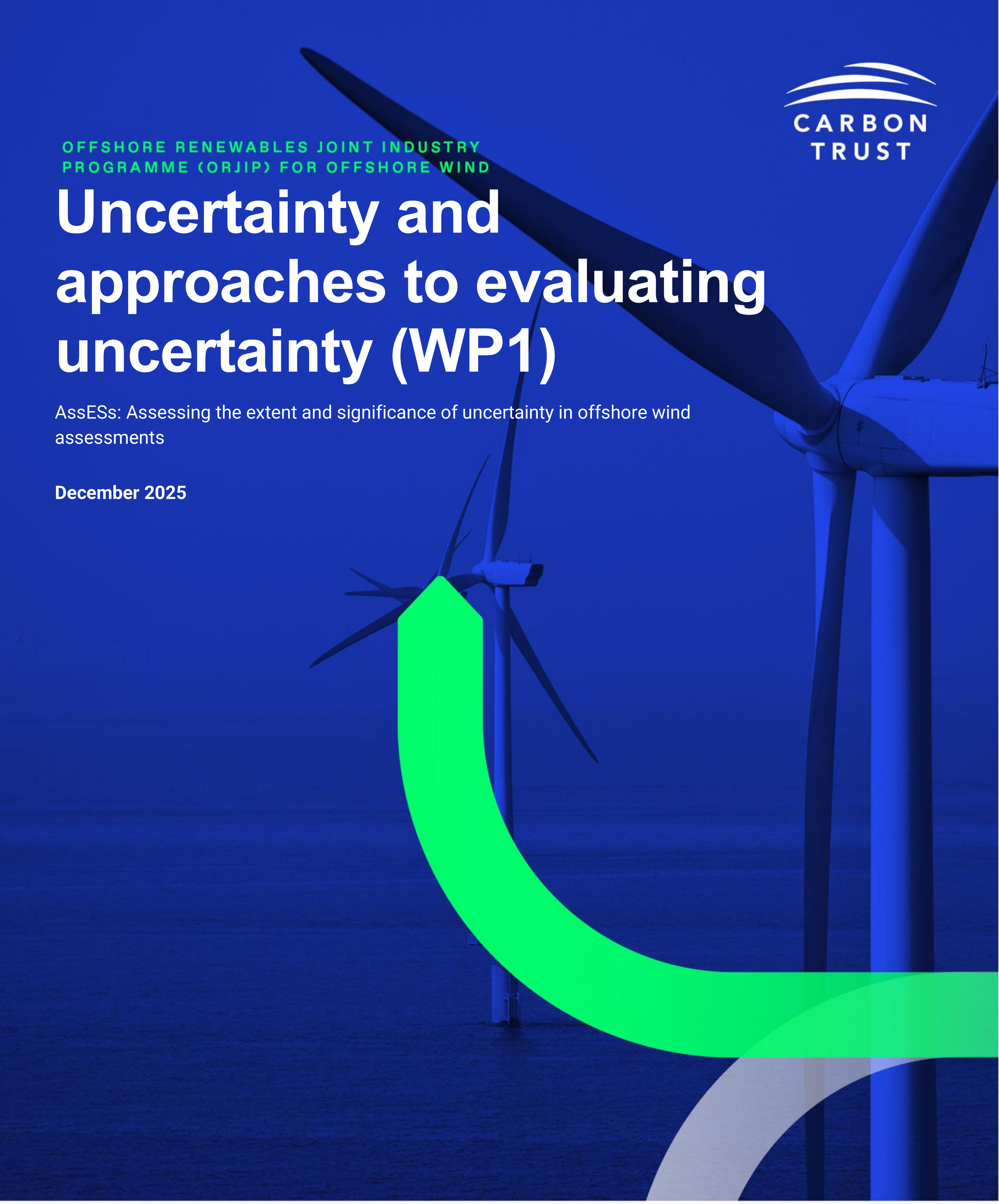
OFFSHORE RENEWABLES JOINT INDUSTRY PROGRAMME (ORJIP) FOR OFFSHORE WIND

Uncertainty and approaches to evaluating uncertainty (WP1)

AssESs: Assessing the extent and significance of uncertainty in offshore wind assessments

December 2025



ORJIP Offshore Wind

The Offshore Renewables Joint Industry Programme (ORJIP) for Offshore Wind is a collaborative initiative that aims to:

- Fund research to improve our understanding of the effects of offshore wind on the marine environment.
- Reduce the risk of not getting, or delaying consent for, offshore wind developments.
- Reduce the risk of getting consent with conditions that reduce viability of the project.

The programme pools resources from the private sector and public sector bodies to fund projects that provide empirical data to support consenting authorities in evaluating the environmental risk of offshore wind. Projects are prioritised and informed by the ORJIP Advisory Network which includes key stakeholders, including statutory nature conservation bodies, academics, non-governmental organisations and others.

The current stage is a collaboration between the Carbon Trust, EDF Energy Renewables Limited, Ocean Winds UK Limited, Equinor ASA, Ørsted Power (UK) Limited, RWE Offshore Wind GmbH, Shell Global Solutions International B.V., SSE Renewables Services (UK) Limited, TotalEnergies OneTech, Crown Estate Scotland, Scottish Government (acting through the Offshore Wind Directorate and the Marine Directorate) and The Crown Estate Commissioners.

For further information regarding the ORJIP Offshore Wind programme, please refer to the Carbon Trust website, or contact Žilvinas Valantiejus (zilvinas.valantiejus@carbontrust.com) and Ivan Savitsky (ivan.savitsky@carbontrust.com).

Acknowledgements

This document was produced on behalf of ORJIP Offshore Wind by MacArthur Green (part of SLR Consulting), the UK Centre for Hydrology (UKCEH) and Biomathematics and Statistics Scotland (BioSS). The report was authored by Prof. Bob Furness (MacArthur Green), Dr. Kate Searle (UKCEH), Prof. Francis Daunt (UKCEH), Dr. Esther Jones (BioSS), Ms. Katie Delacombe (BioSS) and Dr. Adam Butler (BioSS).

The reviews within Sections 2 and 3 were primarily written by MacArthur Green, with input on specific elements of the text from UKCEH and BioSS. The review within Section 4 was primarily written by UKCEH, with input from BioSS. The Introduction and Conclusions (Sections 1 and 5) were primarily written by BioSS, with input from MacArthur Green and UKCEH.

The project was advised by the ORJIP Offshore Wind Steering Group and the AssESs Project Expert Panel. We would like to thank the following organisations for their advice and support of the project via participation on the Project Expert Panel:

- Department for Environment, Food and Rural Affairs (Defra)
- Joint Nature Conservation Committee (JNCC)
- Natural England
- NatureScot

- Royal Society for the Protection of Birds (RSPB)
- Scottish Government Marine Directorate

This report was sponsored by the ORJIP Offshore Wind programme. For the avoidance of doubt, this report expresses the independent views of the authors.

Who we are

Our mission is to accelerate the move to a decarbonised future.

We have been climate pioneers for more than 20 years, partnering with leading businesses, governments and financial institutions globally. From strategic planning and target setting to activation and communication – we are your expert guide to turn your climate ambition into impact.

We are one global network of 400 experts with offices in the UK, the Netherlands, Germany, South Africa, Singapore and Mexico. To date, we have helped set 200+ science-based targets and guided 3,000+ organisations in 70 countries on their route to Net Zero.

Citation

Furness, R.W., Searle, K., Daunt, F., Jones, E., Delacombe, K. and Butler, A. 2025. *Uncertainty and approaches to evaluating uncertainty (WP1). AssESs - Assessing the extent and significance of uncertainty in offshore wind assessments*. The Carbon Trust. December 2025.

Contents

List of Tables

List of Abbreviations

AEOI	Adverse Effect on Integrity
Accelerate	NERC Ecological Implications of Accelerated Seabed Mobility around Windfarms
ANBS	ORJIP Apportioning in Non-Breeding Season
AppSaS	ORJIP Apportioning seabirds seen-at-sea
ASL	Above Sea Level
AssESs	Assessing the extent and significance of uncertainty in offshore wind assessments
BDMPS	Biologically Defined Minimum Population Scales
BRAIDS	ScotMER Bird Responses to Avian Influenza and Developments at Sea
BTO	The British Trust for Ornithology
CRM	Collision Risk Modelling
CTMM	Continuous Time Movement Modelling
DAS	Digital Aerial Survey
DisNBS	ORJIP Effects of displacement from Offshore Renewable Developments in the non-breeding season
ECOWIND	NERC Ecological Consequences of Offshore Wind
ECOWINGS	NERC Ecosystem Change, Offshore Wind, Net Gain and Seabirds
EIA	Environmental Impact Assessment
EU	European Union
GPS	Global Positioning System
HPAI	Highly Pathogenic Avian Influenza
HRA	Habitats Regulations Assessment
InTAS	ORJIP Integration of Tracking and At-sea Survey Data
IQR	Interquartile Range
JNCC	Joint Nature Conservation Committee
LSE	Likely Significant Effect
MD	Marine Directorate
MDE	Marine Data Exchange
MD-LOT	Marine Directorate Licencing Operations Team
MetaKitti	ORJIP Modelling of Kittiwake Metapopulation Dynamics
MOTUS	OWEC Remote Tracking of Seabirds at Sea
NE	Natural England
NERC	Natural Environment Research Council
ORD	Offshore Renewable Development
ORJIP	Offshore Renewables Joint Industry Programme for Offshore Wind
OW	Offshore Wind
OWF	Offshore Wind Farm
OWEC	Offshore Wind Evidence and Change Programme
PCM	Post-consent monitoring
Pelagio	NERC Physics-to-Ecosystem Level Assessment of Impacts of Offshore Windfarms
POSEIDON	OWEC Planning Offshore Wind Strategic Environmental Impact DecisiONs

PrediCtOr	ORJIP/OWEC Prevalence of Seabird Species and Collision Events in Offshore Wind Farms
PrePARED	OWEC Predators and Prey Around Renewable Energy Developments
ProcBe	OWEC Procellariiform Behaviour and Demographics
PTT	Push to Talk
PVA	Population Viability Analysis
QuMR	ORJIP Quantification of Mortality Rates associated with Displacement
RESCUE	OWEC Reducing Seabird Collisions Using Evidence
RSPB	The Royal Society for the Protection of Birds
ScotMER	Scottish Marine Energy Research
sCRM	Stochastic Collision Risk Model
SD	Standard Deviation
SEATRACK	Seabird Tracking
SG	Scottish Government
SMP	Seabird Monitoring Programme
SNCB	Statutory Nature Conservation Body
SPA	Special Protection Area
SUPERGEN	SUstainable PowER GENeration
WP	Work Package

1. Introduction

The Offshore Renewables sector is expanding rapidly, with the growth of the sector motivated by policies to mitigate anthropogenic climate change and increase energy security, driven by ambitious targets including the delivery of 43-50GW of offshore wind at a UK level by 2030. Assessments of the potential ecological impacts of developments must be undertaken to meet the legislative requirements of the EIA Directive (2011/92/EU), Marine Strategy Framework Directive (EC/2008/56), Habitats Directive (EC/92/43), Birds Directive (EC/79/409) and derived legislation. Ornithological impacts are of particular concern, given the global importance of UK seabird populations and the sensitivity of protected seabird species to offshore wind developments.

Assessments of ornithological impacts are complex and typically involve substantial uncertainty. Within this context, the legislative framework requires a precautionary approach to decision-making. Reduction of uncertainty offers direct potential to reduce consent risk by reducing the necessity for precaution. Improved quantified of uncertainty is also crucial, however, since a failure to properly quantify uncertainty, or effectively use information on uncertainty in decision-making, introduces the risk that precaution will be applied inappropriately. Improved quantification of uncertainty reduces this risk, and allows potential approaches for the reduction of uncertainty (e.g., via additional data collection or novel methods of analysis) to be evaluated and prioritised, leading to the reduction of consent risk as new evidence become available.

The key objective of the AssESs Project ("Assessing the extent and significance of uncertainty in offshore wind assessments") is to evaluate the extent and significance of uncertainty within ornithological offshore wind impact assessments, and subsequently provide recommendations for how priority sources of uncertainty can be reduced and how precaution should be applied within the assessment process. The main objectives of this work are to:

- Identify and assess uncertainty and approaches to evaluating uncertainty in ornithological offshore wind impact assessments.
- Evaluate the significance of parameter uncertainty on the outputs of ornithological impact assessments, including how sources of uncertainty interact through the assessment process for both individual projects and cumulatively.
- Seek stakeholder views on how precaution should be applied, accounting for the extent and significance of uncertainty. Define a set of recommendations to address a) the reduction of uncertainty in assessment methods, and b) the treatment of uncertainty within ornithological offshore wind impact assessments.

The AssESs Project is divided into four work packages. Within Work Package (WP) 1 we review the approaches and tools that are commonly used in ornithological assessments within the UK context, review evidence around levels of uncertainty within inputs to assessments, and provide an updated version of the roadmap for reducing and better quantifying uncertainty developed in Searle et al. (2021, 2023a). WP2 then undertakes sensitivity analyses to evaluate the sensitivity of key outputs from assessments to the values of input parameters, with the structure of these analyses being informed by both stakeholder engagement and the outputs of WP1. The outputs from these two work packages are then used within WP3 to inform a structured process of stakeholder engagement (workshops and interviews) to explore the use of uncertainty and precaution within assessments. Finally, WP4 will develop recommendations around (a) changes to treatment of uncertainty within assessments and (b) future research priorities.

This report summarises WP1. Section 2 ("Commonly used tool and methods") focuses on reviewing Statutory Nature Conservation Body (SNCB) guidance in order to summarise the approaches and tools that are commonly used in ornithological assessments within the UK context. Note that this review was conducted in August 2024, and so is not able to capture changes to guidance that have occurred subsequent to this: in particular, it does not include the updating of NatureScot guidance on collision risk modelling (CRM) to reflect JNCC et al. (2024). Section 3 ("Sources of uncertainty in parameters") reviews sources of evidence that are used to specify the values of biological parameters used within these approaches and tools, and the uncertainty within these values - the focus is both of quantified levels of uncertainty and reporting of the metrics used to character this (standard errors, standard deviations, confidence intervals, ranges, bootstrap samples) and on broader, more qualitative, sources of uncertainty (e.g. around selection of underlying sources of information, evaluation of transferability, and selection of model structures). Section 4 ("Update to uncertainty roadmap for ornithological offshore wind impact assessments") provides an update of the roadmap in Searle et al. (2023a) to account for both research activities that have begun since publication of that paper, and to identify knowledge gaps that have emerged, or increased in importance, since publication of that paper.

2. Commonly used tools and methods

We review tools and methods that are commonly used within the context of UK assessments. Guidance issued by Statutory Nature Conservation Bodies (SNCBs) underpins the approaches used in assessments so in order to identify commonly used tools and methods we review current published SNCB guidance.

2.1. Guidance documents

This review of SNCB guidance (WP1.1) was initially completed on 6 August 2024. On 15 August 2024 JNCC published new guidance on CRM assessment, on behalf of the UK SNCBs. Since the new guidance changes several aspects of CRM, that new guidance (JNCC et al., 2024) has been taken into account in this review and text has been added to incorporate that latest information. Note, however, that this revision was completed on 19 August 2024, prior to the online guidance of NatureScot being altered in line with the new Joint Nature Conservation Committee (JNCC) guidance, and so represents a significant but temporary discrepancy of approach between the current online guidance documents.

JNCC coordinated the SNCBs to prepare guidance on particular aspects of impact assessments, including advising avoidance rates to use in collision risk modelling (SNCB, 2014; JNCC et al., 2024) and advising displacement parameter values to use in displacement impact assessments (SNCB, 2017, 2022a,b).

Natural England has provided a series of guidance documents relating to assessing the impacts of offshore wind on seabirds. Those documents are not openly available but are retained on a server behind a firewall with access made available by Natural England on request and registration. The guidance covers pre-application baseline data (Parker et al., 2022a), pre-application engagement and evidence plan process (Parker et al., 2022b), data analysis and presentation at examination (Parker et al., 2022c), and monitoring and environmental requirements at the post-consent phase (Parker et al., 2022d). The intention is to update these documents when new evidence becomes available, and so it is essential to consider the latest version of each document (version number being given in the document but the date of publication so far being retained as 2022).

Parker et al. (2022c) advised that avoidance rates in SNCB (2014) should be used in CRM, based on the compilation by Cook et al. (2014) but with no suitable avoidance rates being available for use with the extended stochastic CRM models (options 3 or 4) so that those options should not be used. However, new interim advice on updated CRM parameters was provided later the same year by Natural England (2022), which alters and updates advice on CRM that was given by Parker et al. (2022c). The newer guidance advises the interim use of avoidance rates based on a reanalysis by Ozsanlav-Harris et al. (2023) of largely the same terrestrial wind farm collision rate data as used by Cook et al. (2014), together with evidence of higher macro-avoidance of offshore wind turbines by gannets (with an avoidance rate correction for gannet being based on large gull avoidance data from terrestrial wind farms).

