be low during nocturnal hours, potential impacts from attraction to turbine lighting in terms of impacts on breeding success is considered to be of negligible magnitude.

Based on available evidence from published studies, it is considered that seabird species in the marine environment would exhibit no more than a **Medium** sensitivity to offshore lighting associated with the Project.

Magnitude of Impact

Based on available evidence, it is considered that red lighting (e.g., aviation warning lights) may have minimal effects on seabirds, with yellow lighting (e.g., navigational lighting) also having low impacts (Syposz *et al*, 2021). Any impacts on birds in the vicinity are considered to be restricted to the operation and maintenance phase, and to the hours of darkness, when the majority of seabirds are inactive. Survival and reproductive rates of key bird species are very unlikely to be impacted to the extent that the population trajectory would be altered.

The maximum magnitude of any effect on key bird species from aviation and navigation lighting associated with the Project has therefore been assessed as **Low.**

For Impact 9, the magnitude of the impact is deemed to be **Low,** and the overall sensitivity of key bird species to offshore turbine lighting is considered to be **Medium**. Any effect on key bird species from effects associated with Impact 9 is therefore considered a **Slight Negative** effect which is **Not Significant** (Table 11-6).

11.8.7 Decommissioning Phase

11.8.7.1 Impact 9 - Disturbance and displacement within the OAA during decommissioning

A Rehabilitation Schedule has been prepared for the Project (see Appendix 5-18), the details of which will be agreed with the local authority prior to any decommissioning. The Rehabilitation Schedule will be updated prior to the end of the operational period in line with decommissioning methodologies that may exist at the time and will be agreed with the competent authority at that time. It can be assumed that the decommissioning activity will resemble the reverse of the installation and therefore the potential impacts associated with the decommissioning phase are considered to be analogous or likely less than that of the construction phase.

The decommissioning base locations will be out of Foynes, Cork and Belfast. Up to three vessels will be used for WTG removal and up to four tugs for foundation removal. For infrastructure removal the installation process is reversed using vessels to remove the WTGs and then to deballast the foundations and wet tow them from the site. Rock protection used for cables and/or seabed preparation material (e.g. stonebeds) is assumed to be left *in situ*. Decommissioning of the cables will involve removal of any exposed or unburied cable. All rock berms will remain undisturbed. This method has the lowest environmental impact. Further information on the decommissioning process is detailed in Chapter 5: Project Description.

Taking this into consideration, along with the mitigation in Table 11-15, which will also be applicable to decommissioning, the effects associated with the Decommissioning Phase will be no **worse** than **slight and adverse** and **Not Significant** for all offshore ornithology receptors.

11.9 Effects on Designated Sites

The Offshore Site is not located within the boundaries of any Natura 2000 Designated Sites. An Appropriate Assessment screening report and a Natura Impact Statement were prepared to provide the information necessary for the competent authority to complete a screening and an Appropriate Assessment for the Project.

As per the EPA Guidelines *"A biodiversity section of an EIAR, for example, should not repeat the detailed assessment of potential effects on European sites contained in documentation prepared as part of the Appropriate Assessment process, but it should refer to the findings of that separate assessment in the context of likely significant effects on the environment, as required by the EIA Directive"*. This section provides a summary of the key assessment findings with regard to potential impacts on European Sites.

The Offshore AA Screening Report concluded:

'The Project alone or in combination with other plans and projects (i.e. Offshore and Onshore plans and projects) has the potential to have LSE on the following European Sites, in light of their conservation objectives and best scientific information (without the application of mitigation). Sites which have been included solely to ensure consistency with the foreshore licensing approach, are marked with an asterix.

