Morecambe Offshore Windfarm: Generation Assets

Environmental Statement

Volume 5

Appendix 11.2 Marine Mammal Information and Survey Data

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Glossary of Acronyms

ADD	Acoustic deterrent device
AfL	Agreement for Lease Area
ASCOBANS	Agreement on the Conservation of Small Cetaceans of the Baltic and North Seas
ASSI	Area of Special Scientific Interest
BEIS[1]	Department for Business, Energy and Industrial Strategy[1]
BSH	German Federal Maritime and Hydrographic Agency
CBD	Convention on Biological Diversity
CCW	Countryside Council for Wales
CEA	Cumulative Effects Assessment
CGNS	Celtic and Greater North Seas
CI	Confidence Interval
CIS	Celtic and Irish Seas
CITES	Convention on International Trade in Endangered Species of Wild Fauna and Flora
CL	Confidence Limit
CODA	Cetacean Offshore Distribution and Abundance in the European Atlantic
CPOD	Cetacean Porpoise Detectors
CRoW	Countryside and Rights of Way Act
CV	Coefficient of Variation
CWT	Cumbria Wildlife Trust
DCO	Development Consent Order
DECC	Department of Energy and Climate Change
DEFA	Department of Environment, Food and Agriculture (IoM)
DESNZ	Department for Energy Security and Net Zero
DRC	Dose-Response Curve
EDR	Effective Deterrent Ranges
EIA	Environmental Impact Assessment
EPP	Evidence Plan Process
EPS	European Protected Species
ES	Environmental Statement
ETG	Expert Topic Group

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

EU	European Union
FCS	Favourable Conservation Status
GES	Good Environmental Status
GSD	Ground Sample Distance
HRA	Habitat Regulations Assessment
IAMMWG	Inter-Agency Marine Mammal Working Group
ICES	International Council for the Exploration of the Sea
IoM	Isle of Man
iPCoD	Interim Population Consequences of Disturbance
IS	Irish Sea
IWC	International Whaling Commission
JCDP	The Joint Cetacean Data Programme
JCP	Joint Cetacean Protocol
JNCC	Joint Nature Conservation Committee
KDE	Kernel Density Estimation
MAC	Maritime Area Consents
META	Marine Energy Test Area
MMMP	Marine Mammal Mitigation Protocol
MMO	Marine Management Organisation
MNR	Marine Noise Registry
MNRs	Marine Nature Reserves
MOD	Military of Defence
MPS	Marine Policy Statement
MSFD	Marine Strategy Framework Directive
MSR	Marine Strategy Regulations
MU	Management Units
MWDW	Manx Whale and Dolphin Watch
NCMPA	Nature Conservation Marine Protected Areas
NE	Natural England
NI	Northern Ireland
NMFS	National Marine and Fisheries Service
NNR	National Nature Reserve
NOAA	National Oceanic and Atmospheric Administration
NPWS	National Parks and Wildlife Service NPWS
NS	North Sea
NW	North-West

OSPAR	Oslo and Paris Convention for the Protection of the Marine Environment of the North-East Atlantic
PDE	Project Design Envelope
PEIR	Preliminary Environmental Information Report
PEMP	Project Environmental Management Plan
PTS	Permanent Threshold Shift
RA	Risk Assessment
RIAA	Report to Inform Appropriate Assessment
RoI	Republic of Ireland
SCANS	Small Cetaceans in the European Atlantic and North Sea
SCOS	Special Committee on Seals
SD	Standard deviation
SEL	Sound Exposure Level
SEL_{cum}	Sound Exposure Level from cumulative exposure
SEL_{SS}	Sound Exposure Level from single strike
SMRU	Sea Mammal Research Unit
SPA	Special Protection Area
SPL_{peak}	peak Sound Pressure Level
TTS	Temporary Threshold Shift
TWT	The Wildlife Trust
UK	United Kingdom
UXO	Unexploded Ordnance
WS	West Scotland

Glossary of Unit Terms

μPa	Micro Pascal
dB	Decibel
kHz	Kilohertz
km	Kilometre
m	Metre
nm	Nautical mile
s	Second

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Glossary of Terminology

Absolute abundance	The most accurate estimate of population size. In the case of diving birds and mammals, this includes an estimate for the number that are believed to be submerged at the time of survey.
Applicant	Morecambe Offshore Windfarm Ltd
CAVOK	“Ceiling and Visibility OK” – term used for aviation surface weather observation reports.
Coefficient of Variation *CV (%)*	The coefficient of variation is a standard measure that describes the dispersion of data points around the mean. The lower the CV the more precise the estimate. It is calculated as the SD/mean.
Confidence limit *(CL)*	The upper and lower values that define the range of the 95% confidence interval.
Density estimate *(animals/km^2)*	The average number of animals per square km surveyed.
Evidence Plan Process (EPP)	A voluntary consultation process with specialist stakeholders to agree the approach, and information to support, the Environmental Impact Assessment (EIA) and Habitats Regulations Assessment (HRA) for certain topics. The EPP provides a mechanism to agree the information required to be submitted to the Planning Inspectorate as part of the Development Consent Order (DCO) application. This function of the EPP helps Applicants to provide sufficient information in their application, so that the Examining Authority can recommend to the Secretary of State whether or not to accept the application for examination and whether an Appropriate Assessment is required.
Expert Topic Group (ETG)	A forum for targeted engagement with regulators and interested stakeholders through the EPP.
Generation Assets (the Project)	Generation assets associated with the Morecambe Offshore Windfarm. This is infrastructure in connection with electricity production, namely the fixed foundation wind turbine generators (WTGs), inter-array cables, offshore substation platform(s) (OSP(s)) and possible platform link cables to connect OSP(s)
Inter-array cables	Cables which link the WTGs to each other and the OSP(s).
Landfall	Where the offshore export cables would come ashore.
Offshore export cables	The cables which would bring electricity from the offshore substation platform to the landfall.
Offshore substation platform(s) (OSP(s))	A fixed structure located within the windfarm site, containing electrical equipment to aggregate the power from the WTGs and convert it into a more suitable form for export to shore.
Platform link cable	An electrical cable which links one or more OSP(s).
Population estimate *(number)*	The mean number of animals estimated within the survey area.

Relative abundance	In the case of diving birds and mammals, this is the estimated population size based on animals recorded on or above the sea surface and does not account for any that may be diving and thus submerged at the time of survey.
Safety zones	An area around a structure or vessel which should be avoided, as set out in Section 95 of the Energy Act 2004 and the Electricity (Offshore Generating Stations) (Safety Zones) (Application Procedures and Control of Access) Regulations 2007.
Scour protection	Protective materials to avoid sediment being eroded away from the base of the foundations due to the flow of water.
Standard deviation *(SD)* of population estimate	The amount of variation or dispersion of a set of values.
Study area	This is an area which is defined for EIA topic, which includes the offshore development area, as well as potential spatial and temporal considerations of the impacts on relevant receptors. The study area for each EIA topic is intended to cover the area within which an effect can be reasonably expected.
Technical stakeholders	Technical stakeholders are organisations with detailed knowledge or experience of the area within which the Project is located and/or receptors which are considered in the EIA and HRA. Examples of technical stakeholders include the Marine Management Organisation (MMO), local authorities, Natural England (NE) and the Royal Society for the Protection of Birds (RSPB).
Transmission Assets	The transmission assets refers to Morgan and Morecambe Offshore Wind Farms export cables.
Windfarm site	The area within which the WTGs, inter-array cables, OSP(s) and platform link cables will be present.
Wind turbine generators (WTGs)	A fixed structure located within the windfarm site that converts the kinetic energy of wind into electrical energy.
95% confidence interval *(CI)*	A measure of uncertainty in the mean value. If the analysis was repeated, 95% of the time the mean population estimate would fall within this range. The smaller the CI range the more confident we can be that the mean estimate is an accurate reflection of the true population size.

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11.2 The future of renewable energy

A leading developer in Offshore Wind Projects

1 Introduction

1. This Appendix provides additional detail on the marine mammal baseline to support **Chapter 11 Marine Mammals** (Document Reference 5.1.11) of the Environmental Statement (ES) for the Morecambe Offshore Windfarm Generation Assets (the Project).

2. The document sets out additional information in relation to the marine mammal species scoped into the ES, including identifying the study areas applied for each species and relevant policy, legislation and guidance. Additional baseline information and data from the two-year site-specific aerial surveys conducted for the Project were also summarised, along with relevant density and abundance estimates.

1.1 Marine mammal species

3. The following marine mammal species have been scoped into the assessment:

- Harbour porpoise (*Phocoena phocoena*)
- Bottlenose dolphin (*Tursiops truncatus*)
- Common dolphin (*Delphinus delphis*)
- Risso's dolphin (*Grampus griseus*)
- White-beaked dolphin (*Lagenorhynchus albirostris*)
- Minke whale (*Balaenoptera acutorostrata*)
- Grey seal (*Halichoerus grypus*)
- Harbour seal (*Phoca vitulina*)

4. These species were determined from the site-specific aerial surveys (**Section 3**) and other data sources and were discussed and agreed with the marine mammal Expert Topic Group (ETG).

1.1.1 Study area

5. The study area for the marine mammal assessment has been defined on the basis that marine mammals are highly mobile and transitory in nature. It was, therefore, necessary to examine species occurrence, not only within the windfarm site, but also over the wider area.

1.1.1.1 Cetaceans

6. Management Units (MUs) provide an indication of the spatial scales at which the effects of plans and projects alone, and in-combination, need to be assessed for the key cetacean species in United Kingdom (UK) waters, with

consistency across the UK (Inter-Agency Marine Mammal Working Group (IAMMWG), 2023). The study area, MUs and reference populations have been determined, based on the most relevant information and scale at which potential effects from the Project-alone, and together with other plans and projects, could occur.

7. The MUs are defined geographical areas in which individuals of a particular species are found and management of human activity is applied (IAMMWG 2023). For this reason, delineation of cetacean MUs have been, as far as is practical, aligned with the International Council for the Exploration of the Sea (ICES) Subarea and/or Divisions that are used for implementation of fisheries management measures, as recommended by the ICES Working Group of Marine Mammal Ecology.

8. For each marine mammal species, the study areas have been defined based on the relevant MUs as outlined in **Table 1.1,** which provide relevant spatial scale for assessment of environmental impacts (IAMMWG, 2023)**.**

Table 1.1 Marine mammal species relevant management unit

Species	Management unit	Source	Study area Plate reference
Harbour porpoise	Celtic and Irish Sea (CIS) MU	IAMMWG, 2023	**Plate 1.1**
Bottlenose dolphin	Irish Sea (IS) MU	IAMMWG, 2023	**Plate 1.2**
Common dolphin	Celtic and Greater North Seas (CGNS) MU	IAMMWG, 2023	**Plate 1.3**
Risso’s dolphin	CGNS MU	IAMMWG, 2023	**Plate 1.3**
White-beaked dolphin	CGNS MU	IAMMWG, 2023	**Plate 1.3**
Minke whale	CGNS MU	IAMMWG, 2023	**Plate 1.3**

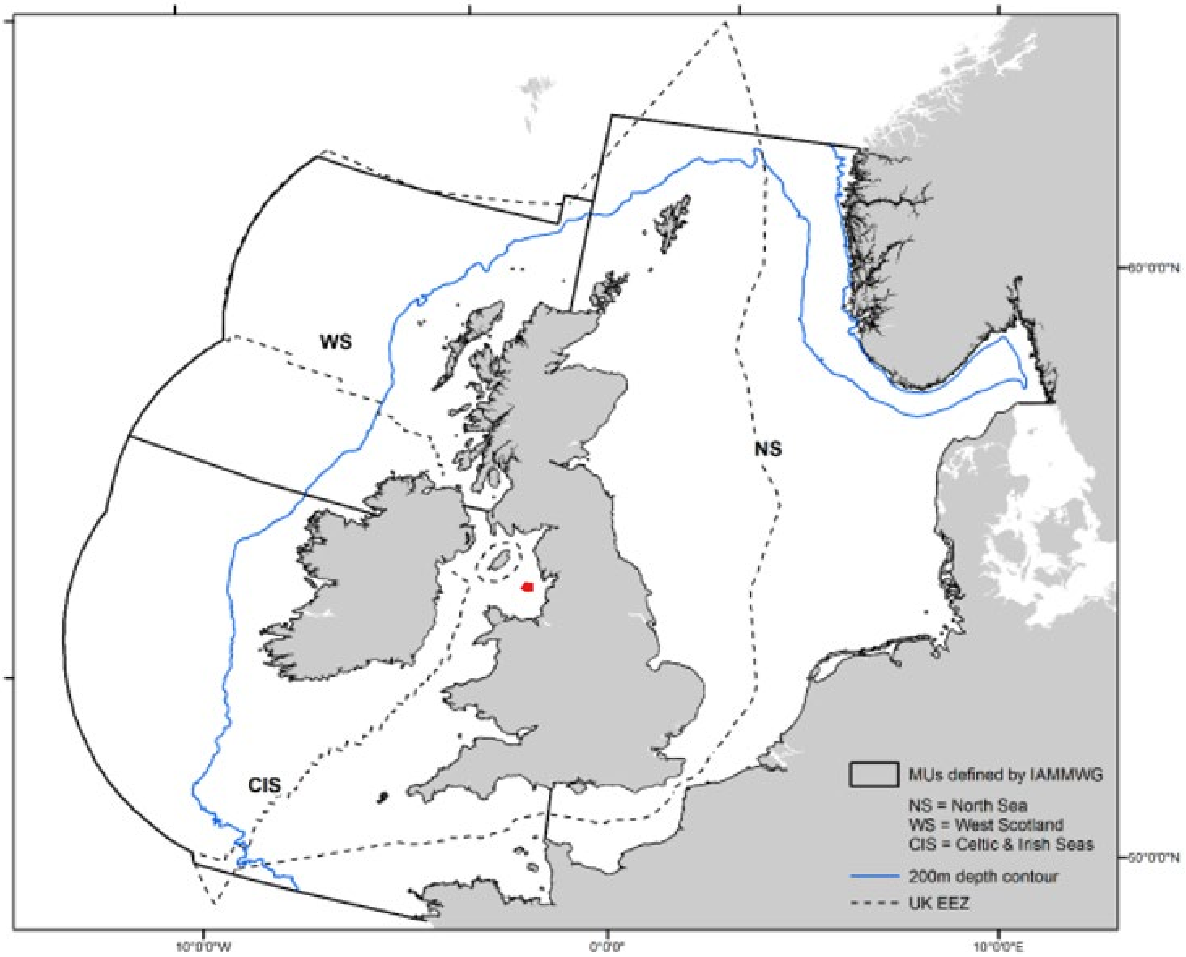


Plate 1.1 Harbour porpoise MUs; Project location is approximate (in red) (IAMMWG, 2023)

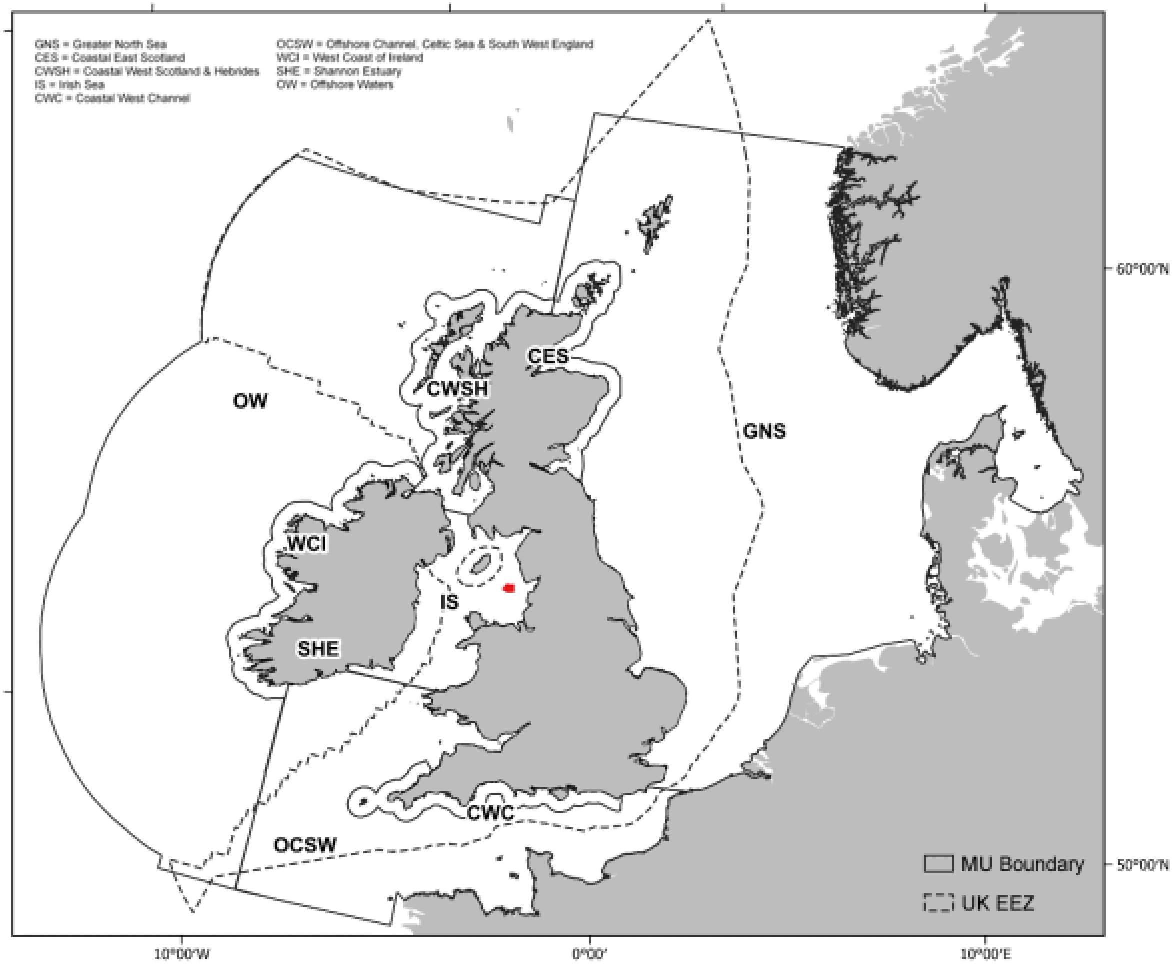


Plate 1.2 Bottlenose dolphin MUs; Project location is approximate (in red) (IAMMWG, 2023)

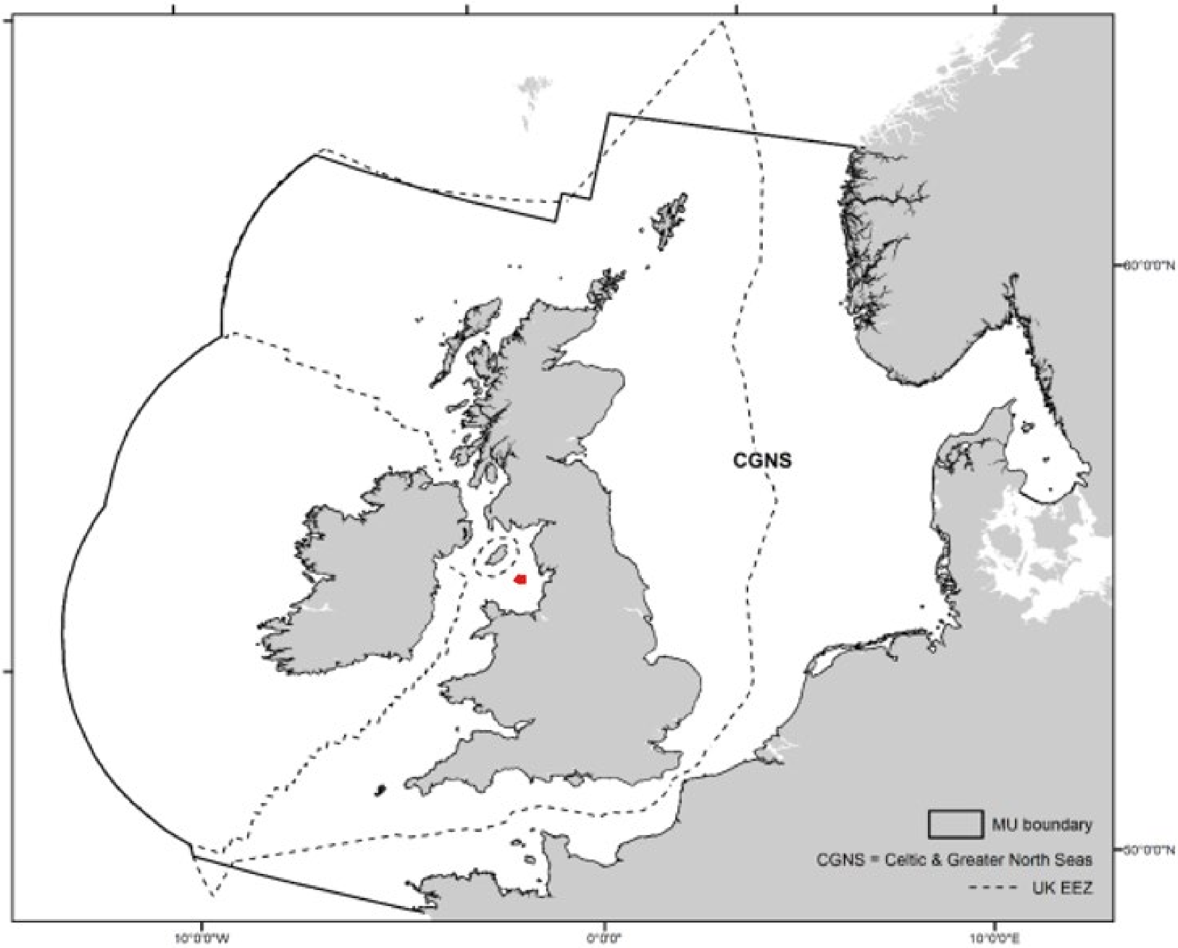


Plate 1.3 MU for common dolphin, Risso's dolphin, white-beaked dolphin and minke whale; Project location is approximate (in red) (IAMMWG, 2023)

1.1.1.2 Pinnipeds

9. Based on the movements of grey seal, and potential connectivity with the Project, the relevant MUs (**Plate 1.4** Special Committee on Seals (SCOS), 2020; **Plate 1.6** National Parks and Wildlife Service (NPWS), 2019) were:

- North-West (NW) England MU (within which the Project is located)
- Wales MU
- Northern Ireland (NI) MU
- Isle of Man (IoM) MU
- Republic of Ireland (RoI) east and southeast MUs

10. **Paragraph 201** provides a brief discussion regarding the use of the Oslo and Paris Convention for the Protection of the Marine Environment (OSPAR) Region III over the use of the seal MUs. For harbour seal, the relevant MUs (**Plate 1.5**; SCOS, 2022) were:

- North-West (NW) England MU
- Northern Ireland (NI) MU

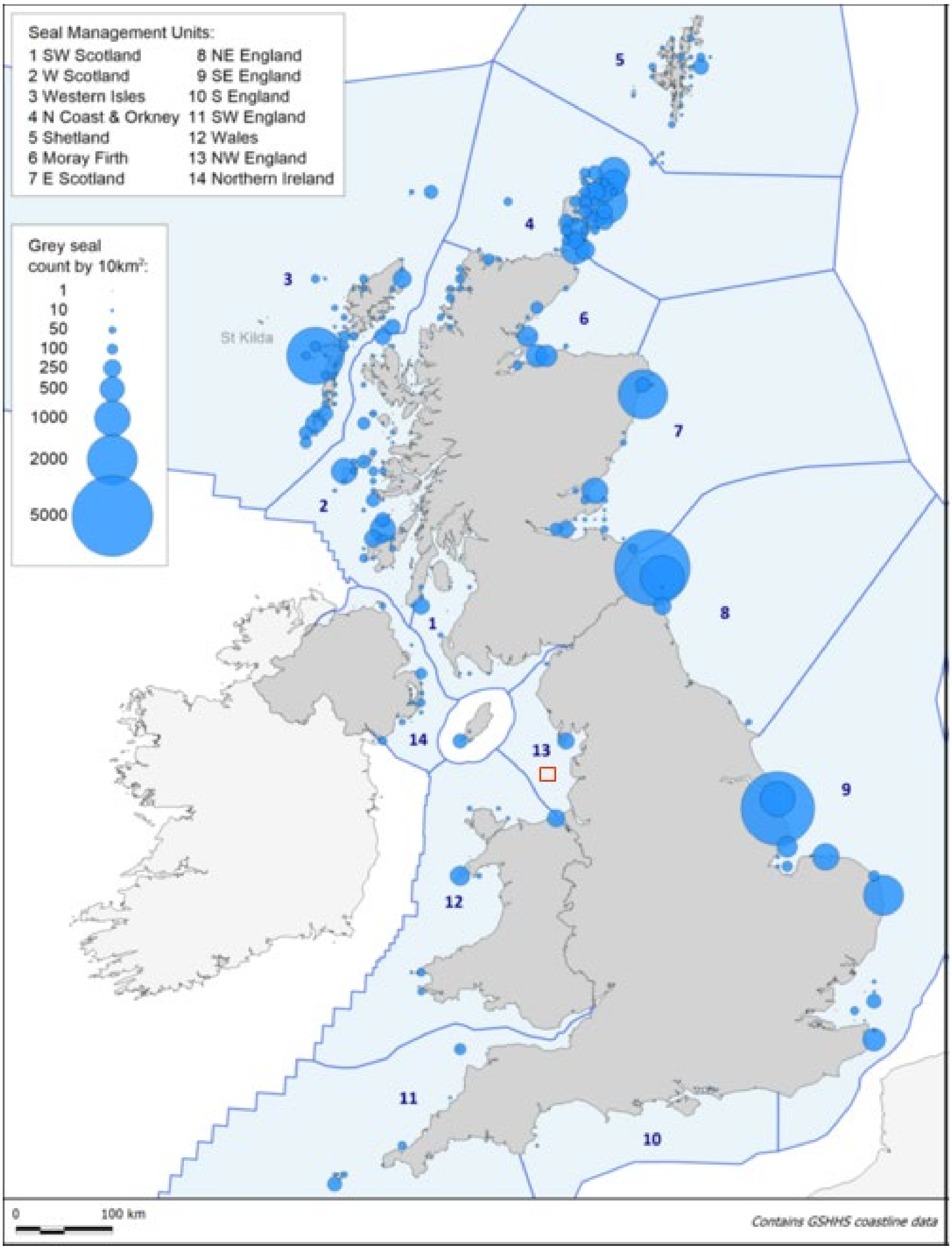


Plate 1.4 Grey seal MUs in the United Kingdom; Project location is approximate (in red) (SCOS, 2022)

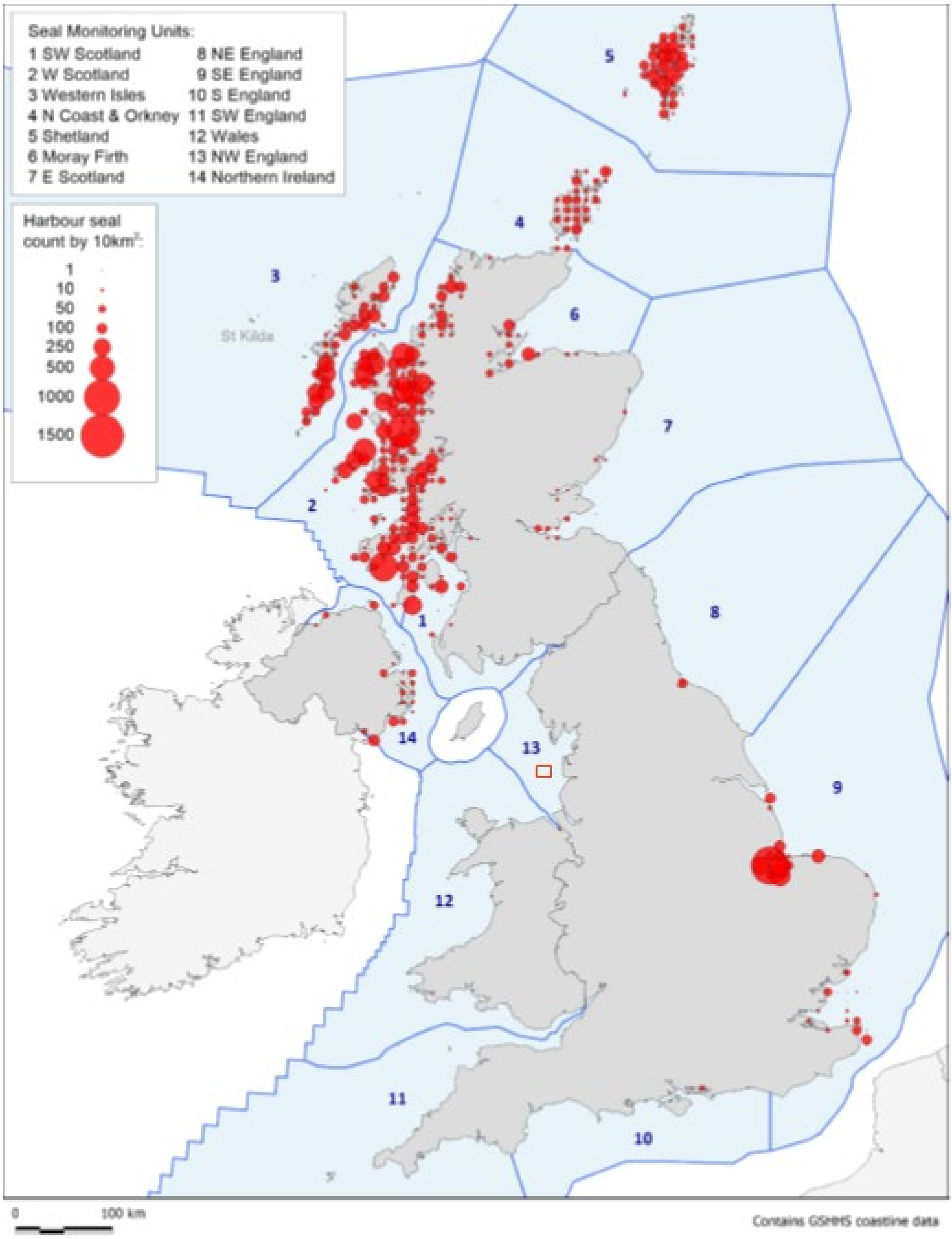


Plate 1.5 Harbour seal MUs in the United Kingdom; Project location is approximate (in red) (SCOS, 2022)

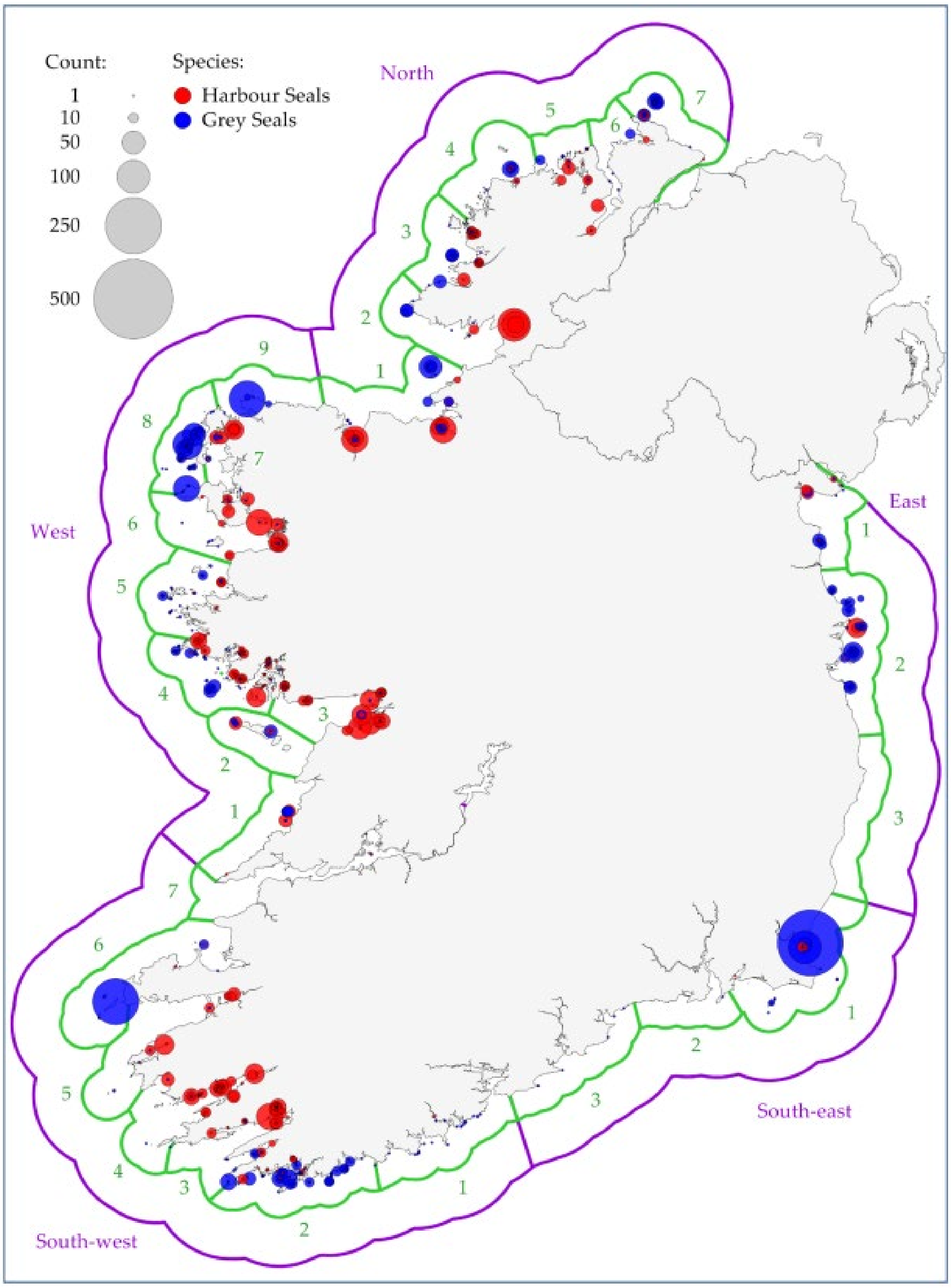


Plate 1.6 Seal MUs in the Republic of Ireland (Morris and Duck, 2019)

2 Policy, legislation, and guidance

2.1 Introduction

11. As outlined in **Chapter 11 Marine Mammals**, and detailed further below, there were a number of pieces of legislation, policy and guidance applicable to the assessment of marine mammals. This information is set out below, under the following:

- National marine policies
- Other national and international legislation for marine mammals
- European Protected Species (EPS) guidance
- Marine Wildlife Licence Requirements
- Legislation under Manx law

2.2 National marine policies and legislation/directives

12. Key national legislation and policy applicable to the marine mammal assessment included:

- The Marine Policy Statement (MPS) (UK Government, 2011)
- The Marine Strategy Framework Directive (MSFD) 2008/56/EC (EC, 2008), transposed into UK law by the Marine Strategy Regulations (MSR) 2010 SI 2010/1627 (United Kingdom (UK) Government, 2010)

2.2.1 The Marine Policy Statement

13. The MPS (UK Government, 2011) provided a high-level approach to marine planning and the general principles for decision making. It set out the framework for environmental, social and economic considerations that need to be taken into account in marine planning. The high-level objective of 'Living within environmental limits' covers the points relevant to marine mammals, which required that:

- Biodiversity is protected, conserved and, where appropriate, recovered and loss has been halted
- Healthy marine and coastal habitats occur across their natural range and are also able to support strong, biodiverse biological communities and the functioning of healthy, resilient and adaptable marine ecosystems
- Our oceans support viable populations of representative, rare, vulnerable and valued species

2.2.2 The Marine Strategy Framework Directive

14. Annex I of the MSFD (EC, 2008) stated that to ensure that good environmental status is met, the following must be considered:

- Biological diversity should be maintained
- The quality and occurrence of habitats, as well as the distribution and abundance of species are in line with prevailing physiographic, geographic and climatic conditions
- All elements of the marine food web, to the extent that they are known, occur at normal abundance and diversity levels capable of ensuring the long-term abundance of the species and the retention of their full reproductive capacity
- Concentrations of contaminants are at levels not giving rise to pollution effects
- Properties and quantities of marine litter do not cause harm to the coastal and marine environment
- Introduction of energy, including underwater noise, is at levels that do not adversely affect the marine environment

2.2.3 The Marine Strategy Regulations

15. The MSR 2010 (as amended) established a framework of measures to achieve or maintain good environmental status (GES) in the marine environment by the year 2020[2]. Qualitative descriptors for determining GES relevant to marine mammals include:

- Biological diversity is maintained. The quality and occurrence of habitats and the distribution and abundance of species are in line with prevailing physiographic, geographic and climatic conditions
- All elements of the marine food webs, to the extent that they are known, occur at normal abundance and diversity and levels capable of ensuring the long-term abundance of the species and the retention of their full reproductive capacity
- Permanent alteration of hydrographical conditions does not adversely affect marine ecosystems
- Concentrations of contaminants are at levels not giving rise to pollution effects
- Introduction of energy, including underwater noise, is at levels that do not adversely affect the marine environment

[2] An update is expected in 2024 following review (including if BES has been achieved)

2.3 Other national and international legislation for marine mammals

16. **Table 2.1** provides an overview of national and international legislation in relation to marine mammals.

17. It should be noted that the Isle of Man, a self-governing British Crown dependency in the Irish Sea, is a signatory to most legislation concerning the UK including: Convention of Biological Diversity, Agreement on the Conservation of Small Cetaceans of the Baltic and North Seas (ASCOBANS), Bonn and Bern Convention, OSPAR and Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES).

Table 2.1 Summary table for national and international legislations relevant for marine mammals

Legislation	Level of protection	Species included	Details
Agreement on the Conservation of Small Cetaceans of the Baltic and North Seas (ASCOBANS)	International	Odontocetes	Formulated in 1992, this agreement has been signed by eight European countries bordering the Baltic and North Seas (including the English Channel) including the UK. Under the Agreement, provision was made for the protection of specific areas, monitoring, research, information exchange, pollution control and increasing public awareness of small cetaceans.
The Bern Convention 1979	International	All cetaceans, grey seal and harbour seal	The Convention conveyed special protection to those species that were vulnerable or endangered. Appendix II (strictly protected fauna): 19 species of cetacean. Appendix III (protected fauna): all remaining cetaceans, grey and harbour seal. Although an international convention, it was implemented within the UK through the Wildlife and Countryside Act 1981 (with any aspects not implemented via that route brought in by the Habitats Directive).
The Bonn Convention 1979	International	All cetaceans All marine turtle species	Protected migratory wild animals across all, or part of their natural range, through international co-operation, and related particularly to those species in danger of extinction. One of the measures identified was the adoption of legally binding agreements, including ASCOBANS.

Legislation	Level of protection	Species included	Details
EC Directive 92/43/EEC, adopted in 1992, known as the Habitats Directive	European	All cetaceans, grey and harbour seal All marine turtle species	Implemented the Convention on the Conservation of European Wildlife and Natural Habitats (the Bern Convention) and The Convention on the Conservation of Migratory Species of Wild Animals (the Bonn Convention). The Directive aimed to conserve natural habitats of wild fauna and flora and was intended to protect biodiversity by requiring Member States to take measures to maintain or restore natural habitats and wild species, including protection for specific habitats listed in Annex I and species listed in Annex II of the Directive. Annex IV also lists species in need of strict protection. The Conservation of Habitats and Species (Amendment) (European Union (EU) Exit) Regulations 2019 (2019 No. 579) set out the changes that applied since the UK left the European Union.
Oslo and Paris Convention for the Protection of the Marine Environment in the North-East Atlantic 1992 (OSPAR)	International	Bowhead whale *Balaena mysticetus*, northern right whale *Eubalaena glacialis*, blue whale *Balaenoptera musculus*, and harbour porpoise	OSPAR has established a list of threatened and/or declining species in the North East Atlantic. These species have been targeted as part of further work on the conservation and protection of marine biodiversity under Annex V of the OSPAR Convention. The list seeks to complement, but not duplicate, the work under the EC Habitats and Birds directives and measures under the Bern Convention and the Bonn Convention.

Legislation	Level of protection	Species included	Details
International Convention for the Regulation of Whaling 1956	International	All cetacean species	This Convention established the International Whaling Commission (IWC) who regulate the direct exploitation and conservation of large whales (in particular sperm and large baleen whales) as a resource and the impact of human activities on cetaceans. The regulation considered scientific matters related to small cetaceans, in particular the enforcing of a moratorium on commercial whaling which came into force in 1986.
Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) 1975	International	All cetacean species All marine turtle species	Prohibited the international trade in species listed in Annex 1 (including sperm whales, northern right whales, and baleen whales) and allowed for the controlled trade of all other cetacean species.
Convention on Biological Diversity (CBD) 1993	International	All marine mammal species	Required signatories to identify processes and activities that were likely to have impacts on the conservation of and sustainable use of biological diversity, inducing the introduction of appropriate procedures requiring an EIA and mitigation procedures.
The Conservation of Habitats and Species Regulations 2017 and The Conservation of Offshore Marine Habitats and Species Regulations 2017	National	All cetaceans, grey and harbour seal All marine turtle species	'The Habitats Regulations'. Provisions of The Habitats Regulations have been described further in **Chapter 11 Marine Mammals** It should be noted that the Habitats Regulations apply within the territorial seas and to marine areas within UK jurisdiction, beyond 12 nautical miles (nm).