In 2023, NatureScot published 11 guidance notes online relating to assessing the impacts of offshore wind farms on seabirds in Scottish waters (NatureScot, 2023a-k). These comprise: (1) an overview (NatureScot, 2023a), (2) baseline characterisation surveys and reporting (NatureScot, 2023b), (3) identifying theoretical connectivity with breeding site Special Protection Areas using breeding season foraging ranges (NatureScot, 2023c), (4) determining connectivity of marine birds with marine Special Protection Areas and breeding seabirds from colony SPAs in the non-breeding season (NatureScot, 2023d), (5) recommendations for marine bird population estimates (NatureScot, 2023e), (6) marine ornithology impact pathways for offshore wind developments (NatureScot, 2023f), (7) advice for assessing collision risk of marine birds (NatureScot, 2023g), (8) advice for assessing the distributional responses, displacement and barrier effects of marine birds (NatureScot, 2023h), (9) advice for seasonal definitions for birds in the Scottish marine environment (NatureScot, 2023i), (10) advice for apportioning impacts to breeding colonies (NatureScot, 2023j), and (11) recommendations for seabird Population Viability Analysis (PVA) (NatureScot, 2023k). Older versions of two of these documents contained errors (such as in the document title) which have been corrected in updated versions. Although dated 2023, guidance note 10 (apportioning) simply states (on 19 August 2024) “Guidance note to be completed shortly” (NatureScot, 2023j). As with Natural England, the intention is to update the guidance documents when new evidence becomes available, and so it is essential to consider the latest version of each document (version number and date being given in the document). However, several of these documents seem to be in need of updating.

The guidance prescribes methodology for key processes in impact assessment: abundance estimation from survey data, CRM, displacement impact assessment, apportioning assessed impacts to Special Protected Area (SPA) populations, and PVA analysis of the consequence for population trajectory.

Guidance from SNCBs is very welcome. It ensures consistency of approach by developers and their consultants. However, there are two contradictory issues that are difficult to resolve. Ideally, guidance will remain the same, so that all planning applications are treated consistently. However, it is also important to ensure that evidence-based policy takes account of new evidence when that becomes available. That is especially important where there is a lack of evidence and high uncertainty, as is inevitable in subject areas that are novel and difficult to study as in the case of offshore wind farm impacts on seabirds.

2.2. Collision risk modelling

Collision risk modelling (CRM) is a key tool used to assess the impact of wind farms on birds flying through a wind farm (Masden & Cook, 2016). In the UK, one of the versions of the Band model (Band, 2012) is the primary CRM in use. These models are sensitive to some key parameters including flight heights, flight speeds, bird densities (Masden et al., 2021) and in particular to the avoidance rate (Black et al., 2019). There is therefore a need to ensure that the avoidance rate is as accurate as possible and that, in the context of offshore wind farms, it reflects the true behaviour of seabirds (JNCC, 2024). According to JNCC (2024) "The avoidance rate is typically thought of as quantifying active avoidance behaviour in response to wind farms." That statement is correct about the viewpoint of people but is not how it is treated in the Band CRM model, where it is a correction term that accounts for error in the predicted numbers of collisions based on evidence of actual numbers of collisions. This is appreciated by some SNCBs. For example, NatureScot (2023g) states "We note that use of site-specific flight height for sCRM requires recalculations of avoidance rates".

Seabird density, measured as the monthly mean density of flying seabirds of focal species determined from digital aerial surveys, is used to derive monthly flux rate of birds passing turbines based on recommended flight speeds for each seabird species. A significant source of uncertainty is that when the model is used for assessments, flux is calculated based on density data from pre-construction aerial surveys. There is therefore uncertainty around how representative pre-construction surveys are of densities post-construction.

A large part of the discrepancy between observed and Band model predicted numbers of collisions is a consequence of bird behaviour to avoid a collision, but numerous other factors are also tied up within the avoidance rate correction. It is unfortunate that this correction factor is termed "avoidance rate" as it is more than that. It is important to understand this because any change to any parameter in the Band model (such as a change in flight speed used for a particular species) will result in the need to adjust the avoidance rate value to reflect the fact that a change in an input parameter to the model will result in a change in the predicted number of collisions, and therefore will alter the difference between predicted numbers and actual numbers. In particular, the estimated avoidance rate is especially sensitive to error in the estimated number of birds passing through the turbine array during the study period, and to the estimated proportion of birds passing through the rotor-swept area (Ozsanlav-Harris et al., 2023).

JNCC et al. (2024) state "The calculated avoidance rates recommended …. incorporate elements of error in relation to both the data used and the model itself (Band, 2012). The incorporation of this error means that the avoidance rates used by Band (2012) or McGregor et al. (2018) are likely to be lower than those measured empirically". This is inaccurate. The component of error due to the parameter values used in the Band model could either over-estimate or under-estimate the likely number of collisions, resulting in a calculated avoidance rate correction that could be either too high or too low. For example, the predicted number of collisions is affected by flight speed. If flight speed is over-estimated then the model will normally predict more collisions. If flight speed is under-estimated then the model will normally predict fewer collisions. Comparing prediction with observed numbers of collisions will result in the avoidance rate being higher in one case and lower in the other. The estimate is not biased towards being over-estimated but could be either over or under-estimated depending on the direction of the error in the parameter value in the model.

For terrestrial wind farms, actual numbers of collisions can be monitored by carcass surveys corrected for survey effort and carcass detection probability. Band model predicted numbers of collisions can then be compared with empirically observed numbers of collisions based on carcass surveys, allowing the avoidance rate correction factor to be calculated. For offshore wind farms it is not practical to carry out carcass surveys. Until very recently there has been no effective technological solution to measure actual collision rates of individual seabird species at offshore wind farms, although development of integrated tracking radar and camera systems (Skov et al., 2018; Tjørnløv et al., 2023) now make that a possibility. In the future there will be empirical data on collision rates that will allow avoidance rates to be quantified at offshore wind farms. That was not possible when offshore wind farms were first constructed, so as a result, precautionary estimates of the likely avoidance rate correction factor were initially used in impact assessments for offshore wind based on evidence for a range of bird taxa at terrestrial wind farms. A small step forward was made by assuming that data on avoidance rates of gulls quantified at terrestrial wind farms could be applied for gulls at offshore wind farms and that other seabird taxa at offshore wind farms that do not occur in terrestrial environments might show similar avoidance rates to gulls (Cook et al., 2014).

There are differences in behaviour of gulls when they are in terrestrial and in marine habitats. For example, tracking data show that gulls tend to fly higher over land than they do over the sea (Corman and Garthe, 2014), and this difference was also evident in data from terrestrial and offshore wind farm studies (Ozsanlav-Harris et al., 2023). There are also differences between wind turbines typically found at terrestrial wind farms and at offshore wind farms. For example, turbines at offshore wind farms tend to be larger. Many of the studies of avoidance rates of gulls at terrestrial wind farms were made some years ago when installed turbine sizes were smaller than those typical of recently constructed terrestrial wind farms. Therefore, data from terrestrial wind farms are not very good proxies for avoidance rate corrections that would be appropriate for offshore wind farms.

SNCB guidance is to use the flux rate of flying birds as input to the CRM model, adjusted by the proportion of birds flying at collision risk height, with a species-appropriate avoidance rate (model correction factor) based on terrestrial wind farm data. NatureScot (2023g) recommends the use of the stochastic collision risk model (sCRM) developed by Masden (2015) and expanded into a sCRM tool by McGregor et al. (2018), which was subsequently developed into the Caneco version of the tool, the last being the most recent version so now the most appropriate to use.

2.2.1. Avoidance rates advised by SNCBs for use in CRM

NatureScot (2023g) states that avoidance rates published in SNCB (2014) should be used in all assessments of seabird collision risk at offshore wind farms in Scottish waters, but that this guidance will be updated once the report by Ozsanlav-Harris et al. becomes publicly available. However, although Ozsanlav-Harris et al. was published in March 2023 and draft versions had been available to SNCBs earlier, no update from NatureScot has yet been published online. As of October 2024 the online NatureScot (2023g) still states "*The Joint Response SNCB to the Marine Scotland Science Avoidance Rate Review guidance note (2014) on avoidance rates should be used with +/- 2 standard deviations*". SNCB (2014) provided the recommended avoidance rate corrections listed in Table 1, which remain those advised by NatureScot according to the current online guidance. Obviously, the data published in 2014 are based on studies carried out in earlier years, mostly in the 2000s, and completely exclude evidence gathered since 2014.

Table 1. SNCB (2014) guidance on appropriate avoidance rate corrections.

Species	Model	Avoidance rate in SNCB, (2014) guidance
Northern gannet	Basic Band model	0.989 (SD 0.002)
Black-legged kittiwake	Basic Band model	0.989 (SD 0.002)
Great black-backed gull	Basic Band model	0.995 (SD 0.001)
Great black-backed gull	Extended Band model (1)	0.989 (SD 0.002)
Lesser black-backed gull	Basic Band model	0.995 (SD 0.001)
Lesser black-backed gull	Extended Band model (1)	0.989 (SD 0.002)
Herring gull	Basic Band model	0.995 (SD 0.001)
Herring gull	Extended Band model (1)	0.990 (SD 0.002)
All other species	Basic Band model	0.98

(1) SNBCs do not encourage use of the extended Band model but details of that model are given in Band (2012) and Cook (2021).

However, the new guidance published by JNCC et al. (2024) in August 2024 advises the use of avoidance rates in Ozsanlav-Harris et al. (2023). Natural England no longer support use of avoidance rates in SNCB (2014) but provide updated interim guidance based on the review by Ozsanlav-Harris et al. (2023). The guidance provided (Natural England, 2022):

- supports the use of the stochastic CRM (sCRM, McGregor et al., 2018);
- states that the Extended Band Model is no longer recommended for any species;
- makes some changes to recommended nocturnal activity factors;
- introduces a novel approach for northern gannet in which a 70% (or 65% to 85%) macro-avoidance rate is used to reduce the density of birds in flight, followed by using the “all gulls” avoidance rate correction for the remaining birds that do not show macro-avoidance;
- updates the avoidance rate corrections to be used in assessments (Table 2a).

Table 2. Recommended avoidance rate corrections of Natural England (2022) and JNCC et al. (2024).

a) Natural England (2022)

Species	Avoidance rate correction for Basic Band model	Avoidance rate correction for Basic sCRM model
Northern gannet	0.992 after 70% macro-avoidance correction to flux rate	0.993 (SD 0.0003) after 70% macro-avoidance correction to flux rate
Black-legged kittiwake	0.992	0.993 (SD 0.0003)
Great black-backed gull	0.994	0.994 (SD 0.0004)
Lesser black-backed gull	0.994	0.994 (SD 0.0004)
Herring gull	0.994	0.994 (SD 0.0004)
Common gull	0.995	0.995 (SD 0.0002)
Black-headed gull	0.995	0.995 (SD 0.0002)
All other species	0.990	0.991 (SD 0.0004)

b) JNCC et al. (2024)

Species	Avoidance rate correction for Basic Band model	Avoidance rate correction for Basic sCRM model
Northern gannet	0.9923 (SD 0.0001)	0.9929 (SD 0.0003)
Black-legged kittiwake	0.9923 (SD 0.0001)	0.9929 (SD 0.0003)
Great black-backed gull	0.9936 (SD 0.0002)	0.9940 (SD 0.0004)
Lesser black-backed gull	0.9936 (SD 0.0002)	0.9940 (SD 0.0004)
Herring gull	0.9936 (SD 0.0002)	0.9940 (SD 0.0004)
Common gull	0.9947 (SD 0.0003)	0.9949 (SD 0.0003)
Black-headed gull	0.9947 (SD 0.0003)	0.9949 (SD 0.0003)
All other species including Sandwich tern	0.9902 (SD 0.0001)	0.9908 (SD 0.0004)

Avoidance rate corrections being advised by Natural England (Natural England, 2022) and JNCC et al. (2024) therefore now differ considerably from those advised by NatureScot (NatureScot, 2023g). However, the evidence used by Natural England (2022) and by JNCC et al. (2024) is also mostly based on less appropriate data than are now available with newly developed technologies such as combined radar and camera systems deployed at offshore wind farms. The review by Ozsanlav-Harris et al. (2023) is mostly based on data collected at terrestrial sites before 2010. Of 19 study sites included in their review, one was studied in the 1990s, ten in the 2000s, eight in the 2010s, and there are no data from any study site for the 2020s. It is unclear why more recent data have not been used by SNCBs and by reviews they have commissioned. If avoidance rate corrections are to be based on data from terrestrial wind farms it would seem logical to use available recent data rather than excluding such studies, as numerous studies are being carried out not only in the UK but world-wide, following best practice guidelines (IFC, EBRD and KfW, 2023). Furthermore, SNCB guidance has not so far taken into account the most relevant recent study from Aberdeen Offshore Wind Farm: an interim report based on the first year of study was published in 2021 (but which was not included in the review by Ozsanlav-Harris et al. (2023), and the final report containing two year's study data was published in February 2023 (Tjørnløv et al., 2023). It is not clear why that study was omitted from the review by Ozsanlev-Harris et al. (2023) except that their review aimed to consider difficulties with data sets that had previously been included in earlier assessments of avoidance rates so focused on those earlier studies. The data from Aberdeen Offshore Wind Farm (Tjørnløv et al., 2023) avoid the problem of applying terrestrial wind farm data as a poor proxy for offshore wind farms, because their study quantified collisions at an offshore wind farm.

There are other limitations to the data used by Ozsanlav-Harris et al. (2023) that have been taken as the basis for prescribing avoidance rate corrections by Natural England (2022) and by JNCC et al. (2024). Of the 18 study sites in the review for which collision data were published, three sites, Zeebrugge (a harbour site with data collected 2000-2005), Oosterbierum (a terrestrial site with data collected 1990-1991), and Slufterdam (a terrestrial site with data collected 2012), recorded a corrected total of 368.2 carcasses. The other 15 sites recorded a corrected total of 164 carcasses. Thus, three sites (17% of the sites) were responsible for 70% of the carcasses. This suggests that collision risk is strongly influenced by individual site (since all studies carried out appropriate carcass correction calculations taking account of search effort, season, terrain, predator-presence etc and the calculations of Ozsanlav-Harris et al. (2023) used corrected data). Furthermore, the two sites with the largest numbers of carcasses (Zeebrugge and Oosterbierum) were the two studies carried out longest ago (so likely to be the least appropriate proxies for OWFs), which further calls into question the decision within SNCB guidance to not add data from studies carried out more recently.

Tjørnløv et al. (2023) recorded video of 781 northern gannet approaches to a turbine at Aberdeen Offshore Wind Farm and found 100% avoidance behaviour by those birds. From this they calculated an overall meso-avoidance of 1 (SD 0) for northern gannet. For 3388 "large gull" approaches, they calculated a meso-avoidance of 0.87. For 2624 approaches by herring gulls they calculated a meso-avoidance of 0.69. For 321 approaches by great black-backed gulls they calculated a meso-avoidance of 0.71. For 2178 approaches by black-legged kittiwake they calculated a meso-avoidance of 0.31 (Tables 10-1 and 10-2 in Tjørnløv et al., 2023). For those individuals that did not show meso-avoidance behaviour, 54 "large gulls" showed micro-avoidance of 0.963; 68 herring gulls showed micro-avoidance of 0.971; 35 "small gulls" showed micro-avoidance of 1; 28 black-legged kittiwakes showed micro-avoidance of 1 (Table 11-3 in Tjørnløv et al. 2023). No birds monitored by radar and video collided with turbine blades, so the overall Band model avoidance rate correction for each study species at Aberdeen Offshore Wind Farm was 1 (SD 0). Although an avoidance rate of 1 (as estimated for northern gannet at Aberdeen OWF) is improbable and the true avoidance is presumably less than 1, the true avoidance may well be closer to that estimated at Aberdeen OWF than to the "all gulls" proxy from terrestrial wind farms. This evidence should be incorporated into the SNCB guidance on avoidance rates at offshore wind farms, and deserves greater weight than evidence collected many years ago from terrestrial wind farms, although that evidence may still need to be used to an extent until further studies from OWF become available.