- *Inishmore Island SAC,*
- *Kilkieran Bay and Islands SAC,*
- *Lower River Shannon SAC,*
- *Slyne Head Peninsula SAC,*
- *Slyne Head Islands SAC,*
- *West Connacht Coast SAC,*
- *Galway Bay Complex SAC,*
- *Blasket Islands SAC,*
- *Duvillaun Islands SAC,*
- *Connemara Bog Complex SAC,*
- *Twelve Bens/Garraun Complex SAC,*
- *Maumturk Mountains SAC,*
- *Lough Corrib SAC,*
- *Mweelrea/Sheeffry/Erriff Complex SAC,*
- *Inishmaan Island SAC,*
- *Carrowmore Point to Spanish Point and Islands SAC,*
- *Carrowmore Dunes SAC,*
- *Kilkee Reefs SAC,*
- *Kenmare River SAC*,*
- *Hook Head SAC*,*
- *Belgica Mound Province SAC*,*
- *Roaringwater Bay and Islands SAC*,*
- *Gweedore Bay and Islands SAC*,*
- *Bunduff Lough and Machair/Trawalua/Mullaghmore SAC*,*
- *St John's Point SAC*,*
- *Carnsore Point SAC*,*
- *Blackwater Bank SAC*,*
- *Lough Swilly SAC*,*
- *Codling Fault Zone SAC*,*
- *Rockabill to Dalkey SAC*,*
- *North Channel SAC*,*
- *West Wales Marine / Gorllewin Cymru Foro SAC*,*
- *Bristol Channel Approaches / Dynesfeydd Môr Hafren SAC*,*
- *Mers Celtiques - Talus du golfe de Gascogne SCI*,*
- *North Anglesey Marine / Gogledd Môn Foro SAC*,*
- *Lambay Island SAC*,*
- *Nord Bretagne DH SAC*,*
- *Ouessant-Molène SAC*,*
- *Abers -Côte des legends SAC*,*
- *Chaussée de Sein SAC*,*

- *Côte de Granit rose-Sept-Iles SAC**,*
- *Baie de Morlaix SAC**,*
- *Côtes de Crozon SAC**,*
- *Récifs et landes de la Hague SAC**,*
- *Anse de Vauville SAC**,*
- *Banc et récifs de Surtainville SAC**,*
- *Baie du Mont Saint-Michel SAC**,*
- *Estuaire de la Rance SAC**,*
- *Baie de Lancieux SAC, Baie de l'Arguenon, Archipel de Saint Malo et Dinard SAC**,*
- *Cap d'Erquy-Cap Fréhel SAC**,*
- *Baie de Saint-Brieuc SAC**,*
- *Tregor Goëlo Es SAC**,*
- *Mid-Clare Coast SPA*
- *Slyne Head to Ardmore Point Islands SPA*
- *Inishmore SPA*
- *Cruagh Island SPA*
- *River Shannon and River Fergus Estuaries SPA*
- *Cliffs of Moher SPA*
- *Illaunonearaun SPA*
- *High Island, Inishark and Duvillaun SPA*
- *Inner Galway Bay SPA*
- *Illaunnanoon SPA*
- *Magharee Islands SPA*
- *Clare Island SPA*
- *Loop Head SPA*
- *Bills Rock SPA*
- *Dingle Peninsula SPA*
- *Duvillaun Islands SPA*
- *Inishglora and InisKeeragh SPA*
- *Blasket Islands SPA*
- *Puffin Islands SPA*
- *Iveragh Peninsula SPA*
- *Skelligs SPA*
- *Stages of Broadhaven SPA*
- *Eirk SPA*
- *The Gearagh SPA*
- *Deenish Island and Scariff Island SPA*
- *Clonakilty SPA*
- *Illanmaster SPA*
- *The Bull and The Cow Rocks SPA*
- *Beara Peninsula SPA*
- *Aughris Head SPA*
- *West Donegal Coast SPA*
- *Tory Island SPA*
- *Horn Head to Fanad Head SPA*
- *Saltee Islands SPA*
- *Mingulay and Berneray SPA*
- *Skomer, Skokholm and the Seas off Pembrokeshire / Sgomer, Sgogwm a Moroedd Penfro SPA*
- *Rum SPA*
- *Seas off St Kilda SPA*
- *St Kilda SPA*
- *Copeland Islands SPA*
- *Glannau Aberdaron ac Ynys Enlli/ Aberdaron Coast and Bardsey Island SPA*