Legislation	Level of protection	Species included	Details
The Wildlife and Countryside Act 1981 (as amended)	National	All cetaceans All marine turtle species	Schedule five: All cetaceans are fully protected within UK territorial waters. This protects them from killing or injury, sale, destruction of a particular habitat (which they use for protection or shelter) and disturbance. Schedule six: Short-beaked common dolphin, bottlenose dolphin and harbour porpoise; prevents these species from being used as a decoy to attract other animals. This schedule also prohibits the use of vehicles to take or drive them, prevented nets, traps or electrical devices from being set in such a way that would injure them and prevents the use of nets or sounds to trap or snare them.
The Countryside and Rights of Way Act (CRoW) 2000	National	All cetaceans	Under the CRoW Act 2000, it is an offence to intentionally or recklessly disturb any wild animal included under Schedule 5 of the Wildlife and Countryside Act.
Conservation of Seals Act 1970 (as amended)	National	Grey and harbour seal	As of 1st March 2021, a person would commit an offence if they intentionally or recklessly kill, injure or take a seal. The legislative changes in England and Wales, amended the Conservation of Seals Act 1970, prohibiting the intentional or reckless killing, injuring or taking of seals and removed the provision to grant licences for the purposes of protection, promotion or development of commercial fisheries or aquaculture activities. These changes were enacted to ensure compliance with the US Marine Mammal Protection Act Import Provision Rule.

Legislation	Level of protection	Species included	Details
Isle of Man Wildlife Act 1990	National	All cetaceans, seals and marine turtles	The 1990 Act is the primary wildlife protection legislation. It sets out schedules of Manx species of animal and plant that are protected by law from injury or disturbance. It also establishes the legal protection of Areas of Special Scientific Interest (ASSIs), Marine Nature Reserves (MNRs) as well as National Nature Reserves (NNRs).

2.4 European Protected Species guidance

18. All cetacean species listed as European Protected Species (EPS) under Annex IV of the Habitats Directive are protected from the deliberate killing (or injury), capture and disturbance throughout their range. Within the UK, The Habitats Directive was enacted through The Conservation of Habitats and Species Regulations 2017 and the Conservation of Offshore Marine Habitats and Species Regulations 2017. Under these Regulations, it is an offence if wild animals listed in Annex IV(a) (including cetaceans) are deliberately disturbed in such a way as to:

- Deliberately capture, injure or kill any EPS
- Deliberately disturb them
- Deliberately damage or destroy a breeding site or resting place

19. The Joint Nature Conservation Committee (JNCC), Natural England (NE) and the Countryside Council for Wales (CCW) (JNCC *et al.*, 2010) have produced draft guidance concerning the Habitat Regulations on the deliberate disturbance of marine EPS. This guidance provided an interpretation of the regulations in greater detail, including for pile driving operations (JNCC, 2010a), seismic surveys (JNCC, 2017) and the use of explosives (JNCC, 2010b[3]).

20. The draft guidance provided the following interpretations of deliberate injury and disturbance offences under the Habitats Regulations, as detailed in the paragraphs below:

> *"Deliberate actions are to be understood as actions by a person who knows, in light of the relevant legislation that applies to the species involved, and the general information delivered to the public, that his action will most likely lead to an offence against a species, but intends this offence or, if not, consciously accepts the foreseeable results of his action;*
>
> *Certain activities that produce loud sounds in areas where EPS could be present have the potential to result in an injury offence, unless appropriate mitigation measures are implemented to prevent the exposure of animals to sound levels capable of causing injury".*

[3] DRAFT guidelines for minimising the risk of injury to marine mammals from UXO clearance in the marine environment (JNCC, 2023) were issued for consultation in 2023. It is anticipated that the publication of the guidelines will occur after submission of this DCO Application and requirements will be updated accordingly.

21. For the purposes of marine users, the draft guidance stated that a disturbance which can cause offence should be interpreted as:

> *"Disturbance which is significant in that it is likely to be detrimental to the animals of an EPS or significantly affect their local abundance or distribution".*

22. The draft guidelines further stated that a disturbance offence would be more likely where an activity caused persistent noise in an area for long periods of time and highlighted that sporadic "trivial disturbance" should not be considered as a disturbance offence under Article 12.

23. Any action that could increase the risk of a long-term decline of the population, increase the risk of a reduction of the range of the species, and/or increase the risk of a reduction of the size of the habitat of the species could be regarded as a disturbance under the Regulations. For a disturbance to be considered non-trivial, the disturbance to marine EPS would need to be likely to at least increase the risk of a certain negative impact on the species at Favourable Conservation Status (FCS).

24. JNCC *et al*. (2010) stated that:

> *"In any population with a positive rate of growth, or a population remaining stable at what is assumed to be the environmental carrying capacity, a certain number of animals can potentially be removed as a consequence of anthropogenic activities (e.g., through killing, injury or permanent loss of reproductive ability), in addition to natural mortality, without causing the population to decrease in numbers, or preventing recovery, if the population is depleted. Beyond a certain threshold however, there could be a detrimental effect on the population".*

25. As per **Paragraph 17**, the same legislative protection for marine mammals outlined in this section and **Section 2.5** extends across the Irish Sea, and therefore to the IoM.

26. Further discussion on the use of thresholds for significance and the permanent or temporary nature of any disturbance has been considered by defining the magnitude of potential impact in the assessment (Section 11.4 of **Chapter 11 Marine Mammals**

2.5 European Protected Species requirements

27. Under the Habitats Regulations 2017, an EPS licence would be required if the risk of injury or disturbance to cetacean species, from any potential effect (i.e., underwater noise, collision risk) has been assessed as likely, following the application of mitigation. In English waters, this is referred to as marine wildlife licence. If such a licence is required, an application must be submitted, the assessment of which comprises three tests, namely:

- Whether the activity falls within one of the purposes specified in Regulation 55 of the Habitats Regulations. Only the purpose of "preserving public health or public safety or other imperative reasons of overriding public interest, including those of a social or economic nature and beneficial consequences of primary importance for the environment" is of relevance to marine mammals in this context
- That there are no satisfactory alternatives to the activity proposed (that would not incur the risk of offence)
- That the licensing of the activity will not result in a negative impact on the species'/population's FCS

28. A marine wildlife licence would consider all cetacean species at potential risk of injury or disturbance. It is likely that the Project would require a licence for disturbance to cetacean species as a result of the piling activities, dependent on the final design for infrastructure foundations.

29. There is currently no legislation that requires seals to be included under a marine wildlife licence; disturbance was not deemed to be an offence under the Conservation of Seals Act 1970, and in the case of injury to seals, the Marine Management Organisation (MMO) is currently only able to grant licences under very specific circumstances as listed under Section 10(1) of the Conservation of Seals Act 1970, which would not apply in the case that a marine wildlife licence was required for the construction of the Project.

30. Under the definitions of 'deliberate disturbance' in the Habitats Regulations, chronic exposure and/or displacement of animals could be regarded as a disturbance offence. Therefore, if these risks cannot be avoided, then the Applicant is likely to be required to apply for a marine wildlife licence from the MMO in order to be exempt from the offence.

31. If required, the marine wildlife licence application would be submitted post-consent. At that point, the Project Design Envelope (PDE) would have been further refined through detailed design and procurement activities and further detail would be available on the foundation type and techniques selected for the construction of the windfarm, as well as the mitigation measures proposed following the development of the Marine Mammal Mitigation Protocol (MMMP) for piling and Unexploded Ordnance (UXO) clearance.

2.6 Legislation under Manx law

32. Statutory marine and coastal conservation in the IoM is the responsibility of the Department of Environment, Food and Agriculture (DEFA) of the IoM Government. The main legislation available for protected species and habitats is the Wildlife Act (1990) and the Fisheries Act (2012).

33. There are several Marine Nature Reserves (MNRs) in Manx waters (IoM) and this is the main conservation designation available for subtidal sites which

have been designated under the Wildlife Act 1990. Historically many MNRs were closed or restricted areas, established for fisheries management and research. However, in 2008 the IoM MNRs project was founded, using existing data and new survey work to identify the most important marine habitats and species for protection.

34. Ramsey Bay was the first designated MNRs in 2011, which was followed by a further nine MNRs. These were re-designated on the 1st September 2018, and all are located within the 0-3nm boundary of Manx waters (**Plate 2.1**). In total, IoM MNRs cover 430km^2, 52% of the 0-3nm area or 11% of the territorial sea.

Plate 2.1 Isle of Man Marine Nature Reserves 2018

35. Of these ten MNRs, only one, Little Ness, does not include marine mammals as a designated feature. See **Table 2.2** for all MNRs and the marine mammal species that were featured in the designation list. Additional information on transboundary effects with the IoM has been discussed in Section 11.8.1 of **Chapter 11 Marine Mammals**.

Table 2.2 Isle of Man Marine Nature Reserves and their marine mammal designation features

Marine Nature Reserve	Harbour porpoise	Risso's dolphin	Bottlenose dolphin	Grey seal	Harbour seal	Minke whale
Baie ny Carrickey	✓	✓	✓	-	-	-
Calf and Wart Bank	✓	✓	-	-	-	-
Douglas	-	✓	✓	-	-	-
Langness	✓	✓	-	✓	✓	-
Laxey	✓	-	✓	-	-	✓
Little Ness	-	-	-	-	-	-
Niarbyl	✓	-	-	✓	-	-
Port Erin	✓	-	-	-	-	-
Ramsey	-	-	-	✓	✓	-
West Coast	✓	-	-	✓	✓	-

3 Site-specific surveys

3.1 Survey overview

36. In order to provide site specific and up to date information on which to base the impact assessment, site-specific aerial surveys were conducted for marine mammals and seabirds. HiDef Aerial Surveying Limited ('HiDef') collected high resolution aerial digital still imagery for marine megafauna (combined with ornithology surveys) over the survey area which included the windfarm Agreement for Lease Area (AfL)[4] and a custom 4km to 10km buffer. The buffer extended to 10km to the north and east due to proximity to Liverpool Bay Special Protection Area (SPA) for birds. The total survey area was 651km^2 (**Plate 3.1**).

37. Following Preliminary Environmental Information Report (PEIR), the windfarm site development area was reduced to 87km^2 with this revised windfarm site now forming the Application boundary. With the reduction in windfarm site, the survey custom buffer now extends 9km from the windfarm site to the west, 4km to the south and 10km to the north and east.

38. The monthly aerial surveys commenced in March 2021, extending over 24 months. The aerial surveys were conducted along a series of strip transects

[4] The AfL area reflects the boundary assessed in the PEIR and encompasses the windfarm site assessed within the ES, noting the boundary was refined following the PEIR.

(31 strip transects at 1km spacing) across the survey area every month for 24 months. The strip transects extended roughly north-east to south-west, perpendicular to the depth contours along the coast (**Plate 3.1**). Such a design ensured that each transect sampled a similar range of habitats (primarily relating to water depth) and reduced the variation in marine mammal abundance estimates between transects. The surveys were flown along the transect pattern at a height of approximately 550m above sea level.

39. **Appendix 12.2 Aerial Survey Two Year Report March 2021 to February 2023** (Document Reference 5.2.12.2) provides full details of the survey methods as well as the full dataset for the two years of monthly surveys flown between March 2021 and February 2023, providing data from 24 surveys.

40. The surveys were undertaken using an aircraft equipped with four HiDef Gen II cameras with sensors set to a resolution of 2cm Ground Sample Distance (GSD). Each camera sampled a strip of 125m width, separated from the next camera by ~25m, to provide a combined sampled width of 500m within a 575m overall strip. Two of the four cameras were analysed, achieving approximately 25% coverage of the survey area in each flight (see **Appendix 12.2**). The remaining footage has been retained for analysis at a later stage if required.

41. Data analysis followed a two-stage process in which video footage was reviewed (with a 20% random sample used for audit) then the detected objects were identified to species or species group level (again with 20% selected at random for audit). The audit of both stages required 90% agreement to be achieved (see **Appendix 12.2**) for further details).

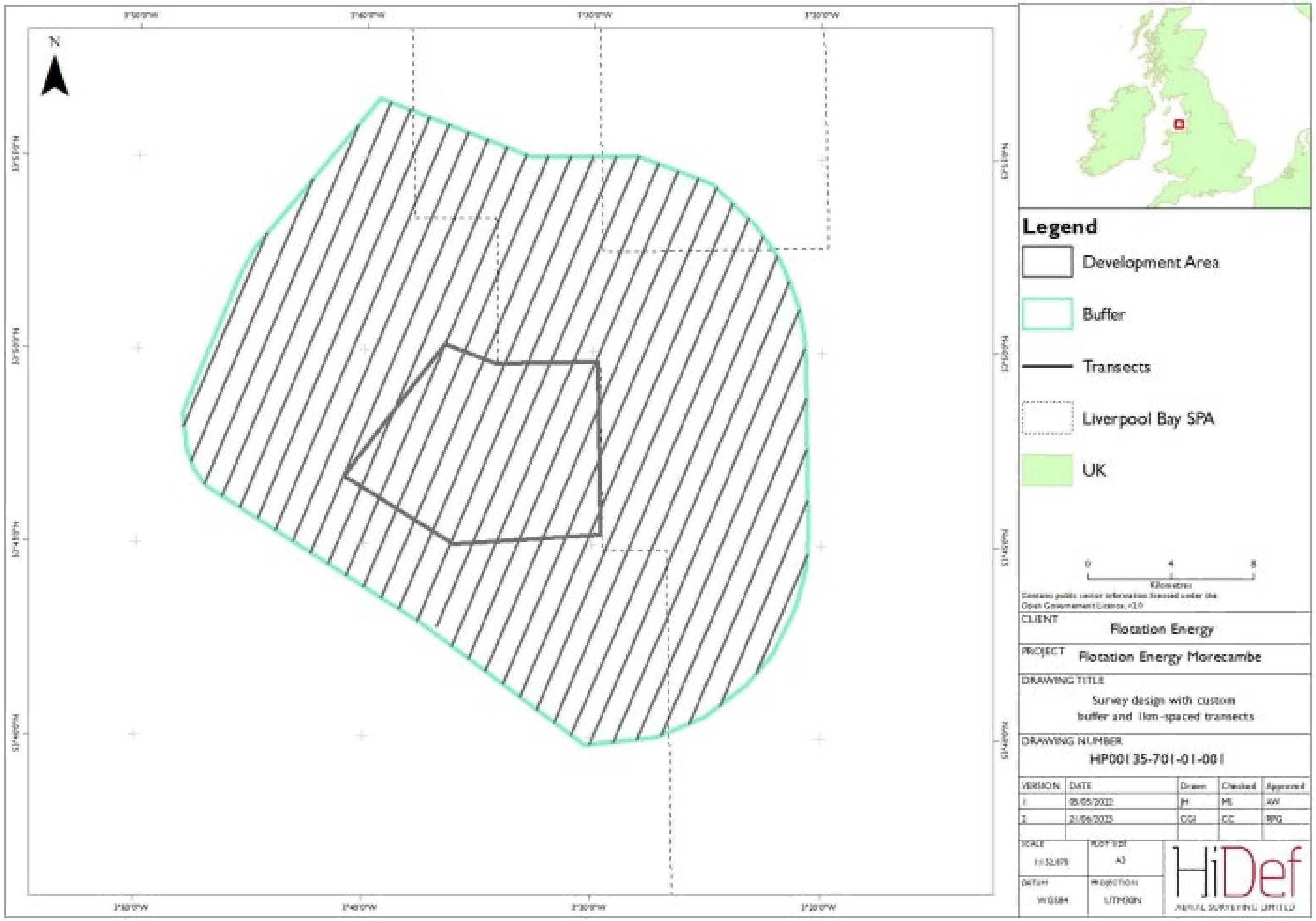


Plate 3.1 Morecambe survey design with 4-10km hybrid buffer with 1km-spaced transects flown between March 2021 and February 2023

42. The environmental conditions per survey month are summarised in **Table 3.1**, more detail can be found in HiDef's Two Year Report (**Appendix 12.2)**. The windspeeds in **Table 3.1** were measured at flight height (550m above sea level) and were typically greater than they were at ground/sea level. The windspeed's greatest effect on the data was via the sea-state, which over the entire 24 months of surveys was 2.6 (smooth - slight) on average.

*Table 3.1 Environmental conditions at flight height reported by Hi-Def in 24 monthly survey reports (CAVOK = Ceiling and Visibility OK) *Average calculated from the cameras reviewed*

Survey date	Wind speed (knots)	Sea state (average*)	Glare (average[5]*)	Cloud base over site (feet)	Turbidity (average*)
March 2021	16	2.03	1.00	2500	0.00
April 2021	3	3.75	1.00	3000	1.00
May 2021	10	1.00	1.00	1800+	0.99
June 2021	20-30	1.01	1.26	CAVOK	0.01
July 2021	5-12	1.00	1.00	2500+	0.00
Aug 2021	5-15	2.82	1.32	CAVOK	0.36
Sept 2021	10-20	2.32	1.85	2500+	1.00
Oct 2021	10-20	2.98	1.00	2500+	1.00
Nov 2021	20-30	4.64	1.00	2500+	1.25
Dec 2021	30	3.14	1.01	2000	1.94
Jan 2022	5-20	2.04	1.55	1800+	0.98
Feb 2022	15-25	3.25	1.00	1800+	1.00
March 2022	5-35	4.42	1.00	1800+	1.18
April 2022	14	2.51	1.04	CAVOK	1.00
May 2022	4-17	2.00	1.00	1800+	1.00
June 2022	5-25	2.01	1.00	1800+	1.00
July 2022	15	3.92	1.59	2000	1.00
Aug 2022	5	0.82	1.00	1800+	1.01
Sept 2022	15-20	1.96	1.00	CAVOK	1.22
Oct 2022	15-25	2.57	1.00	20,000	1.00
Nov 2022	10-18	1.99	1.00	2000+	1.00
Dec 2022	15-24	4.16	1.00	2000+	1.03
Feb (1) 2023	10-20	3.25	1.00	10,000	1.00
Feb (2) 2023	10-15	2.60	1.12	CAVOK	1.81

[5] Sun-glare scoring system 0= not recorded to 4= strong (see Aerial Survey Report, **Appendix 12.2** for details)

43. Key weather effects are noted below, more detail on how the data was treated can be found in HiDef's Two Year Report (**Appendix 12.2)**

- September 2021: high glare was recorded across much of the survey area, hence only data collected in areas with a glare rating of below 3 (out of 4) was used to model population estimates
- October 2021: adverse weather conditions affected several transects to the east of the survey area, hence density and population estimates were calculated for a reduced area
- January 2023: was missed due to lack of available weather windows so two surveys were flown in February 2023 to compensate.

44. **Table 3.2** presents the numbers of marine mammals recorded during the aerial surveys from March 2021 to February 2023. The results indicated harbour porpoise were the most abundant marine mammal species present within the survey area.

45. Apportioning of 'unidentified' seals and cetacean to species level was also undertaken per survey for the purposes of calculating population estimates. The number of unidentified seals or cetacean in each species group were assigned to species where appropriate, based on their respective abundance ratios.

46. There was one unidentified dolphin species in the second year of survey data (February (1) 2023) and four unidentified cetacean species across the survey period. These animals have been apportioned in line with the abundance ratio of other cetaceans identified during the survey.

47. There were three unidentified seal/small cetaceans across the survey period. These could be harbour porpoise or grey seal, however as it was not possible to determine the species, these animals have been apportioned in line with the ratio of other seal and cetacean species during the survey.

48. There were also 59 unidentified seal species. These were most likely to be grey seal based on the ratio of recorded grey and harbour seal during the surveys. Within the survey period, only one harbour seal was identified (in July 2021). These animals have been apportioned in line with the ratio of other seal species identified (largely grey seal except for the one sighting of harbour seal) during the survey.

Table 3.2 Marine mammal species recorded during site-specific HiDef surveys of the windfarm site and buffer (March 2021 to February 2023)

Survey date	Harbour porpoise	Grey seal	Harbour seal	Seal species	Common dolphin	Bottlenose dolphin	Dolphin species	Cetacean species	Seal/ small cetacean species
March 2021	85	0	0	5	0	0	0	0	1
April 2021	13	2	0	3	0	0	0	0	0
May 2021	48	5	0	0	0	0	0	0	0
June 2021	45	4	0	5	0	0	0	0	0
July 2021	39	2	1	2	0	0	0	0	0
Aug 2021	29	2	0	2	0	0	0	1	0
Sept 2021	13	0	0	3	0	0	0	0	0
Oct 2021	25	1	0	0	0	0	0	0	0
Nov 2021	26	2	0	0	0	0	0	0	0
Dec 2021	9	2	0	0	0	0	0	0	0
Jan 2022	19	0	0	2	0	0	0	0	0
Feb 2022	21	1	0	0	0	0	0	0	0
Sub-Total	**372**	**21**	**1**	**22**	**0**	**0**	**0**	**1**	**1**
March 2022	25	4	0	2	0	0	0	0	0
April 2022	18	2	0	1	0	0	0	0	0
May 2022	179	1	0	1	0	0	0	1	0
June 2022	52	1	0	6	0	0	0	0	0
July 2022	6	0	0	1	0	0	0	1	0
Aug 2022	49	4	0	6	32	0	0	0	1
Sept 2022	27	2	0	3	0	0	0	0	0

Survey date	Harbour porpoise	Grey seal	Harbour seal	Seal species	Common dolphin	Bottlenose dolphin	Dolphin species	Cetacean species	Seal/ small cetacean species
Oct 2022	39	0	0	2	0	0	0	0	0
Nov 2022	80	3	0	5	0	0	0	1	1
Dec 2022	28	2	0	2	0	0	0	0	0
Feb (1) 2023	29	1	0	4	0	2	1	0	0
Feb (2) 2023	21	1	0	4	0	0	0	0	0
Sub-Total	**553**	**21**	**0**	**37**	**32**	**2**	**1**	**3**	**2**
Total	**925**	**42**	**1**	**59**	**32**	**2**	**1**	**4**	**3**

49. From the sightings recorded (**Table 3.2**), abundance and density estimates for the survey area were calculated. These were based on confirmed sightings only.

50. Density and abundance estimates have been calculated using strip transect analysis and a statistical technique called kernel density estimation (KDE) to create density surface maps (these are presented in **Plate 3.2** and **Plate 3.3** with further information in **Appendix 12.2**.

51. The density estimate was expressed as the average number of animals per square km in the whole survey area. The population estimate was expressed as the estimated number of animals within the whole survey area. The upper and lower confidence intervals (CIs) define the range that the population estimate fell within with 95% certainty. The coefficient of variance (CV) or the relative standard error is a measure of the precision of the population and density estimates.

52. For species such as marine mammals that dive and therefore spend time underwater, an availability bias or correction factor must be applied in order to account for those individuals that it was not possible to detect as they may have been underwater at the time of image capture. Without these availability biases or correction factors being applied, any abundance or density estimate would be relative only, rather than being an absolute estimate.

53. The depth above which harbour porpoise were available for detection has been estimated to be 2m by Teilmann *et al*. (2013) when correcting for availability bias during visual aerial surveys of harbour porpoise. The correction factors applied for harbour porpoise were dependent on the month and time of day (**Table 3.3**). Further information on the application of the correction factors is provided in **Appendix 12.2**.

Table 3.3 Correction factors used to account for the availability bias for harbour porpoise for different months and times of day (taken from Teilmann et al., 2013)

Month	Behaviour			
	Surface		0 – 2m	
	09:00-15:00	15:00-21:00	09:00-15:00	15:00-21:00
January	0.0490	0.0476	0.4381	0.418614
February	0.0398	0.0384	0.3748	0.355348
March	0.0543	0.0529	0.4637	0.444271
April	0.0646	0.0632	0.5708	0.551331
May	0.0563	0.0549	0.5262	0.506735
June	0.0518	0.0503	0.5093	0.489809
July	0.0493	0.0479	0.5116	0.492099
August	0.0530	0.0516	0.4508	0.431293
September	0.0420	0.0406	0.4468	0.427348
October	0.0413	0.0399	0.4422	0.42276
November	0.0406	0.0392	0.4439	0.424431
December	0.0429	0.0415	0.4790	0.459555

54. **Appendix 12.2** provides a summary of the surfacing behaviour for marine mammals in the survey area between March 2021 and February 2023.

3.2 Density estimates for harbour porpoise

55. To estimate the density of surfacing harbour porpoise, HiDef calculated the proportion of animals as snapshot surfacing. Snapshot surfacing indicated where the dorsal fin was clear of the water surface in the middle frame of the sequence in which the animal was present. This was identified using data from all survey months combined because sample sizes were too small to be accurate when calculating the surfacing proportions in individual months. HiDef then multiplied the calculated density of all harbour porpoise by the proportion of snapshot surfacing encounters in the surveys. The density of surfacing harbour porpoises was then divided by the proportion of surfacing behaviour from Teilmann *et al*. (2013) in **Table 3.3**, to derive the estimates of absolute density and abundance.

56. The monthly absolute density estimates for harbour porpoise for the whole Project survey area, including buffer, are presented in **Table 3.4**. These estimates have been corrected for availability bias based on confirmed harbour porpoise sightings only. The average summer density estimate has been determined based on average of monthly estimates for April to

September. The average winter density estimate has been determined based on average of monthly estimates for October to March. It must be noted that there were two sets of survey results for February 2023, as the survey in January 2023 could not be conducted due to adverse weather. The average annual density estimate has been determined based on the 24 survey months for the first year of site-specific surveys. It is important to note that the density for the summer average has been skewed by a single month of particularly high numbers (May 2022; n= 179; 6.25 animals/km^2). The resulting mean summer density (1.621 animals/km^2) was significantly higher than that of Evans and Waggitt (2023) (0.2 animals/km^2) for the average summer density and the most recent Small Cetaceans in European Atlantic waters and North Sea (NS) (SCANS) IV density of 0.5153 animals/km^2 for the survey block CS-E (Gilles *et al*., 2023).

Table 3.4 Apportioned harbour porpoise absolute density estimates for each month, corrected for availability bias, with summer, winter and annual density estimates for the whole Project survey area including buffer

Month	Maximum absolute density estimate (corrected) for whole survey area (animals/km^2)
March 2021	3.09
April 2021	0.39
May 2021	1.63
June 2021	1.71
July 2021	1.54
August 2021	1.08
September 2021	1.02
October 2021	1.38
November 2021	1.25
December 2021	0.37
January 2022	0.78
February 2022	1.04
March 2022	0.88
April 2022	0.54
May 2022	6.25
June 2022	1.96
July 2022	0.26
August 2022	1.79
September 2022	1.28
October 2022	1.84

Month	Maximum absolute density estimate (corrected) for whole survey area (animals/km^2)
November 2022	3.98
December 2022	1.26
February 2023	1.43
February 2023	1.04
Average for summer period (April-Sept)	**1.621**
Average for winter period (Oct-Mar)	**1.528**
Annual average	**1.574**

3.3 Abundance estimates for harbour porpoise

57. The abundance estimates for harbour porpoise (**Table 3.5**) have been derived in the same way as the density estimates (see **Appendix 12.2**). These are presented in **Plate 3.2** - **Plate 3.5**.

Table 3.5 Apportioned absolute abundance estimates of harbour porpoise within whole Project survey area including buffer, corrected for availability bias

Month	Abundance estimate (corrected) for number of harbour porpoise in survey area	Lower and upper 95% confidence limits for abundance estimates
March 2021	2,026	1,220 - 3,018
April 2021	255	137 - 388
May 2021	1,081	732 – 1,458
June 2021	1,108	747 – 1,499
July 2021	1,010	643 – 1,427
August 2021	718	371 – 1,101
September 2021	648	148 – 1,359
October 2021	898	499 – 1,374
November 2021	820	375 – 1,351
December 2021	266	118 - 466
January 2022	498	278 - 744
February 2022	677	350 -1,043
March 2022	590	304 - 928
April 2022	358	177 – 555
May 2022	4,060	2,196 – 6,481
June 2022	1,285	747 – 1,934
July 2022	180	51 – 309

Month	Abundance estimate (corrected) for number of harbour porpoise in survey area	Lower and upper 95% confidence limits for abundance estimates
August 2022	1,178	766 – 1,639
September 2022	823	415 – 1,306
October 2022	1,197	852 -1,550
November 2022	2,569	1905 – 3,334
December 2022	835	525 – 1,182
February 2023	924	414 – 1,561
February 2023	685	390 - 1,019
Average	**1,028.7**	**598.3 – 1542.8**

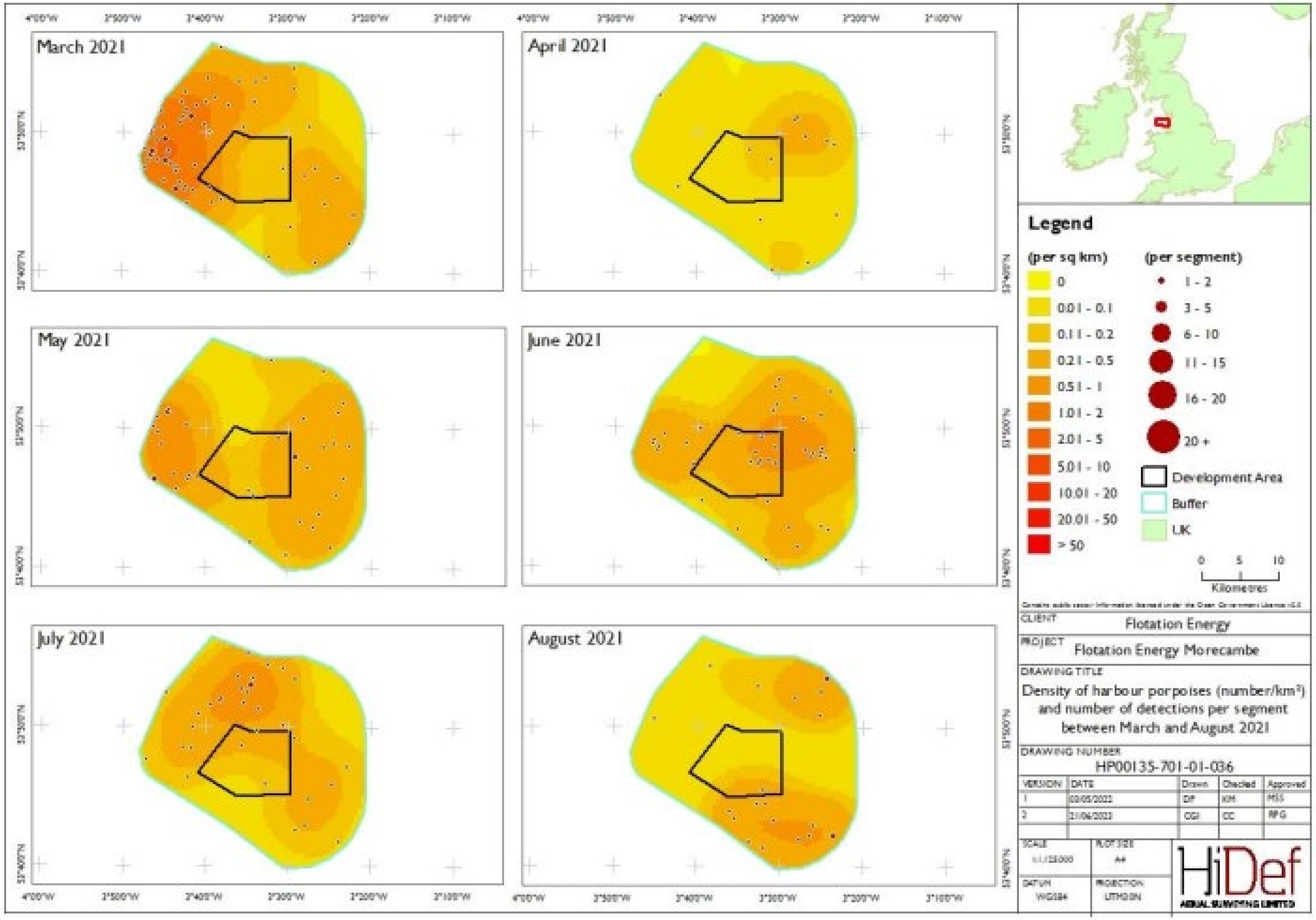


Plate 3.2 Density of harbour porpoise (number/km²) and number of detections per segment in the survey area between March and August 2021

MORECAMBE

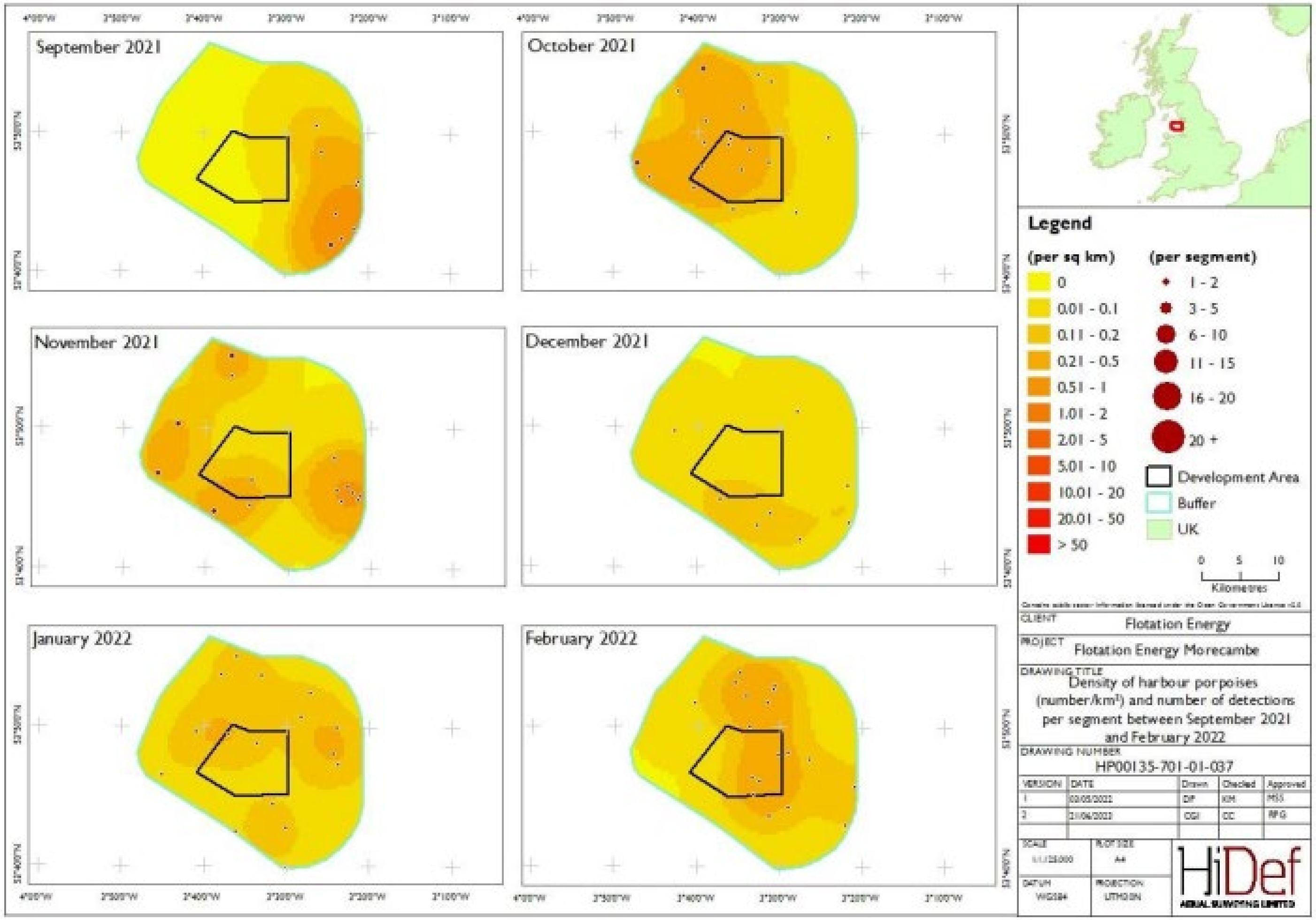


Plate 3.3 Density of harbour porpoise (number/km²) and number of detections per segment in the survey area between September 2021 and February 2022

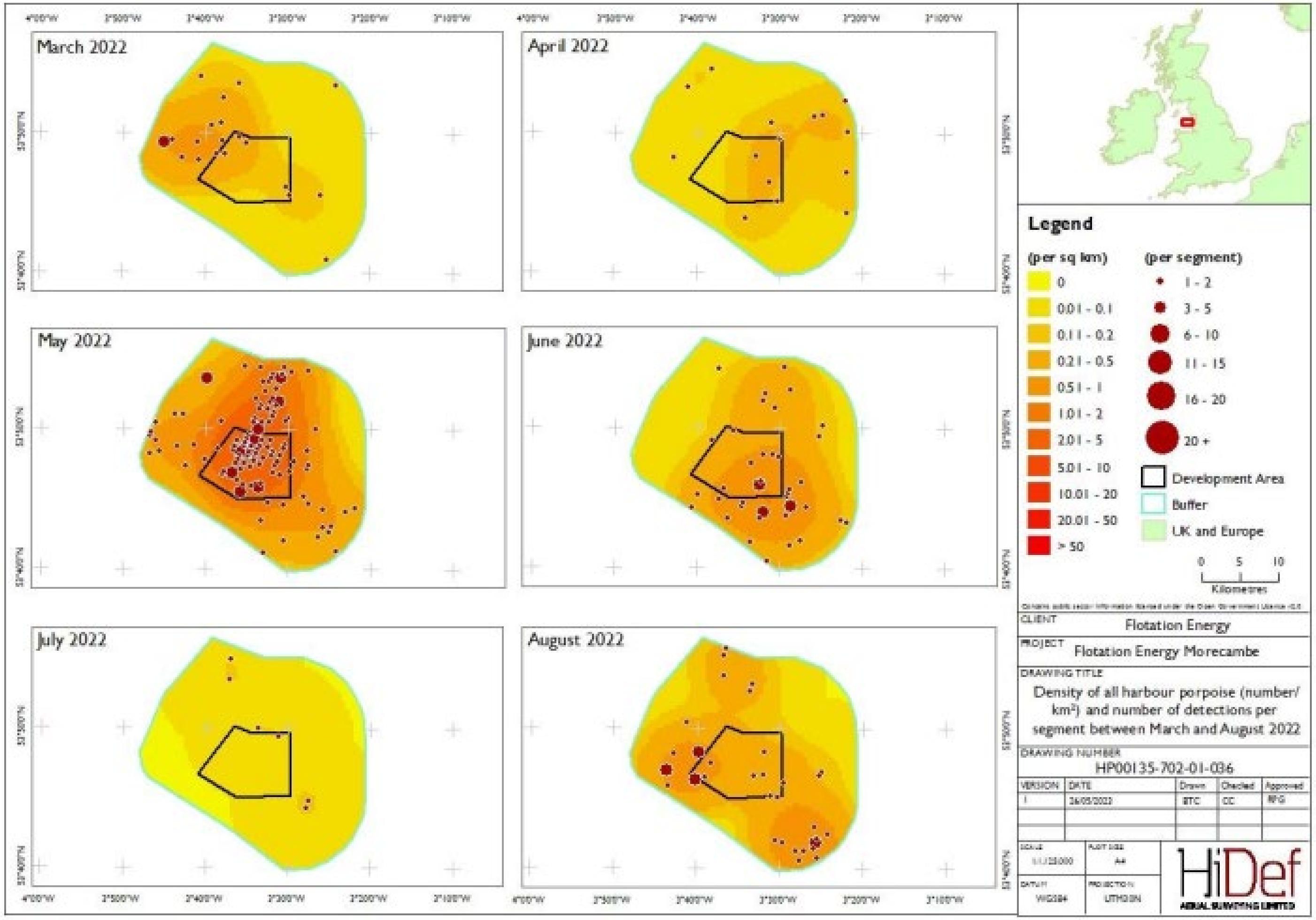


Plate 3.4 Density of harbour porpoise (number/km²) and number of detections per segment in the Morecambe survey area between March and August 2022

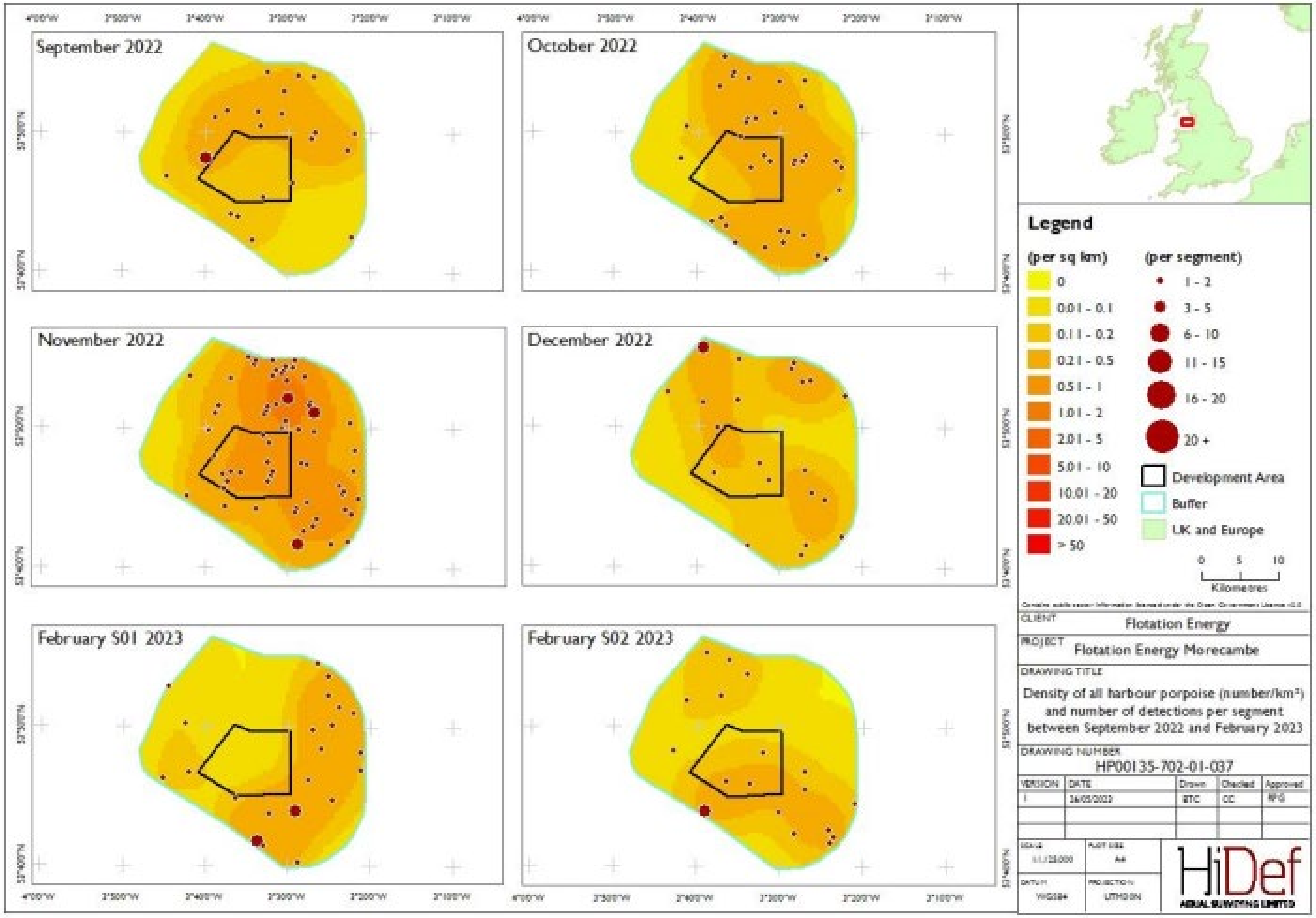


Plate 3.5 Density of harbour porpoise (number/km²) and number of detections per segment in the Morecambe survey area between September 2022 and February 2023

3.4 Density estimates for grey seal

58. The monthly absolute density estimates for grey seal for the whole Project survey area, including buffer, are presented in **Table 3.6**. Unlike for harbour porpoise, an availability bias or correction factor for seals is unavailable.