Natural England (2022) suggest making a 70% macro-avoidance correction to flux rate to account for macro-avoidance by northern gannets (before applying a within-wind farm avoidance rate correction of 0.992). However, JNCC et al. (2024) state that macro-avoidance corrections must be discussed with the relevant SNCB in the case of northern gannet and no figure is given as a recommended correction. The sources of evidence supporting selection of 70% as the macro-avoidance rate are not clear in Natural England (2022), but Peschko et al. (2021) showed that 89% of GPS-tagged breeding adult northern gannets from Helgoland avoided entering offshore wind farms in the German Bight. Subsequently, Natural England commissioned a review of evidence which found nine studies that quantified macro-avoidance by northern gannets at ten offshore wind farms (Pavat et al., 2023). That review recommended application of a macro-avoidance rate of 85.64% (SD 13.34) based on those nine studies. Pavat et al. (2023) found no difference in northern gannet macro-avoidance between the breeding season and non-breeding, but found evidence of site-based variation and could not rule out the possibility of temporal variation. Three published studies of macro-avoidance were not included in the review by Pavat et al. (2023): at Luchterduinen gannet macro-avoidance was 74%, at Prinses Amalia 89% and at Egmond aan Zee 90% (Heinänen and Skov, 2018). It is unclear why those studies were not included. The Aberdeen Offshore Wind Farm study (Tjørnløv et al., 2023), published around the same time as the review by Pavat et al. (2023), indicates that northern gannet meso-avoidance behaviour was 1 (SD 0) for a sample of 781 gannet approaches. The Band model avoidance correction proposed by Natural England for northern gannet (0.992) was made on evidence collected prior to the Aberdeen study and including that new evidence as best available evidence from offshore wind farms would require adjustment of the avoidance rate correction to a greater value.

The availability of high quality data on avoidance behaviour of seabirds at offshore wind farms as a result of combined radar and video recording systems now allows more appropriate use of avoidance rates derived from studies at offshore wind farms rather than from terrestrial sites. There is an urgent need for SNCBs to use the new offshore evidence from Thanet (Skov et al., 2018) as well as Aberdeen (Tjørnløv et al., 2023) to produce updated avoidance rate guidance that gives stronger priority to evidence from offshore wind farms and reduces weight given to evidence from terrestrial wind farms given that those are a poor proxy for OWF collision risk assessment.

Macro-avoidance by species other than northern gannet is also not accounted for in Natural England (2022) or JNCC et al. (2024). However, there is evidence of macro-avoidance by Sandwich terns (Leemans et al., 2022; van Bemmelen et al., 2024).

In assessments of seabird collision mortality impacts of offshore wind farms, a key consideration is the contribution of a particular offshore wind farm to the in-combination impact (the impact of all developments combined) on breeding features of the closest SPA populations of species such as northern gannet, black-legged kittiwake, Sandwich tern, or lesser black-backed gull. Guidance advises that assessment assumes that collision mortality is uniformly distributed across age classes of the focal species' population (Parker et al., 2022c). Thus if 25% of the population in the area comprises breeding adults from the focal SPA, then 25% of the assessed collision mortality should be allocated to that SPA population. However, there is strong evidence that birds of breeding adult status have learned to avoid many mortality risks and that accidental mortality is much more likely to affect the younger age classes than to affect birds of breeding age/status (Dagys, 2001; Riotte-Lambert and Weimerskirch, 2013; Afan et al., 2019; Costa et al., 2020). For example, ring recovery data indicate that first and second year European shags are approximately three times as likely (per bird) to be shot than older birds and four to five times more likely (per bird) to be drowned in fishing gear than are adults, while all ring recoveries of European shags drowned in lobster pots were of young juveniles and no ringed adults died that way (Galbraith et al., 1981). It is likely that collision avoidance behaviour by adult seabirds is more successful than by juveniles and immatures, although we have no understanding yet of the mechanisms behind such learning. Based on numerous studies of age-specific development of skills in seabirds it seems likely that adults will perform slightly better than immatures and much better than juveniles. If avoidance rates are higher among adults than among immatures, the impact on the breeding adult component of SPAs will be overestimated if it is assumed that collisions occur pro rata across age classes. This also raises questions about the use of "all gulls" or "large gulls" categories rather than species-specific data "herring gull" or "lesser black-backed gull" or "great black-backed gull" in estimating appropriate avoidance rate corrections. JNCC et al. (2024) advise the use of "all gull" rate for gannet and kittiwake (on the grounds that although kittiwakes are small gulls they behave differently from other small gulls so possibly the all gull value might be more appropriate), "large gull" rate for herring gull, lesser black-backed gull and great black-backed gull, "small gull" rate for common gull and black-headed gull and "all gulls and terns" rate for Sandwich tern and other species, rather than use of data for these individual species. Since birds identified to species are likely to be predominantly birds in adult plumage whereas the "large gulls" rate is likely to include a high proportion of immature birds (because those cannot easily be identified to species) then the inexperience of immature birds may lead to lower estimates of avoidance. It is, therefore, noteworthy that the avoidance rates calculated by Ozsanlav-Harris et al. (2023) for herring gull (0.9952 (SD 0.0003)), for lesser black-backed gull (0.9954 (SD 0.0003)) and for great black-backed gull (0.9991 (SD 0.0002)) are considerably higher than their calculated avoidance rate correction for "large gulls" (0.9936 (SD 0.0002)). This last is likely to be lower because of the inclusion of a high proportion of juveniles and immatures and so is likely not to be appropriate for application to samples of adults as in Habitat Regulations Assessments (HRA). Use of the "large gull" avoidance rate correction will apparently overestimate collision risk for adult herring gulls, lesser black-backed gulls and great black-backed gulls because age-related changes in behaviour have not been taken into consideration.

2.2.2. Flight heights

NatureScot (2023g) recommends use of flight height distribution data from the corrigendum of Johnston et al. (2014), and these data are the default in the sCRM tool. However, JNCC et al. (2024) now recommend that "robust site-specific flight height data is utilised for proposed offshore wind developments, if available".

The flight height data in Johnston et al. (2014) are derived from large quantities of data from numerous offshore survey areas, but were obtained predominantly by observer estimates from boats, without the use of accurate measuring equipment. Furthermore, the raw data were often recorded into a small number of height bands, so required statistical manipulation to merge disparate data sets using different numbers and sizes of height bands. More recently, several researchers have measured seabird flight heights, using one or more of digital aerial survey image analysis, laser rangefinder, radar, LiDAR, GPS tag deployment, or altimeter tags. All of these methods have advantages and disadvantages.

There have been ongoing problems with attempting to estimate flight heights of seabirds from digital aerial surveys, and a new method to try to resolve these issues has been proposed, using relative change in size in video images to estimate flight height (Humphries et al., 2023). However, that novel methodology has been criticised and may be flawed according to these critics (Boersch-Supan et al., 2024; Forster et al., 2024).

LiDAR carried on survey aircraft may potentially allow accurate measurement of seabird flight height but data have not yet been published beyond the proof of concept stage (Cook et al., 2018) and further work on this may be ongoing. Laser rangefinders may also allow accurate measurement of seabird flight height but are difficult to use, especially where seabirds are flying close to the sea surface, and the inbuilt compass can be affected by metal of ships or at-sea platforms (Skov et al., 2018; Fijn and Collier, 2022). Vertical radar can measure bird flight height but cannot easily identify bird species and cannot accurately measure flight height or even presence of birds at wave level (Fijn et al., 2015). Tracking radar can be combined with a thermal camera system to provide a video recording from which bird species identification may be possible, but this system is limited in the range over which it can record birds and is less able to detect or track birds close to the sea surface (Skov et al., 2018; Tjørnløv et al., 2023), although post-processing can improve results from radar tracking (van Erp et al., 2024).

Johnston et al. (2023) found flight heights produced from altimeter tags on lesser black-backed gulls breeding at two colonies in the UK to be significantly, although not consistently, higher than heights from GPS tags on the same birds. Comparing data from a variety of GPS tags plus barometric tags deployed on raptors, Schaub et al. (2023) found that barometric height was affected by bias but with low noise, whereas flight heights estimated from GPS tag data were affected by noise but little bias. They concluded that barometric altimetry may provide more accurate height data than standard low-frequency GPS tracking, but it involves risk of systematic error that can arise from the lack of local pressure estimates at a height of zero. High-frequency GPS tracking provided greatest accuracy in flight height estimation. As such, further study of the errors and biases associated with altimeter and GPS tags are needed, and what secondary data are required alongside these to maximise accuracy.

Schneider et al. (2024) describe an autonomous thermal tracking system that can be deployed on buoys at sea to measure flight heights and to track flight paths of seabirds, and systems of that type may be suitable to provide seabird baseline data surveys in future.

2.2.3. Other parameters in the Band model

Associated with avoidance rate corrections required for basic or stochastic collision risk modelling using the Band model, Natural England (2022) also advise the use of specific input parameter values for the CRM or sCRM (Table 3 and Table 4).

Table 3. Natural England's recommended summary data for the basic Band CRM model (Natural England, 2022). According to Natural England (2022) nocturnal activity factor for northern gannet is from Furness et al. (2018) and for all other species based on Garthe and Hüppop (2004). According to Natural England (2022) flight speed data are taken from Alerstam (1997) [(1)] except for northern gannet from Pennycuick (1987) [(2)] and Sandwich tern from Fijn and Gyimesi (2018). According to Natural England (2022) body length and wing span data are from Snow and Perrins (1987) [(3)]. Subsequent alterations to these data by JNCC et al. (2024) are shown in bold below the value advised by Natural England (2022).

Species	Flight speed m/s and (SD)	Nocturnal activity factor	Body length (m)	Wing span (m)	Flight type	% of flights upwind
Northern gannet	14.9	8% **14%**	0.94	1.72	Flapping	50
Black-legged kittiwake	13.1	25-50% **40%**	0.39	1.08	Flapping	50
Great black-backed gull	13.7	25-50%	0.71	1.58	Flapping	50
Lesser black-backed gull	13.1	25-50% **30%**	0.58	1.42	Flapping	50
Herring gull	12.8	25-50%	0.6	1.44	Flapping	50
Common gull	Consult SNCB	See Garthe & Hüppop (2004)	Consult SNCB	Consult SNCB	Flapping	50
Black-headed gull	Consult SNCB	See Garthe & Hüppop (2004)	Consult SNCB	Consult SNCB	Flapping	50
Sandwich tern	10.3	See Garthe & Hüppop (2004)	0.38	1.00	Flapping	50

(1) Listed in Natural England (2022) references as Alerstam et al. (2007).

(2) Listed in Natural England (2022) references as Pennycuick (1997).

(3) Listed in Natural England (2022) references as Snow and Perrins (1998).

It is not clear from Natural England (2022) whether the data tabulated in the guidance have been extracted from references that are listed in the reference list in the guidance, or from references listed in the tables in the guidance that are not in the reference list. Specifically, there do not appear to be any papers by T. Alerstam as sole author (or indeed as first author in an "et al." paper) that were published in 1997, nor are there publications by Snow and Perrins in 1987. However, Pennycuick (1987) could refer to a paper on flight speeds of seabirds studied by ornithodolite in Shetland which may be a more appropriate reference for seabird flight speeds than the Pennycuick (1997) reference that is listed in the guidance.

NatureScot (2023g) present almost exactly the same table for use with the basic Band model, except that they classify northern gannet flight as gliding rather than flapping. NatureScot (2023g) also give identical incorrect references to those given by Natural England (2022).

Table 4. Natural England's recommended summary data for the stochastic CRM model (Natural England, 2022). According to Natural England (2022) nocturnal activity factor for northern gannet is from Furness et al. (2018) and for all other species based on Garthe and Hüppop (2004). According to Natural England (2022) flight speed data are taken from Alerstam (1997) [(1)] except for northern gannet from Pennycuick (1987) [(2)] and Sandwich tern from Fijn and Gyimesi (2018). According to Natural England (2022) body length and wing span data are from Snow and Perrins (1987) [(3)]. Subsequent alterations to these data by JNCC et al. (2024) are shown in bold below the value advised by Natural England (2022).

Species	Flight speed m/s and (SD)	Nocturnal activity factor	Body length (m)	Wing span (m)	Flight type	% of flights upwind
Northern gannet	14.9 (0.0)	0.08 (0.10) **0.14 (0.10)**	0.94 (0.0325)	1.72 (0.0375)	Flapping	50
Black-legged kittiwake	13.1 (0.40)	0.375 (0.0637) **0.40 (0.12)**	0.39 (0.005)	1.08 (0.0625)	Flapping	50
Great black-backed gull	13.7 (1.20)	0.375 (0.0637)	0.71 (0.035)	1.58 (0.0375)	Flapping	50
Lesser black-backed gull	13.1 (1.90)	0.375 (0.0637) **0.30 (0.18)**	0.58 (0.03)	1.42 (0.0375)	Flapping	50
Herring gull	12.8 (1.80)	0.375 (0.0637)	0.6 (0.0225)	1.44 (0.03)	Flapping	50
Common gull	Consult SNCB	See Garthe & Hüppop (2004)	Consult SNCB	Consult SNCB	Flapping	50
Black-headed gull	Consult SNCB	See Garthe & Hüppop (2004)	Consult SNCB	Consult SNCB	Flapping	50
Sandwich tern	10.3 (3.4)	See Garthe & Hüppop (2004)	0.38 (0.005)	1 (0.04)	Flapping	50

[(1)] Listed in Natural England (2022) references as Alerstam et al. (2007).

[(2)] Listed in Natural England (2022) references as Pennycuick (1997).

[(3)] Listed in Natural England (2022) references as Snow and Perrins (1998).

For the sCRM model parameters NatureScot (2023g) also repeat the citation of the three incorrect references as in Natural England (2022) and most of the table is the same as in Natural England (2022) except that northern gannet flight is again classified as gliding rather than flapping, and for nocturnal flight activity the quantitative data that are in Natural England (2022) are replaced by "Consult Nature Scot" except for northern gannet where the data are the same as in Natural England (2022).

2.2.4. Flight speed

Flight speed enters the Band model twice; once by determining the relationship between seabird density and flux rate through the rotor-swept area, and once by determining the probability of a bird passing through the rotor-swept area being hit by the rotating turbine blade. Higher flight speed increases number of predicted collisions by increasing flux rate but decreases the probability of an individual being hit while transiting the rotor-swept area. As a consequence, these two effects tend to partly cancel each other. However, the effect of flight speed estimate on flux rate has a stronger influence on predicted numbers of collisions than effects on probability of collision, such that collision numbers will be overestimated if flight speed is overestimated (Masden et al. 2021). Natural England (2022) and NatureScot (2023g) and JNCC et al. (2024) require use of generic flight speed data (Table 3 and Table 4), although JNCC et al. (2024) indicate that site or region-specific evidence might be used in consultation with the relevant SNCB if such data were appropriate. NatureScot (2023g) points out that any change in flight speed used in CRM will require a revision of the avoidance rate correction used, so that there is a strong incentive to retain the generic flight speed data. There are some recent studies that provide new flight speed estimates for seabirds that are a focus of CRM. Those include studies using tracking radar (e.g. Skov et al., 2018; Tjørnløv et al., 2023), and studies that deployed GPS tags on breeding adult seabirds (e.g. Amélineau et al., 2014; Collins et al., 2020; Fijn et al., 2022).

Tags have been deployed on seabirds in order to estimate flight speed. However, tags can affect seabird behaviour, including their flight speed and/or energy costs (Vandenabeele et al., 2012; Heggøy, 2013; Heggøy et al., 2015; Bodey et al., 2017; Symons and Diamond, 2019; Evans et al., 2020). Aerodynamic theory of bird flight and empirical measurement shows that birds, including seabirds, tend to fly at air speeds between the minimum power speed and the maximum range speed, the latter being the speed that allows the greatest distance of travel per unit of energy expenditure (Pennycuick, 1987, 1997; Pennycuick et al., 2012, 2013). In particular, the maximum range speed is strongly influenced by drag and by body mass (Pennycuick et al., 2012, 2013). An increase in drag and mass caused by a tag is likely to reduce flight speed because this increases energy cost of flight (Vandenabeele et al., 2012). A controlled experimental study of birds equipped with tags flying in a wind tunnel demonstrated experimentally that the drag coefficient of a bird carrying a back-mounted tag may increase by nearly 50% relative to the drag of an untagged bird (Pennycuick et al., 2012). This means that data from GPS-tagged birds must be evaluated with great care, where the tag may increase drag (Pennycuick et al., 2012, 2013) or where the tag represents a relatively high percentage of body mass (Vandenabeele et al., 2012), although effects on behaviour can arise even with tags weighing less than 1.5% of body mass (Bodey et al. 2017; Symons and Diamond, 2019). Nevertheless, recent studies seem to provide the potential to improve on the understanding of seabird flight speeds and how those vary with environmental conditions and behaviour. A study of tag effects on flight speed of seabirds may help to resolve this difficulty.