- *Shiant Isles SPA*
- *Flannan Isles SPA*
- *Lambay Island SPA*
- *Ouessant-Molène SPA (France)*
- *Handa SPA*
- *Cape Wrath SPA*
- *Cote de Granit Rose-Sept Iles SPA*
- *Camaret SPA*
- *North Rona and Sula Sgeir SPA*
- *North Caithness Cliffs SPA*
- *Hoy SPA*
- *Cap d'Erquy-Cap Fréhel SPA (France)*
- *Rousay SPA*
- *West Westray SPA*
- *Copinsay SPA*
- *East Caithness Cliffs SPA*
- *Calf of Eday SPA*
- *Iles Houat-Hoedic SPA (France)*
- *Falaise du Bessin Occidental SPA (France)*
- *Seas off Foula SPA*
- *Fair Isle SPA*
- *Littoral seino-marin SPA*
- *Troup, Pennan and Lion's Heads SPA*
- *Foula SPA*
- *Sumburgh Head SPA*
- *Buchan Ness to Collieston Coast SPA*
- *Noss SPA*
- *Hermaness, Saxa Vord and Valla Field SPA*
- *Fetlar SPA*
- *Tullaher Lough and Bog SAC*

As a result, an Appropriate Assessment is required, and a Natura Impact Statement has been prepared.'

The Offshore NIS concluded:

'This NIS (Volumes 1 and 2) has assessed the impacts of the construction, operations and maintenance and decommissioning of the Project on European Sites and their relevant QI to determine whether the Project will have an adverse effect on the integrity of European Sites, either alone or in combination with other plans or projects and in light of the conservation objectives of the sites. The assessment concluded that there will be no adverse effect on the integrity of the

- *Inishmore Island SAC,*
- *Kilkieran Bay and Islands SAC,*
- *Lower River Shannon SAC,*
- *Slyne Head Peninsula SAC,*
- *Slyne Head Islands SAC,*
- *West Connacht Coast SAC,*
- *Galway Bay Complex SAC,*
- *Blasket Islands SAC,*
- *Duvillaun Islands SAC,*
- *Connemara Bog Complex SAC,*
- *Twelve Bens/Garraun Complex SAC,*
- *Maumturk Mountains SAC,*
- *Lough Corrib SAC,*
- *Mweelrea/Sheeffry/Erriff Complex SAC,*

- *Inishmaan Island SAC,*
- *Carrowmore Point to Spanish Point and Islands SAC,*
- *Carrowmore Dunes SAC,*
- *Kilkee Reefs SAC,*
- *Kenmare River SAC,*
- *Hook Head SAC,*
- *Belgica Mound Province SAC,*
- *Roaringwater Bay and Islands SAC,*
- *Gweedore Bay and Islands SAC,*
- *Bunduff Lough and Machair/Trawalua/Mullaghmore SAC,*
- *St John's Point SAC,*
- *Carnsore Point SAC,*
- *Blackwater Bank SAC,*
- *Lough Swilly SAC,*
- *Codling Fault Zone SAC,*
- *Rockabill to Dalkey SAC,*
- *North Channel SAC,*
- *West Wales Marine / Gorllewin Cymru Foro SAC,*
- *Bristol Channel Approaches / Dynesfeydd Môr Hafren SAC,*
- *Mers Celtiques - Talus du golfe de Gascogne SCI,*
- *North Anglesey Marine / Gogledd Môn Foro SAC,*
- *Lambay Island SAC,*
- *Nord Bretagne DH SAC,*
- *Ouessant-Molène SAC,*
- *Abers -Côte des legends SAC,*
- *Chaussée de Sein SAC,*
- *Côte de Granit rose-Sept-Iles SAC,*
- *Baie de Morlaix SAC,*
- *Côtes de Crozon SAC,*
- *Récifs et landes de la Hague SAC,*
- *Anse de Vauville SAC,*
- *Banc et récifs de Surtainville SAC,*
- *Baie du Mont Saint-Michel SAC,*
- *Estuaire de la Rance SAC,*
- *Baie de Lancieux, Baie de l'Arguenon, Archipel de Saint Malo et Dinard SAC,*
- *Cap d'Erquy-Cap Fréhel SAC,*
- *Baie de Saint-Brieuc SAC,*
- *Tregor Goëlo Es SAC,*
- *Mid-Clare Coast SPA*
- *Slyne Head to Ardmore Point Islands SPA*
- *Inishmore SPA*
- *Cruagh Island SPA*
- *River Shannon and River Fergus Estuaries SPA*
- *Cliffs of Moher SPA*
- *Illaunonearaun SPA*
- *High Island, Inishark and Duvillaun SPA*
- *Inner Galway Bay SPA*
- *Illaunnanoon SPA*
- *Magharee Islands SPA*
- *Clare Island SPA*
- *Loop Head SPA*
- *Bills Rock SPA*
- *Dingle Peninsula SPA*
- *Duvillaun Islands SPA*
- *Inishglora and InisKeeragh SPA*