59. The average winter density estimate has been determined based on the average of monthly estimates for October to March; but due to lack of sightings it was only based on nine densities, whereas the slightly higher average summer density was based on ten density values. Neither of the site-specific (survey) densities were higher than the Project-specific density (revised area and 4km buffer) derived from the data provided by Carter *et al*. (2022) (0.100 animals/km^2); discussed further below in **Section 5.7**.

Table 3.6 Apportioned grey seal absolute density estimates for each month, corrected for availability bias, with summer, winter and annual density estimates for the whole Project survey area including buffer

Month	Maximum absolute density estimate (apportioned) for whole survey area
March 2021	*no sighting*
April 2021	0.03
May 2021	0.03
June 2021	0.06
July 2021	0.02
August 2021	0.02
September 2021	*no sighting*
October 2021	0.01
November 2021	0.01
December 2021	0.01
January 2022	*no sighting*
February 2022	0.01
March 2022	0.04
April 2022	0.02
May 2022	0.01
June 2022	0.04
July 2022	*no sighting*
August 2022	0.06
September 2022	0.03
October 2022	*no sighting*
November 2022	0.05

Month	Maximum absolute density estimate (apportioned) for whole survey area
December 2022	0.03
February 2023	0.03
February 2023	0.03
Average for summer period (April-Sept)	**0.032**
Average for winter period (Oct-Mar)	**0.024**
Annual average	**0.0284**

3.5 Abundance estimates for grey seal

60. The abundance estimates for grey seal (**Table 3.7**) have been derived in the same way as the density estimates (see **Appendix 12.2**). Maps with numbers of less abundant marine mammal (including seal) detections per segment can be found in the **Appendix 12.1 Offshore Ornithology Technical Report** (Document Reference 5.2.12.1).

Table 3.7 Apportioned absolute abundance estimates of harbour porpoise within whole Project survey area including buffer, corrected for availability bias

Month	Abundance estimates for number of grey seal in survey area	Lower and upper 95% confidence limits for abundance estimates
March 2021	*no sighting*	*no sighting*
April 2021	21	4 - 40
May 2021	21	0 - 56
June 2021	37	16 - 64
July 2021	14	3 - 28
August 2021	16	0 - 36
September 2021	*no sighting*	*no sighting*
October 2021	5	0 - 14
November 2021	9	0 - 20
December 2021	9	0 - 20
January 2022	*no sighting*	*no sighting*
February 2022	4	0 - 12
March 2022	24	8 - 43
April 2022	12	0 - 28
May 2022	8	0 - 20
June 2022	28	8 - 55
July 2022	*no sighting*	*no sighting*

Month	Abundance estimates for number of grey seal in survey area	Lower and upper 95% confidence limits for abundance estimates
August 2022	41	20 - 66
September 2022	21	4 - 39
October 2022	*no sighting*	*no sighting*
November 2022	33	16 - 52
December 2022	17	4 - 32
February 2023	20	4 - 36
February 2023	21	5 - 37
Average	**19**	**4.8 – 36.7**

3.6 Geotechnical Marine Mammal Survey Report

61. Between 17th July and 20th October 2023, Gardline conducted a series of deep geotechnical surveys within the proposed Project windfarm site. The surveys were conducted from a motor vessel, and visual monitoring for marine mammals was undertaken by non-dedicated mitigation personnel, in accordance with best practice outlined in the JNCC 'Guidelines for minimising the risk of injury to marine mammals from the geophysical surveys' (JNCC, 2017).

62. Over 1,021 hours were surveyed across the 74 days of the geotechnical survey. During this period common dolphins were seen regularly (17 occasions) throughout August and September 2023 but not October 2023. On one occasion on 9th September 2023 a super pod of 300 animals was observed slow swimming, feeding and breaching, of which 100 were identified as juveniles.

63. On only five separate occasions, bottlenose dolphin were observed spread out over the survey period (July, August, September 2023), of which one was a mother and a calf.

64. Fifteen harbour porpoise were sighted on nine occasions from July to August, in small groups of two or three individuals.

65. Grey seal were also sighted, with 33 individual grey seals sighted on separate occasions throughout all survey months.

66. Furthermore, there were four entries of unidentified dolphin species sightings, and seven entries for unidentified seal species.

4 SCANS surveys

67. A series of large-scale surveys for SCANS was initiated in summer 1994 in the North Sea and adjacent waters (SCANS 1995; Hammond *et al*., 2002).

68. SCANS-II was undertaken in summer 2005 in all shelf waters (SCANS-II 2008; *et al*., 2013) and 2007 in offshore waters (Cetacean Offshore Distribution and Abundance in the European Atlantic (CODA), 2009).

69. SCANS-III was conducted in summer 2016 with the aim to survey all European Atlantic waters, however the final surveyed area excluded offshore waters of Portugal and also excluded waters to the south and west of Ireland which were surveyed by the Irish ObSERVE project (Hammond *et al*., 2017, 2021). The Project lies within the boundaries of block F.

70. In October 2023, the SCANS-IV report was released with data collected during the summer 2022 (Gilles *et al*., 2023), with the aim to inform the then upcoming Marine Strategic Framework Directive (MSFD) in European Atlantic Waters in 2024. This survey included the offshore waters of Portugal which had not been previously surveyed as part of SCANS, but excluded waters south and west of Ireland, which were surveyed by the ObSERVE2, and coastal Norwegian waters north of Vestfjorden. Some of the block boundaries have changed since SCANS-III, but the changes have not affected the block in which the Project lies (block CS-E)

71. With reference to **Plate 4.1** for SCANS-III and **Plate 4.2** for SCANS- IV, pink lettered blocks were surveyed by air and blue numbered blocks were surveyed by ship. Blocks coloured green to the south, west and north of Ireland were surveyed by the Irish ObSERVE2 project. SCANS-III blocks FC and FW coloured yellow were surveyed by the Faroe Islands as part of the North Atlantic Sightings Survey in 2015. The cross-hatched area represents where SCANS-IV blocks BB-3 and BB-A overlapped.

72. Amongst many other sources of information, **Section 5** provides a summary of species abundance and density estimates from SCANS-III and IV wherever possible to inform about changes in species distribution in the relevant survey blocks F and CS-E.

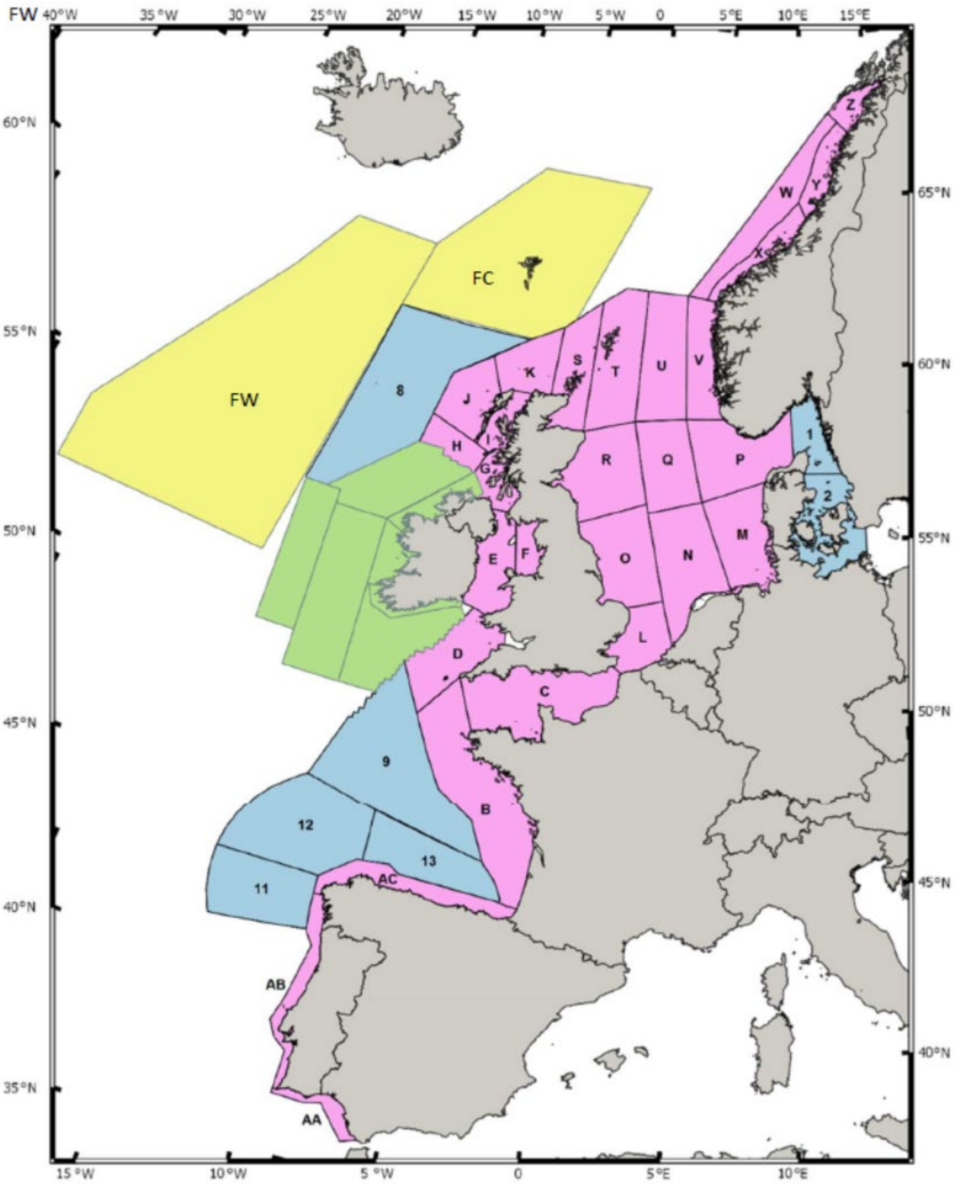


Plate 4.1 Area covered by SCANS-III and adjacent surveys (Hammond et al., 2021). Block colours: blue= ship survey, pink= aerial survey, green=ObSERVE project, yellow=North Atlantic Sightings Survey in 2015)

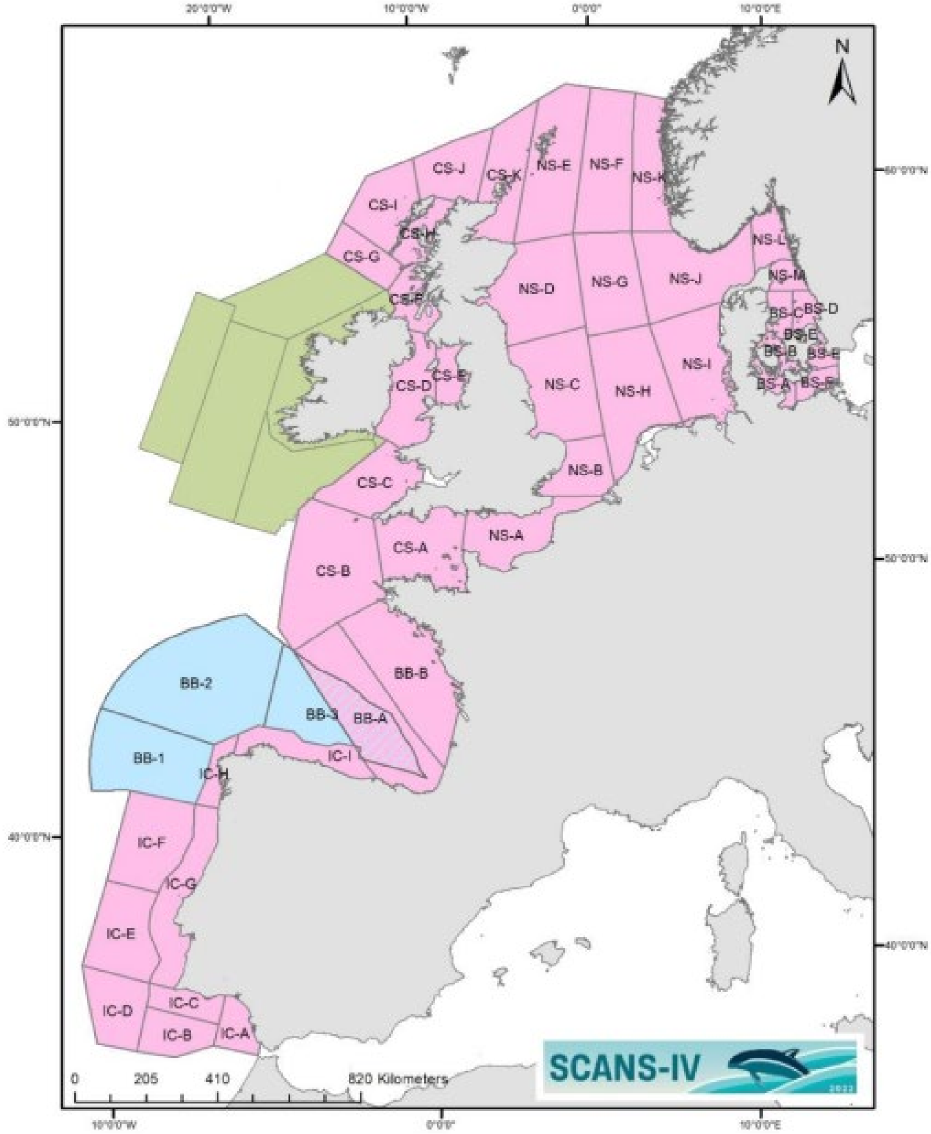


Plate 4.2 Area covered by SCANS-IV and adjacent surveys (Gilles et al., 2023) Block colours: blue= ship survey, pink= aerial survey, green=ObSERVE project)

5 Existing environment

73. The study area for marine mammals has been defined on the basis that marine mammals are highly mobile and transitory in nature. It was, therefore, necessary to examine species occurrence not only within the windfarm site, but also over the wider area. Baseline data from developments and research projects in the wider Northwest have been evaluated to determine species in the wider area of the Project.

74. A series of baseline characterisation aerial surveys were undertaken at Awel y Môr Offshore Windfarm (28.9km from the Project) completing one survey per month for two years, between March 2019 and February 2021. Over the two years of surveys only harbour porpoise were identified to species level, with the remaining sightings being classified as unidentified dolphin, unidentified seal or dolphin/porpoise. There was no seasonal or spatial pattern to the harbour porpoise sightings and a density of 0.13 porpoise/km^2 was recorded.

75. Gwynt y Môr Offshore Windfarm conducted pre- and post-construction marine mammal surveys between 2003 and 2019 by undertaking aerial, boat and land-based surveys. Species recorded included harbour porpoise, bottlenose dolphin, short-beaked common dolphin, grey seal and harbour seal (CMACS Ltd. 2005, 2011, 2013).

76. Morgan Offshore Wind Project Generation Assets began their aerial surveys in April 2021 and finished in March 2023. As presented in their PEIR (Morgan Offshore Wind Ltd, 2023), the only species observed within the first 12 surveys were bottlenose dolphin, grey seal, and harbour porpoise, of which the latter was the most commonly sighted.

77. The aerial surveys for Mona Offshore Wind Project commenced in March 2020 and finished in February 2022. Species recorded, as presented in their PEIR, included harbour porpoise, bottlenose dolphin, short-beaked common dolphin, Risso's dolphin, grey seal and one harbour seal (Mona Offshore Wind Ltd, 2023).

78. The Manx Whale and Dolphin Watch (MWDW) have conducted land-based surveys since 2006 and vessel-based surveys throughout the Manx territorial waters since 2007. Data were available through reports provided on the MWDW website for 2007 - 2016, with additional data obtained for 2018 (Felce, 2014, 2015; Adams, 2017; Clark *et al*., 2019). The surveys, as well as Howe (2018), have reported five main species of marine mammals in Manx territorial waters: harbour porpoise, common dolphin, bottlenose dolphin, Risso's dolphin and minke whale.

79. The Joint Cetacean Data Programme (JCDP) (2022) database was reviewed to ensure all publicly available data sources for the Irish sea had been considered to inform the baseline and existing environment. The main source listed was the SCANS III report (Hammond *et al*., 2021) which has been reviewed alongside the updated SCANS-IV data (Gilles *et al*., 2023).

80. The revised Atlas of the Marine Mammals of Wales (Evans and Waggitt, 2023) became available in June 2023 and provided density maps and species summaries for the five most commonly occurring species in the Irish Sea, including harbour porpoise, bottlenose dolphin, common dolphin, Risso's dolphin and minke whale.

81. It should be noted that, although extremely rare, a humpback whale (*Balaenoptera novaeangliae*) was spotted in July 2023, just 0.5 miles off the west coast of the IoM. This has been the first sighting since 2017, with previous sightings recorded in 2010 and 2013 (Wotton, 2023). No sightings of humpback whales in Liverpool Bay have been recorded by Organisation Cetacea (in the last 30 years), the Sea Watch Foundation (July – August 2023), nor the Hebridean Whale and Dolphin Trust (between 2017 and 2023).

5.1 Harbour porpoise

5.1.1 Distribution

5.1.1.1 Abundance

82. Harbour porpoise within the eastern North Atlantic have been generally considered to be part of a continuous biological population that extends from the French coastline of the Bay of Biscay to northern Norway and Iceland (Tolley and Rosel, 2006; Fontaine *et al*., 2007, 2014; IAMMWG, 2015). However, for conservation and management purposes, it is necessary to consider this population within smaller MUs. MUs provide an indication of the spatial scales at which effects of plans and projects alone, and in-combination, need to be assessed for the key cetacean species in UK waters, with consistency across the UK (IAMMWG, 2015; 2023).

83. IAMMWG defined three MUs for harbour porpoise: North Sea (NS); West Scotland (WS); and the Celtic and Irish Sea (CIS). As outlined in **Section 1.1.1** of this Appendix, the Project is located within the CIS MU (**Plate 1.1**) with an estimated population of 62,517 CV = 0.13) individuals.

84. As outlined in **Section 3**, harbour porpoise was the most commonly sighted marine mammal species during the site-specific surveys, with a total of 925 individuals recorded for the 24-month survey period. Harbour porpoise were recorded in all 24 months and across the entire survey area.

85. Heinänen and Skov (2015) provided the results of detailed analyses of 18 years of survey data in the Joint Cetacean Protocol (JCP) undertaken to inform the identification of discrete and persistent areas of relatively high harbour porpoise density in the UK marine area.

86. Habitat preference modelling for the Celtic and Irish Seas has been conducted by Heinänen and Skov (2015), as well as Lepple (2023 unpublished), in which it was found that high densities of harbour porpoise were typically associated with shallow water depths (ranging between 20-90m). A range of studies (Evans *et al*. 2003, Reid *et al*. 2003, Shucksmith *et al.* 2009, Embling *et al*. 2009, Isojunno *et al*. 2012, Williamson *et al*. 2017) from other sea regions verified the preference for shallow water, possibly linked to distribution and proximity of abundant prey of high nutritional quality (Macleod *et al*. 2003, Johnston *et al.* 2005, Spitz *et al.* 2012).

87. Furthermore, preference for seabed heterogeneity such as headlands with tidal currents and eddies were indicated for harbour porpoise (Shucksmith *et al*. 2009, Heinänen and Skov, 2015, Waggitt *et al.* 2018) as a more complex seabed provided niches for a wide range of species. In contrast, muddier areas were predicted to have lower harbour porpoise densities.

88. The Project windfarm site consists of predominantly muddy sand and sand, whilst at areas along the coast finer sediment is found (as per assessment in **Chapter 9 Benthic Ecology** (Document Reference 5.1.9). The site-specific surveys highlighted consistent numbers of harbour porpoise, which could be related to the presence of favoured prey species (prey has been discussed below in **Section 5.1.2**).

5.1.1.2 Density

89. The predicted densities of harbour porpoise during the summer and winter seasons in the Celtic and Irish Seas showed considerable variation between periods in offshore waters and more persistent patterns in coastal areas. High densities of porpoises were estimated off the northwest and west coasts of Wales during summer, predictions which affirmed the observed densities. Predictions also indicated that the western Bristol Channel supported high densities, as did the area north of the IoM. **Plate 5.1** indicates the predicted high-density areas of harbour porpoise during summer in the Celtic and Irish Seas. **Plate 5.2** indicates the predicted high-density areas of harbour porpoise during winter in the Celtic and Irish Seas.

90. The modelling by Heinänen and Skov (2015) did not predict areas of high harbour porpoise density in or around the Project area during summer or winter (**Plate 5.1** and **Plate 5.2**).

91. The persistent high-density areas of harbour porpoise in the Celtic and Irish Seas identified by Heinänen and Skov (2015) were:

- Three coastal areas off west Wales (Pembrokeshire and Cardigan Bay), and northwest Wales (Anglesey, Llŷn Peninsula), and part of the Bristol Channel (Camarthen Bay)
- Smaller areas north of the IoM (winter) and on the Northern Irish coast near Strangford Lough
- Western Channel off Start Point, Cornwall (summer).

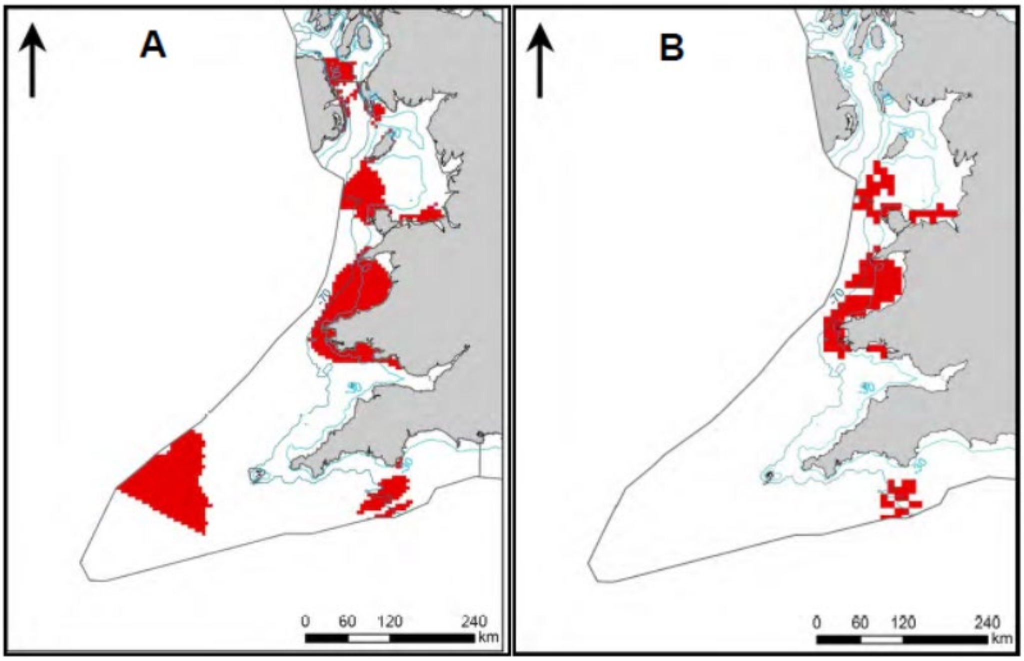


Plate 5.1 Persistent high-density areas identified during summer. In map A the red colours mark areas where persistent high densities as defined by the upper 90th percentile have been identified. In map B the red colours mark persistent high-density areas with survey effort from three or more years [Source: Heinänen and Skov (2015)]

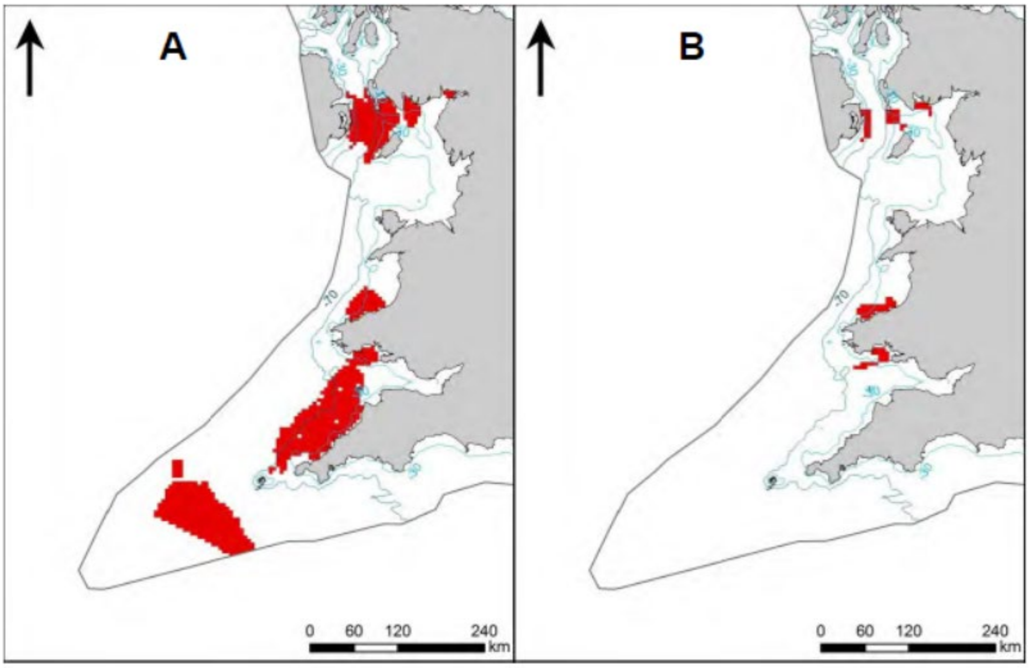


Plate 5.2 Persistent high-density areas identified during winter. In map A the red colours mark areas where persistent high densities as defined by the upper 90 percentile have been identified. In map B the red colours mark persistent high-density areas with survey effort from three or more years [Source: Heinänen and Skov (2015)]

92. The JCP Phase III Report (Paxton *et al*., 2016) identified an estimated density of 0.2 – 0.4 individuals per km^2 in the vicinity of the Project (0.4 – 0.8 per km^2 97.5% CI; Paxton *et al*., 2016).

93. Harbour porpoise have been the most frequently sighted cetacean throughout IoM territorial waters and have been sighted year-round, with an increase in sightings between April and September. Sightings between 2007 and 2014 comprised 81.3% of boat sightings, 75.1% of land-based survey sightings and 51% of opportunistic sightings (Felce, 2014). Similar results were found during surveys in 2018 where harbour porpoise comprised 73.7% of boat sightings, 71.0% of land-based survey sightings and 46.9% of opportunistic sightings (Clark *et al*., 2019). Using the boat-based survey data (2007–2014), it was estimated that the density of harbour porpoise throughout Manx waters was 0.207/km² (0.137-0.312/km², CV = 21.09%) (Howe, 2018a).

94. Distribution and abundance maps were developed by Waggitt *et al*. (2019) for cetacean species around Europe. These maps were generated based on a collation of survey effort across the northeast Atlantic between 1980 and 2018, with a total of 1,790,375km of survey effort for cetaceans. All survey data was standardized to generate distribution maps at 10km resolution, with maps generated for each species included for each month of the year.

95. The density maps in **Plate 5.3** (Waggitt *et al*., 2019) show a high distribution within the Eastern Irish Sea, and the coasts of northwest England and Wales

for both January and July, however the summer, winter and annual density for the original Project area were similar, rounded to 0.58 animals/km^2.

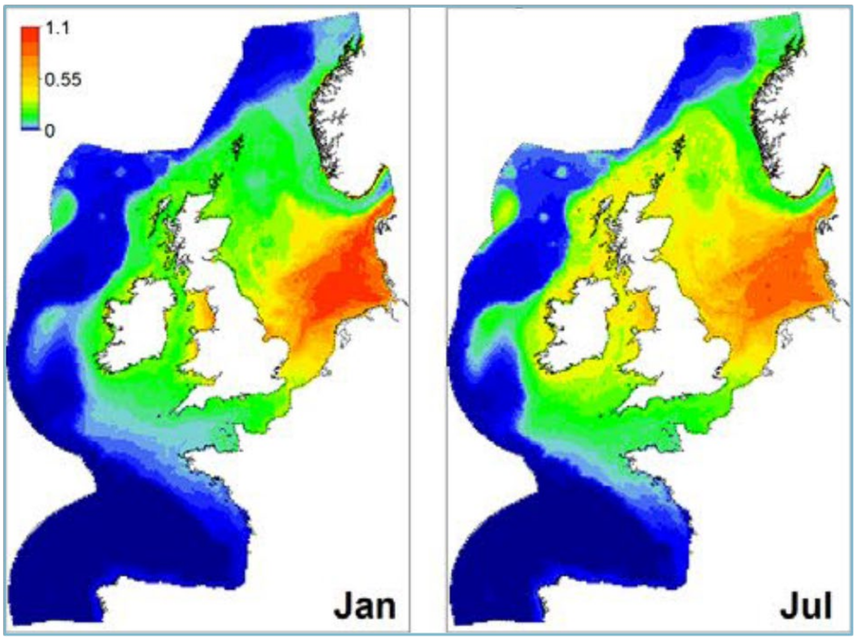


Plate 5.3 Spatial variation in predicted densities (individuals per km of harbour porpoise in January and July in the North-East Atlantic). Values have been provided at 10km resolution (Waggitt et al., 2019)

96. In contrast, the distribution of estimated density over the SCANS-III (Hammond *et al*., 2021) and IV (Gilles *et al*., 2023) survey area indicated that the occurrence of harbour porpoise was greater in western areas of the Irish Sea when compared to eastern areas of the Irish Sea (**Plate 5.4** and **Plate 5.5**).

97. Since SCANS-III, the density of harbour porpoises significantly increased from 0.086 animals/km^2 (block F) to 0.5153 animals/km^2 (CV = 0.250, 95% Confidence Limit (CL) = 3,663 – 10,162) and an estimated abundance of 6,325 individuals (in SCANS-IV; block CS-E).

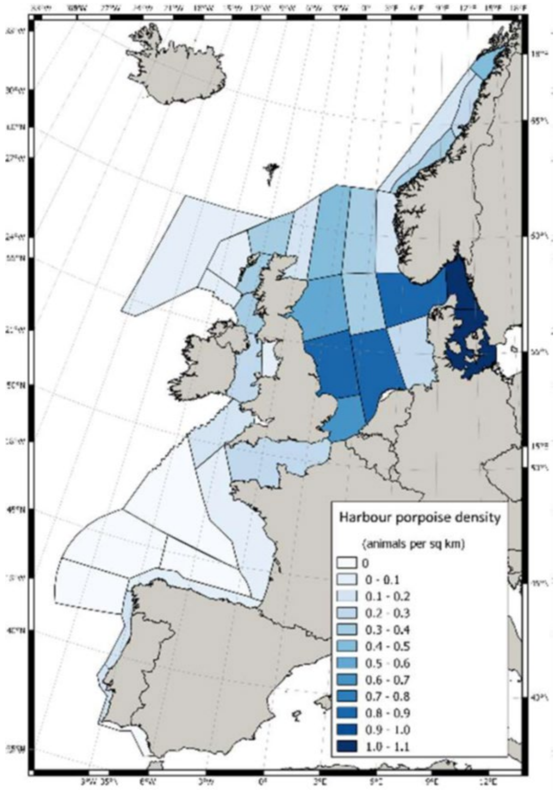


Plate 5.4 Estimated density in each survey block for harbour porpoise from SCANS-III (Hammond et al., 2021)

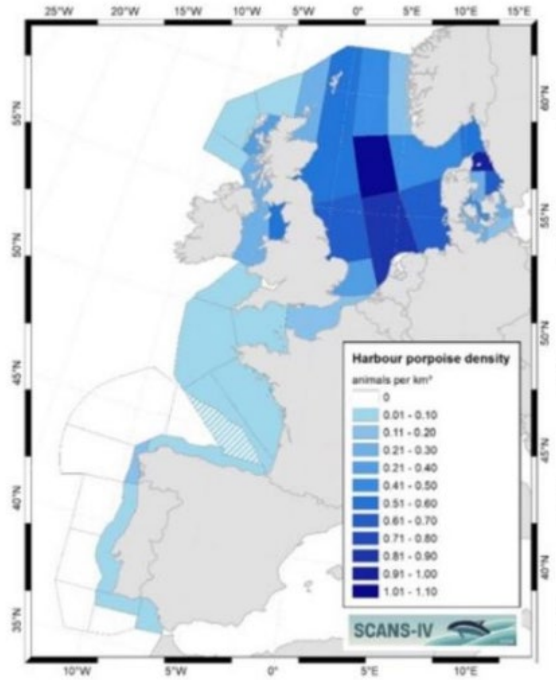


Plate 5.5 Estimated density in each survey block for harbour porpoise from SCANS-IV (Gilles et al., 2023)

98. The most recently available distribution and density maps were developed by Evans and Waggitt (2023). The data underlying these maps were a collation from different sources from three decades (1990 - 2020). The data was modelled to a very fine scale resolution of 2.5km by 2.5km grid cells, making it spatio-temporally more relevant than that of Waggitt *et al*., 2019 (described above) for the Irish Sea. Harbour porpoise densities were high year-round particularly in the relevant study area between north Anglesey and the IoM, as well as the outer part of Cardigan Bay and eastern Ireland. Distribution patterns varied both between seasons and months (**Plate 5.6**), particularly from May to September, with the highest densities overlapping with the breeding season for harbour porpoises, whose births usually peak around June (Lockyer, 1995 and 2003). The average density across the windfarm site and 4km buffer was 0.2 animals/km^2.

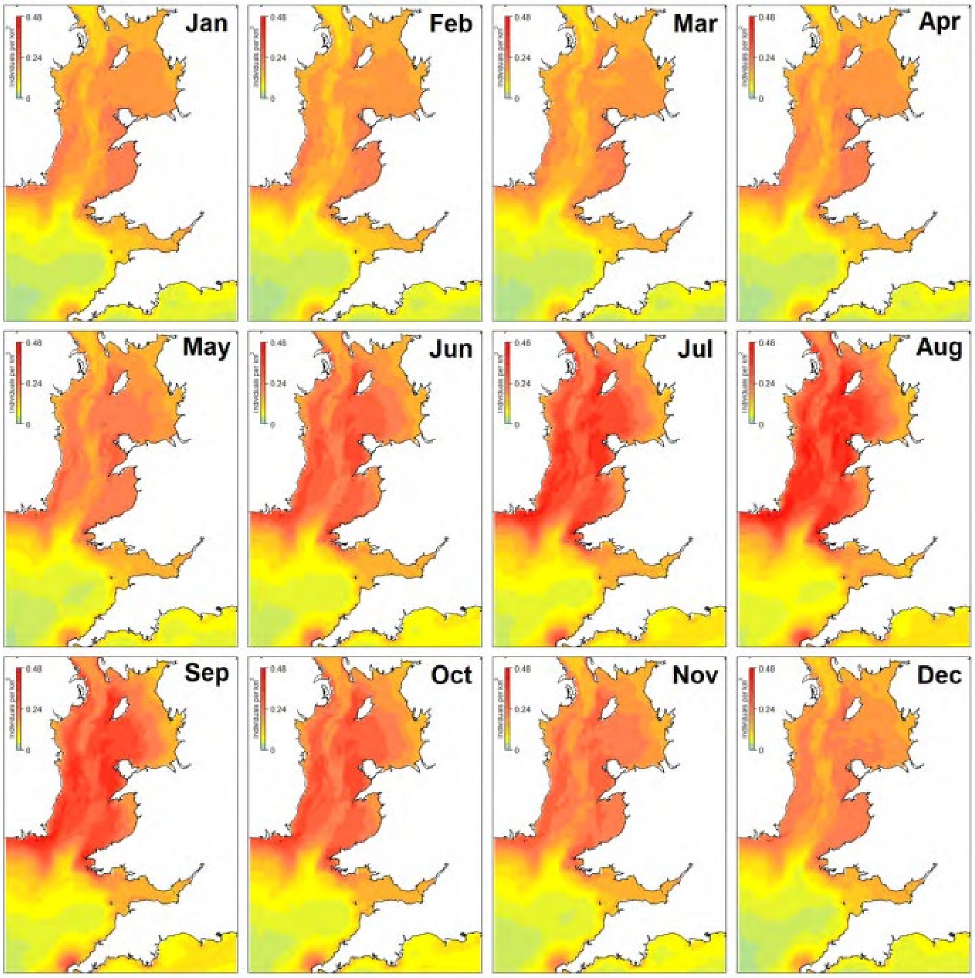


Plate 5.6 Harbour porpoise modelled densities by month (measured as the mean density per cell. Values have been provided at 2.5km resolution (Evans and Waggit, 2023)

5.1.2 Diet

99. The distribution and occurrence of harbour porpoise, as well as other marine mammal species, is considered most likely to be related the availability and distribution of their prey species. They have tended to concentrate their movements in small focal regions (Johnston *et al.,* 2005), which often approximated to particular topographic (Isojunno *et al.* 2012; Brookes *et al.* 2013, Stalder *et al.* 2020) and oceanographic features (Weir and O'Brien 2000, Johnston *et al.,* 2005, Embling *et al.* 2009, Marubini *et al.* 2009, Waggitt *et al.* 2018, Bouveroux *et al*. 2020) that have been associated with prey aggregations (Sveegaard *et al.,* 2012). Consequently, habitat use has been highly correlated with prey density rather than any particular habitat type (e.g. Sveegaard *et al.,* 2012).

100. Harbour porpoise are generalist feeders, and their diet reflects available prey in an area. Therefore, their diet varies geographically, seasonally, and annually, reflecting changes in available food resources and differences in diet between sexes or age classes may also exist. The diet of the harbour porpoise has been found to include a wide variety of fish, including pelagic schooling fish, as well as demersal and benthic species, especially Gadoids, Clupeids and sandeels (Börjesson *et al.,* 2003; Santos and Pierce 2003; Santos *et al.,* 2004; Sveegaard *et al.,* 2012).

101. Harbour porpoise have relatively high daily energy demands and need to capture enough prey to meet their daily energy requirements. It has been noted that they must be near abundant food sources and are driven by the need to feed constantly (Read and Hohn 1995, Johnston *et al.* 2005, Wisniewska *et al.* 2016). However, it has been estimated that, depending on the conditions, harbour porpoise could rely on stored energy (primarily blubber) for three to five days, depending on body condition (Kastelein *et al.,* 1997).

5.2 Bottlenose dolphin

5.2.1 Distribution

5.2.1.1 Abundance

102. Throughout its range, the bottlenose dolphin occurs in a diverse range of habitats, from shallow estuaries and bays, coastal waters, continental shelf edge and deep open offshore ocean waters. However, it is primarily an inshore species, with most sightings within 10km of land, but they can also occur offshore, often in association with other cetaceans[6].

103. In coastal waters, bottlenose dolphin have often been associated with river estuaries (Ingram and Roger, 2002), steep benthic slopes (Wilson *et al.*, 1997, Ingram and Rogan, 2002), headlands or sandbanks, where there is uneven bottom relief and/or strong tidal currents (e.g. Lewis and Evans 1993; Wilson *et al.,*1997; Liret *et al.,* 1998; Liret, 2001; Ingram and Rogan 2002; Reid *et al.,* 2003, Moreno and Mathews, 2018).

104. In the Irish and Celtic Seas, bottlenose dolphins have a predominantly coastal distribution, with higher concentrations off west Wales (particularly Cardigan Bay) and off the coast of Co. Wexford in southeast Ireland. They have also been regularly sighted in summer off the Galloway coast of southwest Scotland and around the IoM (Hammond *et al*., 2005, Baines and Evans, 2012; Department of Energy and Climate Change (DECC), 2016).

105. It has been determined that there are two 'eco-types' of bottlenose dolphin present in Europe, the coastal type and the pelagic type, and that these types were genetically and ecologically different from each other (Louis *et al.,* 2014; Oudejans *et al.* 2015; Department for Business, Energy and Industrial Strategy (BEIS), 2022b).