2.2.5. Nocturnal flight activity

Nocturnal flight activity factors listed in SNCB guidance are derived from Garthe and Hüppop (2004). However, in the SNCB guidance these are expressed as a percentage of daytime flight activity. Garthe and Hüppop (2004) classified seabird species into a 5-point scale from 1 (low) to 5 (high). They specifically stated that at the time of their review "nocturnal flight activity could not be quantified by real data and was thus classified subjectively from 1 (hardly any flight activity at night) to 5 (much flight activity at night)", modulated by expert opinion. Garthe and Hüppop (2004) did not define these scores as covering the full range of percentage of daytime flight activity, nor did they specify whether the categories were of equal size. It is explicit that this was not the case for some metrics. For example, adult survival rate score of 1 was for survival rates from 0 to 0.75, score of 4 was for survival from 0.85 to 0.9 whereas score of 5 was for survival from 0.9 to 1. Therefore, the conversion by SNCBs of the five point scale of Garthe and Hüppop (2004) of nocturnal flight activity into percentages of daytime flight activity with the assumption that a score of 3 for gull species represents 37.5% (even with a large standard deviation) of daytime flight activity (Table 4) is likely to be highly inaccurate. This is further indicated by the fact that Garthe and Hüppop (2004) gave a score of 2 for northern gannet nocturnal flight activity while data from GPS tags deployed on breeding northern gannets indicated nocturnal flight activity representing 8% that of daytime (Furness et al., 2018). With numerous studies that have deployed GPS tags on breeding seabirds, there should be scope to derive evidence-based measures of nocturnal flight activity as has been done for northern gannet (with the caveat of needing to take care in data analysis because of possible tag effects on the data). Use of percentages that have apparently been derived in an arbitrary fashion from the 5-point scores of subjective scoring by Garthe and Hüppop (2004) represents a weak approach. However, it seems likely that nocturnal flight activity correction in the Band model is probably not one of the strongest influences on the uncertainty or bias of the output from the model.

2.2.6. Use of collision numbers in assessments

Guidance (Parker et al. 2022c; NatureScot 2023g) advises that the resulting estimated number of collisions is then apportioned to relevant SPA populations where the species is a breeding feature, and the resulting estimated collision mortality to the SPA population is input, summed with any estimated displacement impact to the same SPA population, into a density-independent PVA.

2.3. Displacement impact assessment

SNCB guidance (Parker et al., 2022c; NatureScot, 2023h) is to (1) use the seasonal mean of peak monthly counts of birds that may be subject to displacement (determined from digital aerial survey baseline characterisation data), (2) multiply that number by the displacement rate (% displaced), (3) multiply that output by the % mortality of birds subject to displacement, (4) apportion that total to relevant SPA populations where the species is a breeding feature, and (5) input the resulting displacement mortality to the SPA population, summed with any estimated collision mortality impact to the same SPA population, into a density-independent PVA. This calculation has uncertainty and precaution at each of these five stages, which will tend to compound over the computation process. NatureScot (2023h) state "displacement and barrier effects should be assessed using the Joint SNCB Interim Advice on the Treatment of Displacement for Red-throated Diver (SNCB, 2022b) matrix methods and the SeabORD tool (Searle et al. 2018) as appropriate to species and season. For black-legged kittiwake, common guillemot, razorbill or Atlantic puffin, during the chick-rearing period only, it is advised to use the SeabORD tool to estimate the impact of displacement (Searle et al. 2018; NatureScot, 2023h). For all other seasonal periods and any other species the impact must be assessed using a matrix approach (SNCB, 2017, 2022a,b; NatureScot, 2023h).

2.3.1. Seasonal mean of peak monthly counts

Baseline characterisation involves monthly digital aerial surveys flying transects over the proposed wind farm site plus appropriate buffer over a period of two years (Parker et al., 2022a-c; NatureScot, 2023b). From these surveys the number (and density) of each seabird species on the water, in flight, and in total, can be calculated for each month. For displacement impact assessments, guidance is to focus on priority species: divers, sea ducks, common guillemot, razorbill, Atlantic puffin and northern gannet (SNCB 2022a,b updates to SNCB 2017), noting that NatureScot (2023h) also require an assessment of kittiwake displacement whereas Natural England do not. Guidance is then to define seasonal periods (breeding season, non-breeding season, or more than these two where relevant) and to take the mean of the highest (i.e. peak) counts in each of the survey years for that defined season as the metric for assessing how many birds may be subject to a displacement impact. Because most of the priority seabird species tend to show aggregated behaviour, either congregating where foraging opportunities are good, or commuting in flocks, or resting on the sea in flocks, the seasonal mean peak count tends to be much higher than the mean or median of the monthly counts for the defined season because aggregations may happen to be present on one occasion but not on others. Furthermore, variability of the defined mean peak count is much higher (because it is based on the average of two extreme counts) than variability around the median or mean of monthly counts which are based on a larger number of individual counts. However, it is also important to note that there is no measure of turnover of individual birds in this assessment. Turnover will mean that larger numbers of birds are exposed to potential displacement effects than the number present on any single survey count. How that turnover should be taken into account in analyses of at-sea survey data is unclear.

2.3.2. Displacement rate

Using a matrix approach requires input of a value for the percentage of birds that are displaced from an offshore wind farm area and buffer (the buffer normally being defined as 2 km around the wind farm, but 10 km for red-throated diver (under most but not all circumstances) and 4 km for other species of diver and for sea ducks). SNCB (2017, 2022a) states that they "acknowledge that in reality there is likely to be a gradient in the reduction in density with increasing distance from OWF site, but the evidence regarding the slope of this gradient is limited".

NatureScot (2023h) advises the use of 60% displacement of auks (common guillemot, razorbill, Atlantic puffin), 70% displacement of northern gannet, and 30% displacement of black-legged kittiwake (although in the same document black-legged kittiwake is not listed as a "priority species" for displacement assessment). However, the guidance provides no evidence or references to support the validity of those percentages or for inclusion of black-legged kittiwake. SNCB (2017, 2022a) states "it is unlikely that cormorant and gull species will need to be routinely assessed for displacement, as a number of empirical studies have demonstrated these species can also be attracted as well as display no noticeable reaction to the presence of OWFs (e.g. Leopold et al., 2013; Vanermen et al., 2015; Petersen et al., 2006; Mendel et al., 2014)". It is not explicit that SNCB (2017, 2022a) recognise black-legged kittiwake to be a species of gull, and therefore not normally requiring displacement assessment, but that appears to be implicit. SNCB (2017, 2022a) states that assessments of displacement levels should "use published indices of disturbance (e.g. Furness et al., 2013) to assign a range of displacement levels for each species individually", and "for auk species the SNCBs would typically advise a displacement level of 30-70% (Guillemot and Razorbill have a 'Disturbance Susceptibility' score of 3)". There is no explanation as to why a 'Disturbance Susceptibility' score of 3 (taken from Furness et al. (2013) equates to 30-70% displacement. This assumption appears to be without empirical evidence to support it, while in Furness et al. (2013) the disturbance susceptibility scores that are on a 5 point scale based on expert opinion rather than quantitative data are not considered to equate to particular bands of percentages of displacement. The use of this approach within SNCB guidance is problematic when there is empirical evidence of the extent of displacement in the literature (e.g. Dierschke et al., 2016; Pavat et al., 2023; Trinder et al., 2024).

2.3.3. % mortality of birds subject to displacement

Using a matrix approach requires input of a value for the percentage of birds that die as a consequence of being displaced. NatureScot (2023h) advises the use of 3% or 5% mortality of displaced auks (common guillemot, razorbill, Atlantic puffin) in the breeding season, 1% or 3% mortality in the non-breeding season; 1% or 3% mortality of displaced northern gannets in the breeding season or non-breeding season; and 1% or 3% mortality of displaced black-legged kittiwakes in the breeding season or non-breeding season (although in the same document black-legged kittiwake is not listed as a "priority species" for displacement assessment). However, the guidance provides no evidence or references to support the validity of those percentages, other than to suggest that the values have been selected to "more closely match SeabORD outputs" (NatureScot 2023h). It should be noted that (a) displacement mortality rates estimates generated by SeabORD are not directly comparable to displacement mortality rates used as inputs to the Displacement Matrix, and (b) outputs from SeabORD depend critically on the chosen input value for proportion of birds displaced and therefore precautionary estimates of displacement rate fed into SeabORD will most likely overestimate the impact output estimate from SeabORD. SNCB (2017, 2022a) states that assessments of displacement mortality should "use published indices of habitat flexibility (e.g. Furness et al., 2013), other empirical evidence if available, and discussions with SNCBs; to agree appropriate levels of likely adult mortality associated with particular displacement levels, for each species individually (acknowledging data very limited at this time)".

SNCB (2022a) notes the consequence of displacement as "Individual fitness may be impacted due to immediate increases in energy expenditure and/or reduced energy intake as a result of relocating to other foraging grounds and experiencing increased competition (an indirect impact resulting from localised habitat loss). Individual fitness may thus be impacted over longer time frames due to negatively affected energy budgets if birds have to relocate to alternative habitat. This impact might operate through increased intra-/inter-specific competition due to a higher density of individuals competing for the same resources and/or through a lower quality/quantity of prey". Such density-dependent processes are widespread in ecosystems and have been studied in particular detail in estuarine shorebirds where prey depletion over the winter results in increased competition and as a result the loss of estuarine foraging habitat can cause decreases in the carrying capacity of estuarine birds. Similar processes might occur in the marine environment but there are important differences which seem to make this unlikely. Firstly the impact of seabirds away from breeding colonies on their prey stock biomass is very small (estimated across five ecosystems to average about 1% of the primary forage fish being consumed by all seabird species (Saraux et al., 2020)) and most predation of forage fish is by predatory fish rather than by seabirds (Barrett et al., 2002), even when prey stocks are particularly low so that predation impact by seabirds increases (Saraux et al., 2020) so prey depletion is very unlikely to be caused by an increase in seabird density except in the case of prey depletion close to large breeding colonies during the breeding season (Birt et al., 1987; Weber et al., 2021). This differs from estuarine systems where shorebirds cause strong depletion of prey biomass through winter. Secondly, forage fish stock biomass varies enormously from year to year and over decadal time scales, whereas seabird population sizes change much more slowly. This means that changes in prey abundance/density determine resource availability to seabirds rather than changes in seabird abundance/density and therefore prey depletion by seabirds is likely to be a much smaller influence than changes in overall prey stock biomass. For example, sandeel total stock biomass in the ICES sandeel area 1r (Dogger Bank and surroundings) was 1,811,450 tonnes in 2010 but fell to only 15% of that level (264,658 tonnes) in 2012 (ICES, 2023). North Sea sprat SSB was 260,337 tonnes in 2020 but only about half as abundant (131,539 tonnes) in 2021 (ICES, 2023). Such fluctuations are typical of forage fish species because their survival is very low and recruitment can be very high but is highly variable from year to year. These fluctuations in prey abundance will determine food availability to seabirds whereas changes in seabird density at sea caused by displacement from offshore wind farms will have an influence that is extremely small by comparison at least over large spatial scales. Thirdly, much of the emphasis in displacement impact assessment is on the impact to SPA breeding populations. Breeding adult seabirds have gained years of experience and are more competitive than juveniles or immatures. That is evident from the much higher survival rates of adults compared to juveniles and young immature age classes, but also from studies of age-specific foraging success in seabirds (e.g. Greig et al., 1983). Any increase in competition for food resources will be likely to affect the least competitive component of the population most, i.e. juveniles and immatures, rather than adults. These aspects of seabird ecology suggest that in years when food availability is good, the effect of any displacement of birds from offshore wind farms is likely to have less impact on survival of adults, whereas in years when birds are stressed by a low abundance of prey, the small additional impact of displacement may increase mortality, especially of younger, less competitive age classes, but the main driver of mortality resulting from competition for food is likely to be the huge decreases in forage fish abundance that occur in some years, driven either by environmental factors or fishing pressure (Carroll et al., 2017; Lindegren et al., 2018).

2.4. Apportioning to SPA populations

NatureScot (2023h) state that non-breeding season "predicted mortality impacts should be considered in the context of the regional populations as defined by the Biologically Defined Minimum Population Scales (BDMPS) (Furness, 2015)" and this point is also made in SNCB (2017, 2022a). It is important to note that the BDMPS numbers were derived for populations as they were thought to be in the early 2000s. Furness (2015) specifically states "This report will soon become out of date. It will be necessary to update seabird population estimates and seabird movement patterns to take account of new data and to take account of changes in environmental conditions". Seabird breeding numbers in Britain and Ireland have changed considerably between Seabird 2000 (1998-2002) and Seabirds Count (2015-2021) (Burnell et al., 2023). Furthermore, populations have in some cases altered much more in particular regions, changing the spatial distributions as well as abundance. For example, black-legged kittiwake numbers of Apparently Occupied Nests in England remained relatively stable from 1998-2002 to 2015-2021, while numbers in Scotland fell by 57% in the same period, such that England held 18% of the population in 1998-2002 but held 30% in 2015-2021 (Burnell et al., 2023). Common guillemot numbers decreased by 31% in Scotland between 1998-2002 and 2015-2021, whereas numbers in east England increased by 106%, in Wales increased by 76% and in south-west England increased by 264% (Burnell et al., 2023). As a consequence, England held 6% of common guillemots in Britain and Ireland in 1998-2002 but this increased to 13% in 2015-2021 (Burnell et al., 2023). Atlantic puffin numbers in Scotland fell by 32% between 1998-2002 and 2015-2021, whereas numbers in England increased by 50% and in Wales increased by 104% (Burnell et al., 2023). Migrations and wintering areas of seabird populations may also have shifted over decadal time scales in response to changes in environmental drivers. For example, in the early 1980s common guillemots from colonies in Shetland tended to remain throughout the non-breeding period around Shetland where there was a large stock of sandeels during that period, returning to breeding colonies as early as October or November (Pennington et al., 2004). After the Shetland sandeel stock collapsed in the late 1980s and sprats moved south so were no longer present at Shetland, a large proportion of the Shetland common guillemot population began to move across the North Sea from Shetland to Scandinavia after the breeding season, spending the autumn and winter in Norwegian and Danish waters, with very few remaining overwinter in the Shetland area and colony reoccupation not tending to occur until January to March (Pennington et al., 2004). Using the BDMPS data from the early 2000s is highly likely to be inappropriate if evaluating impacts on populations in the 2020s.

When apportioning breeding season impacts to particular SPA populations, three methods could be used to split impacts between the breeding adult component of the population (as required for assessment of impact on an SPA population) and immatures. Firstly, the impact could be split between age classes based on survey data of proportions of different age classes. Secondly, the impact could be split based on population model calculations of the proportions of the population in each age category (which can take account of features such as rate of growth of the population). Thirdly, the impact could be allocated entirely to the breeding adult category as a precautionary approach where the proportions of age classes are uncertain. Recent SNCB practice in case work seems to favour the third of these approaches (Mark Trinder, pers. comm.), which is certainly the most precautionary.

Use of age classes in survey data would allow accurate identification of the proportion of black-legged kittiwakes less than one year old in the survey area, as the juvenile plumage of black-legged kittiwake is easy to identify. However, kittiwakes generally do not breed until four years old, and plumages of birds aged one to four years old are very difficult to distinguish from adult plumage unless the birds is viewed from a close distance. From digital aerial surveys the resolution of images is inadequate to discriminate between kittiwakes aged one to four years old and birds of adult age, so no meaningful apportioning of impact between adult and those immature age classes can be made from survey data. For auks this problem is even more severe, as it is not possible to determine the age of auks once juveniles have reached full body size unless the bird can be inspected very closely indeed. For large gulls and northern gannets immature plumages can be distinguished but it is likely that older immature birds will be difficult to distinguish from adult plumage. Using population models would be straightforward, but there is uncertainty about the geographical distributions of immature seabirds and so it would be unwise to assume that all age classes are represented in a defined geographical area in proportion to their abundance in the global population. This leaves SNCBs advocating the use of a precautionary (but inaccurate) apportioning of all of the assessed impact onto the breeding adult component of the population. That is further compounded by then assuming in PVA that additional mortality that has been allocated 100% onto adults is also allocated at that same rate onto immature age classes by the PVA tool (Mark Trinder, pers. comms.).

NatureScot developed an apportioning tool (NatureScot (SNH), 2018) based on the assumption that colony-specific seabird density at sea during the breeding season declined with the square of distance from the colony. For gull species “distance” is typically assumed to be the straight line (Euclidean) distance, on the assumption that birds may fly over either sea or land. “Distance” is assumed for non-gull species to be “distance by sea” (avoiding land), and the tool adjusts for the inverse of the proportion of area within the foraging range that is sea. This approach allows the proportions of birds in a survey area (such as an offshore wind farm (OWF)) originating from different colonies to be estimated based on colony sizes and distance. Whether seabird species overlap in breeding season foraging ranges from neighbouring colonies seems to vary with species. At one extreme, tracking data show that northern gannets remain within largely exclusive colony-based foraging areas, whereas Manx shearwaters from many colonies forage together at hot spots where frontal systems aggregate prey. Gannets appear to prefer to forage individually where possible, whereas Manx shearwaters forage in large flocks, and perhaps gain benefits from the disturbance of prey fish by aggregations of foraging conspecifics. It is less clear where other seabirds fit into that spectrum.