- *Blasket Islands SPA*
- *Puffin Islands SPA*
- *Iveragh Peninsula SPA*
- *Skelligs SPA*
- *Stages of Broadhaven SPA*
- *Eirk SPA*
- *The Gearagh SPA*
- *Deenish Island and Scariff Island SPA*
- *Clonakilty SPA*
- *Illanmaster SPA*
- *The Bull and The Cow Rocks SPA*
- *Beara Peninsula SPA*
- *Aughris Head SPA*
- *West Donegal Coast SPA*
- *Tory Island SPA*
- *Horn Head to Fanad Head SPA*
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- *Skomer, Skokholm and the Seas off Pembrokeshire / Sgomer, Sgogwm a Moroedd Penfro SPA*
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- *Seas off St Kilda SPA*
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- *Hermaness, Saxa Vord and Valla Field SPA*
- *Fetlar SPA*
- *Tullaher Lough and Bog SAC*

either as a result of the Project alone or in combination with other plans or projects, provided that the mitigation listed is adhered to.

Therefore, it can be objectively concluded, following an examination, analysis and evaluation of the relevant information, including in particular the nature of predicted impacts from the Project, that the Project, individually or in combination with other plans or projects, will not adversely affect the integrity of any European Site in light of its conservation objectives and best scientific information, and there is no reasonable scientific doubt in relation to this conclusion.'

As such, it can be concluded that the Offshore Site will not have an adverse impact on any European Sites designated for birds, either alone or in combination with other plans or projects.

11.10 Residual Effects

With the implementation of the measures included in the Project design (Table 11-5), the residual effects are as outlined in the assessment provided in Section 11.8 above.

11.11 Cumulative Effects

This section outlines the cumulative impact assessment on Offshore Ornithology and takes in account the impacts of the Offshore Site alone, together with other plans and projects. As outlined in the Cumulative Impact Assessment Methodology section (Chapter 4), the screening process involved determination of appropriate search areas for projects, plans and activities and Cumulative Study Area for potential cumulative impacts. These were then screened according to the level of detail publicly available and the potential for interactions with regard to the presence of an impact pathway as well as spatial and temporal overlap.

The projects and plans selected as potentially relevant to the assessment of cumulative effects on offshore ornithology receptors were based upon an initial screening exercise undertaken on a long list of existing and reasonably foreseeable projects and plans.

Projects other than OWF projects e.g. dredging activities or port extensions have been screened out of the cumulative effects assessment on the basis that there is low potential for cumulative effects on offshore ornithology with Sceirde Rocks Offshore Wind Farm because the contribution from Sceirde Rocks Offshore Wind Farm in terms of temporary habitat loss/disturbance and increased suspended sediment concentrations (SSCs) is predicted to be small (and even if these occurred at the same time this would not constitute a significant effect).