106. Results of genetic analysis revealed that there were five clusters of genetically distinct coastal bottlenose dolphin populations in the UK and the north of continental Europe (Nykänen *et al.,* 2019). For these five groups, there was the potential for individuals from the east and west Scotland, Wales and Galicia to be present in the Project area, but there was no evidence of connectivity with any other coastal population of bottlenose dolphin in the UK, Ireland, and northern continental Europe. Of these five populations, the migration rates from one population to another were found to be less than 1% in all possible movements, apart from between Wales/West Scotland and East

[6] https://sac.jncc.gov.uk/species/S1349

Scotland (with a migration rate of 25.7%) and between Galicia and East Scotland (with a migration rate of 25.7%; Nykänen *et al*., 2019).

107. The two-year site-specific surveys and the three-months geophysical have indicated that very few bottlenose dolphins seem to utilise the area.

108. As outlined in **Section 1.1.1** of this Appendix, the Project is located within the IS MU (**Plate 1.2**), with an estimated reference population of 293 (CV = 0.54) individuals (IAMMWG, 2023).

5.2.1.2 Density

109. The results of the JCP Phase III Report (Paxton *et al*., 2016) identified that for bottlenose dolphins, densities were low across much of UK waters, with higher densities off the west coast of Wales, and within the Moray Firth. The density of bottlenose dolphin within the Irish Sea was low, with less than 0.1 individuals per km^2 (97.5% CL = 0 – 0.1 per km^2) (Paxton *et al*., 2016).

110. Distribution of estimated density over the SCANS-III and IV survey area indicated that the occurrence of bottlenose dolphin was greater in western areas of the Irish Sea when compared to eastern areas of the Irish Sea (**Plate 5.7** and **Plate 5.8**).

111. During SCANS-III surveys no bottlenose dolphins were recorded in survey block F (Hammond *et al*., 2021), and only very few in block CS-E during SCANS-IV in which the Project is located (Gilles *et al*., 2023). The density in the latter survey was estimated at 0.0104 animals/km^2 (CV = 0.700; 95% CL = 3 – 353) with an abundance of 127 individuals.

112. Being the worst-case, the SCANS-IV block CS-E was taken forward for the impact assessment.

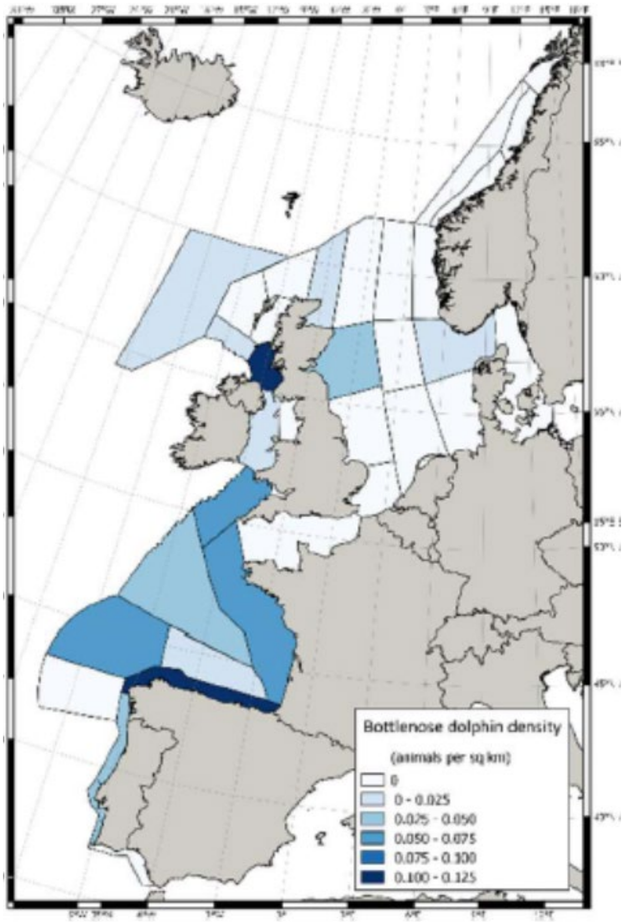


Plate 5.7 Estimated density in each survey block for bottlenose dolphin from SCANS-III (Hammond et al., 2021)

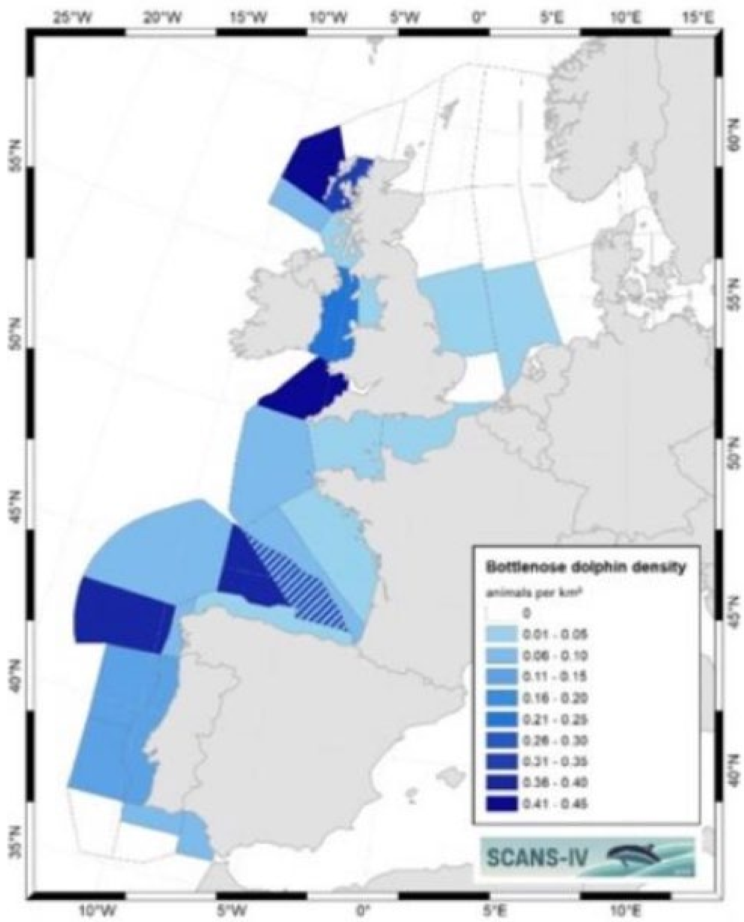


Plate 5.8 Estimated density in each survey block for bottlenose dolphin from SCANS-IV (Gilles et al., (2023)

113. Bottlenose dolphins have been reported throughout MWDW surveys and across the IoM territorial waters, they have been sighted most frequently in the winter months between November and February (60%) and most of the individuals photographed for the ID catalogue have also been photographed in Cardigan Bay. Sightings of bottlenose dolphins comprised of 0.2% of boat-based sightings, 2.2% of land-based sightings and 8.5% of opportunistic sightings between 2007 and 2014 (Felce, 2014).

114. No bottlenose dolphin were recorded in most recent 2018 boat-based and land-based surveys, but 29 opportunistic sightings of the species were reported to MWDW, comprising 5.8% of opportunistic sightings (Clark *et al*., 2019).

115. For bottlenose dolphin, the distribution maps (developed by Waggitt *et al*., 2019) showed a clear pattern of higher density to the western coastal areas of the UK, extending southwards to the Bay of Biscay (**Plate 5.9**; Waggitt *et al*., 2019). The distribution maps indicated a 'corridor' of increased bottlenose dolphin density travelling from west of Scotland, southwards around the west coast of Northern Ireland and the Republic of Ireland, and through the centre

of the Bay of Biscay. Bottlenose dolphin densities in and around the Project windfarm site were low (windfarm site with 4km buffer is 0.0006 animals/km^2).

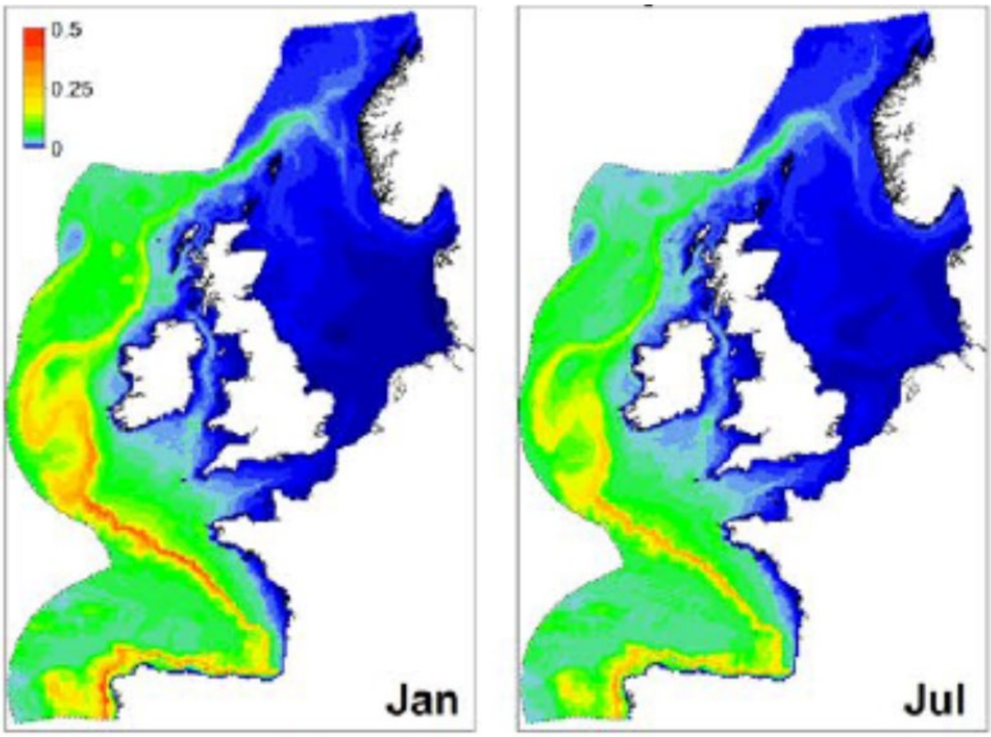


Plate 5.9 Spatial variation in predicted densities (individuals per km of the offshore ecotype bottlenose dolphin in January and July in the North-East Atlantic). Values have been provided at 10km resolution (Waggitt et al., 2019)

116. Whilst Waggitt *et al*. (2019) incorporated the offshore ecotype in their models, the fine-scale distribution maps by Evans and Waggitt (2023) identified clear coastal hotspots from the inshore ecotype. The year-round importance of Cardigan Bay and the Llŷn Peninsula is reflected in **Plate 5.10**. As already indicated from the HiDef monthly surveys, bottlenose dolphins were nearly absent, and the density derived from Evans and Waggitt (2023) confirmed low densities in the windfarm site with 4km buffer:

- Annual: 0.0011 animals/km^2
- Summer: 0.0013 animals/km^2
- Winter: 0.0009 animals/km^2

117. A proportion of the bottlenose dolphins with their summer residency in Cardigan Bay, extended their ranges between October and April to the North Wales coast and as far as the IoM (B. Manley (from MWDT) 2023, personal communication, 27 July 2023; Lohrengel *et al.* 2018).

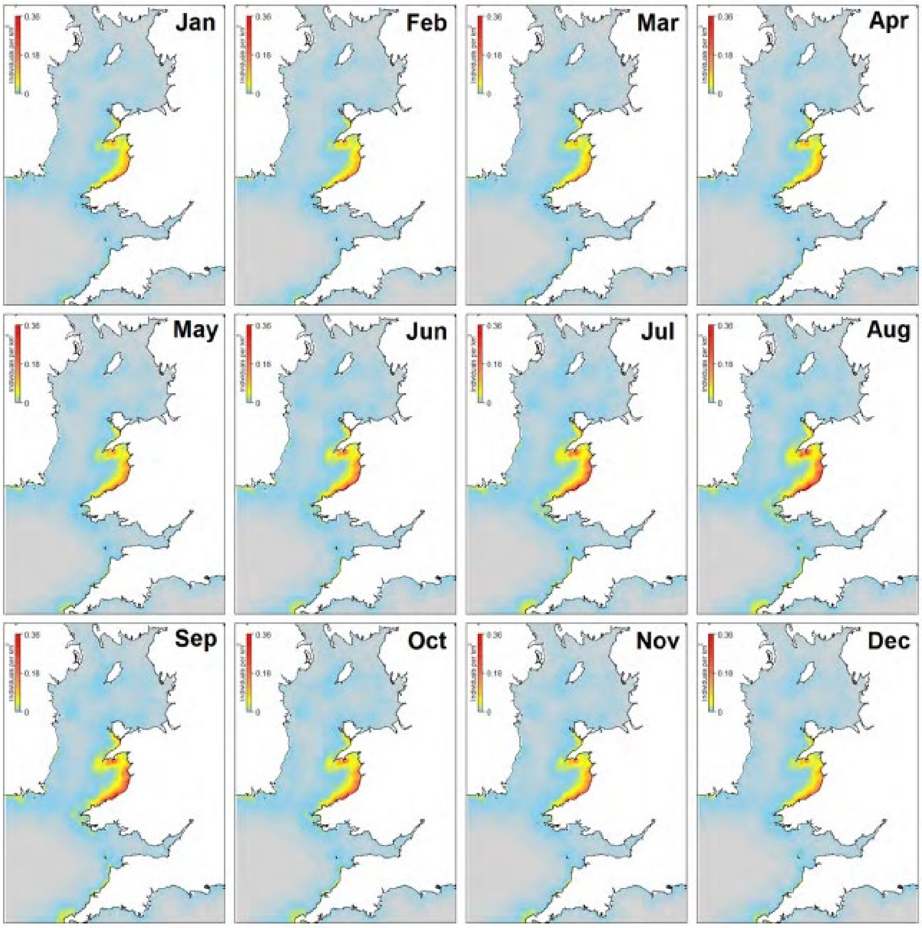


Plate 5.10 Bottlenose dolphin (inshore ecotype) modelled densities by month. Values have been provided at 2.5km resolution (Evans and Waggitt, 2023)

5.2.2 Diet

118. Bottlenose dolphin are opportunistic feeders and take a wide variety of fish and invertebrate species, benthic and pelagic fish (both solitary and schooling species), including:

- Haddock *Melanogrammus aeglefinus*
- Saithe *Pollachius virens*
- Pollock *Pollachius pollachius*
- Cod *Gadus morhua*
- Whiting *Merlangius merlangus*
- Hake *Merluccius merluccius*
- Blue whiting *Micromesistius poutassou*
- Bass *Dicentrarchus labrax*
- Mullet *Mugilidae*
- Mackerel *Scombridae*

- Salmon *Salmo salar*
- Sea trout *Salmo trutta trutta*
- Flounder *Platichthys flesus*
- Sprat *Sprattus sprattus*
- Sandeels (Ammodytidae)

119. Octopus and other cephalopods have also all been recorded in the diet of bottlenose dolphin (Santos *et al*., 2001; Santos *et al*., 2004; Reid *et al*., 2003).

120. Diet analysis has suggested that bottlenose dolphin are selective opportunists and although they may have preference for a type of prey, their diet seemed to be determined largely by prey availability. Research in Australia has shown that when presented with a choice, they would preferentially feed on certain types of prey, particularly those with a high fat content (Corkeron *et al*., 1990).

121. Analysis of the stomach contents of ten bottlenose dolphin in Scottish waters from 1990 to 1999 revealed that the main prey were cod (29.6% by weight), saithe (23.6% by weight), and whiting (23.4% by weight), although other species including salmon (5.8% by weight), haddock (5.4% by weight) and cephalopods (2.5% by weight) were also identified in lower number (Santos *et al*., 2001).

122. In Irish waters, haddock, saithe and pollock were the dominant prey species ingested, followed by whiting, blue whiting, Atlantic mackerel and horse mackerel; cephalopods were also important prey (Hernandez-Milian *et al*., 2015).

5.3 Common dolphin

5.3.1 Distribution

5.3.1.1 Abundance

123. As reviewed in BEIS (2022b), during summer common dolphin were widely distributed throughout the northeast Atlantic, from coastal waters to the mid-Atlantic ridge, from the Azores and the Strait of Gibraltar to Norway, with the majority of sightings having been reported in waters south of 60°N (Murphy *et al*., 2013). Analysis of summer sightings on shelf waters around the UK and adjacent waters showed the vast majority of common dolphins to occur in waters above 14°C in temperature (MacLeod *et al*., 2008; Cañadas *et al*., 2009). Strong seasonal shifts in their distribution have been noted, with winter inshore movements onto the Celtic Shelf and into the western English Channel and St. George's Channel resulting in pronounced concentrations (Northridge *et al*., 2004).

124. The JNCC Cetacean Atlas (Reid *et al*., 2003) reported common dolphin as favouring deep-water habitats. Common dolphin have been recorded in UK waters year-round, and the UK Cetacean Status Review of 2019 reported this species as 'common' in the Southern Irish Sea between May and October (Penrose, 2020).

125. Information on dispersal patterns and site fidelity was scarce, thus the reference population for common dolphin has been based on that of the CGNS MU, as outlined in **Section 1.1.1** (**Plate 1.3**) and was estimated to be 102,656 (CV = 0.29) animals (IAMMWG, 2023).

126. The ObSERVE aerial surveys of Irish waters found common dolphin to be widely distributed in shelf waters off the south and west coasts of Ireland, with higher numbers observed in winter (BEIS, 2022b; Rogan *et al*., 2018). They have also been the most frequently sighted and abundant cetacean recorded during Celtic Sea herring surveys off the south coast of Ireland in October (BEIS, 2022b; O'Donnell *et al*., 2017, 2018).

5.3.1.2 Density

127. Common dolphins have occasionally been sighted in IoM territorial waters, with most sightings being highly seasonal and reported in June, July and August (84%) (Howe, 2018). Sighting in IoM waters comprised 1.4% of boat-based sightings, 3.3% of sightings from land-based surveys and 3.3% of opportunistic sightings between 2007 and 2014 (Felce, 2014). No common dolphins were observed in the 2018 season during boat-based and land-based surveys, but 13 opportunistic sightings were reported, comprising 2.6% of opportunistic sightings (Clark *et al*., 2019).

128. The JCP Phase III Report (Paxton *et al*., 2016) also identified higher density estimates to the West of Ireland and in the Hebrides.

129. Distribution maps developed by Waggitt *et al*. (2019) indicated the highest density in the southwest of the Irish Sea and the Celtic Deep, and lower densities in the Irish Sea and West Scotland. There were also seasonal differences, with higher densities in July compared to January, particularly evident in the Celtic Deep (**Plate 5.11**). The densities were modelled over the Project site with 4km buffer:

- Annual average: 0.019 animals/km^2
- Summer 0.024 animals/km^2
- Winter 0.015 animals/km^2

130. **Plate 5.12** highlights common dolphin abundance in the Celtic Deep within the St George's Channel in the most recently modelled density maps by Evans and Waggitt (2023). Seasonal differenced were highlighted in the densities

derived by Evans & Waggitt (2023), modelled over the Project site with 4km buffer:

- Annual average: 0.00014 animals/km^2
- Summer 0.0002 animals/km^2
- Winter 0.00008 animals/km^2

131. The species has been noted to prefer shelf-edge habitats where water temperatures exceed 15°C. Although common dolphins were observed in one month during the HiDef surveys sightings were rare in the Project area.

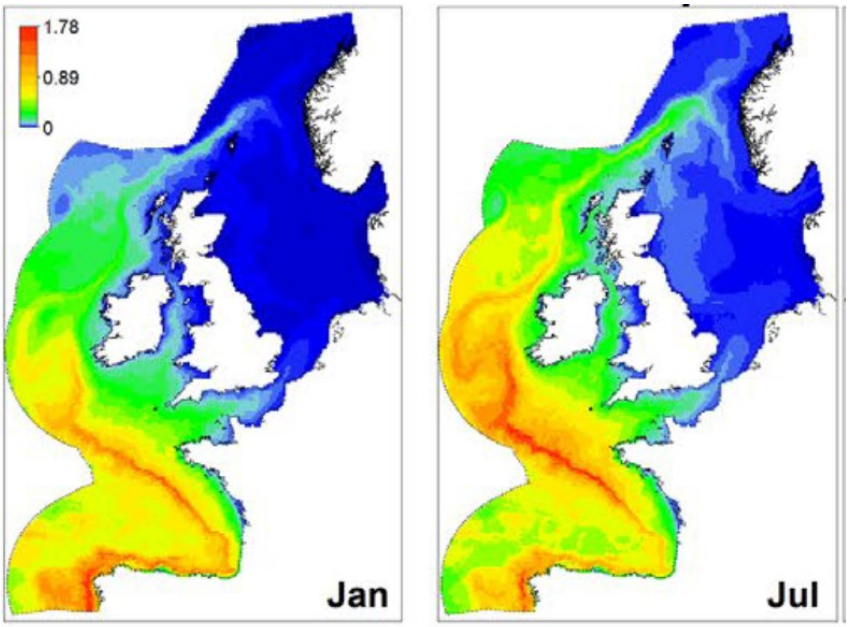


Plate 5.11 Spatial variation in predicted densities (individuals per km of common dolphin in January and July in the North-East Atlantic). Values are provided at 10km resolution (Waggitt et al., 2020)

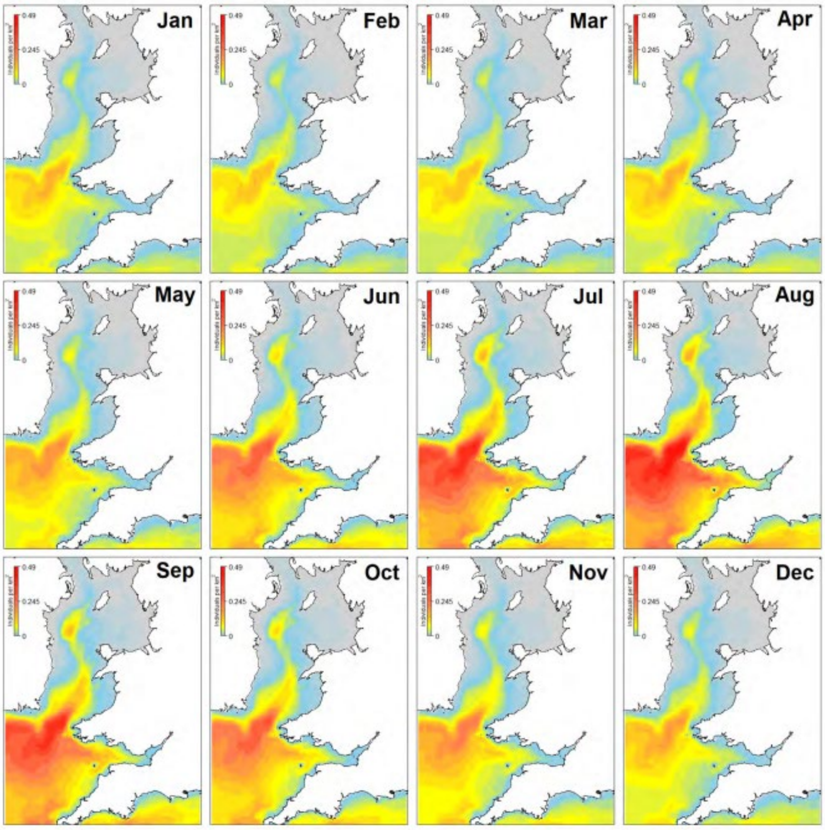


Plate 5.12 Common Dolphin modelled densities by month. Values have been provided at 2.5km resolution (Evans and Waggitt, 2023)

132. No common dolphin were observed in the Irish Sea area during the SCANS-III surveys in July, thus there were no estimated densities for either block F or E (**Plate 5.13**).

133. Similarly, during SCANS-IV (Gilles *et al*., 2023), there were no common dolphin sightings in block CS-E (**Plate 5.14**), but a few in the adjacent block CS-D resulting in a density of 0.0272 animals/km^2 (CV = 0.814; 95% CL = 32 - 2,990) with an abundance estimate of 949 individuals.

134. In order to find a density to best represent the wider area for common dolphin, data from Evans and Waggitt (2023) and Waggitt *et al*. (2019) were applied to the area of SCANS-IV block CS-E.

135. The Waggitt *et al*. (2019) densities averaged across the SCANS-IV block CS-E where the Project is located were:

- Annual: 0.022 animals/km^2
- Summer: 0.028 animals/km^2
- Winter: 0.017 animals/km^2

136. The Evans and Waggitt (2023) densities averaged across the SCANS-IV block CS-E where the Project is located were:

- Annual: 0.00008 animals/km^2
- Summer: 0.00011 animals/km^2
- Winter: 0.00005 animals/km^2

137. Being the worst-case, the mean summer density, derived from Waggitt *et al*. (2019) data over the SCANS CS-E block was taken forward for the impact assessment:

- Mean summer density: 0.028 common dolphin/km^2

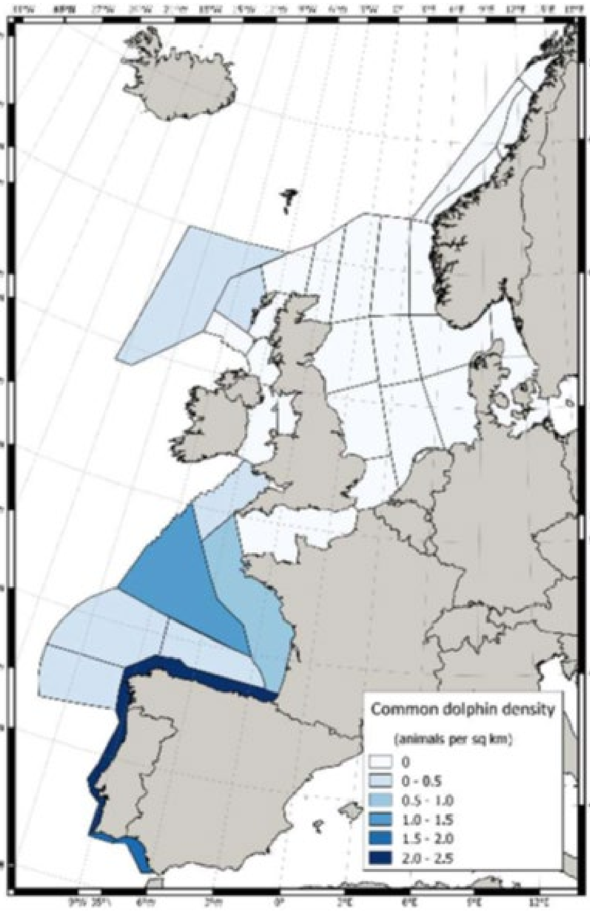


Plate 5.13 Estimated density in each survey block for common dolphin from SCANS-III (Hammond et al., 2021)

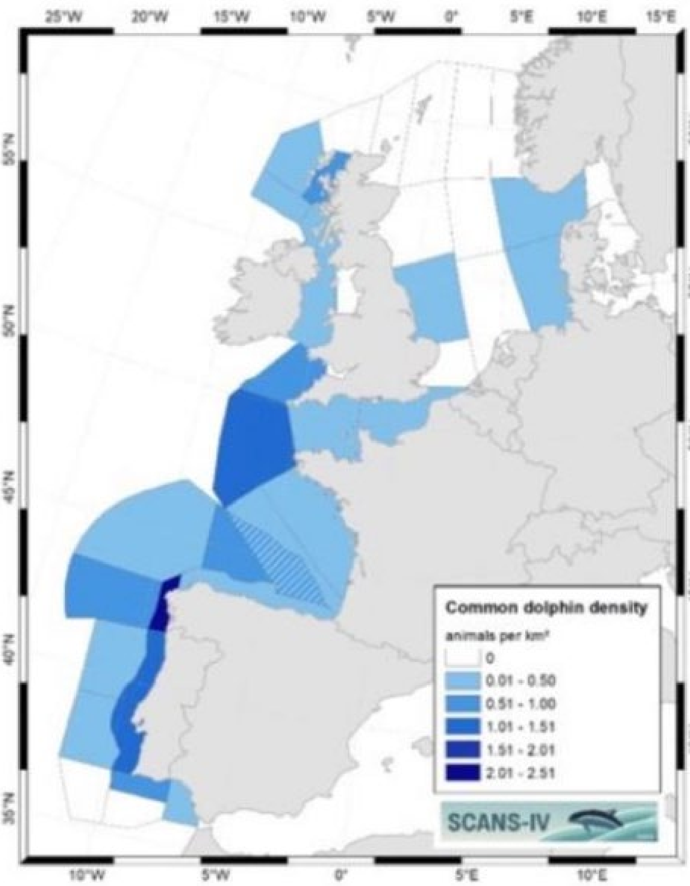


Plate 5.14 Estimated density in each survey block for common dolphin from SCANS-IV (Gilles et al., 2023)

5.3.2 Diet

138. Common dolphin are cooperative feeders, working within a pod to capture prey. They have been noted to have a varied diet of fish including haddock *Melanogrammus aeglefinus*, mackerel *Scomber scombrus*, Atlantic horse mackerel *Trachurus trachurus*, blue whiting *Micromesistius poutassou*, anchovy *Engraulida* spp., and sardine *Sardina pilchardus* (Couperus 1997; Silva 1999; Santos *et al.,* 2013; Marçalo *et al.*, 2018) which have also been targeted by commercial fishers. Other prey items recorded in common dolphins have included cephalopods and crustacea (Brophy *et al.* 2009).

139. Analysis of 63 common dolphin stomach contents from the Bay of Biscay found that their diet was dominated by fish, with mackerel being the preferred fish and cephalopods recorded as a prey of secondary importance (Pusineri *et al.* 2007). Stomach contents of 71 stranded common dolphins along the French coast between 199-2002 contained sardine, anchovy, sprat and horse mackerel (Meynier *et al.*, 2008). This study also highlighted the temporal variations in diet composition, which were attributed to prey availability in the region. It further identified that prey composition and size varied in relation to sex and maturity status of the individual animal. Statistically, common dolphins

were more likely to select high energy prey over low energy prey, which was disregarded, even when highly abundant in the area (Spitz *et al.*, 2010).

5.4 Risso's dolphin

5.4.1 Distribution

5.4.1.1 Abundance

140. Risso's dolphin have been found to be distributed sporadically in UK waters, with individuals commonly recorded around the Hebrides, and seasonally in the Celtic and Irish Seas. The majority of Risso's dolphin sightings in UK waters have been reported around the Hebrides (BEIS, 2022b; Paxton *et al.*, 2014).

141. The JNCC Cetacean Atlas (Reid *et al.*, 2003) indicated this species in northwest Europe was primarily a continental shelf species, and most sightings were in western Scotland around the Outer Hebrides. Clusters of sightings were also recorded in the southern Irish Sea and off southwest Ireland, central and southern North Sea and the Channel.

142. The JCP Phase III Report (Paxton *et al.*, 2016) identified local relative abundance off the west coast of Ireland, the northern Irish Sea and the Hebrides.

143. Risso's dolphin have been the most commonly seen dolphin species in IoM territorial waters, with almost all sightings reported between March and September, located primarily on the east and southern coasts of Manx waters (Howe, 2018). Sightings comprised of 6.5% of sightings from boat-based surveys, 13.2% from land-based surveys and 18.5% of opportunistic sightings between 2007-2014 (Felce, 2014) and 7.9% of sightings from boat-based surveys, 18.7% from land-based surveys and 30.4% of opportunistic sightings in 2019 (Clark *et al.*, 2019).

5.4.1.2 Density

144. Distribution maps by Waggitt *et al.* (2019) indicated higher densities off the west coast of Ireland and the Hebrides. There were also seasonal differences, with higher densities in July than in January, particularly to the north of their range which extended to the North Sea and Irish Sea (**Plate 5.15**; Waggitt *et al.*, 2019). The densities were modelled over the Project site with 4km buffer as follows:

- Annual average: 0.00024 animals/km^2
- Summer 0.0003 animals/km^2
- Winter 0.0002 animals/km^2

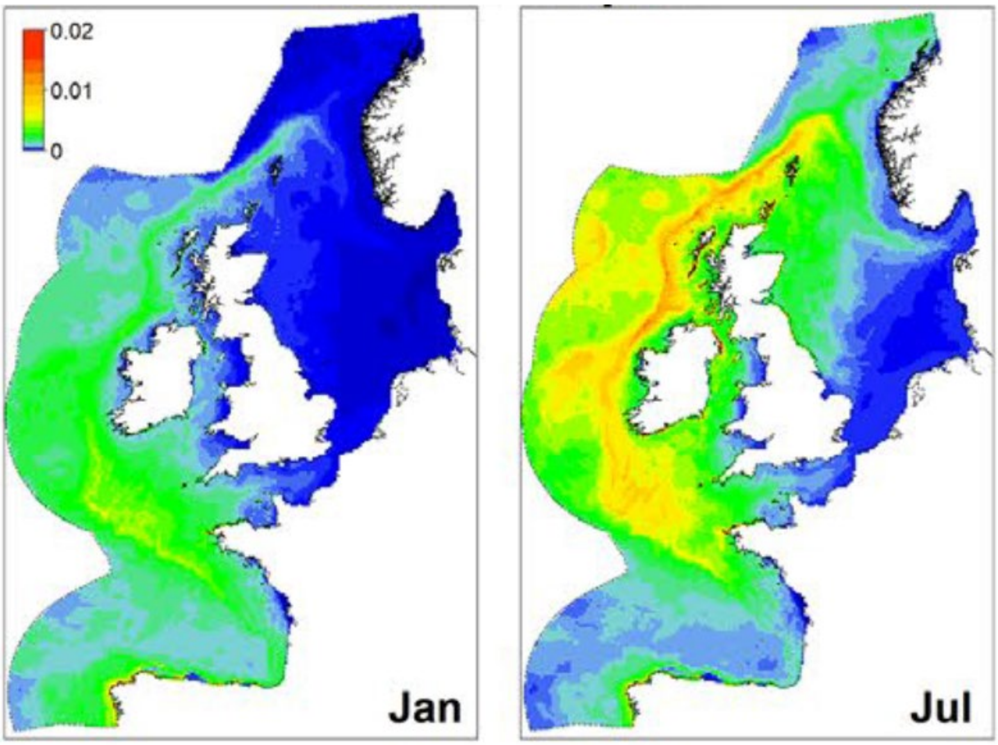


Plate 5.15 Spatial variation in predicted densities (individuals per km of Risso's dolphin in January and July in the North-East Atlantic). Values have been provided at 10 km resolution (Waggitt et al., 2019)

145. As per modelled density maps by Evans and Waggitt (2023), Risso's dolphin sightings occurred mainly between June and October, suggesting that the species moved offshore or out of the region. Densities derived by Evans & Waggitt (2023) were modelled over the Project site with 4km buffer as follows:

- Annual average: 0.00003 animals/km^2
- Summer 0.00005 animals/km^2
- Winter 0.00002 animals/km^2

146. In the same report, the authors stated findings of photo-identified individuals from North Anglesey with matches in southwest Cornwall, south-east Ireland, Pembrokeshire, Bardsey Island and the west Llŷn Peninsula, the IoM, and the Hebrides (citing Stevens (2014) and Mandlik (2021) in Evans and Waggitt, 2023).

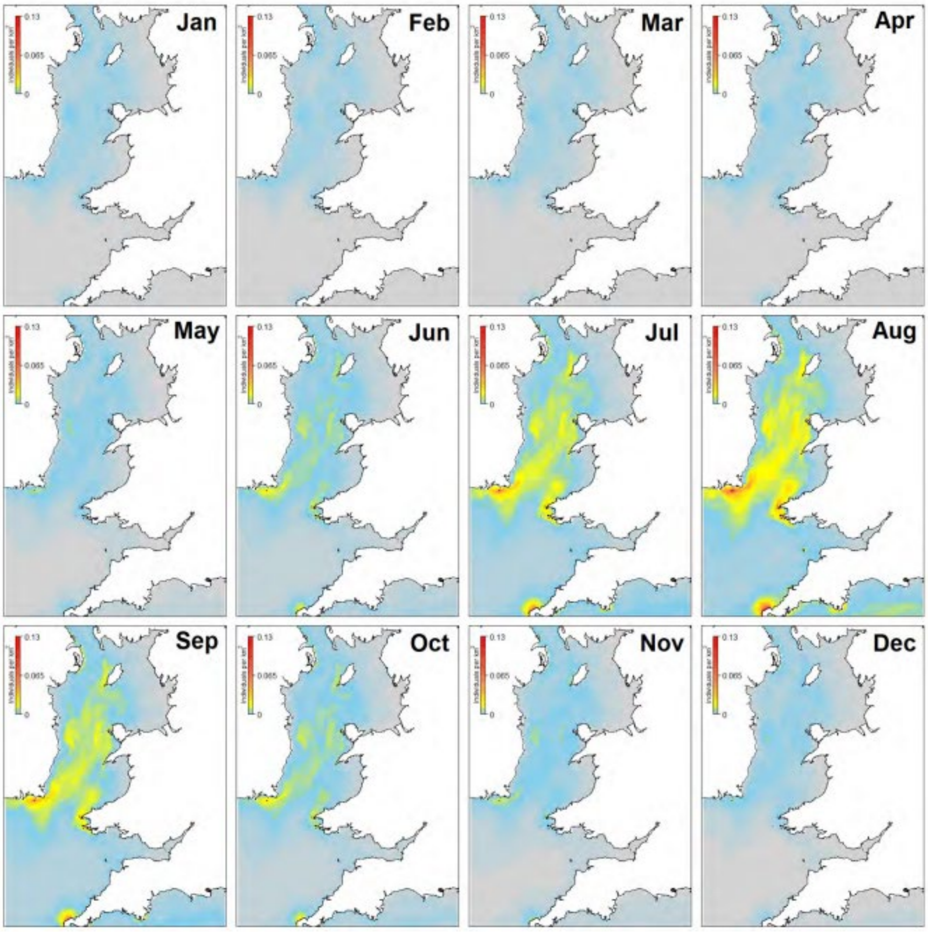


Plate 5.16 Risso's Dolphin modelled densities by month. Values have been provided at 2.5km resolution (Evans and Waggitt, 2023)

147. The SCANS-III survey recorded no Risso's dolphin within survey block F, in which the Project is located (Hammond *et al*., 2021), but in the adjacent block E had an abundance estimate of 1,090 Risso's dolphins (95% CL = 0 – 2,843) with a density estimate of 0.0313/km^2 (CV = 0.686; Hammond *et al*., 2021).

148. During SCANS-IV survey, Risso's dolphin were not seen in block CS-E, but were recorded in the adjacent block CS-D, near the southern tip of the Isle of Man. The density of this block was 0.0022 animals/km^2 (CV = 1.012; 95% CL = 2 – 259) and the abundance estimate was 75 individuals.

149. In order to find a density that represented the wider area for Risso's dolphin, the best data from Evans and Waggitt (2023) and Waggitt *et al*. (2019) were applied to the SCANS-IV block CS-E, where the Project is located.

150. The Waggitt *et al*. (2019) data applied to the SCANS block CS-E derived the following average seasonal densities:

- Annual: 0.0004 animals/km^2
- Summer: 0.0006 animals/km^2
- Winter: 0.0003 animals/km^2

151. The Evans and Waggitt (2023) densities averaged across the SCANS-IV block CS-E were:

- Annual: 0.0002 animals/km^2
- Summer: 0.0003 animals/km^2
- Winter: 0.0001 animals/km^2

152. Being the worst-case of all densities presented, the mean summer density, derived from Waggitt *et al*. (2019) data over the SCANS CS-E block was taken forward for the impact assessment:

- 0.0006 Risso's dolphin/km^2

153. The reference population was based on the population estimate of the CGNS MU of 12,262 (CV=0.46) (CI = 5,227 – 28,764) (IAMMWG, 2023).

5.4.2 Diet

154. Risso's dolphin primarily feed on cephalopods, with some fish and krill. Limited behavioural research was available, but it has been claimed that this species primarily feeds at night. The stomach contents of 11 dolphins stranded between 1992 and 2004 across Scotland were analysed (MacLeod *et al*., 2014) from which seven cephalopod taxa and three fish taxa were identified, however cephalopods made up 98% of the total prey (by weight and number). Analysis of the stomach contents of six stranded Risso's dolphins in the Mediterranean Sea found a total of 578 cephalopod beaks, identified as 386 individuals from 19 different species of Coleoidea cephalopods, one Sepiolida, eight Octopoda, and ten species belonging to the Order Oegopsida (Luna *et al*., 2022).

5.5 White-beaked dolphin

5.5.1 Distribution

5.5.1.1 Abundance

155. White-beaked dolphin have been found in temperate and sub-Arctic seas of the North Atlantic, usually over the continental shelf in waters of 50-100m depth (Reid *et al*., 2003). In UK waters, sightings occurred throughout the year, but were slightly more frequent from July to October (Reid *et al*., 2003).

156. Their distribution has been generally restricted to the northern half of UK waters, with greatest abundance in the central and northern North Sea, Orkney and Shetland and northwest Scotland (BEIS, 2022b).

157. There was only one MU for white-beaked dolphins, the CGNS MU, which was estimated to hold a population of 43,951 individuals (CV = 0.22) (IAMMWG, 2023).

5.5.1.2 Density

158. For white-beaked dolphin, the distribution maps by Waggitt *et al*. (2019) indicated higher densities in the northern North Sea and around the coasts of Scotland, with decreasing densities southwards of Scotland along the east coast of England. There was also a clear seasonal difference in the densities of white-beaked dolphin, with higher densities in July, particularly to the north of their range (**Plate 5.17**; Waggitt *et al*., 2019). Examination of this data, and all 10km grids that overlapped with the windfarm site, indicated an average density estimate for the windfarm site and 4km buffer of:

- Annual: 0.0053 animals/km^2
- Summer: 0.0052 animals/km^2
- Winter: 0.0054 animals/km^2

159. Evans and Waggitt (2023) were unable to provide density maps for this species as they only occurred rarely in the Irish Sea and the Bristol Channel.

160. The SCANS-III and IV surveys recorded no white-beaked dolphin within the survey blocks in which the Project is located, nor in any adjacent survey blocks (Gilles *et al*., 2023; Hammond *et al*., 2021).