The NatureScot apportioning tool assumes a simple mathematical model for the spatial distribution of birds around breeding colonies, and one limitation of this approach is that it does not account for environmental heterogeneity or the effects of competition. Marine Scotland commissioned the development of an alternative tool (Butler et al., 2020) for apportioning birds to SPAs in the breeding season using colony-specific spatial distributions predicted by habitat association models developed from multi-colony GPS tracking data (Wakefield et al., 2017). The models that underpin this tool attempt to explain the spatial distribution of birds in relation to environmental variables, accessibility (via distance to colony) and competition (Wakefield et al., 2017). The model assumptions are assumed to apply to all colonies, but this is a difficult assumption to validate, given that GPS tracking data are only available for a relatively small proportion of colonies. The Marine Scotland tool was developed for four species (kittiwake, guillemot, razorbill and shag), reflecting the species included in Wakefield et al. (2017), but the ORJIP-funded AppSaS project (Apportioning seabirds seen-at-sea) (Carbon Trust, 2024) extended the apportioning tool, and underlying modelling approach, to include a fifth species (lesser black-backed gull). Wakefield et al. (2017) and the associated apportioning tool (Butler et al., 2020) use colony counts derived from Seabird 2000, and there is an acknowledged need to update these to reflect more recent colony count data.

Geolocator tracking data provide a basis for estimating the spatial distribution of auks in the non-breeding season. Buckingham et al. (2022) analysed data from geolocators deployed on 695 common guillemots and 339 razorbills during three breeding seasons (June-July 2017, 2018 and 2019) at twelve breeding colonies around the north of the UK. The geolocators measured light levels, salt water immersion and sea-surface temperature, from which daily locations were estimated throughout the non-breeding season. Kernel density estimation was then used to estimate monthly colony-specific spatial distributions for each tracked colony. Within Work Package 3 of the ORJIP-funded AppSaS project (Carbon Trust, 2024) this approach was extended to include a quantification of uncertainty (which is important given the relatively high levels of locational uncertainty in geolocator tracking data) and translated into an apportioning tool for auks in the non-breeding season (the ANBS Tool; Jones et al., 2024). The relatively high levels of locational uncertainty created challenges for habitat association modelling, so the apportioning tool instead quantifies spatial distributions for untracked colonies using a simple approach based upon a distance weighted average of the distributions for tracked colonies. Users specify a maximum distance threshold, and the tool uses BDMPS for colonies whose distance to the nearest tracked colony exceeds this threshold – the rationale for this hierarchical approach is to avoid extrapolating relationships based on the geolocator tracking data, which are outside of the range that is deemed ecological feasible (by the user of the tool). Additionally, where the BDMPS method of apportioning is reverted to for colonies that are outside of the distance threshold for using information derived from geolocators, uncertainty is added to the apportioning estimates for those colonies to ensure that full uncertainty from all colonies is propagated through to the final estimates of apportioning.

2.5. PVA modelling

Natural England advises that where predicted impacts (the sum of CRM plus displacement impacts) equate to >1% of baseline mortality (Parker et al., 2022c)) while NatureScot advises that where assessed effects exceed a change to the adult survival/mortality of 0.02 (NatureScot 2023k) the Natural England (NE) stochastic density-independent PVA tool (Searle et al., 2019) available on GitHub should be used, using the most recent population count for the focal SPA breeding population and life history parameters taken from Horswill and Robinson (2015).

A counterfactual approach is recommended, to compare population growth rate and population size after 25 and 50 years (NatureScot, 2023k) or 30 years (Parker et al., 2022c) under the impacted and unimpacted scenarios modelling no density dependence. Although the values of demographic parameters entered into a PVA model will affect the absolute population growth rate predicted by the model, in any density-independent version of the seabird population model the counterfactual values are hardly influenced at all by uncertainty or error in demographic inputs because these are matched between the baseline and impacted scenarios. That is a strength of using counterfactuals as the key outputs.

Parker et al. (2022c) state “Density dependent processes are likely to operate on seabird populations”. However, they recommend use of a density-independent model “where there is limited information on population regulation for the focal population” (which is the case for all seabird populations at all colonies). In relation to use of a density-independent model, Parker et al. (2022c) also state “By definition such a model will always predict that an impacted population will have a slower growth rate and smaller final size than an unimpacted population”. The failure to use biologically more realistic density-dependent models (which are particularly relevant where population size is large and so subject to competition as is the case with almost all SPA populations) represents a major weakness in the assessment process (Merrall et al., 2024). Comparing outputs from a density-independent scenario and a range of more realistic density-dependent scenarios would provide greater insight into likely outcomes. This is especially obvious where PVA is used to assess the gain from compensatory measures, where use of a density-independent model often predicts population growth to entirely implausible numbers at the focal SPA colony. For example, any density-independent model with population growth rate greater than 1.00 will predict that colony size will eventually exceed one million breeding pairs of seabirds, despite the fact that density-dependent processes will certainly limit population size far below such numbers. Density-dependence could result in either “compensation” or “depensation”. The former occurs where, for example, competition for resources increases with population size so sets an upper limit to numbers (“carrying capacity”). This form of density-dependence seems particularly likely at large colonies, which includes most of those designated as SPAs for breeding seabirds as those sites have been selected in large part for their large numbers. Depensation occurs where demographic parameters are adversely affected as the local population declines in numbers. A classic example of that is the tendency for breeding success to be reduced by predation impact more strongly in a declining population, which may contribute further to population decline. Therefore, depensation is especially likely in small populations.

For EIA assessments, Parker et al. (2022c) recommend that impacts are assessed against the BDMPS populations defined in Furness (2015) with PVA carried out “if a project is expected to significantly impact a defined population (e.g. UK colonies)”. As stated earlier in relation to apportioning, those numbers are now about 20 years out of date and it is known that many seabird populations have changed considerably in size over that period, so that comparisons with data from two or three decades ago could be misleading.

3. Sources of uncertainty in parameters

We review information on uncertainty within the parameters used in assessment tools and the current SNCB guidance around the use of this information. For each parameter, we identify the available sources of data that provide information relative to the parameter, the estimates provided by these sources, and quantitative sources of information on the uncertainty and/or variability associated with these estimates (including ranges, standard errors and standard deviations). Where no quantitative information on uncertainty/variability is available we note this.

We also review the qualitative uncertainties associated with the selection of data sources that can inform each parameter, evaluating the extent to which different data sources provide transferrable evidence that can be used to inform the values of parameters used in assessments. Transferability involves judgements around geographical, temporal and environmental similarities between the contexts in which data have been collected and the contexts in which they will be applied. Where relevant, we also focus on structural uncertainties around the models used within assessment tools that can limit the transferability of evidence.

We structure the review around key input parameters associated with the commonly used tools and methods identified in Section 2.

3.1. Monthly aerial survey counts

Baseline survey data were extracted from the planning application papers for Berwick Bank OWF (south-east Scotland) (SSE Renewables, 2022), Hornsea Four OWF (Yorkshire) (Orsted, 2021), Dudgeon and Sheringham Extensions OWF (Norfolk) (Equinor, 2022a, 2022b), Awel y Mor (Wales) (APEM, 2022a). These sites were chosen in order to give a geographical spread of the data, and a range of species and abundances at these different sites, but to include projects that have recently entered into the planning process so reflect current practice in terms of baseline assessments. Counts were extracted separately for all birds in the survey area (because it is those data that inform displacement assessment) and for birds in flight in the survey area (because it is those data that inform CRM).

For Berwick Bank OWF breeding season and non-breeding season (as defined for each species by NatureScot) counts were separately tabulated for all species presented in the report: Arctic tern, common gull, common tern, fulmar, gannet, great skua, guillemot, herring gull, kittiwake, lesser black-backed gull, Manx shearwater, puffin, razorbill, and shag, excluding either season if all counts in that season were zero.

For Hornsea Four OWF breeding season and non-breeding season (as defined for each species by Natural England) counts were separately tabulated for all species presented in the report: fulmar, gannet, great black-backed gull, guillemot, herring gull, kittiwake, lesser black-backed gull, puffin, and razorbill, excluding either season if all counts in that season were zero.

For Dudgeon and Sheringham Extensions OWF breeding season and non-breeding season (as defined for each species by Natural England) counts were separately tabulated for all species presented in the report: Arctic skua, Arctic tern, black-headed gull, common gull, common tern, fulmar, gannet, great black-backed gull, guillemot, herring gull, kittiwake, lesser black-backed gull, Manx shearwater, puffin, razorbill, red-throated diver, and Sandwich tern, excluding either season if all counts in that season were zero.

For Awel y Mor OWF breeding season and non-breeding season (as defined for each species by SNCBs) counts were separately tabulated for all species presented in the report: fulmar, gannet, great black-backed gull, guillemot, herring gull, kittiwake, Manx shearwater, and razorbill, excluding either season if all counts in that season were zero.

For each of these four sites, data were tabulated in an Excel spreadsheet for use in later parts of this project. The data show the variability in the survey data from individual survey to individual survey within each defined season, allowing the distribution of the data sets to be described, and means, medians, variability and mean peak numbers to be computed for each species/site/season combination. These are key data for input into displacement assessments and so allow the effect of variability in baseline survey data on displacement impact assessments to be modelled.

3.2. Avoidance rates

In guidance from SNCBs, avoidance rates are typically calculated for species groups, such as "large gulls" rather than individual species. That makes sense because immature large gulls have very similar plumages so are difficult to identify to species unless observed in great detail at close range. Excluding birds of uncertain species to create data sets based only on birds confidently identified to species level could bias estimates, as avoidance rates of adults (which are more easily identified to species) are likely to be higher than avoidance rates of juveniles or young immatures (which are much more difficult to identify to species). We follow that practice here but are careful to ensure that examples listed in the tables below are statistically independent. For example, data for "herring gull" identified to species are not summed with data for "large gulls" but are presented separately where those categories have been presented in results reported in the literature, or are combined but then only presented once, as a single entry in the tables, to ensure that each row within a table is statistically independent.

Avoidance can be macro-avoidance (birds altering flight path in order to avoid entering a wind farm), or meso-avoidance (birds entering the wind farm but flying between rows of turbines in order to avoid entering a rotor-swept area of collision risk) or micro-avoidance (birds approaching close to a turbine but altering flight path to avoid a rotating turbine blade) (Skov et al., 2018). These three processes together result in an avoidance behaviour response that represents a rate similar to the classic Band model avoidance correction, but empirically derived from bird behaviour observations (Skov et al., 2018). Alternatively, a Band model avoidance correction can be derived from observed numbers of collisions in relation to predicted numbers from the Band model. For offshore wind farms, carcass collection is not a practical monitoring process, but radar/video recording of seabird flight paths provides an alternative method of quantifying numbers of collisions in relation to flux rate and predicted numbers based on that (Skov et al., 2018).

Macro-avoidance and displacement/barrier effects on seabird distribution in relation to OWF are often treated as closely similar, with evidence used interchangeably between these concepts. Clearly in collision risk modelling it is only flying birds that are at risk of collision, whereas in displacement impact assessment the displacement may affect birds on the sea as well as (or differently from) flying birds. Macro-avoidance (and displacement/barrier effects) can be quantified from GPS tracking studies, from radar/camera studies at OWFs, or from analysis of spatial distributions of seabirds in the vicinity of OWFs derived from digital aerial surveys (or in the past from boat-based surveys). Tracking studies may be able to discriminate between behavioural responses of different individual birds (some showing macro-avoidance and others not), whereas analysis of spatial distribution of birds cannot identify the role of individual differences in response but only the net effect on the population sample as a whole.

Where data have been presented, we have noted whether or not variability around the estimate (or uncertainty) has also been quantified. Tabulated data provide the measure of variability where that is possible, either as a standard deviation or 95% confidence interval if reported as such, or as differing individual measurements if the paper contains multiple independent measures. However, in many studies the variability is not quantified, and in such cases we note n/a in the table because no quantitative measure of variability around the published estimate is available.

3.2.1. Northern gannet

3.2.1.1. Northern gannet macro-avoidance

Table 5. Macro-avoidance rates shown by northern gannets approaching offshore wind farms.

Macro-avoidance rate	Variability, if assessed	Offshore wind farm	Reference
>0.5	0 (n=1), <0.5% (n=2) >0.5% (n=7)	Robin Rigg, Bligh Bank, Thorntonbank, Prinses Amalia, Egmond aan Zee, Horns Rev 1, Horns Rev 2 (all >50%); North Hoyle, Alpha Ventus (<50%); Thanet (0%)	Dierschke et al. 2016
0.70	<0.6% (n=7), 0.6-0.8% (n=7), >0.8% (n=5)	19 studies	APEM, 2022b
0.64	n/a	Egmond aan Zee	Krijgsveld et al., 2011
0.86	n/a	Horns Rev 2	Skov et al., 2012
1.00	n/a	Alpha Ventus	Mendel et al., 2014
0.9502	n/a	Greater Gabbard	Rehfisch et al., 2014
1.00	n/a	Robin Rigg	Nelson et al., 2015
0.99	n/a	Thorntonbank	Vanermen et al., 2016
0.82	n/a	Bligh Bank	Vanermen et al., 2016
0.617	95% CI 0.259 to 1.00	Lincs	Webb et al., 2016
0.797	SD 0.026 to 0.153 (1)	Thanet	Skov et al., 2018
0.21 (3)	95%CI 0.11 to 0.30	Various in German Bight in 2015	Peschko et al., 2021
0.37 (3)	95%CI 0.28 to 0.45	Various in German Bight in 2016	Peschko et al., 2021
0.79	n/a	Alpha Ventus	Welcker and Nehls, 2016
0.67	n/a	Beatrice	Trinder et al., 2024
0.74 (2)	n/a	Luchterduinen	Heinänen and Skov, 2018
0.89 (2)	n/a	Prinses Amalia	Heinänen and Skov, 2018
0.90 (2)	n/a	Egmond aan Zee	Heinänen and Skov, 2018
0.37	n/a	Luchterduinen	Leemans et al., 2022
0.75	n/a	Belgian North Sea	Vanermen et al., 2023

(1) Skov et al. (2018) calculate variability in the track data as standard deviation 0.026 but uncertainty taking account of other factors totalling SD of 0.153

(2) At all three sites macro-avoidance was reported to be zero in the buffer around the OWF

(3) This study measured macro-avoidance up to 15 km from the OWF boundary so may underestimate macro-avoidance of the OWF footprint

Based on an analysis of the underlying datasets for some of the data in Table 5 but with data from the first two rows (Dierschke et al., (2016); APEM 2022b) and last five rows excluded, Pavat et al. (2023) estimated macro-avoidance by northern gannet to average 0.8564 (95% CI 0.5349 to 0.9736). Interestingly, Peschko et al. (2021) noted from tracking breeding adults that whereas 89% of tracked birds avoided OWFs, the other 11% "frequently entered them when foraging or commuting" suggesting strong individual variation in macro-avoidance behaviour and possible habituation by some individuals. Pavat et al. (2023) incorrectly reported Peschko et al. (2021) as indicating 89% avoidance. That was incorrect based on the data in Peschko et al. (2021) because it failed to take account of the observed fact that a minority of the tracked birds were attracted to OWFs. Whereas 89% of gannets avoided OWFs, 11% apparently were attracted to OWFs. Spatial analysis would therefore show less than 89% macro-avoidance because of the 11% that increased their use of OWF areas. Modelling by Peschko et al. (2021) indicated a reduced average use by tracked northern gannets of OWF habitat relative to habitat away from OWFs of 21% in 2015 (CI 11% to 30%) and 37% in 2016 (CI 28% to 45%). However, studies not included in Pavat et al. (2023) generally also reported high macro-avoidance (as in Heinänen and Skov 2018 but see also Leemans et al. (2022)).

The evidence from tracking of individual differences in behavioural responses of northern gannets to OWFs shows the value of tracking as a tool to investigate this question.

3.2.1.2. Gannet meso-avoidance

Table 6. Published estimates of northern gannet meso-avoidance

Empirical meso-avoidance rate	Birds in meso-zone	Offshore wind farm	Reference
1.00 (SD 0.0)	781	Aberdeen	Tjørnløv et al., 2023
0.9205 (SD 0.137)	1485	Thanet	Skov et al., 2018

3.2.1.3. Gannet micro-avoidance

Table 7. Published estimates of northern gannet micro-avoidance

Empirical meso-avoidance rate	Birds in micro-zone	Offshore wind farm	Reference
1.00 (SD 0.0)	10	Aberdeen	Tjørnløv et al., 2023
1.00 (SD 0.0)	33	Thanet	Skov et al., 2018

3.2.1.4. Northern gannet avoidance rate correction for Band model

Table 8. Published estimates of northern gannet within-wind farm avoidance rate corrections. Data in bold are from offshore wind farms. Data in italics are not for gannet but are suggested applications that could be used as proxy values for that species.