For the breeding season Cumulative Study Area only consented or submitted OWF projects within the Offshore Ornithology Regional Study Area (509.4 km) were considered to have the potential to add any direct or indirect cumulative impact to offshore ornithology receptors in the breeding season. The 509.4 km distance is the breeding season mean maximum (+1S.D.) foraging range for gannet, and this is considered appropriate to use here as gannet is considered a key species in terms of potential collision and displacement impacts. Although other species such as Manx shearwater and fulmar have larger foraging ranges during the breeding season, these species are not considered to be at risk of potential displacement or collision effects, based on reviews of evidence from operational OWFs (e.g. Dierschke et al., 2016). OWF projects at greater distances were screened out on the basis of the very low likelihood of seabirds from breeding colonies beyond this distance foraging within the Sceirde Rocks Offshore Wind Farm array area in the breeding season (Table 11-58). Future projects that have yet to submit an EIAR were also screened out on the basis of there being insufficient data publically available to undertake any assessment.

In the non-breeding season, a similar cumulative study area was considered, with all operational, consented or submitted OWF projects within Irish waters and west coast of the UK included in the CEA.

Table 11-58 Distances of other OWF projects considered within the CEA for Offshore Ornithology

Project	Status	Distance from Sceirde Rocks	Screened IN/OUT of CEA
Arklow Bank Phase I	Operational	611.7 km	OUT
Arklow Bank Phase II	Submitted	612.5 km	OUT
Codling Wind Park	Submitted	645.5 km	OUT
Dublin Array	Submission Due	665.9 km	OUT
NISA	Submitted	681.2 km	OUT
Oriel Wind Farm	Submitted	663.0 km	OUT
Gwynt y Mor	Operational	760.73	OUT
Burbo Bank Extension	Operational	782.39	OUT
Burbo Bank	Operational	785.23	OUT
Walney 2	Operational	711.91	OUT
Walney 1	Operational	719.43	OUT
West of Duddon Sands	Operational	729.74	OUT
Barrow	Operational	734.32	OUT
Ormonde	Operational	725.93	OUT
Rhyl Flats	Operational	766.21	OUT
North Hoyle	Operational	770.13	OUT
TwinHub	Consented	589.41	OUT
Awel y Môr	Consented	753.49	OUT
Erebus	Consented	568.68	OUT
Mona	Submitted	723.93	OUT
Morecambe	Submitted	740.90	OUT
Morgan	Submitted	716.84	OUT
Whitecross	Submitted	600.97	OUT

Project	Status	Distance from Sceirde Rocks	Screened IN/OUT of CEA
West of Orkney	Submitted	856.85	OUT

As there are no operational, consented or submitted OWF projects within 509.4 km, it is considered that there will be no cumulative effects on offshore ornithology arising in the breeding season. Similarly in the non-breeding season, when seabirds are not linked to their breeding colonies, it is considered that the distance between Sceirde Rocks Offshore Wind Farm and other operational, consented or submitted OWF projects will make the potential for any significant cumulative interactions very unlikely. Therefore, cumulative effects between Sceirde Rocks Offshore Wind Farm and other operational, consented or submitted OWF projects in Irish and west coast UK are not considered further in this assessment.

11.12 Conclusion

In conclusion, the offshore ornithology impact assessment has assessed the potential effects resulting from: disturbance and displacement on key bird species as a result of increased vessel activity and other construction/decommissioning activity, indirect effects on foraging seabirds as a result of habitat loss/displacement of prey species, disturbance and displacement on key bird species as a result of increased vessel activity and other maintenance activities, displacement and barrier effects on key bird species within the OAA and appropriate buffer from offshore infrastructure, mortality of key bird species as a result of collision with offshore wind turbines, and disturbance from aviation and navigation lighting over the lifetime of the Project. A number of seabird species have been considered within the assessment. The assessment has concluded that the effect pathways would be Not Significant for all offshore ornithology receptors. This includes the conclusions of the cumulative effects assessment.