161. In order to find a density to best represent the wider area for white-beaked dolphin, data Waggitt *et al*. (2019) were applied to the area of SCANS-IV block CS-E (where the Project is located) to give the following results:

- Annual: 0.00680 animals/km^2
- Summer: 0.00682 animals/km^2
- Winter: 0.00679 animals/km^2

162. Being the worst-case, the mean summer density, derived from Waggitt *et al*. (2019) data over the SCANS CS-E block was taken forward for the impact assessment as follows:

- Mean summer density: 0.007 white-beaked dolphin /km^2.

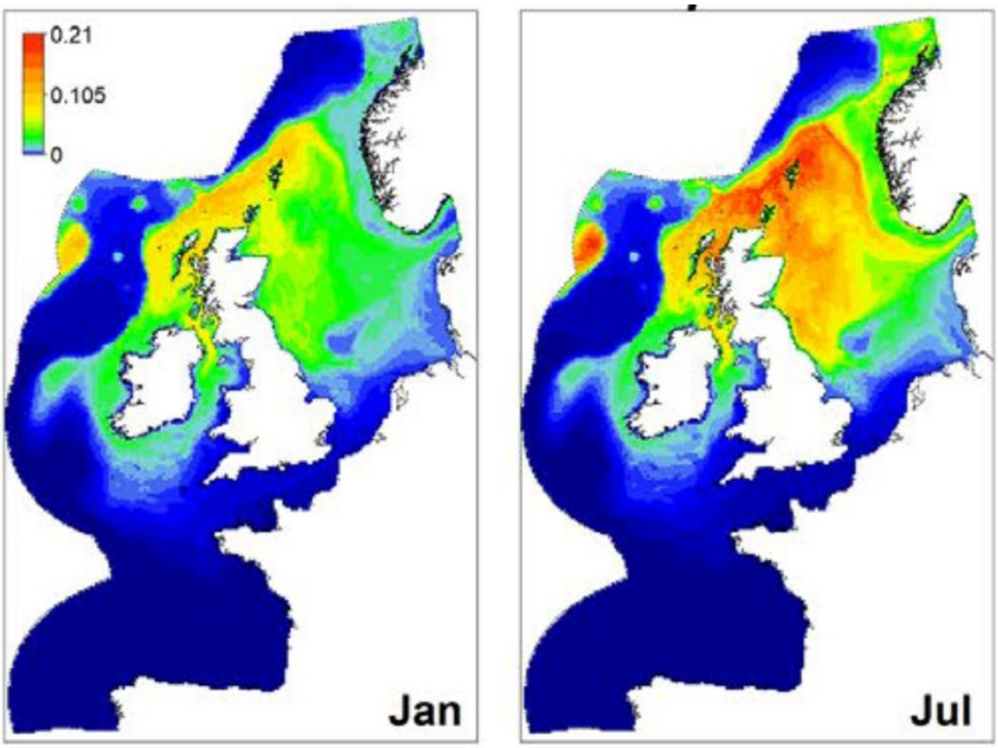


Plate 5.17 Spatial variation in predicted densities (individuals per km of white-beaked dolphin in January and July in the North-East Atlantic). Values have been provided at 10km resolution (Waggitt et al., 2019)

5.5.2 Diet

163. Dietary analysis for white-beaked dolphin stranded between 1992 and 2003 around the UK (Canning *et al.* 2008) and between 1968 and 2005 along the Dutch coast (Jansen *et al.* 2010) found that while a wide variety of prey species were identified, the majority of prey were Gadidae (cod and whiting), haddock and gobies. Canning *et al.* (2008) further identified that herring *Clupea harengus* and mackerel *Scomber scombrus* were also found in the stomachs and this was in line with older research that observed white-beaked dolphins associated with herring and mackerel shoals (Harmer, 1927; Fraser, 1946; Evans, 1980). Anecdotal evidence from fishers in Scotland suggested that individuals seen inshore may have coincided with mackerel appearing in the same areas (Canning *et al.* 2008).

5.6 Minke whale

5.6.1 Distribution

5.6.1.1 Abundance

164. Within UK waters, minke whale have most frequently been sighted in the western central-northern North Sea and west of Scotland around the Hebrides (BEIS, 2022b). They were primarily a seasonal visitor to UK waters, with

increased sightings from May to October, although some animals may remain in coastal waters year-round (BEIS, 2022b; Reid *et al*., 2003).

165. Minke whale have been sighted regularly in IoM territorial waters in the summer, they were highly seasonal and have been sighted mainly in the summer months, with 97.2% being reported between May and November. Sightings comprised 8.5% of boat-based sightings, 9.7% of sightings from land-based surveys and 14.2% of opportunistic sightings between 2007 and 2014 (Felce, 2014). In 2018 they comprised18.4% of boat-based sightings, 10.3% of sightings from land-based surveys and 12.5% of opportunistic sightings (Clark *et al*., 2019). Both the seasonality and the distribution of minke whale in IoM territorial waters were considered to reflect the seasonality and distribution of their main prey (Atlantic herring) (Howe, 2018).

166. Some genetic differentiation among individuals has been reported (e.g., Andersen *et al*., 2003), but this did not appear to be caused by geographic structuring within the north-east Atlantic (Anderwald *et al*., 2011). Minke whale of the North Atlantic were likely to be a single genetic population (Anderwald *et al*., 2012). Therefore, IAMMWG (2023) considered a single MU as appropriate for minke whale in UK waters which held an estimated population of 20,118 individuals (CV = 0.18).

5.6.1.2 Density

167. For minke whale, the distribution maps by Waggitt *et al*. (2019) indicated higher densities in the northern North Sea, around Scotland and Ireland, including the Celtic Sea area, with decreasing densities southwards of Scotland along the east coast of England (**Plate 5.18**). There were relatively low densities in and around the Project windfarm site (0.0019 animals/km^2), compared to other areas in UK waters. There was a clear seasonal difference in the densities of minke whale, with higher densities in July, which was particularly evident in the north of their range (**Plate 5.18,** Waggitt *et al*., 2019).

168. In addition, the distribution maps indicated a ‘corridor’ of increased minke whale density from north of Orkney, around the north and west coasts of the UK to Northern Ireland (**Plate 5.18**). Whilst the density of minke whales in the Project area in January was close to zero, it slightly increased in July, but the overall densities were relatively low. Densities across the windfarm site and 4km buffer were:

- Annual: 0.0019 animals/km^2
- Summer: 0.0026 animals/km^2
- Winter: 0.0013 animals/km^2

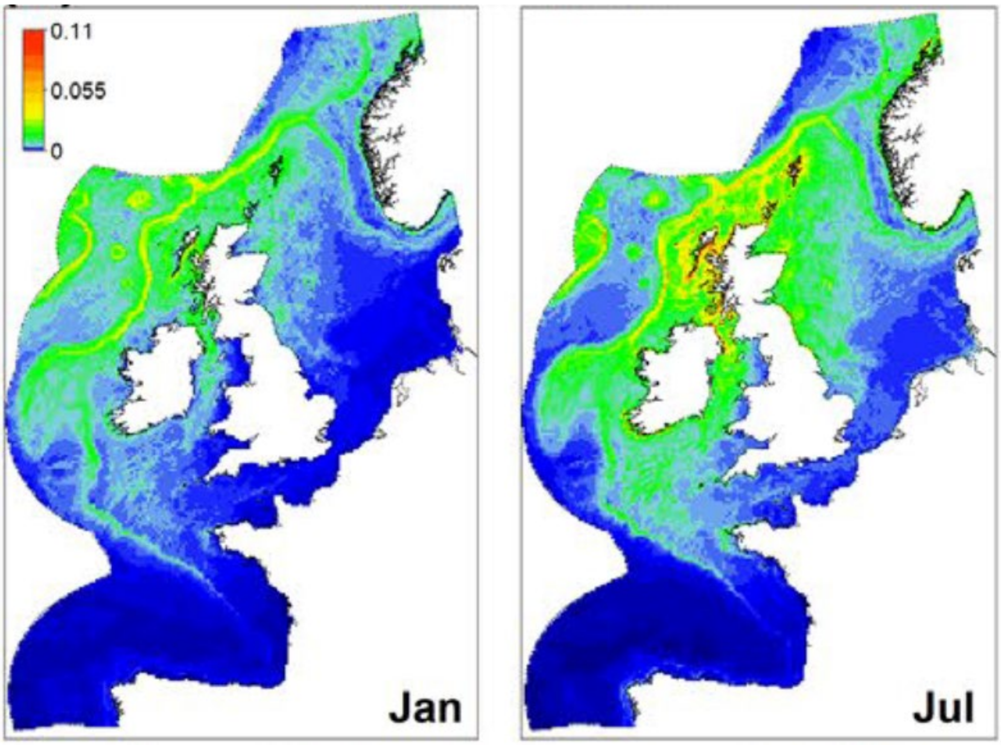


Plate 5.18 Spatial variation in predicted densities (individuals per km of minke whale in January and July in the North-East Atlantic). Values have been provided at 10km resolution (Waggitt et al., 2020)

169. Modelled distribution maps from Evans and Waggitt (2023) indicated a clear seasonal utilisation of the Celtic Deep Channel westwards from Pembrokeshire across the Celtic Deep to Co. Wexford, and from Co. Dublin north-eastwards to around the IoM between July and September (**Plate 5.19**). The densities across the site with a 4km buffer were:

- Annual: 0.0003 animals/km^2
- Summer: 0.0005 animals/km^2
- Winter: 0.0001 animals/km^2

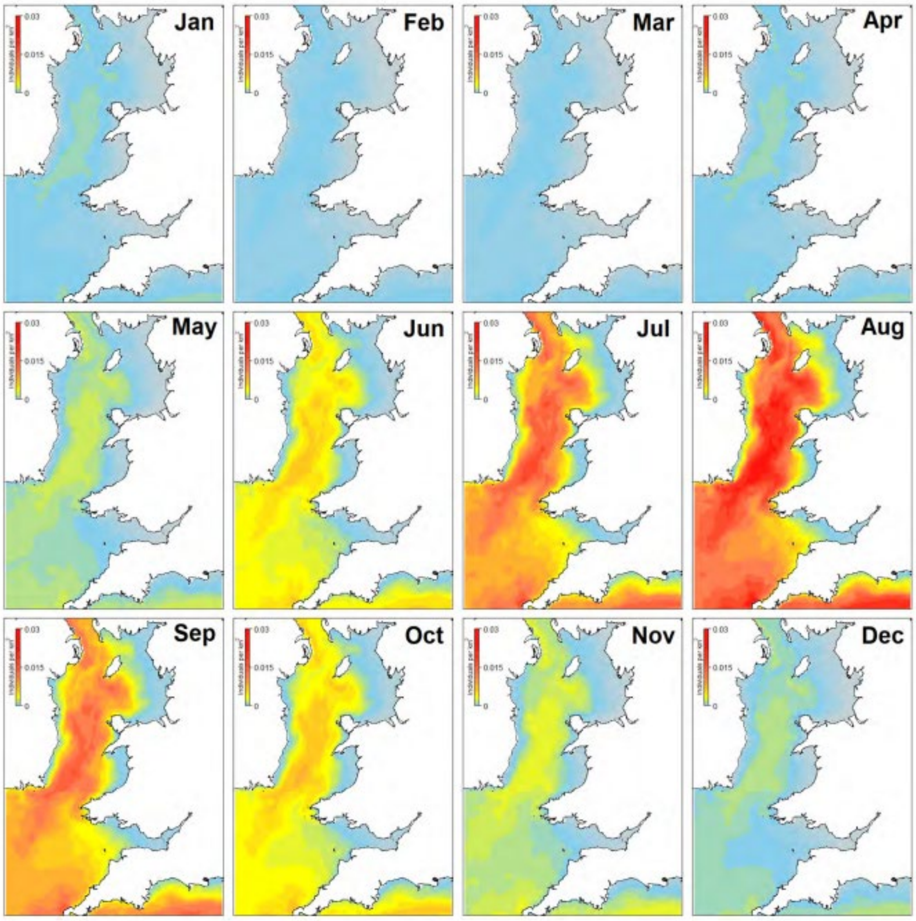


Plate 5.19 Minke whale modelled densities by month. Values provided at 2.5km resolution (Evans and Waggitt, 2023)

170. During the SCANS-III surveys, no minke whale were recorded within block F in which the Project is located. SCANS-III survey block E (located to the west of block F) had an abundance estimate of 603 minke whales (95% CL = 134 –1,753), with a density estimate of 0.0173 animals/km^2 (CV = 0.618; Hammond *et al*., 2021).

171. Only few minke whales were sighted during SCANS-IV, resulting in a low density:

- 0.0088 animals/km^2 (CV = 1.145)

172. The population abundance was estimated at 108 minke whales (95% CL = 1 - 491) in block CS-E (Gilles *et al*., 2023) (see **Plate 5.20**).

173. In order to find a density to best represent the wider area for minke whale, data from Evans and Waggitt (2023) and Waggitt *et al*. (2019) were applied were applied to the SCANS-IV block CS-E, where the Project is located.

174. The Waggitt *et al*. (2019) data applied to the SCANS block CS-E derived the following average seasonal densities:

- Annual: 0.003 animals/km^2

- Summer: 0.004 animals/km^2
- Winter: 0.002 animals/km^2

175. The Evans and Waggitt (2023) densities averaged across the SCANS-IV block CS-E where the Project is located were:

- Annual: 0.0006 animals/km^2
- Summer: 0.001 animals/km^2
- Winter: 0.0002 animals/km^2

176. Being the worst-case, the following density for SCANS-IV block CS-E was taken forward for the impact assessment:

- 0.0088 minke whale/km^2.

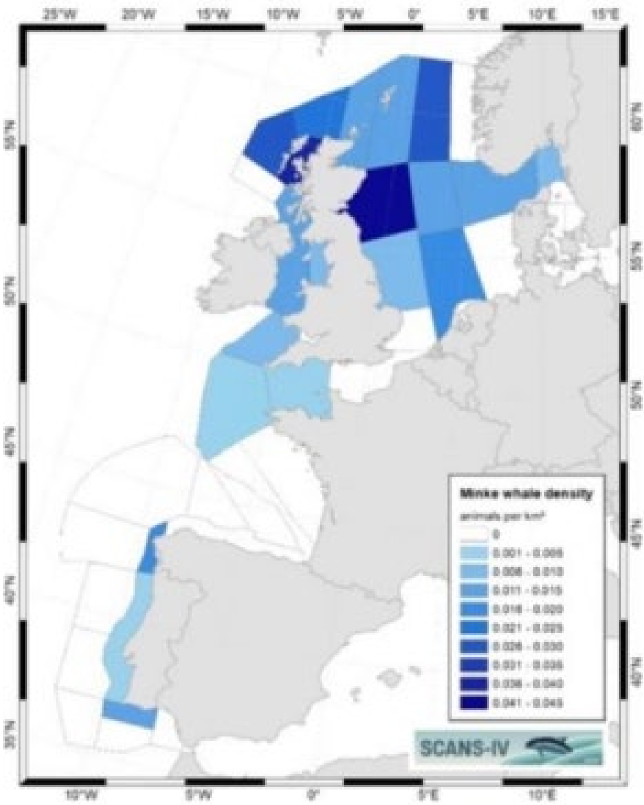


Plate 5.20 Estimated density in each survey block for minke whale from SCANS-IV (Gilles et al., 2023)

5.6.2 Diet

177. Minke whale feed on a variety of fish species, including herring, cod and haddock. They feed by engulfing large volumes of prey and water, the water is then 'sieved' out through their baleen plates and the remaining prey are swallowed whole.

178. A study into the diet of minke whale in the north-eastern Atlantic sampled a total of 210 minke whales forestomach contents from 2000 to 2004, with a

total of 37 forestomach samples analysed within the northern North Sea. Within this area, minke whale were found to prey upon a number of different species at the population level, however, 84% of individuals were found to prey upon only one species. Sandeels (56% of total prey by biomass) and mackerel (30% of total prey by biomass) were found to be the most dominant prey species for minke whale in the northern North Sea (Windsland *et al*., 2007).

5.7 Grey seal

5.7.1 Distribution

179. Grey seal have only been recorded in the North Atlantic, Barents and Baltic Sea with their main concentrations on the east coast of Canada and United States of America and in northwest Europe (SCOS, 2022).

180. Approximately 35% of the world's grey seals breed in the UK. 80% of these breed at colonies in Scotland, with the main concentrations in the Outer Hebrides and in Orkney. There were also breeding colonies in Shetland, on the north and east coasts of mainland Britain and in southwest England and Wales (SCOS, 2022). The IoM has provided a regionally important haul out and resting location for transient as well as resident grey seal (Howe, 2018a).

181. Grey seal have been recorded as wide ranging and able to breed and forage in different areas (Russell *et al*., 2013). They generally travelled between known foraging areas and back to the same haul-out site but also moved to new sites (Russel, 2016).

182. Carter *et al*. (2020, 2022) provided grey seal movement maps for foraging trips only (the tagging data was cleaned to remove data during the grey seal breeding season). This is shown in **Plate 5.21**, with grey seal foraging movements being primarily located along Ramsey and Skomer Islands, Bardsey Island, and the Dee Estuary with some movement offshore.

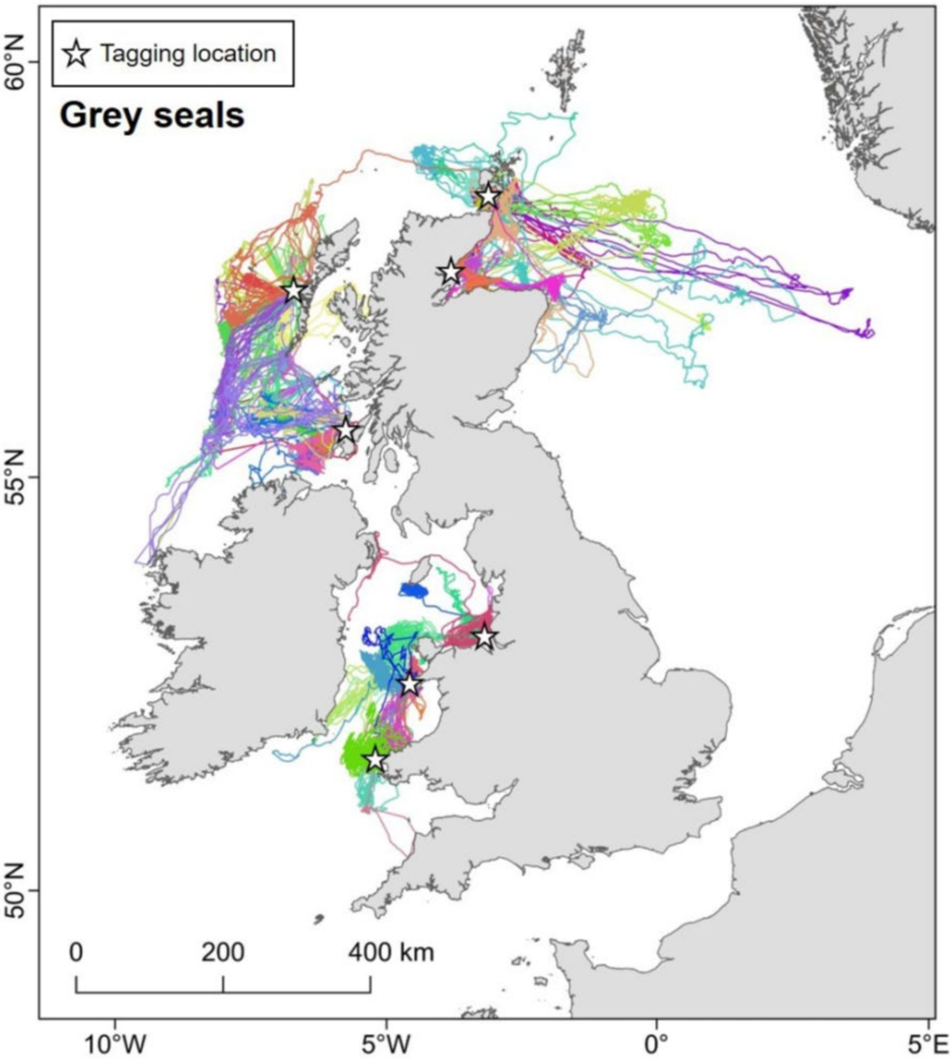


Plate 5.21 Grey seal tagging data, colour-coded by habitat preference region (Carter et al., 2020)

183. Grey seal typically forage in the open sea and they may range widely to forage and frequently travelled over 100km between haul-out sites (SCOS, 2022). Foraging trips can last anywhere between one and 30 days. Tracking of individual grey seals has shown that most foraging probably occurs within 100km of a haul-out site, although they have also been recorded feeding up to several hundred kilometres offshore (SCOS, 2022). The grey seal maximum foraging range has been estimated to be 448km based on tracking data (Carter *et al.*, 2022).

5.7.2 Haul-out sites

184. Compared with other times of the year, grey seal in the UK spent longer hauled out during their annual moult (between December and April) and during their breeding season (between August and December) (SCOS, 2020).

185. In the north and west Scotland, pupping occurred mainly between September and late November, whereas in eastern England it occurs between early November to mid-December (SCOS, 2022). Pups were typically weaned 17 to 23 days after birth, when they moulted their white natal coat, and then remained in the breeding colony for up to two or three weeks before going to sea. Mating occurred at the end of lactation and then adult females departed to sea and provided no further parental care (SCOS, 2022).

186. Two main haul-out sites of grey seals have been identified in northwest England; one in the Dee Estuary on the Welsh-English border (Hilbre Island) and one at South Walney. At South Walney, Cumbria Wildlife Trust (CWT) and Walney Bird Observatory have historically conducted counts of the seals primarily during the breeding and moulting seasons. These data indicated that grey seal abundance was steadily increasing (SCOS, 2020; CTW, 2023).

187. Starting in 2019, CWT have conducted low tide counts in August to provide Sea Mammal Research Unit (SMRU) with numbers comparable to those used in the independent estimate of grey seal abundance. From the unpublished data supplied by CWT (2023), numbers steadily increased with some fluctuations in the surveys from 2019 to 2023, with counts of 482 (2019), 315 (2020), 518 (2021), 287 (2022) and 466 (2023). Pups were recorded for the first time in 2015 but since then numbers of new pups have remained low, ranging between 2 in 2015 and 10 in 2017. Since then, an average of 6 pups have been born each year (CWT, 2023).

188. The Calf of Man, a small island southwest of the Isle of Man was considered to be the most important haul-out and pupping site in the territory, although there were other locations where pupping has been observed along its coast (Howe, 2018a). Since 2009, when annual dedicated pup surveys commenced, the number of pups increased from 27 to 84 in 2016; in 2021 the survey counted 62 pups of which four were confirmed dead (Stokes *et al*. 2021).

5.7.3 Abundance and density estimates for grey seal

5.7.3.1 Seal density maps

189. The following sections provide the grey seal at-sea density estimates from a grey seal mapping dataset (Carter *et al*., 2022). **Figure 5.1** shows the relative abundance of grey seals in the wider Project area as a percentage of the total UK population.

190. The relative seals at-sea abundance maps have been used to calculate grey seal density estimates for the windfarm site. The Carter *et al*. (2022) density maps were an update to the Russell *et al*. (2017) mapping and included updated tagging studies. These density maps only included tagging studies from the UK.

191. The resultant density of seals at-sea maps (Carter *et al*., 2022) differed from the Russell *et al*. (2017) maps, in that they showed the relative density of seals in each 5km-by-5km grid cell. Each grid cell showed the percentage of the overall seal population within the British Isles, which could then be related to the current best population estimate for each species. This ensured that the relative densities could be updated based on overall population level changes.

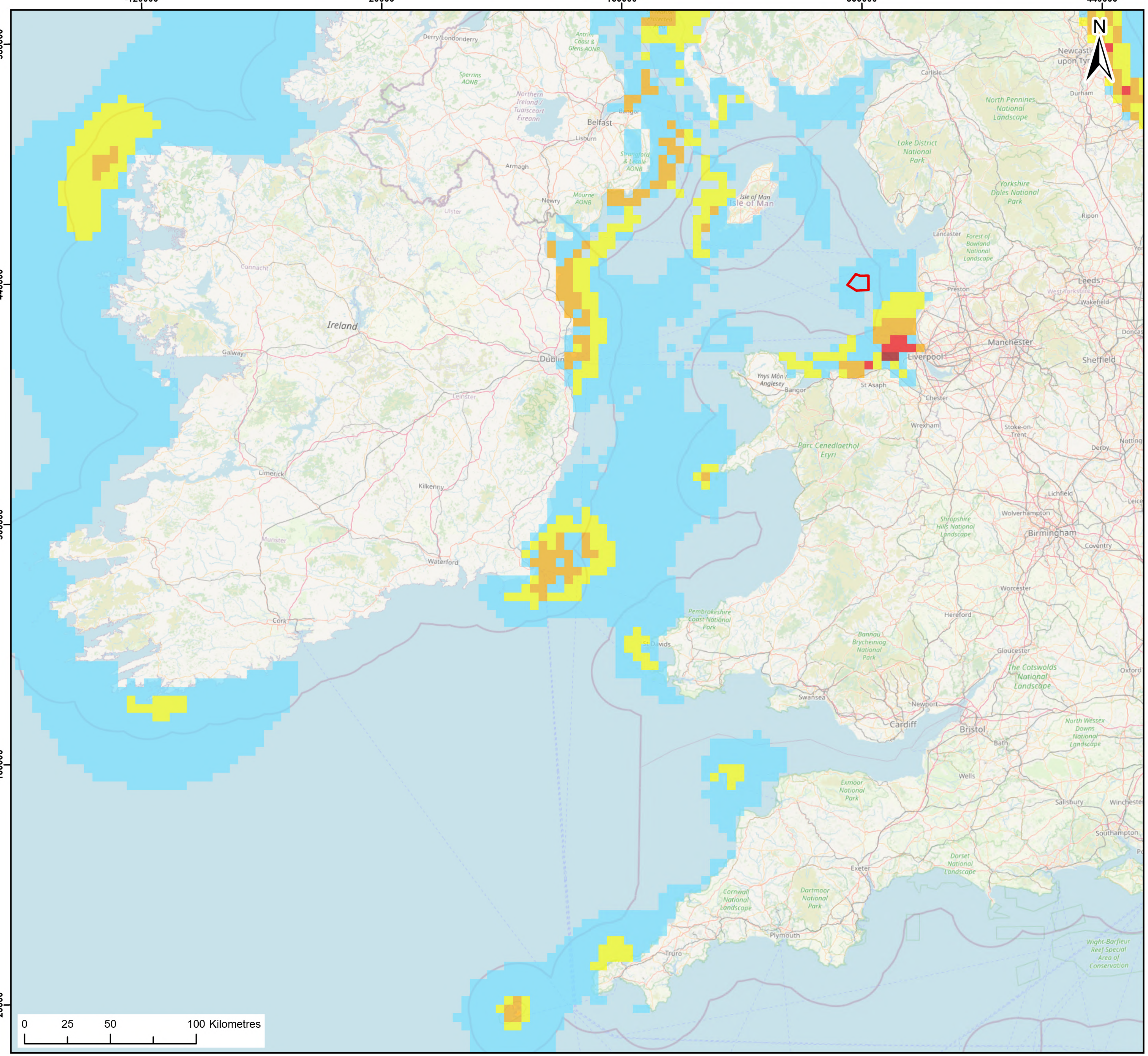
192. To calculate a density estimate for assessments from the Carter *et al*. (2022) data, the latest at-sea population of each species was used. A correction factor was also applied to the overall population level to take account of those individuals that were estimated to be on land (**Plate 5.22** shows mean percentage of at-sea population estimated to be present in each 5 km x 5 km grid square at any one time).

193. The total grey seal population in the British Isles, at-sea, was approximately 162,000 individuals. This at-sea estimate has been based on the most recent available (SCOS, 2022) grey seal August counts of 44,833 for the UK and RoI[7], which has been corrected for both those individuals that were not available to count (0.2515; SCOS-BP 21/02 in SCOS, 2021), and for those individuals that would be at-sea at any one time (0.8616; Russel *et al*., 2015). This was the population estimate used with the Carter *et al*. (2022) data to calculate density estimates for the windfarm site (see **Section 5.7.3.2**) The grey seal density estimates for the windfarm site have been calculated from the latest seal at-sea maps produced by SMRU (Carter *et al*., 2022), based on the 5km x 5km grids that overlap with the Project area.

194. The mean at-sea density estimate derived from Carter *et al*. (2022) data over the area of the windfarm site and 4km buffer was taken forward in the impact assessment:

- Mean at-sea density estimate of 0.100 grey seal/km^2

[7] Based on the latest grey seal counts provided by the 2021 grey seal counts (http://www.smru.st-andrews.ac.uk/scos/scos-data/august-seal-counts/august-seal-counts-england/) and the 2019 counts (SCOS, 2020).



Legend:

Morecambe Offshore Windfarm Site

Grey Seals Relative Abundance (Carter et al 2022)

% British Isles At-sea Pop per 25km2

- ≤ 0.001 (Transparent)
- 0.001 - 0.005
- 0.005 - 0.01
- 0.01 - 0.025
- 0.025 - 0.05
- 0.05 - 0.1

Report: Morecambe Offshore Windfarm: Generation Assets Environmental Statement

Title: Grey seal at-sea distribution. Maps show mean percentage of at-sea population estimated to be present in each 5km x 5km grid square at any one time (Carter et al., 2022)

Figure: 5.1 | Drawing No: PC1165-RHD-ES-OF-DG-Z-0101

Revision:	Date:	Drawn:	Checked:	Size:	Scale:
P01	16/11/2023	SB	GC	A3	1:2,250,000
P02	03/04/2024	JH	SB	A3	1:2,250,000

Co-ordinate system: WGS 1984 UTM Zone 30N

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5.7.3.2 Grey seal population counts

195. Grey seal population trends have been assessed from the counts of pups born during the autumn breeding season, when females congregate on land to give birth (SCOS, 2022). The pup production estimates have been converted to estimates of total population size (1+ aged population) using a mathematical model and projected forward (SCOS, 2022).

196. The most recent surveys of the principal grey seal breeding sites in Scotland, Wales, Northern Ireland and England resulted in an estimate of 67,850 pups (in 2019; 95% CL = 60,500 – 75,200; SCOS, 2022).

197. The estimated adult UK grey seal population size in regularly monitored colonies in 2022 was 162,000 (approximate 95% CL = 146,700-178,500; SCOS, 2022). This estimate was based on 2019 pup production and represented the total population alive on the first day of the 2022 breeding season.

198. The most recent counts of grey seal in the August surveys in 2016-2021, estimated that the minimum count of grey seals in the UK was 41,135 (SCOS, 2022).

199. In order to take account of the grey seals that were not available for counting during these surveys, a population scalar was added to provide a more accurate population estimate. Approximately 0.2515 grey seals were available to count within the August surveys (i.e., were hauled out), and therefore this has been used as a correction factor (SCOS-BP 21/02 in SCOS, 2021) to derive a more accurate population estimate of grey seal within each MU (rather than the number counted). The adjusted reference population estimates for relevant MUs for grey seal were therefore derived as shown in **Table 5.1.**

Table 5.1 Grey seal counts and population estimates

Population area	Grey seal haul-out count	Source of haul-out count data	Correction factor for seals not available to count	Grey seal total population
NW England MU	300	SCOS (2022)	0.2515	1,193
Wales	900	SCOS (2022)	0.2515	3,579
SW Scotland	517	SCOS (2022)	0.2515	2,056
NI MU	549	SCOS (2022)	0.2515	2,182
IoM	400	Howe (2018a)	-	400
E RoI MU	418	Morris and Duck (2019)	0.2515	1,662
SE RoI MU	556	Morris and Duck (2019)	0.2515	2,211
Total wider reference population	**3,640**			**13,283**

200. The total wider reference population taken forward to the assessment was 13,283 grey seals. Assessments have been undertaken in the context of the combined NW England MU and IoM population estimates (1,593 grey seal), as well as the wider reference population (13,283 grey seal). As a worst-case, it was assumed that all seals were from the nearest MUs (i.e. the combined NW England MU and IoM populations), although a more realistic assessment has also been presented based on the wider reference population, which took into account the movement of seals.

5.7.3.3 Limitations of approach

201. The inclusion of the aforementioned MUs was deemed sufficient for the impact assessment as they reflected known grey seal movements and distributions, and this was agreed to by Natural England during Scoping and via the Marine Mammal Ecology ETG process.

202. On the contrary, NRW's position on this matter was that all grey seals in the much larger OSPAR Region III: Celtic Seas area (**Plate 5.22**) should be used as the appropriate interim MU (NRW, 2021). The OSPAR III region held a population of 60,780 grey seals, whereas the wider reference population from the MUs included within the assessment totalled 13,283.

203. Should the OSPAR III region population be used in the impact assessment, the increase in population numbers would cause a dilution of animals affected in the assessment and was likely to underestimate effects. As such the most

precautionary approach (to use the reference population set out in **Section 5.7.3.2** above) has been taken.

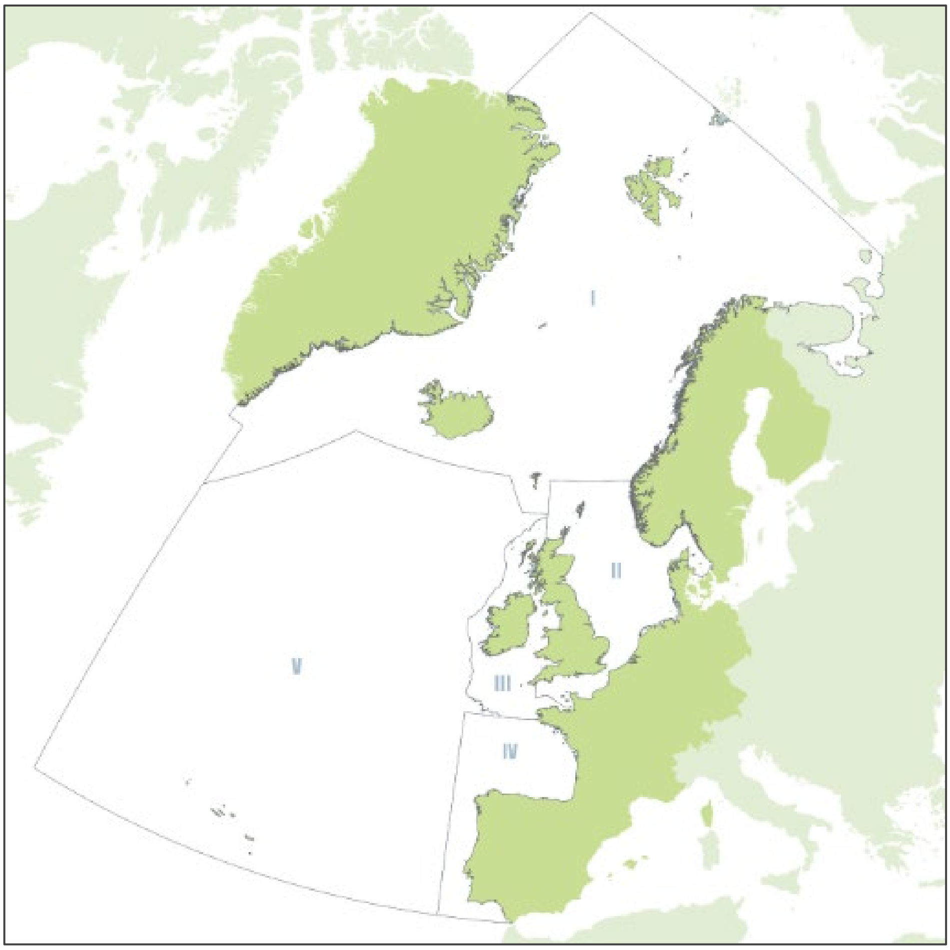


Plate 5.22 The north-east Atlantic divided into OSPAR region I: Arctic Waters, II: Greater North Sea, III: Celtic Seas, IV: Bay of Biscay and Iberian Coast, V: Wider Atlantic (Source: www.ospar.org)

5.7.4 Diet and foraging

204. Grey seals will typically forage in the open sea and return regularly to land to haul-out, although they may frequently travel up to 100km between haul-out sites. Foraging trips generally occurred within 100km of their haul-out sites, although grey seal can travel up to several hundred kilometres offshore to forage (SCOS, 2020). Grey seal generally travel between known foraging areas and back to the same haul-out site but will occasionally move to a new site. For example, movements have been recorded between haul-out sites on the east coast of England and the Outer Hebrides (SCOS, 2020).

205. Individual grey seals based at a specific haul-out site often make repeated trips to the same region offshore but will occasionally move to a new haul-out site and begin foraging in a new region (SCOS, 2020). Telemetry studies of grey seal in the UK have identified a highly heterogeneous spatial distribution

with a small number of offshore 'hot spots' continually utilised (Matthiopoulos *et al*., 2004; Russell *et al*., 2017).

206. Grey seals are generalist feeders, feeding on a wide variety of prey species (SCOS, 2020; Hammond and Grellier, 2006). Diet varies seasonally and from region to region (SCOS, 2020).

207. Principal prey items were sandeel, whitefish (such as cod, haddock, whiting and ling *Molva molva*) and flatfish (plaice *Pleuronectes platessa*, sole *Solea solea*, flounder, and dab *Limanda limanda*) (Hammond and Grellier, 2006). Amongst these, sandeels were typically the predominant prey species.

208. Food requirements depend on the size of the seal and fat content (oiliness) of the prey, but an average consumption estimate of an adult was 4 to 7kg per seal per day depending on the prey species (SCOS, 2020).

5.8 Harbour seal

5.8.1 Distribution

209. Harbour seals have a circumpolar distribution in the Northern Hemisphere and are divided into five sub-species. The population in European waters represents one sub-species *Phoca vitulina vitulina* (SCOS, 2020).

210. On the west coast of Britain harbour seal distribution has been generally restricted, with a total of five harbour seal counts in the NW England MU from 2016-2019 (SCOS, 2020). There was a total of 818 harbour seal recorded in counts in August 2021 in the NI MU (SCOS, 2022).

211. SMRU, in collaboration with others, deployed 344 telemetry tags on harbour seals around the UK between 2001 and 2012. The spatial distributions indicated harbour seals persisted in discrete regional populations, displayed heterogeneous usage, and generally stayed within 50km of the coast (Russell and McConnell, 2014). Tagged harbour seals were observed to have a more coastal distribution than grey seals and did not travel as far from haul-outs (Russell and McConnell, 2014).

212. Harbour seal tags, deployed between 2006 and 2017, were cleaned and analysed, and maps of tracks for all individuals included in a habitat preference analysis (n= 239) are shown in **Plate 5.23** (Carter *et al*., 2020).

213. Harbour seals generally made smaller foraging trips than grey seals, typically travelling 40-50km from their haul-out sites to foraging areas (SCOS, 2020). Tracking studies have shown that harbour seals travelled 50-100km offshore and could travel 200km between haul-out sites (Lowry *et al*., 2001; Sharples *et al.,* 2012). The range of these trips varied depending on the location and surrounding marine habitat. The typical and average foraging range for harbour seal was 50-80km (SCOS, 2021). Tracking data analysed in Carter

et al. (2022) produced a radius based on the maximum geodesic distance of 273km for harbour seals representing the species' maximum foraging range.

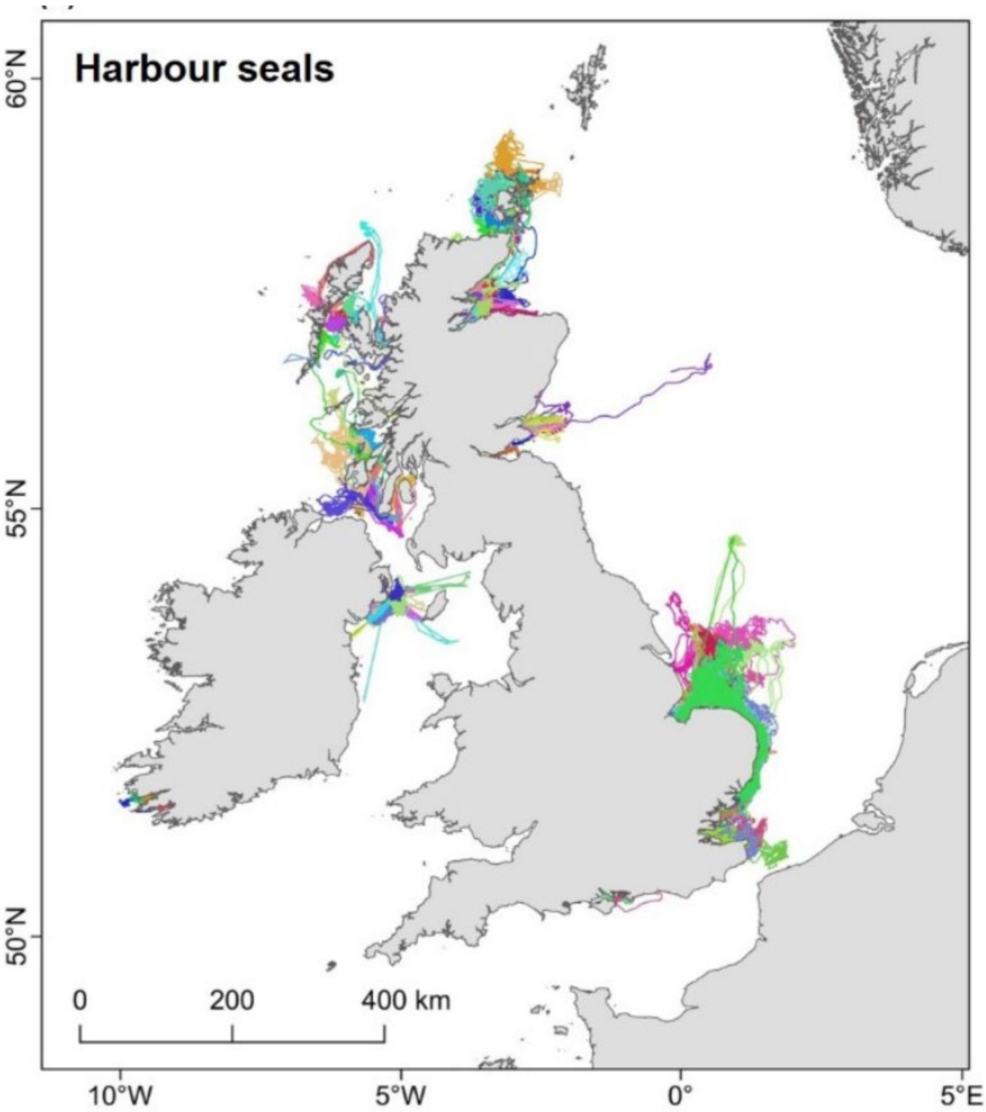


Plate 5.23 GPS tracking data for harbour seals available for habitat preference models. (Carter et al., 2020)

5.8.2 Haul-out sites

214. Harbour seal come ashore in sheltered waters, typically on sandbanks and in estuaries, but also in rocky areas. They regularly haul-out on land in a pattern that is often related to the tidal cycle (SCOS, 2020). Harbour seals give birth to their pups in June and July and pups can swim almost immediately after birth (SCOS, 2022). Harbour seals moult in August and spend a higher proportion of their time on land during the moult than at other times (SCOS, 2022).