Empirical total avoidance rate	Number of collisions	Number approaches observed	Offshore wind farm	Reference
1.00 (SD 0.0)	**0**	**781**	**Aberdeen**	**Tjørnløv et al., 2023**
0.999 (SD 0.003)	**0**	**1485**	**Thanet**	**Skov et al., 2018**
0.9956 (SD 0.0004) (1)			Terrestrial sites	Cook et al., 2014

Empirical total avoidance rate	Number of collisions	Number approaches observed	Offshore wind farm	Reference
0.9893 (SD 0.0008) [(2)]			Terrestrial sites	Cook et al., 2014
0.993 (SD 0.0003) [(3)]				Natural England, 2022
Ozsanlav-Harris et al., 2023 "all gulls" rate (0.993 (SD 0.0003)) with reduced flux rate based on macroavoidance as specified by local SNCB				JNCC et al., 2024

[(1)] Based on "large gull" collisions including immatures as well as adults

[(2)] Based on "all gulls" collisions which is predominantly for small gulls rather than large gulls (to be more precautionary, but no reason is given why gannets should be considered more like small gulls than like large gulls, which seems highly improbable)

[(3)] Based on "all gulls" rate derived by Ozsanlev-Harris et al., 2023 from mostly the same data set as used by Cook et al., 2014 and to be used after reducing flux rate by 70% or 65-85%.

3.2.2. Black-legged kittiwake

3.2.2.1. Black-legged kittiwake macro-avoidance

Dierschke et al. (2016) noted that kittiwake numbers were strongly reduced (>50%) post-construction at two sites (Thorntonbank and Horns Rev 2), slightly reduced (<50%) at two sites (Prinses Amalia and Alpha Ventus), unaltered at five sites (North Hoyle, Thanet, Bligh Bank, Egmond aan Zee, Horns Rev 1), slightly increased (<50%) at one site (Gunfleet Sands) and greatly increased (>50%) at one site (Robin Rigg). At Alpha Ventus, kittiwake numbers were significantly higher at one side of the wind farm than within the wind farm, but numbers on the other three sides were not significantly different from within the wind farm (Welcker and Nehls, 2016). From tracking data, Skov et al. (2018) estimated a macro-avoidance of 0.566 (SD 0.169) for black-legged kittiwake at Thanet OWF. Leemans et al. (2022) reported no macro-avoidance of Luchterduinen by black-legged kittiwakes from visual observations of numbers inside and outside that wind farm. Pollock et al. (2024) found a very slight attraction of tracked kittiwakes to offshore wind farms in the North Sea. Compared to numbers outside wind farm areas, Peschko et al. (2020a) found that the relative density of black-legged kittiwakes was 45% lower inside offshore wind farms in the German Bight in the breeding season but was not significantly different during the non-breeding season.

3.2.2.2. Black-legged kittiwake meso-avoidance

Table 9. Published estimates of black-legged kittiwake meso-avoidance

Empirical meso-avoidance rate	Birds in meso-zone	Offshore wind farm	Reference
0.31 (no SD reported)	2178	Aberdeen	Tjørnløv et al., 2023
0.916 (SD 0.0339)	205	Thanet	Skov et al., 2018

3.2.2.3. Black-legged kittiwake micro-avoidance

Table 10. Published estimates of black-legged kittiwake micro-avoidance

Empirical meso-avoidance rate	Birds in micro-zone	Offshore wind farm	Reference
1.0 (SD 0.0)	28	Aberdeen	Tjørnløv et al., 2023
0.95 (SD 0.0128)	299	Thanet	Skov et al., 2018 (1)

(1) Based on "all seabirds" data

3.2.2.4. Black-legged kittiwake avoidance rate correction for Band model

Table 11. Published estimates of black-legged kittiwake within-wind farm avoidance rate corrections. Data in bold are from offshore wind farms. Data in italics are not for black-legged kittiwake but are suggested applications that could be used as proxy values for that species.

Empirical total avoidance rate	Number of collisions	Number approaches observed	Offshore wind farm	Reference
1.00 (SD 0.0)	**0**	**2178**	**Aberdeen**	**Tjørnløv et al., 2023**
0.998 (SD 0.006)	**1**	**Not reported**	**Thanet**	**Skov et al., 2018**
0.9979 (SD 0.0013)				Ozsanlav-Harris et al., 2023
0.9921 (SD 0.0015) (1)			Terrestrial sites	Cook et al., 2014
0.989 (SD 0.002) (1)			Terrestrial sites	NatureScot, 2023g
0.993 (SD 0.0003) (2)			Terrestrial sites	Natural England, 2022
"all gulls" rate from Ozsanlev-Harris et al., 2023 (*0.993 (SD 0.0003)*)				JNCC et al., 2024

(1) Based on "small gull" collisions at terrestrial wind farms

(2) Based on all gulls rate in Ozsanlev-Harris et al., 2023

3.2.3. Great black-backed gull

3.2.3.1. Great black-backed gull macro-avoidance

Dierschke et al. (2016) noted that great black-backed gull numbers were slightly reduced (<50%) post-construction at one site (Gunfleet Sands), unaltered at five sites (Thanet, Prinses Amalia, Egmond aan Zee, Horns Rev 2and Nysted), slightly increased (<50%) at one site (Horns Rev 1) and greatly increased (>50%) at four sites (Robin Rigg, Bligh Bank, Thorntonbank, and Alpha Ventus). The authors concluded that although there was some variation among sites, the tendency was for great black-backed gulls to be attracted to offshore wind farms, and so they inferred that this evidence implied that macro-avoidance must be very low or non-existent. However, from tracking data, Skov et al. (2018) estimated a macro-avoidance of 0.464 (SD 0.196) for great black-backed gull at Thanet OWF. Leemans et al. (2022) reported no macro-avoidance of Luchterduinen by great black-backed gulls from visual observations of numbers inside and outside that wind farm. Vanermen et al. (2023) reported that great black-backed gulls were strongly attracted to offshore wind farm turbines in the Belgian North Sea. Tracking studies may resolve whether this species also shows strong variation in individual behavioural responses as shown by northern gannets (Peschko et al. 2021).

3.2.3.2. Great black-backed gull meso-avoidance

Table 12. Published estimates of great black-backed gull meso-avoidance

Empirical meso-avoidance rate	Birds in meso-zone	Offshore wind farm	Reference
0.71 (no SD reported)	321	Aberdeen	Tjørnløv et al., 2023 (1)
0.8423 (SD 0.0394)	294	Thanet	Skov et al., 2018
0.8937 (SD 0.0174)	1073	Thanet	Skov et al., 2018 (1)

(1) Based on combined video data for great black-backed gull and lesser black-backed gull

3.2.3.3. Great black-backed gull micro-avoidance

Table 13. Published estimates of great black-backed gull micro-avoidance

Empirical meso-avoidance rate	Birds in micro-zone	Offshore wind farm	Reference
0.963 (no SD reported)	54	Aberdeen	Tjørnløv et al., 2023 (1)
0.95 (SD 0.0128)	299	Thanet	Skov et al., 2018 (2)

(1) Based on "large gull" category not split into individual species

(2) Based on "all seabirds" data

3.2.3.4. Great black-backed gull avoidance rate correction for Band model

Table 14. Published estimates of great black-backed gull within-wind farm avoidance rate corrections. Data in bold are from offshore wind farms. Data in italics are not for great black-backed gull but are suggested applications that could be used as proxy values for that species.

Empirical total avoidance rate	Number of collisions	Number approaches observed	Offshore wind farm	Reference
1.00 (SD 0.0)	**0**	**321**	**Aberdeen**	**Tjørnløv et al., 2023 (1)**
1.00 (SD 0.0)	**0**	**6012**	**Aberdeen**	**Tjørnløv et al., 2023 (2)**
0.996 (SD 0.011)	**Not certain**	**Not reported**	**Thanet**	**Skov et al., 2018**
0.9991 (SD 0.0002)				Ozsanlev-Harris et al., 2023
0.9956 (SD 0.0004) (3)			terrestrial sites	Cook et al., 2014
0.995 (SD 0.001)			Based on Cook et al., 2014 large gulls	NatureScot, 2023g

0.994 (SD 0.0004)			Based on large gulls rate in Ozsanlev-Harris et al., 2023	Natural England, 2022 and JNCC et al., 2024

(1) Based on observed birds identified to "Great and lesser black-backed gull" group so probably mostly adults rather than immatures

(2) Based on observed birds identified to "large gull" group but excluding "Great and lesser black-backed gull" group so statistically independent of that result but probably including many immature birds

(3) Based on "large gulls" including immatures and other species

3.2.4. Herring gull

3.2.4.1. Herring gull macro-avoidance

Dierschke et al. (2016) noted that herring gull numbers were unaltered post-construction at nine sites (North Hoyle, Thanet, Thorntonbank, Prinses Amalea, Egmond aan Zee, Alpha Ventus, Horns Rev 1, Nysted, Lillgrund), and greatly increased (>50%) at three sites (Robin Rigg, Gunfleet Sands, Bligh Bank). They concluded that although there was some variation among sites, the tendency was for herring gulls to be slightly attracted to offshore wind farms, and these authors suggested on that basis that macro-avoidance must be very low or non-existent. However, from tracking data, Skov et al. (2018) estimated a macro-avoidance of 0.422 (SD 0.199) for herring gull at Thanet OWF. Leemans et al. (2022) reported no macro-avoidance of Luchterduinen by herring gulls from visual observations of numbers inside and outside that wind farm. Vanermen et al. (2023) reported that herring gulls were slightly attracted to offshore wind farm turbines in the Belgian North Sea. Tracking studies may resolve whether this species also shows strong variation in individual behavioural responses as shown by northern gannets (Peschko et al., 2021).

3.2.4.2. Herring gull meso-avoidance

Table 15. Published estimates of herring gull meso-avoidance

Empirical meso-avoidance rate	Birds in meso-zone	Offshore wind farm	Reference
0.69 (no SD reported)	2624	Aberdeen	Tjørnløv et al., 2023
0.9614 (SD 0.174)	272	Thanet	Skov et al., 2018
0.9134 (SD 0.008)	4183	Thanet	Skov et al., 2018 (1)

(1) Based on observed birds identified to "large gull" group but excluding birds identified to species

3.2.4.3. Herring gull micro-avoidance

Table 16. Published estimates of herring gull micro-avoidance

Empirical meso-avoidance rate	Birds in micro-zone	Offshore wind farm	Reference
0.971 (no SD reported)	68	Aberdeen	Tjørnløv et al., 2023
0.95 (SD 0.0128)	299	Thanet	Skov et al., 2018 (1)

(1) Based on “all seabirds” data

3.2.4.4. Herring gull avoidance rate correction for Band model

Table 17. Published estimates of herring gull within-wind farm avoidance rate corrections. Data in bold are from offshore wind farms. Data in italics are not for herring gull but are suggested applications that could be used as proxy values for that species.

Empirical total avoidance rate	Number of collisions	Number approaches observed	Offshore wind farm	Reference
1.00 (SD 0.0)	**0**	**2624**	**Aberdeen**	**Tjørnløv et al., 2023** (1)
1.00 (SD 0.0)	**0**	**3709**	**Aberdeen**	**Tjørnløv et al., 2023** (2)
0.999 (SD 0.005)	**Not certain**	**Not reported**	**Thanet**	**Skov et al., 2018**
0.9952 (SD 0.0003)				Ozsanlev-Harris et al., 2023
0.9959 (SD 0.0006)			Terrestrial sites	Cook et al., 2014
0.995 (SD 0.001)			Based on Cook et al., 2014 large gulls	NatureScot, (2023g)
0.994 (SD 0.0004)			Based on large gulls rate in Ozsanlev-Harris et al., 2023	Natural England, 2022 and JNCC et al., 2024

(1) Based on observed birds identified to species

(2) Based on observed birds identified to “large gull” group but excluding birds identified to species so statistically independent of that result

3.2.5. Lesser black-backed gull

3.2.5.1. Lesser black-backed gull macro-avoidance

Dierschke et al. (2016) noted that lesser black-backed gull numbers were unaltered post-construction at six sites (Thanet, Gunfleet Sands, Thorntonbank, Prinses Amalia, Egmond aan Zee, and Alpha Ventus), increased but not significantly at four sites (North Hoyle, Kentish Flats, Horns Rev 1, and Nysted) and greatly increased (>50%) at one site (Bligh Bank). They concluded that although there was some variation among sites, the tendency was for lesser black-backed gulls to be slightly attracted to offshore wind farms, and these authors suggested on that basis that macro-avoidance must be very low or non-existent. However, from tracking data, Skov et al. (2018) estimated a macro-avoidance of 0.619 (SD 0.198) for lesser black-backed gull at Thanet OWF. Leemans et al. (2022) reported no macro-avoidance of Luchterduinen by lesser black-backed gulls from visual observations of numbers inside and outside that wind farm. Vanermen et al. (2023) reported that lesser black-backed gull density within offshore wind farms in the Belgian North Sea was only 25% that found in areas outside the wind farms, but it was unclear whether the difference represented macro-avoidance or the absence of fishing activity within the wind farms. Tracking studies may resolve whether this species also shows strong variation in individual behavioural responses as shown by northern gannets (Peschko et al., 2021).

3.2.5.2. Lesser black-backed gull meso-avoidance

Table 18. Published estimates of lesser black-backed gull meso-avoidance

Empirical meso-avoidance rate	Birds in meso-zone	Offshore wind farm	Reference
0.87 (no SD reported)	3388	Aberdeen	Tjørnløv et al., 2023 (1)
0.894 (SD 0.174)	51	Thanet	Skov et al., 2018
0.9134 (SD 0.008)	4183	Thanet	Skov et al., 2018 (1)

(1) Based on "large gull" category not split into individual species

3.2.5.3. Lesser black-backed gull micro-avoidance

Table 19. Published estimates of lesser black-backed gull micro-avoidance

Empirical meso-avoidance rate	Birds in micro-zone	Offshore wind farm	Reference
0.963 (no SD reported)	54	Aberdeen	Tjørnløv et al., 2023 (1)
0.95 (SD 0.0128)	299	Thanet	Skov et al., 2018 (2)

(1) Based on "large gull" category not split into individual species

(2) Based on "all seabirds" data

3.2.5.4. Lesser black-backed gull empirical avoidance rate correction for Band model

Table 20. Published estimates of lesser black-backed gull within-wind farm avoidance rate corrections. Data in bold are from offshore wind farms. Data in italics are not for lesser black-backed gull but are suggested applications that could be used as proxy values for that species.

Empirical total avoidance rate	Number of collisions	Number approaches observed	Offshore wind farm	Reference
1.00 (SD 0.0)	**0**	**321**	**Aberdeen**	**Tjørnløv et al., 2023 (1)**
1.00 (SD 0.0)	**0**	**6012**	**Aberdeen**	**Tjørnløv et al., 2023 (2)**
0.998 (SD 0.006)	**Not certain**	**Not reported**	**Thanet**	**Skov et al., 2018**
0.9954 (SD 0.0003)				Ozsanlev-Harris et al., 2023
0.9982 (SD 0.0005)			Terrestrial sites	Cook et al., 2014
0.995 (SD 0.001)			Based on Cook et al., 2014 large gulls	NatureScot, (2023g)
0.994 (SD 0.0004)			Based on large gulls rate in Ozsanlev-Harris et al., 2023	Natural England, 2022 and JNCC et al., 2024

[1] Based on observed birds identified as "great and lesser black-backed gull"

[2] Based on observed birds identified to "large gull" group but excluding birds identified to species so statistically independent of that result

3.3. Flight heights

Flight height data advised by SNCBs to be used in CRM are from the Corrigendum table of Johnston et al. (2014) and are listed below in Table 21. Variability is indicated by the reported confidence intervals associated with each estimate.

Table 21. Proportions of seabirds estimated by Johnston et al. (2014 corrigendum) to be flying at collision risk height at offshore wind farms (assuming no behavioural change due to presence of turbines) and proportions estimated to be flying through the rotor-swept area assuming either a homogeneous distribution of flight heights within the at risk band, or assuming flight height distribution follows an empirically described spline-fitted relationship.

Species	Proportion flying at risk height [1] and 95% confidence limits	Proportion within rotor-swept area assuming homogeneous distribution	Proportion within rotor-swept area assuming cubic spline-fitted heterogeneous distribution
Red-throated diver	0.062 (0.015; 0.323)	0.049 (0.012; 0.254)	0.028 (0.006; 0.216)
Northern fulmar	0.010 (0.000; 0.092)	0.008 (0.000; 0.072)	0.004 (0.000; 0.044)
Manx shearwater	0.000 (0.000; 0.000)	0.000 (0.000; 0.000)	0.000 (0.000; 0.000)
Northern gannet	0.126 (0.062; 0.200)	0.099 (0.049; 0.157)	0.064 (0.028; 0.112)
Arctic skua	0.026 (0.017; 0.100)	0.020 (0.013; 0.079)	0.010 (0.006; 0.049)
Great skua	0.059 (0.035; 0.179)	0.046 (0.028; 0.141)	0.026 (0.015; 0.097)
Black-legged kittiwake	0.150 (0.117; 0.173)	0.117 (0.029; 0.136)	0.079 (0.058; 0.095)
Black-headed gull	0.139 (0.057; 0.255)	0.109 (0.045; 0.201)	0.072 (0.025; 0.153)
Common gull	0.219 (0.190; 0.301)	0.172 (0.150; 0.236)	0.126 (0.105; 0.186)
Lesser black-backed gull	0.282 (0.203; 0.431)	0.221 (0.159; 0.338)	0.172 (0.114; 0.294)
Herring gull	0.319 (0.252; 0.412)	0.251 (0.198; 0.324)	0.201 (0.149; 0.278)
Great black-backed gull	0.325 (0.285; 0.428)	0.255 (0.224; 0.336)	0.206 (0.175; 0.294)
Sandwich tern	0.070 (0.061; 0.149)	0.055 (0.048; 0.117)	0.032 (0.027; 0.078)
Common tern	0.074 (0.044; 0.099)	0.058 (0.034; 0.077)	0.034 (0.019; 0.048)
Arctic tern	0.040 (0.006; 0.143)	0.032 (0.004; 0.112)	0.017 (0.002; 0.074)
Common guillemot	0.004 (0.000; 0.102)	0.003 (0.000; 0.080)	0.001 (0.000; 0.050)
Razorbill	0.027 (0.000; 0.137)	0.021 (0.000; 0.108)	0.011 (0.000; 0.071)
Atlantic puffin	0.000 (0.000; 0.068)	0.000 (0.000; 0.053)	0.000 (0.000; 0.031)

[1] Assuming that the turbine blades are from 20 to 120 m above sea level

LiDAR measurements of northern gannet flight heights resulted in an estimated median height of 6.1 m with an Interquartile Range (IQR) 3.1 to 7.8 m (Cook et al., 2018). These values are biased against birds flying below 2 m above sea level, so overestimate the proportion of birds at greater altitudes by an uncertain but probably small amount (Cook et al., 2018). Nevertheless, these results suggest much lower flight than estimated from visual observations (Johnston et al., 2014). Deployment of altimeter tags on breeding adult northern gannets at the Bass Rock indicated median flight height of 22 m with an IQR from

10.1 to 40.0 m (Cleasby et al., 2015). Subsequent deployments at the same colony (Lane et al., 2019) gave median flight heights of commuting adults with a headwind as 12.6 m (IQR 3.8 to 29.2 m), foraging adults with a headwind as 27.8 m (IQR 9.4 to 47.3), commuting adults with a tailwind as 25.6 m (IQR 9.6 to 46.1 m), foraging adults with a tailwind as 28.3 m (IQR 7.7 to 48.7 m). It is unclear whether the much higher estimates from altimeter tags reflect error in altimeter tag data, or differences between location, age class or season (Cook et al., 2018). However, Johnston et al. (2023) also found that altimeter tags gave significantly higher flight height estimates than GPS tags deployed on the same breeding lesser black-backed gulls, with the mean difference between altimeters and 10-second GPS data being 11.45 m (range 0.6 to 16.28 m). Median flight height estimates also differed between GPS tags programmed at a rate of 5 minutes (median 13 m) and 10 seconds (median 22 m). There is evidence that variation in flight heights of lesser black-backed gulls may be related to season, year, diel period, behaviour (foraging or commuting), weather conditions and environment (Johnston et al., 2023) and this is likely to be generally the case for seabirds. Flight heights of seabirds at Aberdeen OWF were strongly influenced by wind and turbulence conditions as well as by distance from a turbine (with birds approaching turbines moving either up or down to avoid the rotor-swept area) and whether the bird was commuting or foraging (Tjørnløv et al., 2023).

LiDAR measurements of black-legged kittiwake flight heights resulted in an estimated median height of 4.4 m with an IQR 3.2 to 6.6 m (Cook et al., 2018). These values may slightly overestimate flight height due to difficulties caused by waves interfering with measurement for lowest-flying seabirds (Cook et al., 2018). Furthermore, there was clear evidence of spatial variation in kittiwake flight heights, with birds flying higher towards the coast (Cook et al., 2018).

Borkenhagen et al. (2018) measured flight heights of seabirds using laser rangefinders and also compared data from rangefinder and from GPS tag deployment on lesser black-backed gulls. Their measurements with GPS tags suggested that 24.5% of flight was between 20 and 120 m asl and so at collision risk height, whereas laser rangefinder measurements indicated 53.7% to be at collision risk height (predominantly at the bottom of that range of heights). However, Borkenhagen et al. (2018) emphasised that the laser rangefinder method tends to exclude birds flying close to the sea surface so provides an overestimate of risk. Corman and Garthe (2014) noted that GPS tags showed 89% of recorded fixes of lesser black-backed gulls flying over the sea to be below 20 m asl.

3.4. Flight speed

Flight speeds can be measured as either ground speeds or air speeds, the latter requiring the wind speed and direction to be taken into account during measurement. Seabird ground speeds tend to be faster with a tailwind but airspeeds tend to decrease with a tailwind and/or increase in a headwind (e.g.Collins et al., 2020). Airspeed of European shags increased in headwinds, but was otherwise consistent (Kogure et al., 2016). It is the ground speed that is relevant to modelling collision risk in the Band model, not the airspeed.

Flight speeds may also change as seabirds approach a turbine as part of the behavioural response to collision risk, with flight speed slowing closer than about 200 to 100 m to the rotating blades (Tjørnløv et al., 2023). Very few studies have reported seabird flight speeds at operating OWFs and so application of flight speeds recorded in other contexts may not be directly appropriate for CRM use.

Flight speeds advised by Natural England (2022) and NatureScot (2023g) to use in sCRM are listed in Table 22. Variability is indicated by standard deviation metrics associated with each estimate. However, the SNCB guidance to use a standard deviation of zero for gannet flight speed appears to be an error, as

Pennycuick (1987) in which these flight speed data were originally reported includes a high standard deviation (2.0) around the same mean flight speed.

Table 22. Flight speeds of seabirds for use in CRM with the sCRM tool as advised by Natural England (2022) and NatureScot (2023g). For any other species details must be discussed with the SNCB.

Species	Mean speed (m/sec)	SD	Source	Number measured	Airspeed or groundspeed
Northern gannet	14.9	0.0	Pennycuick, 1987	32	Airspeed
Black-legged kittiwake	13.1	0.40	Alerstam et al., 2007	2 for total of 660 seconds	Airspeed
Lesser black-backed gull	13.1	1.90	Alerstam et al 2007	11 for total of 3150 seconds	Airspeed
Herring gull	12.8	1.80	Alerstam et al., 2007	18 for total of 7210 seconds	Airspeed
Great black-backed gull	13.7	1.20	Alerstam et al., 2007	4 for total of 700 seconds	Airspeed
Sandwich tern	10.3	3.4	Fijn and Gyimesi, 2018	221 trips by 27 birds with GPS tags	Groundspeed

It is noteworthy that SNCB guidance to use a flight speed of 14.9 m/sec (SD 0) for northern gannet (Table 22) is based on data collected by Colin Pennycuick that presents flight speeds as air speeds, not as ground speeds. The original data also show clearly that there was considerable individual variation in flight speed of the 32 gannets in the sample studied by Pennycuick (1987), with a standard deviation of 2 around the mean airspeed measurement (Table 23), rather than 0.0 as advised by SNCBs. Flight speeds of northern gannets were about 12% higher measured as air speeds (mean 16.0 m/sec) than as ground speeds (mean 14.2 m/sec) (Mateos-Rodríguez and Bruderer, 2012 in Table 23). Similarly, speeds presented by Alerstam et al. (2007) are airspeeds derived from tracking radar measurements that have been corrected for wind speed effects. The measurements that should be used in CRM should be the corrected ground speed equivalents or, if available, the original ground speeds measured by the radar. In the case of lesser black-backed gull that correction reduced the predicted numbers of collisions by about 2% (Masden et al. 2021), but the magnitude of the effect may vary among species depending on flight characteristics such as wing loading which will influence effects of wind speed and direction.

Table 23. Flight speeds of northern gannets measured in different published studies. Data from OWFs shown in bold.

Study	Method	Air or ground speed	Sample size	Mean (m/sec)	SD
Pennycuick, 1987	Ornithodolite	Air	32	14.9	2
Skov et al., 2018	**Rangefinder**	**Ground**	**>3000**	**13.33**	**4.24**
Tjørnløv et al., 2023	**Radar**	**Ground**	**755**	**5.63**	**2**
Garthe et al., 2007	GPS tag	Ground	11 trips by 7 birds	16.2	0.53

Study	Method	Air or ground speed	Sample size	Mean (m/sec)	SD
Mateos-Rodríguez and Bruderer, 2012	Radar	Ground	120 tracks for 17075 seconds	14.2	2.7
Mateos-Rodríguez and Bruderer, 2012	Radar	Air	120 tracks for 17075 seconds	16.0	2.8
Lane et al., 2019	GPS tag	Ground	188 trips	15	3

Northern gannet flight speeds measured by Pennycuick (1987) were for birds observed during the breeding season but included birds of all age classes present. Northern gannet flight speeds measured by Tjørnløv et al. (2023) were also for birds observed during the breeding season including birds of all age classes present, but not separated into different age classes.

Table 24. Flight speeds of black-legged kittiwakes measured in different published studies. Data from OWFs shown in bold.

Study	Method	Air or ground speed	Sample size	Mean (m/sec)	SD
Pennycuick, 1987	Ornithodolite	Air	18	13.1	1
Alerstam et al., 2007	Radar	Air	2 (660 sec)	13.1	0.40
Tjørnløv et al., 2023	**Radar**	**Ground**	**2286**	**6.3**	**3**
Collins et al., 2020	GPS tags (1)	Air	47	9.7	3
Collins et al., 2020	GPS tags (1)	Ground	47	9	3
Skov et al., 2018	**Rangefinder**	**Ground**	**>2000**	**8.71**	**3.16**
Elliott et al., 2014	GPS 17 g tags (2)	Ground	10 birds	10.6	n/a
Kotzerka et al., 2010	GPS 11 g tags	Ground	16 trips by 9 birds	9.2	3.61
Götmark, 1980	Following by car on road parallel to flight line	Ground		11	n/a
Redfern and Bevan 2014	GPS tags (1)	Ground	19 trip sections of 4 birds	11.5	2.2

(1) Tags deployed weighed up to 5% of body mass so may have influenced flight speed

(2) Tags weighed ca 3.5% of body mass and the authors state that they "assumed that they affected behaviour similarly across individuals because all were equipped similarly"

Table 25. Flight speeds of lesser black-backed gulls measured in different published studies. Data from OWFs shown in bold.

Study	Method	Air or ground speed	Sample size	Mean (m/sec)	SD
Alerstam et al., 2007	Radar	Air	11 (3150 secs)	13.1	1.9

Study	Method	Air or ground speed	Sample size	Mean (m/sec)	SD
Skov et al., 2018	**Rangefinder**	**Ground**	**>1000**	**10.13**	**3.93**
Masden et al., 2021 Barrow	**GPS tags**	**Ground**	**Upwind but n not given**	**9.16**	**0.11**
Masden et al., 2021 Barrow	**GPS tags**	**Ground**	**Downwind but n not given**	**14.57**	**0.11**
Masden et al., 2021 Barrow	**GPS tags**	**Ground**	**Crosswind but n not given**	**10.90**	**0.11**
Masden et al., 2021 Ormonde	**GPS tags**	**Ground**	**Upwind but n not given**	**8.66**	**0.10**
Masden et al., 2021 Ormonde	**GPS tags**	**Ground**	**Downwind but n not given**	**14.08**	**0.10**
Masden et al., 2021 Ormonde	**GPS tags**	**Ground**	**Crosswind but n not given**	**10.40**	**0.10**
Masden et al., 2021 W of Duddon Sands	**GPS tags**	**Ground**	**Upwind but n not given**	**8.69**	**0.19**
Masden et al., 2021 W of Duddon Sands	**GPS tags**	**Ground**	**Downwind but n not given**	**14.11**	**0.19**
Masden et al., 2021 W of Duddon Sands	**GPS tags**	**Ground**	**Crosswind but n not given**	**10.44**	**0.19**
Masden et al., 2021 Walney1	**GPS tags**	**Ground**	**Upwind but n not given**	**8.33**	**0.16**
Masden et al., 2021 Walney1	**GPS tags**	**Ground**	**Downwind but n not given**	**13.75**	**0.16**
Masden et al., 2021 Walney1	**GPS tags**	**Ground**	**Crosswind but n not given**	**10.06**	**0.16**
Masden et al., 2021 Walney2	**GPS tags**	**Ground**	**Upwind but n not given**	**8.10**	**0.03**
Masden et al., 2021 Walney2	**GPS tags**	**Ground**	**Downwind but n not given**	**13.51**	**0.03**
Masden et al., 2021 Walney2	**GPS tags**	**Ground**	**Crosswind but n not given**	**9.84**	**0.03**

Using GPS-measured flight speeds of lesser black-backed gulls, predicted numbers of collisions at five offshore wind farms were 10.2 to 16.3% lower than predicted by SNCB guidance-based flight speeds (Masden et al., 2021). Thermal soaring behaviour by lesser black-backed gulls at sea during suitable weather conditions in summer can increase the time spent in the collision risk area and rotor-swept zone (van Erp et al., 2023).

Table 26. Flight speeds of herring gulls measured in different published studies. Data from OWFs shown in bold.

Study	Method	Air or ground speed	Sample size	Mean (m/sec)	SD
Pennycuick, 1987	Ornithodolite	Air	16	11.3	0.5
Alerstam et al., 2007	Radar	Air	18 (7210 secs)	12.8	1.8
Tjørnløv et al., 2023	**Radar**	**Ground**	**13307**	**6.44**	**3**

Study	Method	Air or ground speed	Sample size	Mean (m/sec)	SD
Skov et al., 2018	**Rangefinder**	**Ground**	**>2000**	**9.68**	**3.47**

Table 27. Flight speeds of great black-backed gulls measured in different published studies. Data from OWFs shown in bold.

Study	Method	Air or ground speed	Sample size	Mean (m/sec)	SD
Pennycuick, 1987	Ornithodolite	Air	25	12.4	2
Alerstam et al., 2007	Radar	Air	4 (700 secs)	13.7	1.2
Skov et al., 2018	**Rangefinder**	**Ground**	**>2000**	**9.78**	**3.65**

Table 28. Flight speeds of Sandwich terns measured in different published studies

Study	Method	Air or ground speed	Sample size	Mean (m/sec)	SD
Fijn and Gyimesi, 2018	GPS tracking	Ground	7238 GPS fixes from 221 foraging trips by 27 breeding adults	10.25	3.42
Wakeling and Hodgson, 1992	Visual observation during headwinds	Air	60	14.3	2.1
Wakeling and Hodgson, 1992	Visual observation during tailwinds	Air	6	9.2	2.3
Wakeling and Hodgson, 1992	Visual observation during crosswinds	Air	56	12.1	2.3
Perrow et al., 2010 in Fijn and Gyimesi, 2018	Boat-based following	Ground	Not stated	9.44	0.08
Perrow et al., 2011	Boat-based following	Ground	23	10.83	1.28

Table 29. Flight speeds of great skuas measured in different published studies

Study	Method	Air or ground speed	Sample size	Mean (m/sec)	SD
Pennycuick, 1987	Ornithodolite	Air	72	14.9	2
Mateos-Rodríguez and Bruderer, 2012	Radar	Ground	12 tracks for 1550 seconds	15.8	3.9

Study	Method	Air or ground speed	Sample size	Mean (m/sec)	SD
Mateos-Rodríguez and Bruderer, 2012	Radar	Air	12 tracks for 1550 seconds	17.2	4.6

Table 30. Flight speeds of Arctic skuas measured in different published studies

Study	Method	Air or ground speed	Sample size	Mean (m/sec)	SD
Pennycuick, 1987	Ornithodolite	Air	20	13.3	1
Alerstam et al., 2007	Radar	Air	7 (1500 secs)	13.8	2.2

Table 31. Flight speeds of northern fulmars measured in different published studies

Study	Method	Air or ground speed	Sample size	Mean (m/sec)	SD
Pennycuick, 1987	Ornithodolite	Air	104	13.0	2

Table 32. Flight speeds of Arctic terns measured in different published studies

Study	Method	Air or ground speed	Sample size	Mean (m/sec)	SD
Alerstam et al., 2007	Radar	Air	2 (500 secs)	10.9	0.9

Table 33. Flight speeds of common gulls measured in different published studies

Study	Method	Air or ground speed	Sample size	Mean (m/sec)	SD
Alerstam et al., 2007	Radar	Air	36 (14050 secs)	13.4	2.9

Table 34. Flight speeds of black-headed gulls measured in different published studies

Study	Method	Air or ground speed	Sample size	Mean (m/sec)	SD
Alerstam et al., 2007	Radar	Air	19 (6190 secs)	11.9	1.6

3.5. Nocturnal flight activity

Northern gannet flight activity at night can be derived from studies that deployed various tag types on breeding adults (such as GPS tags, Argos satellite Push to Talk (PTT)s, geolocator tags, depth and wing-beat monitoring tags, and stomach temperature monitor tags (Furness et al., 2018)). Most tag types only provide data for breeding birds during the breeding season, but geolocator tags allow activity to be determined throughout the nonbreeding season. These studies provide estimates of the amount of flight activity from sunset to sunrise as a percentage of the amount of flight activity during daytime, and/or of the percentage of foraging plunge-dives that occurred at night (Table 35).