215. Visits from harbour seal on the IoM are rare, but a small number haul out along the coast around The Sound and Maughold Head. Unlike for grey seal,

harbour seal are not a designated feature of any UK marine protected areas (Howe, 2018a).

5.8.3 Abundance and density estimates for harbour seal

5.8.3.1 Seal density maps

216. Impact assessments were based on densities derived from desk-based sources. Carter *et al*. (2022) provided habitat-based predictions of at-sea distribution for harbour seal around the British Isles. The habitat preference approach predicted estimates per species, on a 5km x 5km grid, of relative at-sea density for seals hauling-out in the British Isles.

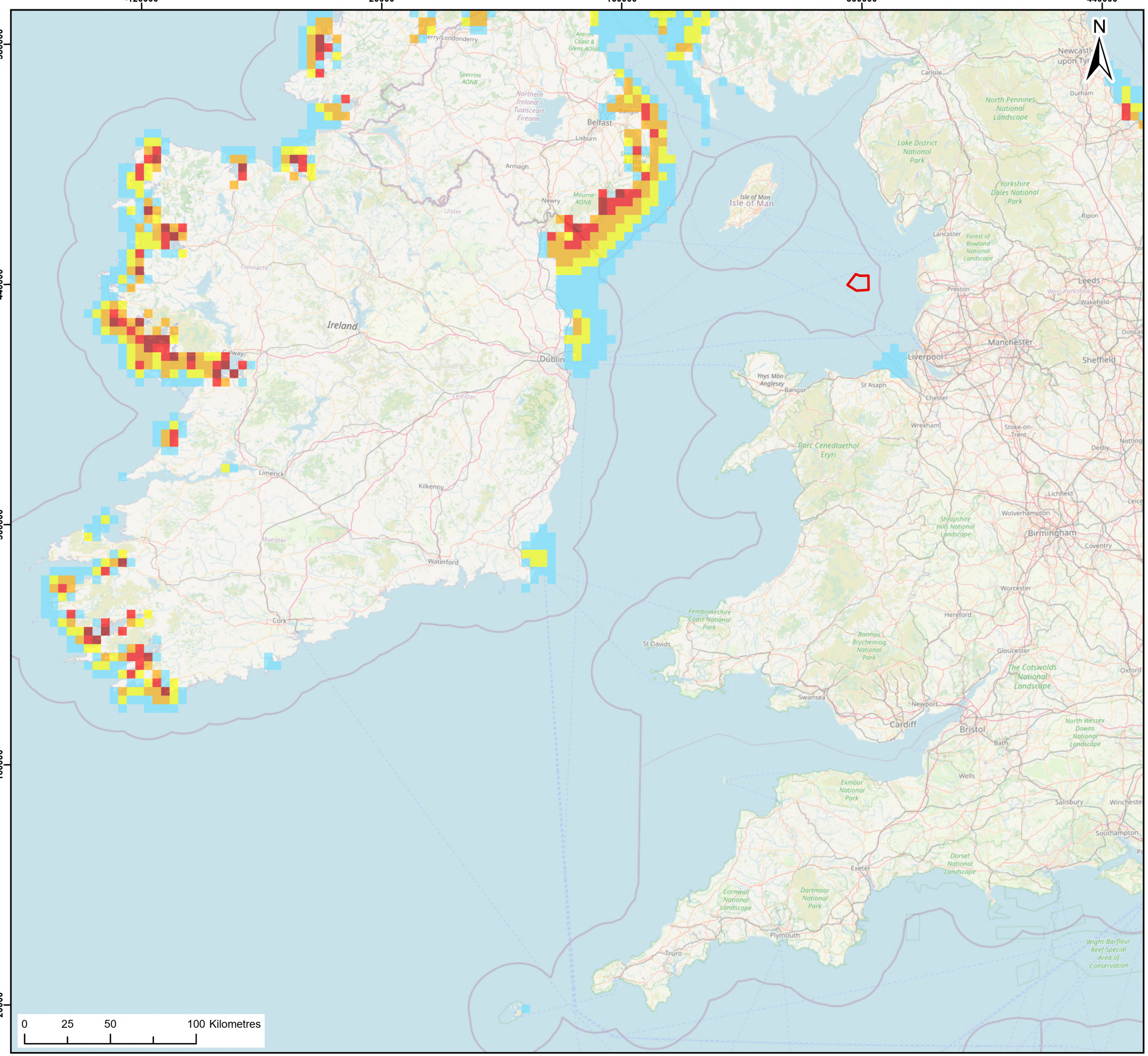
217. To calculate a density estimate to be used in the impact assessments from the Carter *et al*. (2022) data, the current at-sea population of each species was used. A correction factor was also applied to the overall population level to take account of those individuals that were estimated to be on land**. Figure 5.2** shows the mean percentage of at-sea population estimated to be present in each 5km x 5km grid square at any one time (Carter *et al*., 2022)).

218. The total at-sea harbour seal population in the British Isles was approximately 48,419 individuals. The estimate is based on the correction factors for the number of harbour seals available to count during the haul-out counts (0.72; Lonergan *et al*., 2013). The harbour seal density estimates for the windfarm site were calculated from the latest seal at-sea maps produced by SMRU (Carter *et al*., 2022), based on the 5km x 5km grids that overlap with the Project area and the estimated portion of this population expected to be at-sea (using the correction factor 0.8236; Russell *et al.,* 2015), using the most recent haul-out counts for the UK and RoI (total of 39,878 individuals; SCOS, 2022).

219. The mean at-sea density estimate derived from Carter *et al*. (2022) data over the area of the windfarm site and 4km buffer was taken forward in the impact assessment:

- Mean at-sea density of 0.00011 harbour seal/km^2.

220. Unlike the many sightings of grey seals during the 24-months of site-specific surveys, there was only one harbour seal sighted within the survey area (in July 2021) over the two year survey period and therefore no relevant densities were derived from this single sighting.



Legend:

Morecambe Offshore Windfarm Site

Harbour Seals Relative Abundance (Carter et al 2022)

% British Isles At-sea Pop per 25km2

- ≤ 0.001 (Transparent)
- 0.001 - 0.005
- 0.005 - 0.01
- 0.01 - 0.025
- 0.025 - 0.05
- 0.05 - 0.1

Report: Morecambe Offshore Windfarm: Generation Assets Environmental Statement

Title: Harbour seal at sea distribution. Maps show mean percentage of at-sea population estimated to be present in each 5km x 5km grid square at any one time (Carter et al., 2022)

Figure: 5.2 Drawing No: PC1165-RHD-ES-OF-DG-Z-0100

Revision:	Date:	Drawn:	Checked:	Size:	Scale:
P01	16/11/2023	SB	GC	A3	1:2,250,000
P02	02/04/2024	JH	GC	A3	1:2,250,000

Co-ordinate system: WGS 1984 UTM Zone 30N

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5.8.3.2 Harbour seal population counts

221. Harbour seal were counted while they were on land during their August moult, giving a minimum estimate of population size (SCOS, 2022). Combining the most recent counts (2022) gave a total of 30,855 counted in the UK. Scaling this by the estimated proportion hauled out (0.72 (95% CL = 0.54-0.88)) produced an estimated total population for the UK in 2019 of 42,854 harbour seal (approximate 95% CL = 35,062 – 57,139; SCOS, 2022).

222. The SCOS (2022) data showed abundance of harbour seals within the NW England MU remained below six from 1996-2019 (two seals from 1996-1997; and five seals from 2000-2006). Since 2000, the numbers of harbour seal in the MU have been stable at five to seven harbour seals (SCOS, 2022), but they have not been surveyed in recent years, thus it was unclear how many harbour seals were present.

223. Tagging maps showed that harbour seals that were present were most likely from neighbouring MUs and that the population was not independent from others, based on the available data (Carter *et al*., 2020, 2022). Further, no significant harbour seal breeding or haul out sites were identified in the NW England MU (SCOS, 2022).

224. Despite the low harbour seal population number within the NW England MU, it was considered as the reference population within the impact assessment, reflecting a precautionary approach to the assessment. Considering the Carter *et al*. (2020, 2022) tracking data that showed harbour seal movements most likely occur with neighbouring MU, the NW England MU and the NI MU, were considered most suitable to represent the wider reference population, and provide a more realistic wider reference population within the impact assessment.

225. The wider reference population estimates for harbour seal, based on the most recent estimates, are shown in **Table 5.2.**

Table 5.2 Harbour seal counts and population estimates

Population area	Harbour seal haul-out count	Source of haul-out count data	Correction factor for seals not available to count	Harbour seal total population
NW England MU	5	SCOS, 2022	0.72	7
Northern Ireland	818	SCOS, 2022	0.72	1,136
Total wider reference population	823		0.72	**1,143**

5.8.4 Diet and foraging

226. Harbour seal take a wide variety of prey including sandeels, gadoids, herring, sprat, flatfish and cephalopods. Diet varies seasonally and regionally, prey diversity and diet quality also showed some regional and seasonal variation (SCOS, 2020). It has been estimated that harbour seals eat 3-5kg per adult seal per day depending on the prey species (SCOS, 2020) and the likely daily ration suggested approximately 3kg of fatty fish or up to 5kg of whitefish per day (BEIS, 2022b)

227. The range of foraging trips varies depending on the surrounding marine habitat. Telemetry studies have indicated that the tracks of tagged harbour seals had a more coastal distribution than grey seals and did not travel as far from haul-outs.

6 Review of potential disturbance from underwater noise during piling

228. There were no agreed thresholds or criteria for the behavioural response and disturbance of marine mammals at the time of writing, therefore it was not possible to conduct underwater noise modelling to predict impact ranges.

229. Therefore, a review of most recent available information on the potential disturbance of marine mammals during piling has been conducted to get a better understanding of the potential effects and inform the assessment set out in **Chapter 11 Marine Mammals**.

230. The JNCC *et al*. (2010) guidance proposed that "any action that is likely to increase the risk of long-term decline of the population(s) of (a) species could be regarded as disturbance under the Regulations."

231. The JNCC *et al*. (2010) guidance indicated that a score of 5 or more on the Southall *et al*. (2007) behavioural response severity scale could be significant (see **Table 6.1**). The more severe the response on the scale, the less time animals will likely tolerate the disturbance before there could be significant negative effects on life functions, which would constitute a disturbance.

Table 6.1 Southall et al. (2007) Severity Scale for Ranking Observed Behavioural Responses of Free-Ranging Marine Mammals

Response score	Corresponding behaviours in free-ranging subjects
0	No observable response.
1	Brief orientation response (investigation/visual orientation).
2	Moderate or multiple orientation behaviours Brief or minor cessation/modification of vocal behaviour Brief or minor change in respiration rates
3	Prolonged orientation behaviour Individual alert behaviour Minor changes in locomotion speed, direction, and/or dive profile but no avoidance of sound source Moderate change in respiration rate Minor cessation or modification of vocal behaviour
4	Moderate changes in locomotion speed, direction, and/or dive profile but no avoidance of sound source Brief, minor shift in group distribution Moderate cessation or modification of vocal behaviour

Response score	Corresponding behaviours in free-ranging subjects
5	Extensive or prolonged changes in locomotion speed, direction, and/or dive profile but no avoidance of sound source Moderate shift in group distribution Change in inter-animal distance and/or group size (aggregation or separation) Prolonged cessation or modification of vocal behaviour
6	Minor or moderate individual and/or group avoidance of sound source Brief or minor separation of females and dependent offspring Aggressive behaviour related to sound exposure (e.g., tail/flipper slapping, fluke display, jawclapping/gnashing teeth, abrupt directed movement, bubble clouds) Extended cessation or modification of vocal behaviour Visible startle response Brief cessation of reproductive behaviour
7	Extensive or prolonged aggressive behaviour Moderate separation of females and dependent offspring Clear anti-predator response Severe and/or sustained avoidance of sound source Moderate cessation of reproductive behaviour
8	Obvious aversion and/or progressive sensitisation Prolonged or significant separation of females and dependent offspring with disruption ofacoustic reunion mechanisms Long-term avoidance of area Prolonged cessation of reproductive behaviour
9	Outright panic, flight, stampede, attack of conspecifics, or stranding events Avoidance behaviour related to predator detection

232. It should be noted that a behavioural response does not mean that the individuals will avoid the area. In addition, the maximum predicted ranges for behavioural response have been based on the maximum hammer energy at the worst-case location for noise propagation. In reality, the duration of any piling at maximum energy would be less (if this energy is reached at all) and noise propagation would vary considerably with location (i.e., be less than the worst case).

6.1.1 Behavioural response of harbour porpoise to piling

233. The study of harbour porpoise at Horns Rev II (Brandt *et al.*, 2011), found that at closer distances (2.5 to 4.8km) there was 100% avoidance. However,

avoidance decreased significantly moving away from the pile driving activity, such that at distances of 10.1 to 17.8km, avoidance occurred in 32 to 49% of the population, and at 21.2km, harbour porpoise abundance reduced by just 2%. This suggests that an assumption of behavioural displacement of all individuals would be unrealistic and that in reality not all individuals would move out of the area. To take this into account within the marine mammal assessments, the proportion of harbour porpoise that may show a behavioural response has been calculated by assuming 75% or 50% could respond. This approach was consistent with the response at distances of 10.1 to 17.8km indicated by the Brandt *et al*. (2011) study (**Plate 6.1**), at which approximately 50% of individuals present could respond at the maximum predicted level as suggested by the dose-response curve (DRC) in Thompson *et al*. (2013).

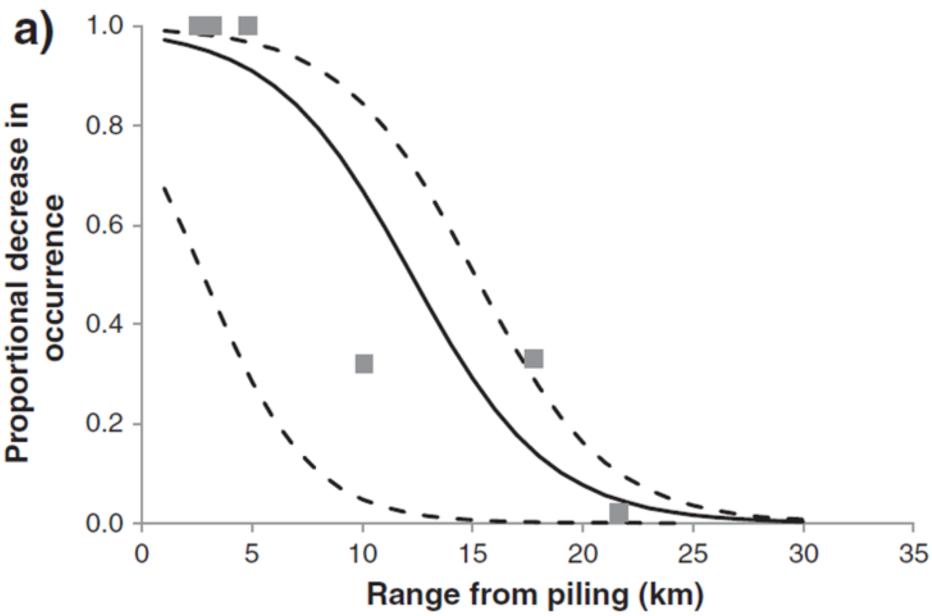


Plate 6.1 Predicted harbour porpoise dose response curve based on the monitoring of piling activity at Horns Rev II (based on data from Brandt et al., 2011, as presented in Thompson et al. (2013))

234. During the construction of two Scottish wind farms (Beatrice Offshore Wind Farm and Moray East Offshore Wind Farm), a set of cetacean porpoise detectors (CPODs) were deployed to monitor harbour porpoise presence during construction (Benhemma-Le Gall *et al*., 2021). In addition, the broadband noise levels were recorded and monitored, together with vessel Automatic Identification System (AIS) data. The purpose of this study was to assess the response of harbour porpoise to both the changes in the baseline noise level due to impact piling at the two wind farms, and due to an increase in vessel activity. Piling at Beatrice was for 2.2m jacket pin piles. The study demonstrated that there was an 8-17% decline in porpoise presence during

impact piling and other construction activities, compared to baseline levels (Benhemma-Le Gall *et al*., 2021).

235. An increase in broadband noise levels due to piling led to a significant reduction in porpoise presence. When piling was not occurring, porpoise detections decreased by 17% as the noise levels increased (from 102dB re 1 µPa (sound pressure level; SPL) to 159dB re 1 µPa (SPL)) (**Plate 6.2**; Benhemma-Le Gall *et al*., 2021). During piling, porpoise detections decreased by 9% as noise levels increased (from 102dB to 159dB). A similar reduction in buzz vocalisations was also evident; the presence of buzz vocalisations can be attributed to foraging behaviours. When piling was not taking place, buzz vocalisations decreased by 41.5% as the noise levels increased (from 104dB re 1 µPa (SPL) to 155dB re 1 µPa (SPL)). During piling, porpoise detections decreased by 61.8% as noise levels increased (from 104dB to 155dB re 1 µPa (SPL)) (Benhemma-Le Gall *et al*., 2021).

236. Harbour porpoise buzz vocalisations increased by 4.2% during Moray East piling compared to the baseline levels. At this point, Beatrice foundations were constructed, and the introduction of hard substrates were likely to have improved the fine-scale habitat for key harbour porpoise prey species, with the potential for increasing prey resources (Benhemma-Le Gall *et al*., 2021).

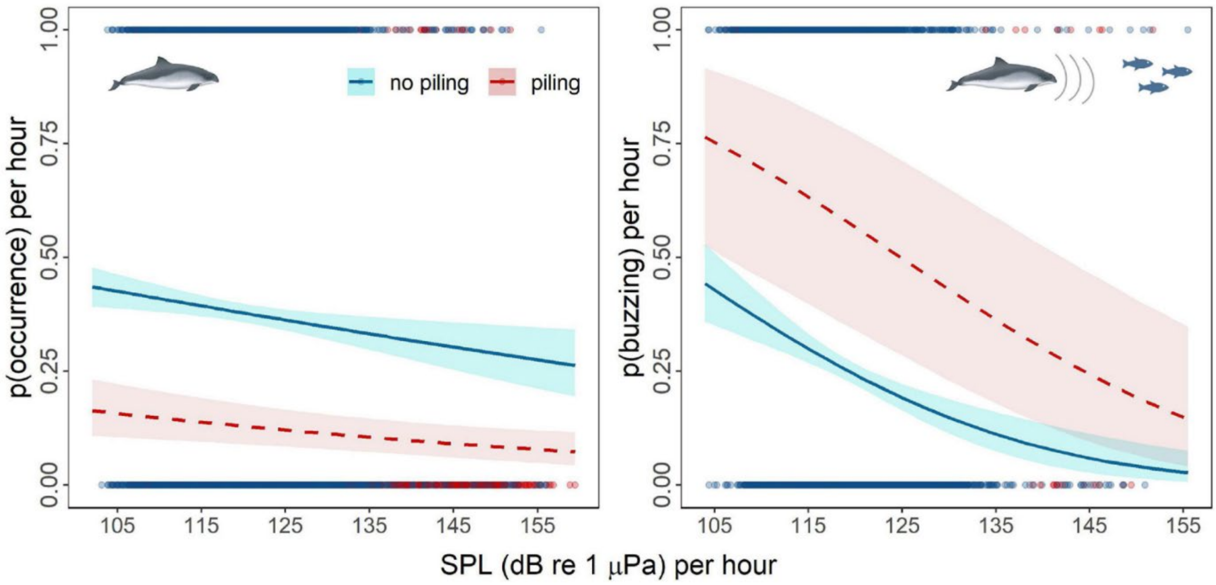


Plate 6.2 [Left] The probability of harbour porpoise presence in relation to the SPL (Red = during piling, Blue = outside of piling time, and [Right] the probability of buzzing activity per hour in relation to the SPL (Red = during piling, Blue = outside of pilin

237. A more recent study demonstrated that harbour porpoise started to leave the area in the two days leading up to a piling event, when pre-piling installation activities and vessel presence increased (Benhemma-Le Gall *et al*., 2023). The study found a 33% decline in acoustic click detections during the 48hrs prior to piling, which provided evidence that porpoises were displaced for a longer time period than just the piling event itself.

6.1.2 Behavioural response of dolphins to piling

238. There is limited information on the behavioural response of any dolphin species to piling.

239. Within the Southall et al. (2007) paper, a review of the data available for mid-frequency cetaceans (which included species other than dolphins, such as sperm whale *Physeter macrocephalus* and beluga *Delphinapterus leucas*) indicated that a significant response was observed at a SPL of 120dB to 130dB re 1µPa (root mean square (rms)), although the majority of individuals did not display a significant behavioural response until exposed to a level of 170dB to 180dB re 1µPa (rms). Other mid-frequency species were observed to have no behavioural response even when exposed to a level of 170dB to 180dB re 1µPa (rms). It should be noted that few of the reviewed studies were based on dolphin species.

240. Graham et al. (2017a) studied the responses of bottlenose dolphins due to both impact and vibration pile driving noise during harbour construction works in northeast Scotland. The study used passive acoustic monitoring (PAM) devices to record cetacean activity, and noise recorders to measure and predict received noise levels. Local abundance and patterns of occurrence of bottlenose dolphins were also compared with a five-year baseline. The median peak-to-peak source level estimated for impact piling was 240dB re 1µPa (single-pulse SEL (sound exposure level) 198dB re 1µPa^{2}s), and the rms source level for vibration piling was 192dB re 1µPa (Graham *et al.*, 2017a).

241. The results of the study found that bottlenose dolphin were not excluded from sites in the vicinity of impact piling or vibration piling; nevertheless, some small effects were detected, where bottlenose dolphins spent a reduced period of time in the vicinity of construction works during both impact and vibration piling (Graham *et al.*, 2017a). Dolphins generally showed a weak behavioural response to impact piling, reducing the amount of time they spend around the construction activity during piling (Graham *et al.*, 2017a). Observed fine-scale behavioural responses by dolphins during this study to piling occurred at predicted received single-pulse SEL values of between 104 and 136.2dB re 1µPa^{2}s for impact piling (Graham *et al.*, 2017a).

242. During the Beatrice wind farm piling campaign in 2017, dolphin detections decreased by 50% in the Impact Areas (minimum of 53km from the piling site) and decreased by 14% in the Reference Area (minimum of 80km from the piling site), compared to baseline years (Fernandez-Betelu *et al*., 2021). When impact piling was conducted at Moray East Offshore Wind Farm in 2019, no significant difference in dolphin detections between the study areas (Impact Area at a minimum of 45km from the piling site; Reference Area at a minimum of 78km from the piling site) was found in comparison to baseline years (Fernandez-Betelu *et al*., 2021).

243. The southern coast of the Moray Firth would have been the closest area to the offshore activities within this bottlenose dolphin population's range, with piling at Beatrice taking place 50–70km from the studied population, and Moray East 40–70km from the population. The analyses showed that dolphins continued using the southern coast of the Moray Firth during the seismic survey and impact pile-driving and therefore the species was not significantly affected at this distance of 40-70km (Fernandez-Betelu *et al*., 2021). While displacement distances were available for other marine mammal species (such as harbour porpoise), there were no such studies conducted for bottlenose dolphins. However, as dolphins were generally less sensitive than harbour porpoises to underwater noise, shorter ranges of displacement would be expected (Fernandez-Betelu *et al*., 2021).

244. While displacement distances were available for other marine mammal species (such as harbour porpoise), there were no such studies conducted for bottlenose dolphins. However, as dolphins were generally less sensitive than harbour porpoises to underwater noise, shorter ranges of displacement would be expected (Fernandez-Betelu *et al*., 2021).

245. It is possible that pile-driving noise can be perceived by dolphins for a minimum of 10km, and up to 40km away and interfere with dolphin communication, echolocation, and breeding. Depending on the communication, clicks can be masked up to 6km, whereas whistles have the potential to be masked up to 40km away.

246. While there was limited evidence as to the potential disturbance ranges of dolphin species due to impact piling, the above presented information indicates that the presence of dolphins may reduce due to piling works, however, there was no indication of a significant disturbance response, with individuals remaining in the vicinity of piling works. It was expected that dolphin species were less sensitive to disturbance from underwater noise than other species (such as harbour porpoise), however, due to the limited availability of evidence for dolphin species, as a precautionary approach, they were assumed to have the same sensitivity as harbour porpoise (medium).

6.1.3 Behavioural response of minke whale to piling

247. There is limited information on the behavioural response of minke whale to piling. Southall *et al.* (2007) recommended that the most appropriate way to assess the disturbance effect of a noise source on marine mammals was the use of empirical studies. The same paper presented a severity scale to apply to observed behavioural responses, and subsequent JNCC guidance indicated that a score of five or more on this behavioural response severity scale could be significant (see **Table 6.1**). A score of five relates to extensive changes in swim speed and direction, or dive pattern, but no avoidance of the

noise source, or a moderate shift in distributions, a change in group size, aggregations and separation distances, and a prolonged cessation in vocal behaviours. The higher the behavioural response score, the more likely the associated noise source would result in a significant disturbance effect.

248. Southall *et al.* (2007) included a summary of the observed behavioural responses from noise sources. However, the majority of the studies included were based on the responses to seismic surveys. These studies contained some relevant information for whale species behavioural responses.

249. Whale species were typically observed to respond significantly at a received level of 150dB to 160dB re 1µPa (rms) (Malme *et al.,* 1983, 1984; Richardson *et al.,* 1986; Ljungblad *et al.,* 1988; Todd *et al.,* 1996; McCauley *et al.,* 1998), with behavioural changes including:

- Visible startle responses
- Extended cessation or modification of vocal behaviour
- Brief cessation of reproductive behaviour
- Brief and minor separation of females and dependent offspring

250. During migration periods, avoidance behaviours of bowhead whales, *Balaena mysticetus,* were observed at distances of more than 20km from seismic sources (Koski & Johnson, 1987; Richardson *et al.,* 1999). However, during foraging periods, bowhead whales did not respond at greater than 6km from the source (Richardson *et al.,* 1986; Miller *et al.,* 2005). Richardson *et al.* (1986) concluded that due to a single airgun, avoidance and behavioural response was observed once noise levels reached more than 160dB re 1µPa.

251. For a migrating bowhead whale study, most individuals avoided a seismic survey source at distances of up to 20km (the seismic surveys used airgun arrays of up to 16 guns, and total volume of 560 to 1,500 cu. in.), with significantly reduced bowhead whale presence between 20 and 30km from the source, with estimated received noise levels of 120 to 130dB re 1µPa ($_{rms}$) at that distance (Richardson *et al.,* 1999).

252. However, during foraging periods, bowhead whales did not respond at greater than 6km from the source (Richardson et al., 1986; Miller et al., 2005). Observations of behavioural changes in baleen whale species have shown avoidance reactions of up to 10km for a seismic survey, with a noise source level of 143dB re 1µPa (peak to peak) (Macdonald *et al.,* 1995).

253. Dose-response functions for avoidance responses of grey whales *Eschrichtius robustus* to both continuous and impulsive noises were developed for vessel noise and seismic air guns by Malme (1984). For continuous noise sources, avoidance of minke whale started at a received level of 110-119dB re 1µPa ($L_{peak,\ rms}$), with more than 80% of individuals

responding at 130dB re 1 µPa ($L_{peak, rms}$), and 50% at 120dB re 1 µPa ($L_{peak, rms}$).

254. Higher noise levels were required for an avoidance response due to the impulsive noise source (seismic airguns), with 10% of migrating grey whales responding at 164dB re 1 µPa ($L_{peak, rms}$), 50% at 170dB re 1µPa ($L_{peak, rms}$), and 90% at 180dB re 1µPa ($L_{peak, rms}$) (Malme, 1984 *cited in* Tyack and Thomas, 2019). A secondary study (Malme *et al.*, 1987) using 100 cu. in. air guns (with a source level of 226dB re 1µPa) for foraging grey whales found a response level (where individuals would cease foraging activities) of 50% at 173dB re 1µPa ($L_{peak, rms}$), and 10% at 163dB re 1µPa ($L_{peak, rms}$).

255. There was limited information on the potential disturbance ranges of minke whale to piling, however, there were some studies that provide observed disturbance of baleen whale species to seismic surveys. Baleen whale species have been observed to respond at up to 20km during migration, with disturbance observed up to 30km from a seismic source. One study found that baleen whales were more sensitive to disturbance from continuous sources than from impulsive sources. Typically, baleen whales have been reported to avoid and respond at impulsive noise levels of 150-160 re 1µPa (rms) (Malme *et al.*, 1983, 1984; Richardson *et al.*, 1986; Ljungblad *et al.*, 1988; Todd *et al.*, 1996; McCauley *et al.*, 1998), with 50% of individuals responding at 170dB to 173dB re 1µPa ($L_{peak, rms}$) (Malme, 1984; Malme *et al.*, 1987).

256. The studies summarised above suggest that baleen whale species (including minke whale) may be similarly sensitive to disturbance from underwater noise as harbour porpoise, and therefore a sensitivity of medium was appropriate.

6.1.4 Behavioural response of seals to piling

257. There was limited data on seal species presented within the Southall *et al.*, 2007 paper. Although these species are not found in UK waters, one included study was for ringed seals *Pusa hispida*, bearded seals *Erignathus barbatus,* and spotted seals *Phoca largha* (Harris *et al.,* 2001), which found the onset of a significant response at a received noise level of 160 to 170dB re 1µPa (rms), although a larger proportion of individuals showed no response at noise levels of up to 180dB re 1µPa (rms). Only at much higher sound pressure levels (190 to 200dB re 1µPa (rms)) did significant numbers of seals exhibit a significant disturbance response.

258. Tagged harbour seals in the Wash indicated that seals were not excluded from the vicinity of the Lincs Offshore Wind Farm during the overall construction phase but that there was clear evidence of avoidance during pile driving, with significantly reduced levels of seal activity at ranges of up to 25km from piling sites (Russell *et al.,* 2016). However, within two hours of cessation of piling, seal distribution returned to pre-piling levels (Russell *et al.,* 2016).

6.1.5 Dose response curves

259. As per current best practice guidance (Southall *et al*., 2021), a behavioural disturbance dose-response analysis has been carried out for those species for which appropriate dose-response evidence existed within the scientific literature. In case of absence of such evidence, a fixed behavioural threshold approach (that was used in most assessments) has been applied.

260. The dose-response relationship used for harbour porpoise was developed by Graham *et al*. (2017b) using data collected on harbour porpoises during Phase 1 of piling at the Beatrice Offshore Wind Farm. This dose response relationship is displayed in **Plate 6.3**. Following the development of this dose-response relationship, further study revealed that the responses of harbour porpoises to piling noise diminished over the construction period (Graham *et al*., 2019). Therefore, the use of the dose-response relationship related to an initial piling event for all piling events in the ES marine mammal assessment can be considered conservative.

261. In the absence of species-specific dose-response data for dolphins or whales, harbour porpoise was the only species of cetacean that this analysis was applied to. Due to differences in audiograms and behaviour, it would not be appropriate to extrapolate the findings of Graham *et al*. (2017b) to other cetacean species.

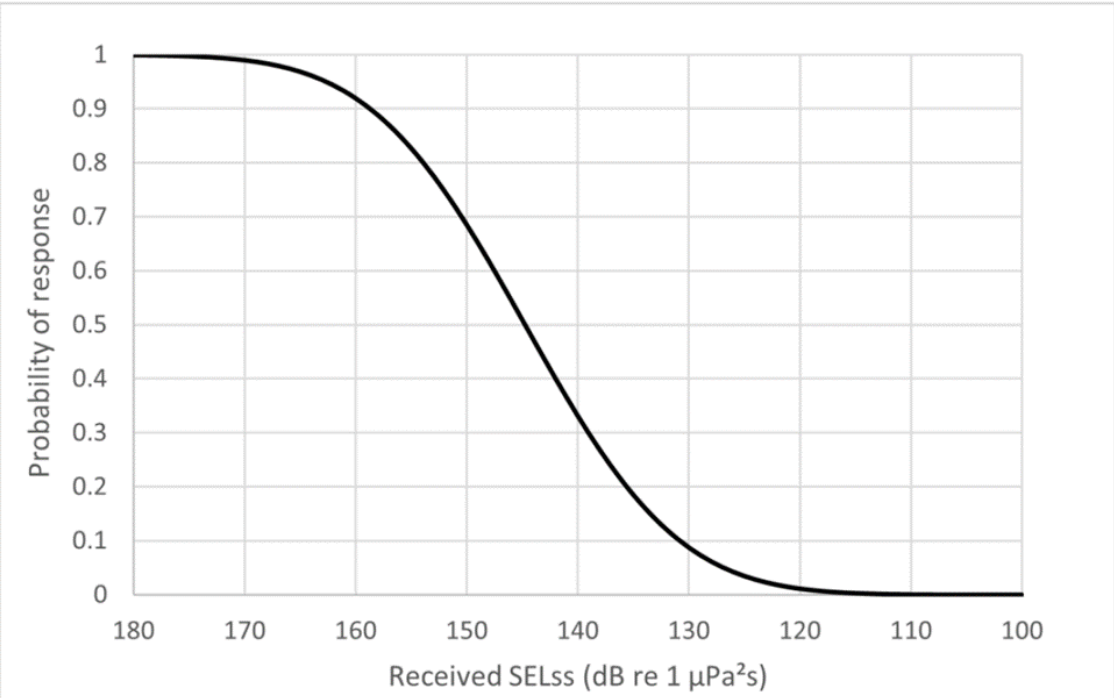


Plate 6.3 Dose-response relationship developed by Graham et al. (2017b) used for harbour porpoise in the assessment

262. For both harbour seal and grey seal, a dose-response relationship that was derived from harbour seal telemetry data collected during several months of piling at the Lincs Offshore Wind Farm has been used (Whyte *et al*., 2020). As seen in **Plate 6.4**, the greatest Sound Exposure Level from single strike (SEL_{SS}) considered in the Whyte *et al*. (2020) study was 180 dB re 1μPa2s. The ES marine mammal assessment has therefore conservatively assumed that at SEL_{SS} > 180dB re 1μPa^{2}s all seals would be disturbed. The dose-response curve for harbour seal has been used for grey seal, as both species have similar hearing audiograms.

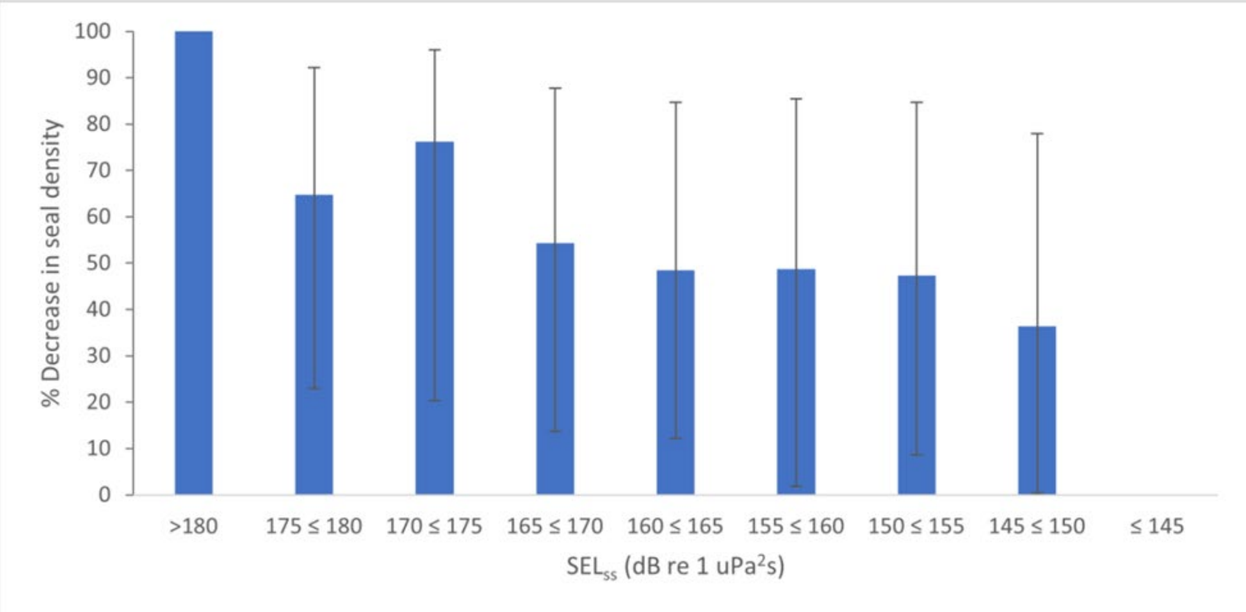
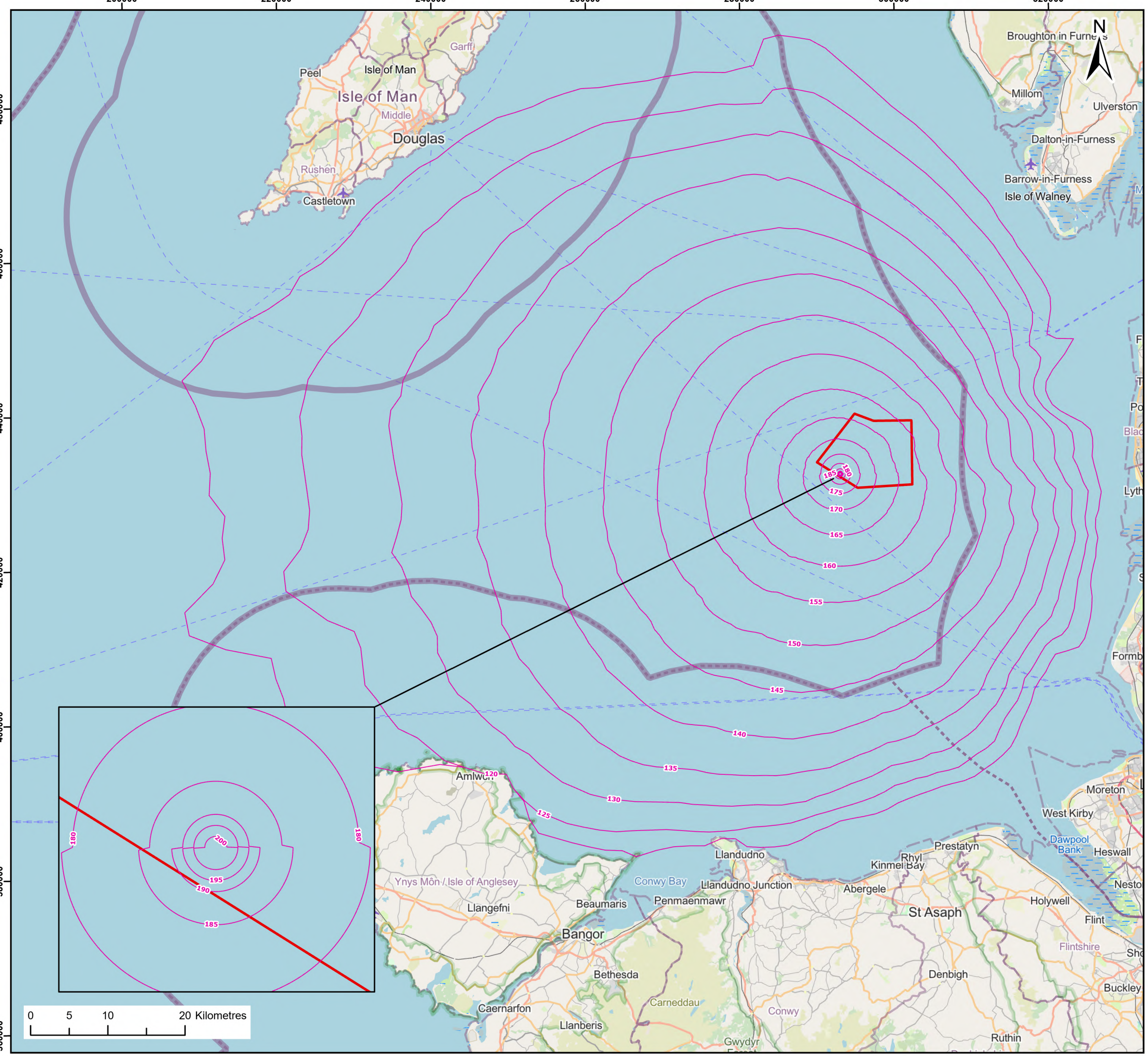


Plate 6.4 Dose-response behavioural disturbance data for harbour seal derived from the data collected and analysed by Whyte et al. (2020). This data has been used for harbour and grey seals in the assessment.

263. To estimate the number of animals disturbed by piling, SEL_{SS} contours at 5dB increments (generated by the noise modelling – see **Figure 6.1** and **Figure 6.2**) were overlain on the relevant species density surfaces, to quantify the number of animals receiving each SEL_{SS}, and, subsequently, the number of animals likely to be disturbed, based on the corresponding dose-response curve.



Legend:

- Morecambe Offshore Windfarm Site
- 5dB contours for South West (worst-case) location using Unweighted SELss for monopiles

Report: Morecambe Offshore Windfarm: Generation Assets Environmental Statement

Title: 5dB contours for the South West (worst-case) location using Unweighted SELss for monopiles

Figure: 6.1 | Drawing No: PC1165-RHD-ES-OF-DR-Z-0099

Revision:	Date:	Drawn:	Checked:	Size:	Scale:
P01	16/11/2023	JH	SB	A3	1:500,000
P02	03/04/2024	JH	SB	A3	1:500,000

Co-ordinate system: WGS 1984 UTM Zone 30N

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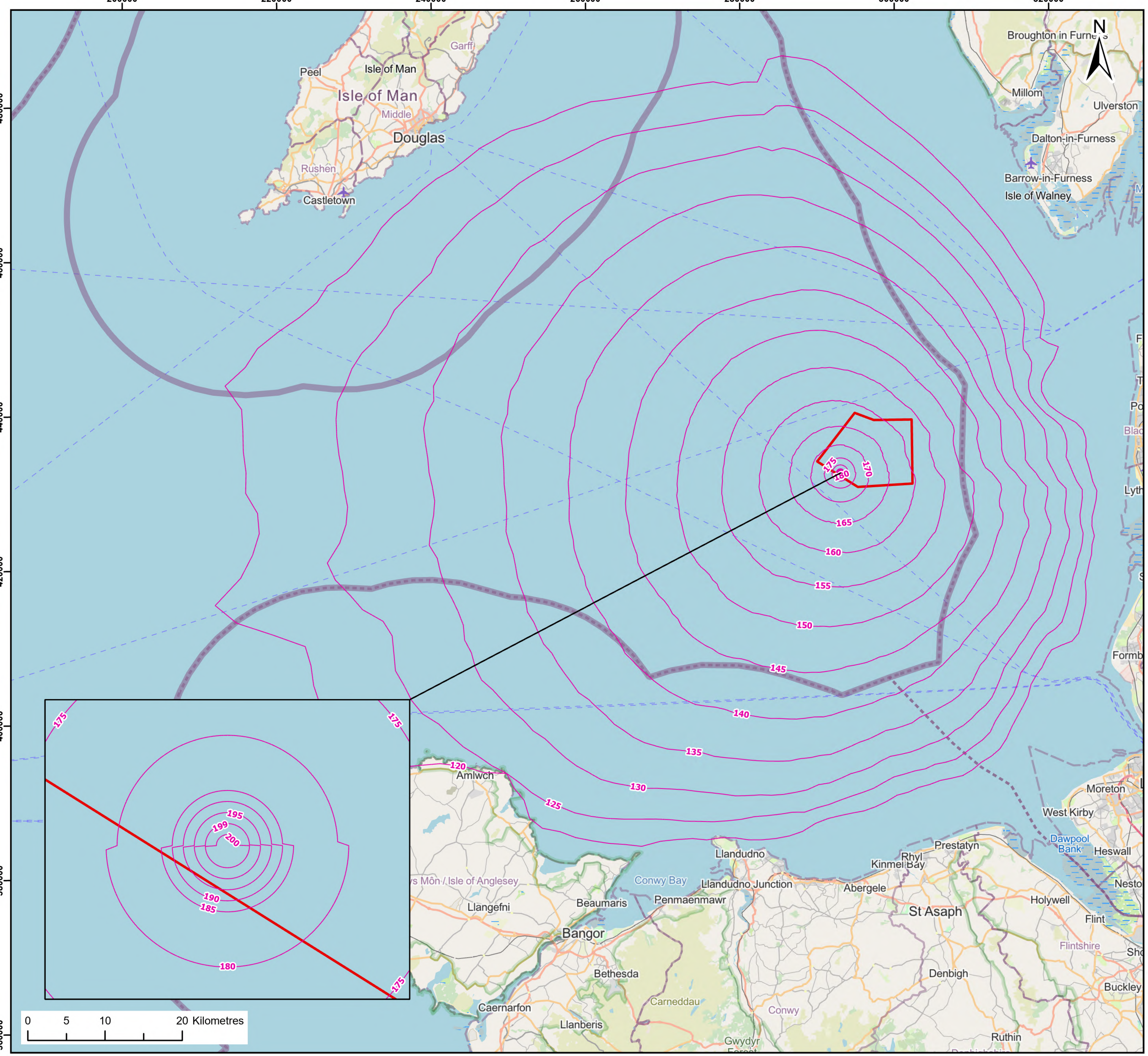

Legend:

Morecambe Offshore Windfarm Site

5dB contours for the South West (worst-case) location using Unweighted SELss for pinpiles

Report: Morecambe Offshore Windfarm: Generation Assets Environmental Statement

Title: 5dB contours for the South West (worst-case) location using Unweighted SELss for pinpiles

Figure: 6.2 | Drawing No: PC1165-RHD-ES-OF-DR-Z-0106

Revision:	Date:	Drawn:	Checked:	Size:	Scale:
P01	16/11/2023	JH	SB	A3	1:500,000
P02	03/04/2024	JH	SB	A3	1:500,000

Co-ordinate system: WGS 1984 UTM Zone 30N

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6.1.5.1 Assumptions and limitations

264. There was a lack of empirical data on bottlenose dolphin, minke whale or grey seal responses to pile driving to derive species-specific dose-response curves for these species. For grey seal, the harbour seal dose response curve has been used as a reasonable proxy since both species were of the same hearing group. For the remaining species, all dolphins and minke whale, the harbour porpoise dose-response curve was used although there were uncertainties regarding the use of this proxy since the species have all been classified as being in different hearing groups, and thus in reality their response to the same sound source was unlikely to be the same.

265. The use of the dose-response relationship for harbour seal from Whyte *et al*. (2020) in conjunction with the modelling results presented here was conservative. The exact drivers behind the dose response relationship were unknown and were likely to be influenced by a combination of distance from the sound source and the received level. Yet the dose-response curve presented in Whyte *et al*. (2020) was based upon received level only. Responses of animals were not only elicited by the received level but also by other factors, such as signal shape. The shape of a signal with the same SEL from the same sound source differs depending on distance. Piling noise has been noted to lose its impulsive character with distance (Southall *et al*. 2007, Hastie *et al*. 2019, Southall *et al.* 2019b; **Plate 6.5**), and therefore animals were expected to react less strongly to piling noise with the same received levels when exposed at larger distances. Such an effect has been quantified for blue whales with regard to military sonar, where a received level of 170dB SEL from cumulative exposure (SEL_{cum}) at 1km resulted in a probability response of >0.5 at severity score 4-6[8] whereas the same received level of 170dB SEL_{cum} at 5km resulted in a probability of response of <0.1 at severity score 4-6 (Southall *et al.* 2019a). This is important to note, since the original dataset in Whyte *et al.* (2020) showed that "predicted seal density significantly decreased within 25 km or above SEL_{ss} 145dB re 1μPa²s".

[8] Severity score 4-6 denotes "moderate severity"

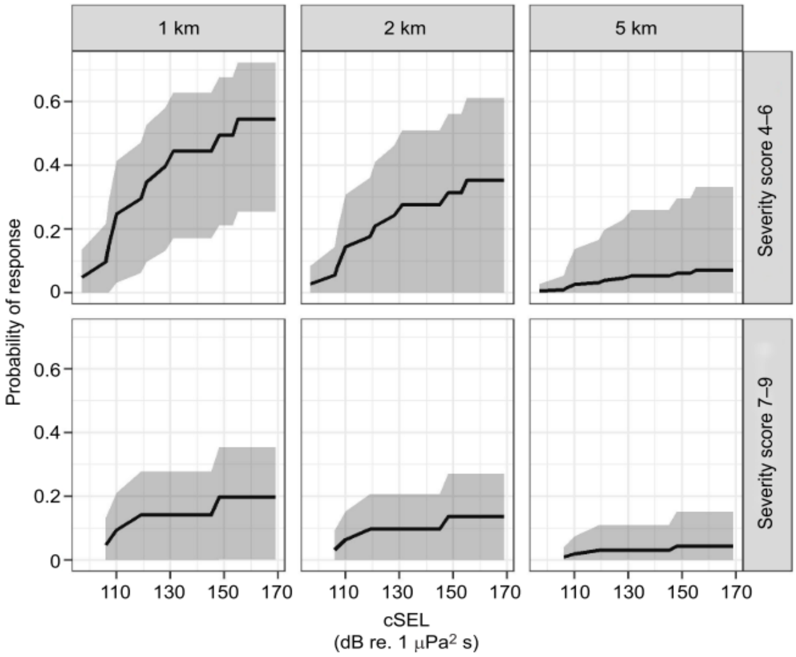

Plate 6.5 Behavioural response probability for blue whales exposed to military sonar as a function of received level and distance from the sound source. Severity score 4-6 denotes 'moderate severity 'and 7-9 denotes 'high severity'. Image taken from Southall et al. (2019)

266. In addition to these issues, it should be recognised that estimates of received noise levels were likely to be extremely conservative given they have been based on the maximum hammer energy. In practice, pile driving at other UK offshore wind farms has often been completed using much lower than the predicted hammer energies as shown for other OWFs (DOWE, 2016).

6.1.6 Beatrice offshore wind farm

267. During the piling campaign at Beatrice Offshore Wind Farm in 2017, an array of underwater noise recorders was deployed to determine noise levels associated with the piling campaign, alongside a separate array of acoustic recorders to monitor the presence of harbour porpoise during piling (Graham *et al*., 2019). Piling at Beatrice comprised four pin piles at each turbine or sub-station structure, with a 2.2m pile diameter and a maximum hammer energy of 2,400kJ. The sound levels recorded were then used to determine the sound level at each of the acoustic recorders.

268. This study assumed that a change in the number of harbour porpoise present at each location was based on the number of positive identifications of porpoise vocalisations (Graham *et al*., 2019). These two data sets (the harbour porpoise presence and the perceived sound level at each location) were then analysed to determine any disturbance impacts as a result of the piling activities and at what sound level impacts were observed. Harbour porpoise presence was measured over a period of 48 hours prior to piling being undertaken and continued following the cessation of piling to ensure that

any change in porpoise detections could be observed (a total period of 96 hours was recorded for each included piling event, with a total of 17 piling events included within this analysis) (Graham *et al*., 2019).

269. The results of the study at Beatrice Offshore Wind Farm (Graham *et al*., 2019) found that at the start of the piling campaign, there was a 50% chance of a harbour porpoise responding to piling activity, within a distance of 7.4km, during the 24 hours following piling. By the middle of the piling campaign, this 50% response distance had reduced to 4.0km, and by the end of the piling had reduced further to 1.3km. The response to audiogram weighted SEL noise levels reduced over time, with a 50% response being observed at sound levels of 54.1dB re 1µPa2s at the first location during the first 24 hours following piling, increasing to 60.0dB re 1µPa2s during the middle of the campaign, and to 70.9 dB re 1µPa2s by the end of the piling activities. Similarly, the response to unweighted SEL noise levels reduced over time, with a 50% response being observed at sound levels of 144.3dB re 1µPa2s at the first location during the first 24 hours following piling, increasing to 150.0dB re 1 1µPa2s during the middle of the campaign, and to 160.4dB re 1µPa2s by the end of the piling activities (Graham *et al*., 2019).

270. Additional comparisons were made through this study (Graham *et al*., 2019) to assess the difference in harbour porpoise presence where Acoustic Deterrent Devices (ADD) were used and where they were not, as well as relating to the number of vessels present within 1km of the piling site. A significant difference was observed in the presence of harbour porpoise where ADDs were used compared to where they were not, but only in the short-term (less than 12 hours following piling), and there was no significant difference when considering a longer time period from piling. 50% response distances for pile locations with ADD use were recorded as up to 5.3km (during 12 hours after piling), and up to 0.7km with no ADD in use in the 12 hours following piling. It should be noted however that only two locations used in the analysis deployed ADD, and therefore the sample number in this analysis was small (Graham *et al*., 2019).

271. Overall, this study showed that the response of harbour porpoise to piling activities reduced over time, suggesting a habituation effect occurred. In addition, there has been some indication that the use of ADDs would reduce the presence of harbour porpoise in the short term. Also, the higher levels of vessel activity increased the potential for a response by harbour porpoise. Harbour porpoise response to piling activity was best explained by the distance from the piling location, or from the received noise levels (taking into account weighting for their hearing) (Graham *et al*., 2019).

6.1.7 Gescha 2

272. The Gescha 2 study (Effects of noise-mitigated offshore pile driving on harbour porpoise abundance in the German Bight 2014-2016; Rose *et al*., 2019) analysed the impact from the construction of eleven offshore wind farms in Germany on harbour porpoise in the German North Sea and adjacent Dutch waters from 2014 to 2016. The study also included analysis of previously completed surveys within the Gescha 1 study, which studied the impact from the construction of eight German offshore wind farms from 2009 to 2013. The study involved the deployment of CPODs and digital aerial surveys to monitor harbour porpoise presence and abundance during the construction of these projects, alongside the measurement of noise levels associated with piling at both 750m and 1,500m from source. The piling activities monitored in this study were mostly undertaken with noise abatement systems deployed to reduce disturbance impacts on harbour porpoise.

273. The Gescha 2 study (Rose *et al*., 2019) found that noise levels recorded during piling were predominantly below the limit of 160dB at 750m (the German Federal Maritime and Hydrographic Agency (BSH) mandatory noise limit for German waters) and were 9dB lower than the noise levels recorded during the Gescha 1 study, due to advancement in noise abatement methods. The study also found that noise levels were 15dB less using noise abatement than for noise levels from unmitigated piling. It was expected that the improved efficiency of noise abatement for piling, and therefore the overall reduced noise levels, would lead to a reduction in disturbance impacts on harbour porpoise, however, this was not the case.

274. The range of disturbance impact of harbour porpoise to piling within the Gescha 2 study (Rose *et al*., 2019) was 17km (Standard Deviation (SD) 15-19km), and the duration of disturbance (i.e., the time it took for harbour porpoise to return to baseline levels) was between 28 and 48 hours, as shown by CPOD data. The impact range was found to be between 11.4 and 19.5km based on aerial data (at least 12 hours after piling) (Rose *et al*., 2019). These results were similar to those reported in the Gescha 1 study (with a disturbance range of 15km (Standard deviation (SD) 14-16km) and duration of disturbance of 25 to 30 hours), which showed higher piling noise levels (Rose *et al*., 2019). This suggested that the noise level of the piling was not the only determining factor when discussing the potential for disturbance.

275. Analysis of the CPOD data collected in the Gescha 2 study (Rose *et al*., 2019) indicated that there was no correlation between noise levels received and the range at which harbour porpoise become disturbed, for noise that was below 165dB at 750m from source. This could have been due to individuals maintaining a certain distance from noisy activities, irrespective of the actual noise levels, provided that noise level was above a certain threshold for that

individual (Rose *et al*., 2019). It should be noted however that this study recorded noise levels up to 20kHz only, and therefore there may have been higher frequency noise associated with piling that these results did not take into account.

276. A reduction in harbour porpoise presence was seen for all offshore wind farms (for both the Gescha 1 and 2 studies) up to 24 hours prior to any noisy activity occurring, which could have been due to the increased vessel activity at the pile location prior to piling taking place (Rose *et al*., 2019). However, the displacement during pile driving was noted to be larger than for the period prior to piling. In Gescha 2, a decrease in detection rates was found in the three hours prior to piling activity at a distance up to 15km from the piling location, with no difference in detection rates observed at a distance of 25km (Rose *et al*., 2019).

7 Population modelling

277. In **Chapter 11 Marine Mammals**, the assessment results for disturbance (Section 11.6.3.2), revealed that elevations in subsea noise due to piling could potentially lead to the behavioural disturbance of a large number of individuals of the key species identified within the marine mammal study area.

278. Population modelling has therefore been conducted for harbour porpoise, bottlenose dolphin, minke whale, harbour and grey seal. The interim Population Consequences of Disturbance (iPCoD) framework (Harwood *et al*. 2014, King *et al*. 2015) was used to predict the potential medium- and long-term population consequences of the predicted amount of disturbance resulting from the piling at the Project.

279. iPCoD used a stage-structured model of population dynamics with nine age classes and one stage class (adults 10 years and older). The model was used to run a number of simulations of future population trajectories with and without the predicted level of impact to facilitate an understanding of the potential future population-level consequences of predicted behavioural responses and auditory injury.

7.1 Methodology

7.1.1 Piling parameters

280. The amount of piling required for the Project would be dependent on the foundations selected and the final piling schedule. The worst-case scenario (monopiles with the highest strike rate) was taken forward for modelling in iPCoD.

281. Whilst for the underwater noise modelling (**Appendix 11.1 Underwater Noise Assessment** (Document Reference 5.2.11.1) the worst-case with regard to modelled impact ranges was presented where piling for the Project could occur sequentially (up to three monopiles or four pin-piles in a 24h period); for population modelling the worst-case assumed that only one pile would be installed in each 24h period, thereby maximising the number of days in which disturbance could occur over the construction phase.

282. At this stage, uncertainty exists around the exact piling schedule that would be used for construction of the Project, however the periods during which piling is likely to occur are known. Therefore, the required number of piling days for each project construction scenario have been distributed randomly within the known piling periods.

283. The piling parameters used in the iPCoD modelling for the Project-alone scenario is detailed in **Table 7.1**.

Table 7.1 Piling scenario used for iPCoD modelling for the Project

Parameter	Value
Number of WTGs	35
Number of OSP(s)	2
Number of piles	Monopiles: 35 (WTG) and 2 (OSP) Pin-pile: 140 (WTG) and 8 (OSP)
Number of piling days	37 (assumed 1 pile per day)
Piling window	Q2 and Q3 2027 (WTG/OSP monopiles)
Piling schedule	Q2 and Q3 2027: 37 monopile days (distributed randomly)

284. The piling parameters used in the iPCoD modelling for the projects in the cumulative assessment scenario are detailed in **Table 7.2**. Further information on the projects included has been provided in **Appendix 11.4 Marine Mammal CEA Screening** (Document Reference 5.2.11.4).

Table 7.2 Piling parameters for other projects screened into the cumulative iPCoD modelling

Project	Number of piling days	Piling schedule
Awel y Môr	201	Q1 Year 2 - Q4 Year 4 2027-2029
Erebus	18	Q4 2024 – Q4 2026
Morgan	70	2026/27
Mona	70	2026/27
Transmission Assets	6	2026/27
White Cross	5	Q2 2025 - Q3 2027

7.1.2 Model inputs

285. The iPCoD model v5.2[9] was set up using the program R v4.3.2 (R Core Team, 2023) with RStudio as the user interface. To enable the iPCoD model to be run, the following data were provided:

- Demographic parameters for each key species
- User specified input parameters
  - Vulnerable subpopulations
  - Residual days of disturbance

[9] https://www.smruconsulting.com/population-consequences-of-disturbance-pcod

- Number of animals predicted to experience Permenant Thershold Shift (PTS) and/or disturbance during piling
- Estimated piling schedule during the proposed construction programme

7.1.3 Demographic parameters

286. Demographic parameters for the key species assessed in the population model are presented in **Table 7.3**. In the case of harbour seal, evidence for demographic parameters for the English populations was lacking (Sinclair et al., 2020). The SCOS (2022) data showed abundance of harbour seals within the NW England MU remained below six from 1996-2019 (two seals from 1996-1997; and five seals from 2000-2006).

287. Since 2000, the numbers of harbour seal in the NW England MU have been stable at five to seven harbour seals (SCOS, 2022). However, they have not been surveyed in recent years, therefore it was unclear how many harbour seals were present. The NW England MU appeared to be stable, and the MU with demographic information available (and greatest chance for connectivity), namely the NI MU, was also considered to be stable. For this reason, the demographic parameters for the NI MU have been used in the modelling for harbour seal.

Table 7.3 Demographic parameters recommended for each species for the relevant Management Unit (MU)/SMAs (Sinclair et al., 2020)

Species	MU	Age calf/pup becomes independent (age1)	Age of first birth (age2)	Calf/Pup Survival	Juvenile Survival	Adult Survival	Fertility	Growth Rate
Harbour Porpoise	North Sea	1	5	0.8455	0.85	0.925	0.34	1.00
Grey Seal	All UK	1	6	0.222	0.94	0.94	0.84	1.01
Harbour Seal	Northern Ireland	1	4	0.4	0.78	0.92	0.85	1.00
Bottlenose dolphin	All MUs (except East Coast Scotland)	2	9	0.8	0.94	0.94	0.25	1.00
Minke whale	European waters	1	9	0.7	0.77	0.96	0.91	1.00

7.1.4 Reference populations

288. The populations of marine mammal species vulnerable to piling-induced PTS/disturbance were specified in the model as the reference populations against which the effects (i.e. number of animals suffering PTS/disturbed) were assessed in **Chapter 11 Marine Mammals**, Section 11.6.3.2. **Table 7.4** provides the reference populations used in the iPCoD modelling.

Table 7.4 Reference populations used in the iPCoD modelling

Species	Area	Population
Harbour porpoise	Celtic and Irish Sea MU	62,517
Grey Seal	Wider reference population: NW England MU; SW Scotland; IoM count; Wales MU; NI MU; E RoI; SE RoI	13,283
Harbour Seal	Wider reference population: NW England MU and NI MU	1,143
Bottlenose dolphin	Irish Sea MU	293
Minke whale	Celtic and Greater North Sea MU	20,118

7.1.5 Residual days disturbance

289. Empirical evidence from constructed wind farms (e.g. Graham *et al*., 2019; Brandt *et al*., 2011) suggested that the detection of animals returned to baseline levels in the hours following a disturbance from piling and therefore, for the most part, it could be assumed that the disturbance occurred only on the day (24 hours) that piling took place (at least in the case of harbour porpoise which was the focus of these studies). However, the number of residual days of disturbance has, conservatively, been selected as one, meaning that the model assumed that disturbance occurred on the day of piling and persisted for a period of 24 hours after piling ceased.

7.1.6 Vulnerable sub-populations

290. For the purposes of the modelling, it was assumed that the entire population of interest was potentially vulnerable to pile driving disturbance and PTS.

7.1.7 Number of animals with PTS or disturbed

291. The number of animals predicted to experience PTS and/or disturbance during piling was based on the density values identified for harbour porpoise, harbour and grey seal as part of the baseline assessment in **Chapter 11 Marine Mammals**. In the case of disturbance, the estimated number of harbour porpoise and seals affected was based on effective deterrent ranges (EDRs) which are fixed ranges that are based on empirical evidence as opposed to disturbance ranges estimated from noise modelling (JNCC, 2020). The estimated number of bottlenose dolphin and minke whale affected was based on known disturbance ranges (as detailed in **Sections 6.1.2** and **6.1.3**).

292. Whilst **Chapter 11 Marine Mammals**, provided alternative estimates of the number of animals disturbed, based on a dose-response analysis (which could be considered more realistic), the estimates resulting from EDRs were greater, and were therefore used in the iPCoD model as a conservative worst-case.

293. **Table 7.5** presents the number of individuals that could potentially be disturbed due to piling at the Project-alone.

Table 7.5 Estimated number of animals to have PTS or to be disturbed during each piling event

	Number of animals affected during each piling event	
Species	**PTS**	**Disturbance**
Harbour porpoise	243	3,443 (based on 26km EDR)
Bottlenose dolphin	0.001	56.3 (based on DRC)
Minke whale	2.9	24.9 (based on 30km range)
Grey seal	0.2	196.4 (based on 25km EDR)
Harbour Seal	0.0002	0.2 (based on 25km EDR)

294. For cumulative effects assessments (CEA), the number of animals predicted to experience PTS and/or disturbance during piling was based on the density values that were published in the respective PEIR or ES chapters for the projects screened into the CEA.

295. **Table 7.6** presents the number of individuals that could potentially be affected by PTS or be disturbed from piling at the OWF projects screened into the CEA. Note: the Morgan and Morecambe Offshore Windfarms: Transmission Assets are hereafter referred to as ‘Transmission Assets’.

Table 7.6 Estimated number of marine mammals to have PTS or be disturbed from piling at the CEA screened in projects

Number of animals affected by PTS during each piling event					
Projects	**Harbour porpoise**	**Bottlenose dolphin**	**Minke whale**	**Grey seal**	**Harbour Seal**
Awel y Môr OWF	2,112	<1	3	<1	Not assessed
Erebus OWF	<1	<1	<1	<1	Not assessed
Morgan Offshore Wind Project Generation Assets	0	0	<1	0	<1
Mona Offshore Wind Project	0	0	<1	0	<1
Transmission Assets	Not assessed	Not assessed	Not assessed	Not assessed	Not assessed
White Cross OWF	0.92	0.0006	3.5	0.00005	Not assessed
Number of animals disturbed during each piling event					
Awel y Môr OWF	83	23	35	81	Not assessed
Erebus OWF	1,967	310	55	18	Not assessed
Morgan Offshore Wind Project Generation Assets	979	11	69	45	Not assessed
Mona Offshore Wind Project	429	13	69	45	Not assessed
Transmission Assets	1,793	4	69	28	Not assessed
White Cross OWF	649	0.0005	60.5	9.5	Not assessed

7.1.8 Piling schedule

296. As described in **Section 7.1.1**, the piling schedule was developed from the Project Design Envelope (PDE) which provided an estimate of the number of days piling for the WTG and OSP foundations within a defined piling phase, which is scheduled to take place within an overall offshore construction window.

7.2 Assumptions and limitations

297. The iPCoD framework (Harwood *et al*. 2014, King *et al*. 2015), provided by SMRU Consulting, has been used to predict the potential population consequences of the predicted amount of disturbance resulting from Project piling[10].

298. Insufficient empirical evidence exists regarding how alterations in behaviour and hearing sensitivity might impact the survival and reproductive capabilities of individual marine mammals. Therefore, in the absence of empirical data, the iPCoD framework used the results of an expert elicitation process, described in Donovan *et al*. (2016), to predict the effects of disturbance and PTS on survival and reproductive rates. The process generated a set of statistical distributions for these effects and then simulations were conducted using values randomly selected from these distributions that represented the opinions of a 'virtual' expert. This process was repeated many 100s of times to capture the uncertainty among experts. While the iPCoD model was subject to many assumptions and uncertainties relating to the link between impacts and vital rates, the model presented the best available scientific expert opinion at the time of assessment.

299. In the latest update of the iPCoD model there was no elicitation for minke whale (PTS or disturbance) or bottlenose dolphins (disturbance) and the results presented in **Chapter 11 Marine Mammals**, were highly conservative and represented an overestimate of any potential population level effects. There were several precautions built into the iPCoD model that meant that the results were highly precautionary and would over-estimate the true population level effects. These included, but were not limited to, the following three factors:

- The fact that the model assumed a minke whale would not forage for 24 hours after being disturbed
- The lack of density dependence in the model (meaning the population would not respond to any reduction in population size)

[10] iPCoD version 5.2

- The level of environmental and demographic stochasticity in the model

300. The following sections explore the background to each of these factors to illustrate the level of conservatism in this modelling and provide critical context for the evaluation of these results.

7.2.1 Duration of disturbance

301. The iPCoD model for minke whale and bottlenose dolphin disturbance was last updated following the expert elicitation in 2013 (Harwood *et al.,* 2014). When this expert elicitation was conducted, the experts provided responses on the assumption that a disturbed individual would not forage for 24 hours. However, the most recent expert elicitation in 2018 highlighted that this was an unrealistic assumption for harbour porpoises (generally considered to be more responsive than minke whales and bottlenose dolphin) and was amended to assume that disturbance resulted in six hours of non-foraging time (Booth *et al*., 2019).

302. As minke whales and bottlenose dolphins were not included in the updated expert elicitation for disturbance, the iPCoD model still assumed 24 hours of non-foraging time for minke whales and bottlenose dolphin. Given the most recent understanding of marine mammal reactions to pile driving, this scenario appears unrealistic. A recent study estimated energetic costs associated with disturbance from sonar, where it was assumed that 1 hour of feeding cessation was classified as a mild response, 2 hours of feeding cessation was classified as a strong response and 8 hours of feeding cessation was classified as an extreme response (Czapanskiy *et al.,* 2021).

303. The presumption of a 24-hour feeding cessation for minke whale and bottlenose dolphin surpasses what has been typically deemed an extreme response. Hence, it has been regarded as unrealistic and likely to inflate the actual disturbance levels anticipated from the Project.

304. Despite these limitations and uncertainties, this assessment has been carried out according to best practice, using the best available scientific information, and the latest expert elicitation results from Sinclair *et al*. (2020). The information provided was therefore considered to be sufficient to carry out an adequate assessment for harbour porpoise, bottlenose dolphin, minke whale, grey seal, and harbour seal.

7.2.2 Lack of density dependence

305. Density dependence has been described as 'the process whereby demographic rates change in response to changes in population density, resulting in an increase in the population growth rate when density decreases, and a decrease in that growth rate when density increases (Harwood *et al.*

2014). The iPCoD scenario run for bottlenose dolphin assumed no density dependence since there was insufficient data to parameterise this relationship. Essentially, this meant that there would be no ability for the modelled impacted population to increase in size and return to carrying capacity following disturbance.

306. At a recent expert elicitation on bottlenose dolphins, conducted for the purpose of modelling population impacts of the Deepwater Horizon oil spill (Schwacke *et al*. 2021), experts agreed that there would likely be a concave density dependence on fertility, which meant that, in reality, it would be expected that the impacted population would recover to carrying capacity (which was assumed to be equal to the size of unimpacted population – i.e. it was assumed the un-impacted population was at carrying capacity) rather than continuing at a stable trajectory that was smaller than that of the unimpacted population.

7.2.3 Environmental and demographic stochasticity

307. The iPCoD model attempts to model some of the sources of uncertainty inherent in the calculation of the potential effects of disturbance on marine mammal population. This includes demographic stochasticity and environmental variation. Environmental variation has been defined as *'the variation in demographic rates among years as a result of changes in environmental conditions'* (Harwood *et al*., 2014). Demographic stochasticity has been defined as *'variation among individuals in their realised vital rates as a result of random processes'* (Harwood *et al.,* 2014).

308. The iPCoD protocol describes this in further detail: 'Demographic stochasticity is caused by the fact that, even if survival and fertility rates were constant, the number of animals in a population that die and give birth will vary from year to year because of chance events. Demographic stochasticity has its greatest effect on the dynamics of relatively small populations, which has been incorporated into models for all situations where the estimated population within an MU was less than 3000 individuals. One consequence of demographic stochasticity was that two otherwise identical populations that experienced exactly the same sequence of environmental conditions would follow slightly different trajectories over time. As a result, it was possible for a "lucky" population that experienced disturbance effects to increase, whereas an identical undisturbed but "unlucky" population may decrease' (Harwood *et al.* 2014).

309. This was clearly evidenced in the outputs of iPCoD where the un-impacted (baseline) population size varied greatly between iterations, not as a result of disturbance but simply as a result of environmental and demographic stochasticity. In the example provided in **Plate 7.1**, after 25 years of

simulation, the un-impacted population size varied between 176 (lower 2.5%) and 418 (upper 97.5%). Thus, the change in population size resulting from the impact of disturbance was significantly smaller than that driven by the environmental and demographic stochasticity in the model.

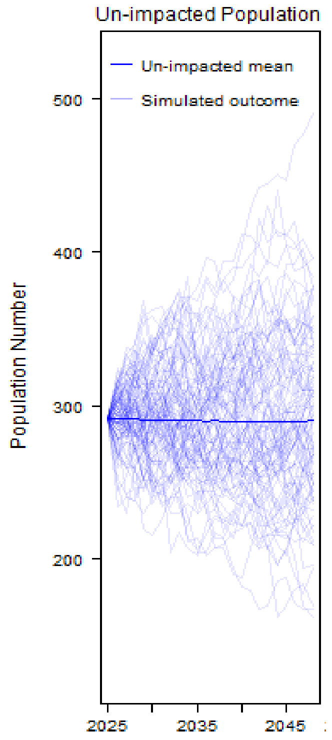

Plate 7.1 Simulated un-impacted (baseline) population size over the 25 years modelled

7.2.4 Summary

310. All of the precautions built into the iPCoD model mean that the results were considered to be highly precautionary. Despite the discussed limitations and uncertainties, this assessment has been carried out according to best practice, using the best available scientific information, and the latest expert elicitation results from Booth and Heinis (2018). The information provided was therefore considered to be sufficient to carry out an adequate assessment for bottlenose dolphin, harbour porpoise, harbour seal and grey seal. Results have also been presented for minke whale, noting the caveat above regarding no update to the expert elicitation for minke whale.

8 Review of potential disturbance from vessel activity

311. Noise levels reported by Malme *et al*. (1989) and Richardson *et al*. (1995) for transiting large surface vessels indicate that physiological damage to auditory sensitive marine mammals would be unlikely. The potential risk of PTS in marine mammals as a result of vessel noise is highly unlikely, as the sound levels would be well below the threshold for PTS (Southall *et al*., 2019b). In general, vessels generate noise in the low frequency range between 10-100 Hz (Erbe *et al*., 2019).

312. Vessel noise has been shown to affect the behaviour of marine mammals, where changes in vocalisation and behavioural state have been observed, in addition to displacement of animals from areas where ships were present.

313. The disturbance impact of displacement has been seen across a variety of marine mammal species. In a large-scale study of harbour porpoise density in UK waters, including the North Sea MU and the Irish Sea MU, increased vessel activity was associated with lower porpoise densities. However, in NW Scottish waters, shipping had little effect on the density of individuals (Heinänen and Skov, 2015). A similar trend was seen with a study of Indo-Pacific bottlenose dolphins, when analysing habitat occupancy along the coast of Western Australia, dolphin density was negatively affected by vessels at one site but had no significant impact at the other (Marley *et al*. 2017a). Displacement was also seen with harbour porpoise detections around a pile driving site, where detections declined several hours prior to the start of pile driving. The decline was assumed to be due to the increase in other construction related activities and vessel presence in advance of the actual pile driving (Brandt *et al*., 2018; Benhemma-Le Gall *et al*., 2020).

314. However, for harbour seals a recent UK telemetry study showed there was no evidence of reduced seal presence as a result of vessel traffic. This was despite distributional overlaps (overlaps were most frequently found within 50km of the coast) between seal and vessel presence and high cumulative sound levels (Jones *et al*., 2017). Another study of grey seal pup tracks in the Celtic Sea and adult grey seals in the English Channel found that no animals were exposed to cumulative shipping noise that exceeded thresholds for TTS (using the Southall *et al.,* 2019b thresholds) (Trigg *et al*., 2020). A study of grey seal pupping beaches around Ramsey Island in Pembrokeshire found that disturbance occurred when vessels were closer than 150m to seal locations (Strong and Morris, 2010). Reduced presence of common dolphins was seen with the construction of a pipeline in NW Ireland due to vessel presence, however patterns suggested disturbance impacts were only short term (Culloch *et al.,* 2016).

315. As well as the potential to have displacement effects, vessel activity has also been shown to elicit other potential behavioural changes. One study between 2012 - 2016 tagged seven harbour porpoises in a region of high shipping density in the inner Danish waters and Belt seas. The tagging of individuals provided data on responses to stressors in the marine environment. High noise levels coincided with erratic behaviour including ‘vigorous fluking’, bottom diving, interrupted foraging, and the cessation of vocalisations. Four out of six of the animals that were exposed to noise levels above 96dB re 1μPa (16kHz third octave levels) produced significantly fewer buzzes at high volumes of vessel noise. In one case, the proximity of a single vessel resulted in a 15 minute cessation in foraging (Wisniewska *et al*., 2018). Studies for bottlenose dolphin have indicated vessel presence has the potential to increase swimming speeds and reduce the time spent for foraging, resting and socialising (Marley *et al*., 2017b; Piwetz 2019). Behavioural changes associated with disturbance have also been seen in common dolphins, due to the presence of vessels. Foraging and resting activity was significantly disrupted by vessel activity and returns to foraging activity took significantly longer than returns to other states (Stockin *et al*., 2008, Meissner *et al*., 2015). Behavioural changes have also been seen in minke whale with vessel interactions including a decrease in foraging activity, increase in swim speeds and energy expenditure (Christiansen *et al*., 2014).

316. Evidence suggests marine mammal species respond to vessel presence in a variety of ways, but all have the potential to be disturbed either through displacement, behavioural changes or both. Responses depended on a range of environmental factors but also the type and size of vessels. Some of the studies mentioned above based findings on fast moving vessels and vessels seeking close proximity to species, such as fast ferries and whale watching vessels (Wisniewska *et al*., 2018; Christiansen *et al*., 2014). Therefore, less of a disturbance effect is likely for the proposed construction vessels which would be slow moving or stationary.

9 References

Adams, J. (2017). Manx Whale and Dolphin Watch land-based surveyor network report 2016. Available at: http://www.mwdw.net/web/wp-content/uploads/2016/09/Land-based-surveyors-project-report-2016.pdf (Accessed December 2023)

Andersen, L.W., Born, E.W., Dietz, R., Haug, T., Oien, N. and Bendixen, C. (2003). Genetic population structure of minke whales Balaenoptera acutorostrata from Greenland, the North East Atlantic and the North Sea probably reflect different ecological regions. *Marine Ecology Progress Series* 247: 263-280.

Anderwald, P., Danielsdottir, A.K., Haug, T., Larsen, F., Lesage, V., Reid, R.J., Vikingsson, G.A. and Hoelzel, A.R. (2011). Possible cryptic stock structure for minke whales in the North Atlantic: implications for conservation and management. *Biological Conservation* 144: 2479-2489.

Anderwald, P., Evans, P.G.H., Dyer, R., Dale, A., Wright, P.J. and Hoelzel, A.R. (2012). Spatial scale and environmental determinants in minke whale habitat use and foraging. *Marine Ecology Progress Series* 450: 259-274.

Baines, M.E. and Evans, P.G.H. (2012). Atlas of the Marine Mammals of Wales. CCW.

BEIS (2022b) UK Offshore Energy Strategic Environmental Assessment 4 (OESEA4) Appendix A1a.8 Marine mammals and otter. https://www.gov.uk/government/consultations/uk-offshore-energy-strategic-environmental-assessment-4-oesea4 (Accessed January 2024)

Benhemma-Le Gall, A., Graham, I.M., Merchant, N.D. and Thompson, P.M. (2021). Broad-Scale Responses of Harbor Porpoises to Pile-Driving and Vessel Activities During Offshore Windfarm Construction. *Front. Mar. Sci.* 8:664724. doi: 10.3389/fmars.2021.664724 (Accessed December 2023)

Benhemma-Le Gall, A., P. Thompson, I. Graham, and N. Merchant. (2020). Lessons learned: harbour porpoises respond to vessel activities during offshore windfarm construction.in Environmental Interactions of Marine Renewables 2020, Online.

Benhemma-Le Gall, A., Thompson, P., Merchant, N. and Graham, I. (2023). Vessel noise prior to pile driving at offshore windfarm sites deters harbour porpoises from potential injury zones. *Environmental Impact Assessment Review*, 103, p.107271.

Booth, C. G., F. Heinis, and Harwood J (2019). Updating the Interim PCoD Model: Workshop Report - New transfer functions for the effects of disturbance on vital rates in marine mammal species. Report Code SMRUC-BEI-2018-011, submitted to the Department for Business, Energy and Industrial Strategy (BEIS), February 2019 (unpublished).

Booth, C., and Heinis, F. (2018). Updating the Interim PCoD Model: Workshop Report - New transfer functions for the effects of permanent threshold shifts on vital rates in marine mammal species. Report Code SMRUC-UOA-2018-006, submitted to the University of Aberdeen and Department for Business, Energy and Industrial Strategy (BEIS), June 2018 (unpublished)

Börjesson, P., Berggren, P. and Ganning, B. (2003). Diet of harbour porpoises in the Kattegat and Skagerrak seas: accounting for individual variation and sample size. *Marine Mammal Science*, 19(1), pp.38-058.

Bouveroux, T., Waggitt, J.J., Belhadjer, A., Cazenave, P.W., Evans, P.G. and Kiszka, J.J., (2020). Modelling fine-scale distribution and relative abundance of harbour porpoises in the Southern Bight of the North Sea using platform-of-opportunity data. *Journal of the Marine Biological Association of the United Kingdom*, 100(3), pp.481-489.

Brandt, M. J., A.-C. Dragon, A. Diederichs, M. A. Bellmann, V. Wahl, W. Piper, J. NabeNielsen, and G. Nehls. (2018). Disturbance of harbour porpoises during construction of the first seven offshore wind farms in Germany. *Marine Ecology Progress Series* 596:213-232

Brandt, M. J., Diederichs, A., Betke, K. and Nehls, G. (2011). Responses of harbour porpoises to pile driving at the Horns Rev II offshore wind farm in the Danish North Sea. *Marine Ecology Progress Series*, 421, 205-216

Brookes KL, Bailey H, Thompson PM (2103) Predictions from harbor porpoise habitat association models are confirmed by long-term passive acoustic monitoring. *J Acoust Soc Am* 134: 2523-2533

Brophy, J. T., Murphy, S., and Rogan, E. (2009). The diet and feeding ecology of the short-beaked common dolphin (Delphinus delphis) in the northeast Atlantic. International Whaling Commission Scientific Committee paper SC/61/SM14, 18.

Cañadas A., Donovan GP., Desportes G., Borchers DL. (2009). A short review of the distribution of short beaked common dolphins (Delphinus delphis) in the central and eastern North Atlantic with an abundance estimate for part of this area. North Atlantic Sightings Surveys. NAMMCO Scientific Publications 7: 201-220.

Canning, S.J., Santos, M.B., Reid, R.J., Evans, P.G., Sabin, R.C., Bailey, N. and Pierce, G.J. (2008). Seasonal distribution of white-beaked dolphins (*Lagenorhynchus albirostris*) in UK waters with new information on diet and habitat use. *Journal of the Marine Biological Association of the United Kingdom*, 88(6), pp.1159-1166.

Carter, M.I., Boehme, L., Duck, C.D., Grecian, J., Hastie, G.D., McConnell, B.J., Miller, D.L., Morris, C., Moss, S., Thompson, D. and Thompson, P. (2020). Habitat-based predictions of at-sea distribution for grey and harbour seals in the British Isles: Report to BEIS, OESEA-16-76, OESEA-17-78.

Carter, M.I.D., Boehme, L., Cronin, M.A., Duck, C.D., Grecian, W.J., Hastie, G.D., Jessopp, M., Matthiopoulos, J., McConnell, B.J., Miller, D.L., Morris, C.D., Moss, S.E.W., Thompson, D., Thompson, P.M. and Russell, D.J.F. (2022). Sympatric

Seals, Satellite Tracking and Protected Areas: Habitat-Based Distribution Estimates for Conservation and Management. *Front. Mar. Sci.* 9:875869.

Christiansen, F., M. H. Rasmussen, and D. Lusseau. (2014). Inferring energy expenditure from respiration rates in minke whales to measure the effects of whale watching boat interactions. *Journal of Experimental Marine Biology and Ecology* 459:96-104.

Clark, A., A. Richter, B. Manley, and J. Adams. (2019). Cetacean Research in Manx Waters 2018. Manx Whale and Dolphin Watch.

CMACS Ltd. (2005). Gwynt y Môr Offshore Wind Farm Marine Ecology Technical Report. Report number: J3004/2005, Report to: npower renewables.

CMACS Ltd. (2011). Gwynt y Môr Offshore Wind Farm Marine Mammal Monitoring Baseline Report.

CMACS Ltd. (2013). Gwynt Y Môr (GYM) Offshore Wind Farm Marine Mammal Mitigation First Year Construction Mitigation Report v1.

CODA (2009). Cetacean Offshore Distribution and Abundance in the European Atlantic (CODA). Final Report. University of St Andrews, UK. http://biology.st-andrews.ac.uk/coda/. (Accessed November 2023)

Corkeron, P.J., Bryden, M.M. and Hedstrom, K.E. (1990). Feeding by bottlenose dolphins in association with trawling operations in Moreton Bay, Australia. In The bottlenose dolphin (pp. 329-336). Academic Press.

Couperus, A.S., 1997. Interactions between Dutch midwater trawl and Atlantic white-sided dolphins (*Lagenorhynchus acutus*) southwest of Ireland. Journal of Northwest Atlantic fishery science, 22.

Culloch, R. M., P. Anderwald, A. Brandecker, D. Haberlin, B. McGovern, R. Pinfield, F. Visser, M. Jessopp, and M. Cronin. (2016). Effect of construction-related activities and vessel traffic on marine mammals. *Marine Ecology Progress Series* 549:231-242.

Cumbria Wildlife Trust (CWT) (2023). 'Annual seal counts and pup data for Marine Mammal EWG*', unpublished raw data obtained through personal communication.*

Czapanskiy, M. F., M. S. Savoca, W. T. Gough, P. S. Segre, D. M. Wisniewska, D. E. Cade, and J. A. Goldbogen (2021). Modelling short-term energetic costs of sonar disturbance to cetaceans using high-resolution foraging data. *Journal of Applied Ecology*.

DECC (2016). UK Offshore Energy Strategic Environmental Assessment 3 (OESEA3): Appendix 1D: Water Environment. Available from: https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/504541/OESEA3_A1d_Water_environment.pdf (Accessed February 2024)

Donovan, C., J. Harwood, S. King, C. Booth, B. Caneco, and C. Walker (2016). Expert elicitation methods in quantifying the consequences of acoustic disturbance from offshore renewable energy developments. Pages 231-237 The Effects of Noise on Aquatic Life II. Springer.

Embling, C.B., Gillibrand, P.A., Gordon, J., Shrimpton, J., Stevick, P.T. and Hammond, P.S. (2009). Using habitat models to identify suitable sites for marine protected areas for harbour porpoises (Phocoena phocoena). *Biological Conservation*, 143(2), pp.267-279.

Erbe, C., S. A. Marley, R. P. Schoeman, J. N. Smith, L. E. Trigg, and C. B. Embling. (2019). The Effects of Ship Noise on Marine Mammals—A Review. *Frontiers in Marine Science* 6.

European Council (EC) (2008). 56/EC of the European Parliament and of the Council of 17 June 2008 establishing a framework for community action in the field of marine environmental policy (Marine Strategy Framework Directive). OJEC L, 164, p.40.

Evans, P.G., Anderwald, P. and Baines, M.E., (2003). UK cetacean status review. Report to English Nature and Countryside Council for Wales, UK.

Evans, P.G.H and Waggitt, J.J. (2023). Modelled Distribution and Abundance of Cetaceans and Seabirds in Wales and Surrounding Waters. NRW Evidence Report, Report No: 646, 354 pp. Natural Resources Wales, Bangor.

Evans, P.G.H., (1980). Cetaceans in British waters. *Mammal Review*, 10(1), pp.1-52.

Felce, T. (2014). Cetacean research in Manx waters 2007-2014.

Felce, T. (2015). Cetacean research in Manx waters 2015.

Fernandez-Betelu, O., Graham, I.M., Brookes, K.L., Cheney, B.J., Barton, T.R. and Thompson, P.M. (2021). Far-field effects of impulsive noise on coastal bottlenose dolphins. *Frontiers in Marine Science*, p.837.

Fontaine, M.C., Tolley, K.A., Siebert, U., Gobert, S., Lepoint, G., Bouquegneau, J.M. and Das, K. (2007). Long-term feeding ecology and habitat use in harbour porpoises Phocoena from Scandinavian waters inferred from trace elements and stable isotopes. *BMC Ecology*, 7, p.1.

Fontaine, M.L.C., Roland, K., Calves, I., Austerlitz, F., Palstra, F.P., Tolley, K.A., Ryan, S., Ferreira, M., Jauniaux, T., Llavona, A. and Ürk, B.Ö. (2014). Postglacial climate changes and rise of three ecotypes of harbour porpoises, *Phocoena phocoena*, in western Palearctic waters. *Molecular Ecology*, 23, pp.3306-3321.

Fraser F.C. (1946) Report on Cetacea stranded on the British coasts from 1933 to 1937. No. 12. London: British Museum (Natural History).

Gilles, A, Authier, M, Ramirez-Martinez, NC, Araújo, H, Blanchard, A, Carlström, J, Eira, C, Dorémus, G, FernándezMaldonado, C, Geelhoed, SCV, Kyhn, L, Laran, S, Nachtsheim, D, Panigada, S, Pigeault, R, Sequeira, M, Sveegaard, S,Taylor, NL, Owen, K, Saavedra, C, Vázquez-Bonales, JA, Unger, B, Hammond, PS. (2023). Estimates of cetacean abundance in European Atlantic waters in summer 2022 from the SCANS-IV aerial and shipboard surveys. Final report published 29 September 2023. 64 pp. https://tinyurl.com/3ynt6swa (Accessed March 2024)

Graham, I. M., E. Pirotta, N. D. Merchant, A. Farcas, T. R. Barton, B. Cheney, G. D. Hastie, and P. M. Thompson. (2017a). Responses of bottlenose dolphins and harbor porpoises to impact and vibration piling noise during harbor construction. *Ecosphere* 8(5):e01793. 10.1002/ecs2.1793

Graham, I. M., Farcas, A., Merchant, N. D., and Thompson, P. (2017b). Beatrice Offshore Wind Farm: An interim estimate of the probability of porpoise displacement at different unweighted single-pulse sound exposure levels. Prepared by the University of Aberdeen for Beatrice Offshore Windfarm Ltd.

Graham, I. M., Merchant, N. D., Farcas, A., Barton, T. R., Cheney, B., Bono, S., & Thompson, P. M. (2019). Harbour porpoise responses to pile-driving diminish over time. *Royal Society Open Science*, 6(6), 190335. https://doi.org/10.1098/rsos.190335 (Accessed January 2024)

Hammond, P.S. and Grellier, K. (2006). Grey seal diet composition and prey consumption in the North Sea. Final Report to Department for Environment Food and Rural Affairs Project MF0319.

Hammond, P.S., Berggren, P., Benke, H., Borchers, D.L., Collet, A., Heide-Jørgensen, M.P., Heimlich, S., Hiby, A.R., Leopold, M.F., Øien, N. (2002). Abundance of harbour porpoises and other cetaceans in the North Sea and adjacent waters. *J. Appl. Ecol.* 39, 361–376.

Hammond, P.S., Lacey, C., Gilles, A., Viquerat, S., Boerjesson, P., Herr, H., Macleod, K., Ridoux, V., Santos, M.B., Scheidat, M., Teilmann, J., Vingada, J. and Øien, N. (2021). Estimates of cetacean abundance in European Atlantic waters in summer 2016 from the SCANS-III aerial and shipboard surveys. June 2021. Available from: https://synergy.st-andrews.ac.uk/scans3/files/2021/06/SCANS-III_design-based_estimates_final_report_revised_June_2021.pdf (Accessed January 2024)

Hammond, P.S., Lacey, C., Gilles, A., Viquerat, S., Boerjesson, P., Herr, H., Macleod, K., Ridoux, V., Santos, M., Scheidat, M. and Teilmann, J. (2017). Estimates of cetacean abundance in European Atlantic waters in summer 2016 from the SCANS-III aerial and shipboard surveys. Wageningen Marine Research.

Hammond, P.S., Macleod, K., Berggren, P., Borchers, D.L., Burt, L., Cañadas, A., Desportes, G., Donovan, G.P., Gilles, A., Gillespie, D., Gordon, J. (2013). Cetacean abundance and distribution in European Atlantic shelf waters to inform conservation and management. Biological Conservation, 164, pp.107-122.

Hammond, P.S., Northridge, S.P., Thompson, D., Gordon J.C.D., Hall, A.I., Aarts, G. and Matthiopoulos, J. (2005). Background information on marine mammals for Strategic Environmental Assessment 6. Sea Mammal Research Unit.

Harmer, S.F. (1927). Report of Cetacea stranded on the British Isles from1913 to 1926. No. 10. London: British Museum (Natural History).

Harwood, J., S. King, R. Schick, C. Donovan, and C. Booth. (2014). A protocol for Implementing the Interim Population Consequences of Disturbance (PCoD) approach: Quantifying and assessing the effects of UK offshore renewable energy developments on marine mammal populations. Report Number SMRUL-TCE-2013-014. *Scottish Marine And Freshwater Science*, 5(2).

Hastie, G., N. D. Merchant, T. Götz, D. J. Russell, P. Thompson, and V. M. Janik. (2019). Effects of impulsive noise on marine mammals: investigating range-dependent risk. *Ecological Applications* 29:e01906

Heinänen, S. and Skov, H. (2015). The identification of discrete and persistent areas of relatively high harbour porpoise density in the wider UK marine area, JNCC Report No.544 JNCC, Peterborough.

Hernandez-Milian G, Berrow S, Santos MB, Reid D and Rogan E (2015). Insights into the trophic ecology of bottlenose dolphins (*Tursiops truncatus*) in Irish waters. *Aquatic Mammals* 41: 226-239.

Howe, L. (2018). Marine Mammals-Cetaceans. In; Manx Marine Environmental Assessment (1.1 Edition - partial update).

Howe, L. (2018a). Marine Mammals-Seals. In: Manx Marine Environmental Assessment (2nd Ed). Isle of Man Government. pp. 21.

IAMMWG (2015). Management Units for cetaceans in UK waters (January 2015). JNCC Report No. 547, JNCC Peterborough.

IAMMWG (2023). Review of Management Unit boundaries for cetaceans in UK waters (2023). JNCC Report 734, JNCC, Peterborough, ISSN 0963-8091.Updated abundance estimates for cetacean Management Units in UK waters. JNCC Report No. 680 (Revised March 2022), JNCC Peterborough, ISSN 0963-8091.

Ingram, S.N. and Rogan, E. (2002). Identifying critical areas and habitat preferences of bottlenose dolphins *Tursiops truncatus*. *Marine Ecology Progress Series*. 244, pp.247-255.

Isojunno S, Matthiopoulos J Evans PGH (2012) Harbour porpoise habitat preferences: robust spatio-temporal inferences from opportunistic data. *Mar Ecol Prog Ser* 448: 155-170

Jansen, O. E., Leopold, M. F., Meesters, E. H., and Smeenk, C. (2010). Are white-beaked dolphins *Lagenorhynchus albirostris* food specialists? Their diet in the southern North Sea. *Journal of the Marine Biological Association of the United Kingdom*, 90(8), 1501-1508.

JCDP. (2022). ICES, Copenhagen, Denmark. https://cetaceans.ices.dk

JNCC (2010a). JNCC guidelines for minimising the risk of injury to marine mammals from using explosives. August 2010.

JNCC (2010b). Statutory nature conservation agency protocol for minimising the risk of injury to marine mammals from piling noise. August 2010.

JNCC (2017). JNCC guidelines for minimising the risk of injury to marine mammals from geophysical surveys. April 2017.

JNCC (2023). DRAFT guidelines for minimising the risk of injury to marine mammals from unexploded ordnance clearance in the marine environment. October 2023.

JNCC, Natural England and CCW (2010). Draft EPS Guidance - The protection of marine European Protected Species from injury and disturbance. Guidance for the marine area in England and Wales and the UK offshore marine area. Joint Nature Conservation Committee, Natural England and Countryside Council for Wales. October 2010.

Johnston DW, Westgate AJ, Read AJ (2005) Effects of fine-scale oceanographic features on the distribution and movements of harbour porpoises Phocoena phocoena in the Bay of Fundy. *Mar Ecol Prog Ser* 295: 279-293

Jones, E., G. Hastie, S. Smout, J. Onoufriou, N. D. Merchant, K. Brookes, and D. Thompson. (2017). Seals and shipping: quantifying population risk and individual exposure to vessel noise. *Journal of Applied Ecology* 54:1930-1940.

Kastelein, R.A., Hardemann, J. and Boer, H. (1997). Food consumption and body weight of harbour porpoises (*Phocoena phocoena*). In The biology of the harbour porpoise Read, A.J., Wiepkema, P.R., Nachtigall, P.E (1997). Eds. Woerden, The Netherlands:De Spil Publishers. pp. 217–234

King, S. L., R. S. Schick, C. Donovan, C. G. Booth, M. Burgman, L. Thomas, and J. Harwood. (2015). An interim framework for assessing the population consequences of disturbance. *Methods in Ecology and Evolution* 6:1150-1158.

Koski, W.R., Johnson, S.R. (1987). Responses of bowhead whales to an offshore drilling operation in the Alaskan Beaufort Sea, Autumn 1986: behavioral studies and aerial photo grammetry. LGL Ltd., King City, ON.

Lewis, E.J. and Evans, P.G.H. (1993). Comparative ecology of bottlenose dolphins (*Tursiops truncatus*) in Cardigan Bay and the Moray Firth, pp.57-62. In: European Research on Cetaceans - 7. Proc. 7th Ann. Conf. ECS, Inverness, ed P.G.H. Evans. European Cetacean Society, Cambridge, England. 306pp.

Liret, C. (2001). Domaine vital, utilisation de l’espace et des ressources: les grands dauphins, *Tursiops truncatus*, de l’île de Sein. Thèse de doctorat de l’Université de Bretagne Occidentale, Brest. 155 p.

Liret, C., Creton, P., Evans, P. G. H., Heimlich-Boran, J. R. and Ridoux, V. (1998). English and French coastal Tursiops from Cornwall to the Bay of Biscay, 1996. Photo-identification Catalogue. Project sponsored by Ministere de l'Environnement, France and Sea Watch Foundation, UK.

Ljungblad, D. K., Würsig, B., Swartz, S. L., and Keene, J. M. (1988). Observations on the behavioral responses of bowhead whales (Balaena mysticetus) to active geophysical vessels in the Alaskan Beaufort Sea. Arctic, 183-194.

Lockyer, C. (1995). Aspects of the biology of the harbour porpoise, Phocoena phocoena, from British waters. In Developments in marine biology (Vol. 4, pp. 443-457). Elsevier Science.

Lockyer, C. (2003) Harbour porpoises (Phocoena phocoena) in the North Atlantic: Biological parameters. NAMMCO Scientific Publications 5: 71-89

Lohrengel, K., Evans, P.G.H., Lindenbaum, C.P., Morris, C.W., Stringell, T.B. (2018) Bottlenose Dolphin Monitoring in Cardigan Bay 2014 - 2016, NRW Evidence Report No: 191, 162pp, Natural Resources Wales, Bangor.

Lonergan, M., Duck, C., Moss, S., Morris, C. and Thompson, D. (2013). Rescaling of aerial survey data with information from small numbers of telemetry tags to estimate the size of a declining harbour seal population. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 23(1), pp.135-144.

Louis, M., Fontaine, M.C., Spitz, J., Schlund, E., Dabin, W., Deaville, R., Caurant, F., Cherel, Y., Guinet, C. and Simon-Bouhet, B. (2014). Ecological opportunities and specializations shaped genetic divergence in a highly mobile marine top predator. *Proceedings of the Royal Society B: Biological Sciences*, 281(1795), p.20141558.

Lowry, L.F., Frost, K.J., Hoep, J.M. and Delong, R.A. (2001). Movements of satellite-tagged subadult and adult harbor seals in Prince William Sound, Alaska. Marine Mammal Science 17(4): 835−861.

Luna, A., Sánchez, P., Chicote, C., and Gazo, M. (2022). Cephalopods in the diet of Risso's dolphin (*Grampus griseus*) from the Mediterranean Sea: A review. *Marine Mammal Science*, 38(2), 725-741.

Macdonald, M.A., Hildebrand, J.A. and Webb, S.C. (1995). Blue and fin whales observed on a seafloor array in the Northeast Pacific. *J Acoust Soc Am*. 98:712-721.

Macleod K, Simmonds MP, Murray E (2003) Summer distribution and relative abundance of cetacean populations off north-west Scotland. *J Mar Biol Assoc* U K 85: 1187- 1192

MacLeod, C.D., Santos, M.B., Burns, F., Brownlow, A. and Pierce, G.J. (2014). Can habitat modelling for the octopus Eledone cirrhosa help identify key areas for Risso's dolphin in Scottish waters? Hydrobiologia, 725, pp.125-136.

MacLeod, C.D., Weir, C.R., Santos, M.B. and Dunn, T.E. (2008). Temperature-based summer habitat partitioning between white-beaked and common dolphins around the United Kingdom and Republic of Ireland. *Journal of the Marine Biological Association of the United Kingdom* 88: 1193-1198.

Malme, C. I., and Miles, P. R. (1983). Acoustic testing procedures for determining the potential impact of underwater industrial noise on migrating gray whales. *The Journal of the Acoustical Society of America*, 74(S1), S54-S54.

Malme, C. I., Miles, P. R., Clark, C. W., Tyack, P., and Bird, J. E. (1984). Investigations of the potential effects of underwater noise from petroleum-industry activities on migrating gray-whale behavior. Phase 2: January 1984 migration (No. PB-86-218377/XAB; BBN-5586). Bolt, Beranek and Newman, Inc., Cambridge, MA (USA).

Mandlik, D.S. (2021) A photo-ID study of the Risso's dolphin (Grampus griseus) in coastal waters of Anglesey, along with the environmental determinants of their spatial and temporal distribution in the Irish Sea. MSc thesis, Bangor University.

Marçalo, A., Nicolau, L., Giménez, J., Ferreira, M., Santos, J., Araújo, H., Silva, A., Vingada, J. and Pierce, G.J. (2018). Feeding ecology of the common dolphin (Delphinus delphis) in Western Iberian waters: has the decline in sardine (Sardina pilchardus) affected dolphin diet?. *Marine Biology*, 165, pp.1-16.

Marley, S., C. S. Kent, and C. Erbe. (2017a). Occupancy of bottlenose dolphins (Tursiops aduncus) in relation to vessel traffic, dredging, and environmental variables within a highly urbanised estuary. *Hydrobiologia* 792:243-263.

Marley, S., C. Salgado-Kent, C. Erbe, and I. M. Parnum. (2017b). Effects of vessel traffic and underwater noise on the movement, behaviour and vocalisations of bottlenose dolphins in an urbanised estuary. *Nature* 7.

Marubini F, Gimona A, Evans PGH, Wright PJ, Pierce GJ (2009) Habitat preferences and interannual variability in occurrence of the harbour porpoise *Phocoena phocoena* off northwest Scotland. *Mar Ecol Prog Ser* 381: 297-310

Matthiopoulos J., McConnell, B.J., Duck, C. and Fedak, M.A. (2004). Using satellite telemetry and aerial counts to estimate space use by grey seals around the British Isles. *Journal of Applied Ecology*. 41(3):476-491.

McCauley, R.D., Jenner, M.N., Jenner, C., McCabe, K.A. and Murdoch, J. (1998). The response of humpback whales (*Megaptera novaeangliae*) to offshore seismic survey noise: preliminary results of observations about a working seismic vessel and experimental exposures. The APPEA Journal, 38(1), pp.692-707.

Meissner, A. M., F. Christiansen, E. Martinez, M. D. Pawley, M. B. Orams, and K. A. Stockin. (2015). Behavioural effects of tourism on oceanic common dolphins, Delphinus sp., in New Zealand: the effects of Markov analysis variations and current tour operator compliance with regulations. *PLoS ONE* 10:e0116962.

Meynier, L., Stockin, K.A., Bando, M.K.H. and Duignan, P.J. (2008). Stomach contents of common dolphin (Deiphinus sp.) from New Zealand waters. *New Zealand Journal of Marine and Freshwater Research*, 42(2), pp.257-268.

Miller, G. W., Moulton, V. D., Davis, R. A., Holst, M., Millman, P., MacGillivray, A., et al. (2005). Monitoring seismic effects on marine mammals – southeastern Beaufort Sea, 2001-2002. In S. L. Armsworthy, P. J. Cranford, and K. Lee (Eds.), Offshore oil and gas environmental effects monitoring: Approaches and technologies (pp. 511-542). Columbus, OH: Battelle Press.

Mona Offshore Wind Ltd (2023) Mona Offshore Wind Project: Preliminary Environmental Information Report.

Moreno, P. and Mathews, M. (2018). Identifying Foraging Hotspots of Bottlenose Dolphins in a Highly Dynamic System: A Method to Enhance Conservation in Estuaries. *Aquatic Mammals*, 44(6).

Morgan Offshore Wind Limited (2023). Morgan Offshore Wind Project: Preliminary Environmental Information Report.

Morris, C.D and Duck,C.D. (2019) Aerial thermal-imaging survey of seals in Ireland, 2017 to 2018. Irish Wildlife Manuals, No. 111 National Parks and Wildlife Service, Department of Culture, Heritage and the Gaeltacht, Ireland.

Murphy, S., Pinn, E.U. and Jepson, P.D. (2013). The short-beaked common dolphin (*Delphinus delphis*) in the North-East Atlantic: distribution, ecology, management and conservation status. Oceanography and *Marine Biology*: An Annual Review 51: 193-280.

Northridge, S., Mackay, A., Sanderson, D., Woodcock, R. and Kingston, A. (2004). A review of dolphin and porpoise bycatch issues in the Southwest of England. Report to the Department for Environment Food and Rural Affairs. Sea Mammal Research Unit, St. Andrews, Scotland, UK.

NRW (2021). NRW's position on the use of Marine Mammal Management Units for screening and assessment in Habitats Regulations Assessments for Special Areas of Conservation with marine mammal features. Marine Programme Board. Adams, J. (2017). Manx Whale and Dolphin Watch land-based surveyor network report 2016.

Nykänen, M., Louis, M., Dillane, E., Alfonsi, E., Berrow, S., O''Brien, J., Brownlow, A., Covelo, P., Dabin, W., Deaville, R. and de Stephanis, R. (2019). Fine-scale population structure and connectivity of bottlenose dolphins, *Tursiops truncatus*, in European waters and implications for conservation. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 29, pp.197-211.

O'Donnell, C., Mullins, E., Lynch, D., Lyons, K., Keogh, N. and O'Callaghan, S. (2018). Celtic Sea Herring Acoustic Survey cruise report 2018. FSS Survey Series 2018/04, 44pp.

O'Donnell, C., O'Malley, M., Lynch, D., Lyons, K., Keogh, N. and O'Driscoll, D. (2017). Celtic Sea Herring Acoustic Survey (CSHAS) cruise report 2017. FSS Survey Series 2017/04, 38pp

Oudejans, M.G., Visser, F., Englund, A., Rogan, E. and Ingram, S.N. (2015). Evidence for distinct coastal and offshore communities of bottlenose dolphins in the North East Atlantic. *PLoS ONE* 10(4): e0122668.

Paxton, C.G.M., Scott-Hayward, L., Mackenzie, M., Rexstad, E. and Thomas, L. (2016). Revised Phase III Data Analysis of Joint Cetacean Protocol Data Resources with Advisory Note, JNCC Report 517, ISSN 0963-8091: http://jncc.defra.gov.uk/page-7201. (Accessed December 2023)

Paxton, C.G.M., Scott-Hayward, L.A.S. and Rexstad, E. (2014). Review of available statistical approaches to help identify Marine Protected Areas for cetaceans and basking shark. Scottish Natural Heritage Commissioned Report No. 573, 47pp.

Penrose, R.S. (2020). Marine Mammal and Marine Turtle Strandings (Welsh Coast). CSIP/Marine Environmental Monitoring Annual Report.

Piwetz, S. (2019). Common bottlenose dolphin (Tursiops truncatus) behavior in an active narrow seaport. *PLoS ONE*.

Pusineri, C., Magnin, V., Meynier, L., Spitz, J., Hassani, S. and Ridoux, V. (2007). Food and feeding ecology of the common dolphin (Delphinus delphis) in the oceanic Northeast Atlantic and comparison with its diet in neritic areas. *Marine Mammal Science*, 23(1), pp.30-47.

R Core Team (2023). _R: A Language and Environment for Statistical Computing_. R Foundation for Statistical Computing, Vienna, Austria. https://www.R-project.org/ (Accessed January 2024)

Read AJ, Hohn AA (1995) Life in the fast lane: the life history of harbour porpoises from the Gulf of Maine. Mar Mamm Sci 11: 423-440

Reid, J.B., Evans, P.G. and Northridge, S.P. eds. (2003). Atlas of cetacean distribution in north-west European waters. Joint Nature Conservation Committee.

Richardson, J., Greene, C.R., Malme, C.I. and Thomson, D.H. (1995). Marine Mammals and Noise. San Diego California: Academic Press.

Richardson, W. J., Miller, G. W., and Greene, C. R., Jr. (1999). Displacement of migrating bowhead whales by sounds from seismic surveys in shallow waters of the Beaufort Sea. *Journal of the Acoustical Society of America*, 106, 2281.

Richardson, W. J., Würsig, B., and Greene Jr, C. R. (1986). Reactions of bowhead whales, B alaenamysticetus, to seismic exploration in the Canadian Beaufort Sea. *The Journal of the Acoustical Society of America*, 79(4), 1117-1128.

Rogan, E., Cañadas, A., Macleod, K., Santos, M.B., Mikkelsen, B., Uriarte, A., Canneyt, O.V., Vázquez, J.A. and Hammond, P.S. (2018). Distribution, abundance and habitat use of deep diving cetaceans in the North East Atlantic. *Deep Sea Research Part II: Topical Studies in Oceanography* 141: 8-19.

Rose, A., Brandt, M., Vilela, R., Diederichs, A., Schubert, A., Kosarev, V., Nehls, G. and Freund, C.K. (2019). Effects of noise-mitigated offshore pile driving on harbour porpoise abundance in the German Bight 2014-2016 (Gescha 2). Report by IBL Umweltplanung GmbH.

Russell, D.J., Hastie, G.D., Thompson, D., Janik, V.M., Hammond, P.S., Scott-Hayward, L.A., Matthiopoulos, J., Jones, E.L. and McConnell, B.J. (2016). Avoidance of wind farms by harbour seals is limited to pile driving activities. *Journal of Applied Ecology*, 53(6), pp.1642-1652.

Russell, D.J., McClintock, B.T., Matthiopoulos, J., Thompson, P.M., Thompson, D., Hammond, P.S., Jones, E.L., MacKenzie, M.L., Moss, S. and McConnell, B.J., (2015). Intrinsic and extrinsic drivers of activity budgets in sympatric grey and harbour seals. *Oikos*, 124(11), pp.1462-1472.

Russell, D.J., McConnell, B., Thompson, D., Duck, C., Morris, C., Harwood, J. and Matthiopoulos, J. (2013). Uncovering the links between foraging and breeding regions in a highly mobile mammal. *Journal of Applied Ecology*, 50(2), pp.499-509.

Russell, D.J.F, Jones, E.L. and Morris, C.D. (2017) Updated Seal Usage Maps: The Estimated at-sea Distribution of Grey and Harbour Seals. *Scottish Marine and Freshwater Science* 8 (25), 25pp.

Russell, D.J.F. and McConnell, B.J. (2014). Seal at-sea distribution, movements and behaviour. Report to DECC. URN: 14D/085. March 2014 (final revision).

Santos, M.B. and Pierce, G.J. (2003). The diet of harbour porpoise (*Phocoena phocoena*) in the North east Atlantic. Oceanography and *Marine Biology*: an Annual Review 2003, 41, 355–390.

Santos, M.B., German, I., Correia, D., Read, F.L., Cedeira, J.M., Caldas, M., López, A., Velasco, F. and Pierce, G.J. (2013). Long-term variation in common dolphin diet in relation to prey abundance. *Marine Ecology Progress Series*, 481, pp.249-268.

Santos, M.B., Pierce, G.J., Learmonth, J.A., Reid, R.J., Ross, H.M., Patterson, I.A.P., Reid, D.G. and Beare, D., (2004). Variability in the diet of harbor porpoises (*Phocoena phocoena*) in Scottish waters 1992–2003. *Marine Mammal Science*, 20(1), pp.1-27.

Santos, M.B., Pierce, G.J., Reid, R.J., Patterson, I.A.P., Ross, H.M. and Mente, E. (2001). Stomach contents of bottlenose dolphins (*Tursiops truncatus*) in Scottish waters. Marine Biological Association of the United Kingdom. *Journal of the Marine Biological Association of the United Kingdom*, 81(5), p.873.

SCANS (1995). Distribution and abundance of the harbour porpoise and other small cetaceans in the North Sea and adjacent waters. Final report under LIFE Nature project LIFE 92-2/UK/027.

SCANS-II (2008). Small cetaceans in the European Atlantic and North Sea. Final Report submitted to the European Commission under project LIFE04NAT/GB/000245, SMRU, St Andrews.

SCANS-III aerial and shipboard surveys (2001). June 2021. Available from: https://synergy.st-andrews.ac.uk/scans3/files/2021/06/SCANS-III_design-based_estimates_final_report_revised_June_2021.pdf (Accessed February 2024)

Schwacke, L. H., T. A. Marques, L. Thomas, C. Booth, B. C. Balmer, A. Barratclough, K. Colegrove, S. De Guise, L. P. Garrison, and F. M. Gomez (2021). Modeling population impacts of the Deepwater Horizon oil spill on a long-lived species with implications and recommendations for future environmental disasters. Conservation Biology.

SCOS (2020). Scientific Advice on Matters Related to the Management of Seal Populations: 2020. Available at: http://www.smru.st-andrews.ac.uk/research-policy/scos/http://www.smru.st-andrews.ac.uk/research-policy/scos/ (Accessed December 2023)

SCOS (2021). Scientific Advice on Matters Related to the Management of Seal Populations: 2021. Available at: http://www.smru.st-andrews.ac.uk/files/2022/08/SCOS-2021.pdfhttp://www.smru.st-andrews.ac.uk/files/2022/08/SCOS-2021.pdf (Accessed December 2023)

SCOS (2022). Scientific Advice on Matters Related to the Management of Seal Populations: 2022. Available at: http://www.smru.st-andrews.ac.uk/files/2023/09/SCOS-2022.pdfhttp://www.smru.st-andrews.ac.uk/files/2023/09/SCOS-2022.pdf (Accessed December 2023)

Shucksmith, R., Jones, N.H., Stoyle, G.W., Davies, A., Dicks, E.F. (2009) Abundance and distribution of the harbour porpoise (Phocoena phocoena) on the north coast of Anglesey, Wales, UK. *Journal of the Marine Biological Association of the United Kingdom*, 89(5), 1051-1058.

Silva, M.A. (1999). Diet of common dolphins, Delphinus delphis, off the Portuguese continental coast. *Journal of the Marine Biological Association of the United Kingdom*, 79(3), pp.531-540.

Sinclair, R. R., Sparling, C. E., & Harwood, J. (2020). Review Of Demographic Parameters and Sensitivity Analysis To Inform Inputs And Outputs Of Population

Consequences Of Disturbance Assessments For Marine Mammals. *Scottish Marine and Freshwater Science*, 11(14), 74. https://doi.org/10.7489/12331-1 (Accessed January 2024)

Southall, B. L., S. L. Deruiter, A. Friedlaender, A. K. Stimpert, J. A. Goldbogen, E. Hazen, C. Casey, S. Fregosi, D. E. Cade, A. N. Allen, C. M. Harris, G. Schorr, D. Moretti, S. Guan, and J. Calambokidis (2019a). Behavioral responses of individual blue whales (Balaenoptera musculus) to mid-frequency military sonar. *The Journal of Experimental Biology*

Southall, B., J. J. Finneran, C. Reichmuth, P. E. Nachtigall, D. R. Ketten, A. E. Bowles, W. T. Ellison, D. Nowacek, and P. Tyack. (2019b). Marine Mammal Noise Exposure Criteria: Updated Scientific Recommendations for Residual Hearing Effects. *Aquatic Mammals* 45:125-232.

Southall, B.L., Bowles, A.E., Ellison, W.T., Finneran, J.J., Gentry, R.L., Greene Jr., C.R., Kastak, D., Ketten, D.R., Miller, J.H., Nachtigall, P.E., Richardson, W.J., Thomas, J.A., and Tyack, P.L. (2007). Marine Mammal Noise Exposure Criteria: Initial Scientific Recommendations. *Aquatic Mammals*, 33 (4), pp. 411-509.

Southall, B.L., Nowacek, D.P., Bowles, A.E., Senigaglia, V., Bejder, L. and Tyack, P.L. (2021). Marine mammal noise exposure criteria: assessing the severity of marine mammal behavioral responses to human noise. *Aquatic Mammals*, 47(5), pp.421-464. DOI 10.1578/AM.47.5.2021.421.

Spitz, J., Mourocq, E., Leauté, J.P., Quéro, J.C. and Ridoux, V. (2010). Prey selection by the common dolphin: Fulfilling high energy requirements with high quality food. *Journal of Experimental Marine Biology and Ecology*, 390(2), pp.73-77.

Spitz, J., Trites, A.W., Becquet, V., Brind'Amour, A., Cherel, Y., Galois, R. and Ridoux, V. (2012). Cost of living dictates what whales, dolphins and porpoises eat: the importance of prey quality on predator foraging strategies. *PLoS ONE*, 7(11), p.e50096.

Stalder, D., van Beest, F.M., Sveegaard, S., Dietz, R., Teilmann, J., Nabe-Nielsen, J. (2020) Influence of environmental variability on harbour porpoise movement. *Mar Ecol Prog Ser* 648: 28 207-219

Stevens A. 2014. A photo-ID study of the Risso's dolphin (Grampus griseus) in Welsh coastal waters and the use of Maxent modeling to examine the environmental determinants of spatial and temporal distribution in the Irish Sea. MSc thesis, University of Bangor. 97pp.

Stockin, K. A., D. Lusseau, V. Binedell, N. Wiseman, and M. B. Orams. (2008). Tourism affects the behavioural budget of the common dolphin Delphinus sp. in the Hauraki Gulf, New Zealand. *Marine Ecology Progress Series* 355:287-295.

Stokes, L., Young, M., Howe L. (2021). *Calf of Man Survey Autumn 2021*. Manx Wildlife Trust. Available at: https://www.mwt.im/sites/default/files/2023-

03/Calf%20Seal%20Report%202021.pdfhttps://www.mwt.im/sites/default/files/2023-03/Calf%20Seal%20Report%202021.pdf. (Accessed August 2023).

Strong, P. and Morris, S.R. (2010). Grey seal (*Halichoerus grypus*) disturbance, ecotourism and the Pembrokeshire Marine Code around Ramsey Island. *J. Ecotourism* 9(2): 117–132.

Sveegaard S, Nabe-Nielsen J, Stæhr KJ, Jensen TF, Mouritsen KN, Teilmann J (2012) Spatial interactions between marine predators and their prey: herring abundance as a driver for the distributions of mackerel and harbour porpoise. *Mar Ecol Prog Ser* 468: 7 245-253

Sveegaard, S., Andreasen, H., Mouritsen, K.N., Jeppesen, J.P., Teilmann, J. and Kinze, C.C. (2012). Correlation between the seasonal distribution of harbour porpoises and their prey in the Sound, Baltic Sea. *Marine Biology*, 159, pp.1029-1037.

Teilmann, J., Christiansen, C.T., Kjellerup, S., Dietz, R. and Nachman, G. (2013). Geographic, seasonal and diurnal surface behaviour of harbour porpoise. *Marine Mammal Science* 29(2): E60-E76.

Thompson, P.M., Brookes, K.L., Graham, I.M., Barton, T.R., Needham, K., Bradbury, G. and Merchant, N.D. (2013). Short-term disturbance by a commercial two-dimensional seismic survey does not lead to long-term displacement of harbour porpoises. *Proc R Soc B* 280: 20132001. http://dx.doi.org/10.1098/rspb.2013.2001.(Accessed October 2023)

Todd, S., Lien, J., Marques, F., Stevick, P., and Ketten, D. (1996). Behavioural effects of exposure to underwater explosions in humpback whales (Megaptera novaeangliae). *Canadian Journal of Zoology*, 74(9), 1661-1672.

Tolley, K.A. and Rosel, P.E. (2006). Population structure and historical demography of eastern North Atlantic harbour porpoises inferred through mtDNA sequences. *Marine Ecology Progress Series*, 327, pp.297-308.

Trigg, L., F. Chen, G. Shapiro, S. Ingram, C. Vincent, D. Thompson, D. Russell, M. I. D. Carter, and C. Embling. (2020). Predicting the exposure of diving grey seals to shipping noise. *The Journal of the Acoustical Society of America* 148.

Tyack, P. L., and Thomas, L. (2019). Using dose–response functions to improve calculations of the impact of anthropogenic noise. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 29, 242-253. https://doi.org/10.1002/aqc.3149 (Accessed September 2023)

UK Government (2011). Marine Policy Statement. Available at: https://www.gov.uk/government/uploads/system/uploads/attachment_data/file/69322/pb3654-marine-policy-statement-110316.pdf (Accessed December 2023)

Waggitt JJ, Cazenave PW, Howarth LM, Evans PGH, Van der Kooij J, Hiddink JG (2018) Combined measurements of prey availability explain habitat selection in foraging seabirds. *Biol Lett*, doi: 10.1098/rsbl.2018.034

Waggitt, J.J., Evans, P.G., Andrade, J., Banks, A.N., Boisseau, O., Bolton, M., Bradbury, G., Brereton, T., Camphuysen, C.J., Durinck, J. and Felce, T. (2019). Distribution maps of cetacean and seabird populations in the North-East Atlantic. *Journal of Applied Ecology*, 57(2), pp.253-269.

Weir CR, O'Brien SH (2000) Association of the harbour porpoise (*Phocoena phocoena*) with the Western Irish sea front. European Research on Cetaceans, 14, 61–65.

Whyte, K. F., Russell, D. J. F., Sparling, C. E., Binnerts, B., and Hastie, G. D. (2020). Estimating the effects of pile driving sounds on seals: Pitfalls and possibilities. *The Journal of the Acoustical Society of America*, 147(6), 3948–3958. https://doi.org/10.1121/10.0001408 (Accessed January 2024)

Williamson, L.D., Brookes, K.L., Scott, B.E., Graham, I.M. and Thompson, P.M., (2017). Diurnal variation in harbour porpoise detection potential implications for management. *Marine Ecology Progress Series*, 570, pp.223-232.

Wilson, B., Thompson, P.M., Hammond, P.S. (1997). Habitat use by bottlenose dolphins: seasonal distribution and stratified movement patterns in the Moray Firth Scotland. The *Journal of Applied Ecology* 34, pp.1365–1374.

Windsland, K., Lindstrom U., Nilssen, K.T. and Haug, T. (2007). Relative abundance and size composition of prey in the common minke whale diet in selected areas of the north-eastern Atlantic during 2000-04. *J. Cetacean Res. Manage*, 9(3), pp.167-178.

Wisniewska, D. M., M. Johnson, J. Teilmann, U. Siebert, A. Galatius, R. Dietz, and P. T. Madsen. (2018). High rates of vessel noise disrupt foraging in wild harbour porpoises (Phocoena phocoena). *Proceedings of the Royal Society B: Biological Sciences* 285:20172314.

Wisniewska, D.M., Johnson, M., Teilmann, J., Rojano-Donate, L., Shearer, J., Sveegaard, S., Miller, L.A., Siebert, U. and Madsen, P.T., (2016). Ultra-high foraging rates of harbor porpoises make them vulnerable to anthropogenic disturbance. *Current Biology*, 26(11), pp.1441-1446.

Wotton, A. (2023). Humpback whale spotted in Manx waters for first time in six years. *BBC*, 14 July. Available at: https://www.bbc.co.uk/news/world-europe-isle-of-man-66205421https://www.bbc.co.uk/news/world-europe-isle-of-man-66205421 (Accessed November 2023).