Table 35. Summary of flight and plunge-diving activity of northern gannets between sunset and sunrise in relation to daytime levels.

Season	Colony	Tag type	Flight activity at night as a percentage of flight activity during the day	Percent of plunge-dives occurring at night	Reference
Breeding	Hermaness	Stomach temperature	21%	0%	Garthe et al., 1999
Breeding	Funk Island	Depth and wing-beat	-	4.3%	Garthe et al., 2000
Breeding	Bass Rock	Argos PTT	0%	0%	Hamer et al., 2000
Breeding	Funk Island	Depth and temperature	6%	0.5%	Garthe et al., 2003
Breeding	Bass Rock	Argos PTT or GPS	0%	-	Hamer et al., 2007
Breeding	Bempton	Argos PTT	0%	-	Langston et al., 2013
Breeding	Bonaventure	GPS and altimeter	-	15%	Garthe et al., 2014
Breeding	Alderney	GPS and accelerometer	-	0%	Warwick-Evans et al., 2015
Breeding	Alderney	GPS and accelerometer	15.7%	-	Warwick-Evans et al., 2018
Breeding	Bass Rock	geolocator	<5%	-	Furness et al., 2018
Breeding	Helgoland	GPS and depth	-	0.7%	Garthe et al., 2017
Autumn	North Sea	geolocator	2.5%	-	Garthe et al., 2012
Autumn	North Sea	geolocator	1%	-	Furness et al., 2018
Autumn	Migrating to west Africa	geolocator	3.8%	-	Furness et al., 2018
Winter	North Sea to West Africa	geolocator	1.9%	-	Garthe et al., 2012
Winter	North Sea	geolocator	2.4%	-	Furness et al., 2018

Furness et al. (2018) concluded from these data that northern gannets in the nonbreeding season show flight activity between sunset and sunrise consistently around 1 to 4% of the amount recorded during daylight. However, there is considerable uncertainty in the estimation of flight behaviour of seabirds from the geolocator data derived from tags that have been used up to now. Breeding adults during the breeding season show flight activity between sunset and sunrise averaging 7% of the daytime level, with diving activity at night about 3% of the daytime level. Higher nocturnal flight activity of breeding adults could reflect high energy demands in the breeding season, but may simply result from the fact that almost all nocturnal flight activity occurs during civil twilight, which lasts longer in summer than in winter.

In their paper on nocturnal flight activity in northern gannets, Furness et al. (2018) state "Tracking data exist for several other seabird species. However, most data sets have not been published in a form that allows nocturnal flight activity to be seen. Given that the evidence-based estimates for gannet represent a large reduction from the value employed by Band (2012), there would be merit in analysing nocturnal flight activity of species such as kittiwake, great black-backed gull and lesser black-backed gull. Evidence-based estimates for those species would also help to reduce uncertainty in environmental impact assessments for offshore wind farms". However, nocturnal flight activity of black-legged kittiwakes is reported in Daunt et al. (2002). In addition, a study that tracked lesser black-backed gulls with GPS tags showed that these birds flew at lower height asl at night when they did fly at night, so that collision risk would be further reduced by their lower flight altitude at night (Corman and Garthe, 2014).

3.6. Displacement rates

Displacement rates are relevant where there is a need to assess the impact on a seabird population of disturbance, displacement from habitat or a barrier effect caused by OWFs. There is a broad consensus that the seabird species in UK/European waters considered to be at most risk of displacement include divers (especially red-throated divers), sea ducks, northern gannet, common guillemot, razorbill, and Atlantic puffin (Petersen et al., 2006, Leopold et al., 2013, Mendel et al., 2014; Vanermen et al., 2015; Dierschke et al., 2016, 2017; SNCB, 2022a,b; Pavat et al., 2023).

NatureScot (2023h) advises the use of 30% displacement of black-legged kittiwake by OWFs (although in the same document black-legged kittiwake is not listed as a "priority species" for displacement assessment). Although SNCB (2022a) states that gull species do not normally require displacement assessment, we include black-legged kittiwake in this section since it seems that NatureScot may wish to consider kittiwakes as susceptible to displacement effects.

There is some very limited evidence that displacement rates may decline over years as a consequence of habituation (Vanermen et al., 2021). If that is confirmed, the long-term impact of displacement of seabirds by offshore wind farms will have been overestimated by modelling based on short-term displacement evidence. However, here we consider the quantified short-term rates of displacement of red-throated diver, black-throated diver, great northern diver, northern gannet, black-legged kittiwake, common guillemot, razorbill, and Atlantic puffin.

In assessing displacement/barrier effects on seabirds, evidence from tracking is useful in a biological sense because processes are distinguished and because tracking provides multiple data points per individual so gives a distribution of behavioural responses, critical for understanding the impacts of sub-lethal effects in displacement assessment. In contrast, surveys of spatial distribution can provide much higher sample sizes, capture all species, age classes and seasons, but combine displacement and barrier effects, and individuals are classed into two groups or a gradient. Displacement assessments to date have relied largely on analyses of survey data, where changes in relative distribution inside and outside the wind farm are assumed to result from displacement or attraction. The benefits that tracking data can provide in understanding individual variation in behavioural changes associated with displacement and barrier effects have been underused historically. However, this balance is now shifting with the increasing use of the SeabORD individual-based model in assessments.

For some species, such as northern gannet, emphasis has been on quantifying macro-avoidance in the context of CRM, whereas for other species, such as divers, sea ducks and auks, emphasis has been on displacement/barrier effects in terms of loss of habitat or increased energy expenditure. However, these terms seem often to be used interchangeably, although macro-avoidance relates only to flying birds and not to birds on the water, whereas displacement may relate more (or at least differently) to birds on the water than to flying birds, while barrier effects typically relate more to flying (especially commuting) birds.

3.6.1. Red-throated diver

Table 36. Summary of displacement evidence for red-throated diver at North Sea OWFs (updated from Allen et al., 2019 and SNCB, 2022b). The published values tend to be single estimates without explicit measures of uncertainty/variation. Distance over which displacement could be detected represents the distance at which birds were found to be present at a significantly lower density and therefore it is inferred that individuals tend to be displaced from that zone, although at lower rates than from the OWF itself.

Wind farm	Distance over which statistically significant displacement could be detected (km)	Magnitude of displacement from within OWF (%)	Magnitude of displacement in buffer zone(s) around OWF (%)	Reference
Horns Rev 1	2-4 km	100	-	Petersen et al., 2006
Nysted	0 km	-	-	Petersen et al., 2006
Kentish Flats	3 km	95	87% to 500 m 76% to 1 km 61% to 2 km 63% to 3 km	Percival, 2010
Thanet	OWF footprint	73	0% to 2 km	Percival, 2013
Kentish Flats	OWF footprint	89-94	0% to 3 km	Percival, 2014
Alpha Ventus	-	100	-	Mendel et al., 2014
Horns Rev 2	5-6 km according to authors but possibly up to 13 km	-	-	Petersen et al., 2014
London Array	2-4 km	-	-	APEM, 2015
Alpha Ventus	1.5 to 2 km	90	-	Welcker and Nehls, 2016
German Bight (several)	20 km	-	-	Heinänen et al., 2016

Wind farm	Distance over which statistically significant displacement could be detected (km)	Magnitude of displacement from within OWF (%)	Magnitude of displacement in buffer zone(s) around OWF (%)	Reference
German Bight (several)	20 km	-	-	Zydelis et al., 2016
Lincs, Lynn, Inner Dowsing	8 km	83	55% to 4 km 34% to 8 km	Webb et al., 2016
Liverpool Bay	3.8 km	-	-	Burt et al., 2017
Egmond aan Zee	2 km	90	50% to 2 km	Heinänen and Skov, 2018
German Bight (several)	16.5 km	-	94% to 3 km 84% to 10 km	Mendel et al., 2019
German Bight (several)	8.5 to 10.2 km	-	-	Vilela et al., 2020
German Bight (several)	10 to 15 km	>90	90% to 5 km	Heinänen et al., 2020
London Array	11.5 km	55%	39% to 2 km 36% to 4 km 35% to 6 km 41% to 8 km 40% to 10 km 13% to 11.5 km	APEM, 2021

3.6.2. Black-throated diver

It is likely that a small proportion of the divers included in the red-throated diver data set (Table 36) were black-throated divers, and there is a broad assumption that behavioural responses of black-throated divers are similar to those of red-throated divers. However, there do not appear to be any quantitative data on the extent to which black-throated divers are displaced by OWFs.

3.6.3. Great northern diver

Great northern divers do not respond in the same way as red-throated divers to disturbance. Whereas the latter tend to fly away from approaching boats and show strong avoidance of structures at sea such as offshore wind farms, great northern divers often approach close to artificial structures while foraging in winter (David Jardine pers. comm.) and tend to dive rather than fly when disturbed (Jarrett et al., 2018). However, there do not appear to be any quantitative data on the extent to which great northern divers are displaced by OWFs.

3.6.4. Northern gannet

For gannets, displacement by OWFs is normally expressed as a macro-avoidance rate, which is the percentage of birds approaching an OWF that divert their flight (or swimming) to avoid the wind farm

(Pavat et al., 2023). Reviewing 19 studies, APEM (2022b) assessed that the mean displacement/macro-avoidance rate was 70%, with half of the studies estimating displacement/macro-avoidance of 60-80%. A review by Pavat et al. (2023) considered ten studies which indicated mean macro-avoidance of 85.64% (95% CI 53.49 to 97.36%). Subsequent studies and studies not included in the two reviews gave further estimates of 37, 67, 74, 89 and 90% macro-avoidance (Table 5). Tracking provided interesting insight into the fact that macro-avoidance by northern gannets appears to show strong individual variation, with a large majority of individuals strongly avoiding OWFs but a minority apparently being attracted to OWFs (Peschko et al., 2021).

3.6.5. Black-legged kittiwake

In a review of evidence published up to early 2016, Dierschke et al. (2016) reported that black-legged kittiwakes showed on average no displacement or attraction in relation to offshore wind farms, with five sites showing no displacement (North Hoyle, Thanet, Bligh Bank, Egmond aan Zee, Horns Rev 1), two sites showing displacement >50% (Thorntonbank, Horns Rev 2), two sites showing displacement <50% (Prinses Amalia, Alpha Ventus), one site showing attraction <50% (Gunfleet Sands), one site showing attraction >50% (Robin Rigg) and one site showing presence close to turbines that suggested possible attraction (Kentish Flats). More recent studies show similar results to those reviewed by Dierschke et al. (2016) (Table 37). See also evidence listed in Section 3.2.2.

Table 37. Summary of displacement evidence for black-legged kittiwake at North Sea OWFs

OWF	Displacement (%)	Uncertainty	Reference
12 sites	Mean 0	n/a	Dierschke et al., 2016
Luchterduinen	0	n/a	Heinänen and Skov, 2018
Prinses Amalia	0	n/a	Heinänen and Skov, 2018
Egmond aan Zee	0	n/a	Heinänen and Skov, 2018
Beatrice	0	n/a	Trinder et al., 2024
Luchterduinen	Not statistically significant	n/a	Leemans et al., 2022
Belgian North Sea	Slight attraction	n/a	Vanermen et al., 2023
North Sea	Slight attraction	n/a	Pollock et al., 2024
German North Sea	45% reduction within wind farm during breeding season but zero reduction at other times of year	n/a	Peschko et al. 2020a

3.6.6. Common guillemot

In a review of evidence published up to early 2016, Dierschke et al. (2016) reported that on average (based on data from 12 offshore wind farms) common guillemots showed weak avoidance of offshore wind farms. Of the 12 sites, avoidance was strong (>50%) at five, weak (<50%) at four, nil at two and birds were strongly attracted (>50% increase) at one site. At three other sites where the change was not quantified, guillemots were reported to be present in good numbers close to turbines whereas at one site they were reported to be mostly absent from areas close to turbines.

Table 38. Summary of displacement evidence for common guillemots at North Sea OWFs

OWF	Displacement (%)	Uncertainty	Reference
12 sites	Mean <50%	n/a	Dierschke et al., 2016
Robin Rigg	0	n/a	Vallejo et al., 2017
Egmond aan Zee	0	n/a	Leopold, 2018
Prinses Amalia	0	n/a	Leopold, 2018
Robin Rigg	Very small	n/a	Leopold, 2018
Luchterduinen	52 (1)	n/a	Heinänen and Skov, 2018
Prinses Amalia	70 (2)	n/a	Heinänen and Skov, 2018
Egmond aan Zee	28 (1)	n/a	Heinänen and Skov, 2018
Beatrice	0	n/a	Trinder et al., 2024
Luchterduinen	43	n/a	Leemans et al., 2022
German Bight	63	n/a	Peschko et al., 2020b
German Bight	44 (summer)	n/a	Peschko et al., 2020a
German Bight	63 (spring)	n/a	Peschko et al., 2020a
Belgian North Sea	20	n/a	Vanermen et al., 2023
German Bight	91 (autumn)	95% CI 84-94	Peschko et al., 2024
German Bight	67 (winter)	95% CI 53-77	Peschko et al., 2024

(1) These authors reported that displacement from the 2 km buffer around the wind farm was zero

(2) These authors reported that displacement from the 2 km buffer around the wind farm was <50%

3.6.7. Razorbill

In a review of evidence published up to early 2016, Dierschke et al. (2016) reported that, on average (based on data from seven offshore wind farms), razorbills showed weak avoidance of offshore wind farms. Of the seven sites, avoidance was strong (>50%) at two, weak (<50%) at three, nil at two. At one other site where the change was not quantified, razorbills were reported to be present in good numbers close to turbines.

Table 39. Summary of displacement evidence for razorbills at North Sea OWFs

OWF	Displacement (%)	Uncertainty	Reference
7 sites	Mean <50%	n/a	Dierschke et al., 2016
Luchterduinen	52 (1)	n/a	Heinänen and Skov, 2018
Prinses Amalia	72 (1)	n/a	Heinänen and Skov, 2018
Egmond aan Zee	Not statistically significant	n/a	Heinänen and Skov, 2018
Beatrice	0	n/a	Trinder et al., 2024
Luchterduinen	Not statistically significant	n/a	Leemans et al., 2022
Belgian North Sea	Slight attraction	n/a	Vanermen et al., 2023

(1) these authors reported that displacement from the 2 km buffer around the wind farm was zero

3.6.8. Atlantic puffin

In a review of evidence published up to early 2016, Dierschke et al. (2016) found that the behavioural response of puffins had not been quantified at any OWF sites (because puffin at-sea densities were almost invariably too low to quantify any effect) but the one site reporting on puffin response noted that there were good numbers present close to turbines.

Table 40. Summary of displacement evidence for Atlantic puffins at North Sea OWFs

OWF	Displacement (%)	Uncertainty	Reference
Beatrice	0	n/a	Trinder et al., 2024

3.6.9. Sandwich tern

In a review of evidence published up to early 2016, Dierschke et al. (2016) reported that on average (based on data from six offshore wind farms) Sandwich terns showed weak avoidance of offshore wind farms. Of the six sites, avoidance was strong (>50%) at two (Alpha Ventus and Kentish Flats), weak (<50%) at one (Sheringham Shoal), nil at three (Egmond aan Zee, Prinses Amalia and Horns Rev 1).

Table 41. Summary of displacement/macro-avoidance evidence for Sandwich terns at North Sea OWFs

OWF	Displacement/macro-avoidance (%)	Uncertainty	Reference
6 sites	Mean <50%	n/a	Dierschke et al., 2016
Egmond aan Zee	70	n/a	Krijgsveld et al., 2011
Luchterduinen	60	n/a	Leemans et al., 2022
Several near Scolt Head colony	54	95% CI 35 to 70	van Bemmelen et al., 2024
Several near De Putten colony	41	95% CI 21 to 56	van Bemmelen et al., 2024
Belgian North Sea	50	n/a	Vanermen et al., 2023

3.7. Percentage mortality of displaced birds

The possibility that displacement might lead to mortality has been considered based on ecological theory, empirical observation of density-dependent effects on bird populations, and modelling. However, no published studies have reported quantitative empirical data on the mortality of seabirds displaced by OWFs.

3.8. Demographic parameters for PVA modelling

Since the SNCBs advise use of a density-independent population model with a counterfactual approach to compare the rate of growth with and without OWF impact, the exact values of demographic parameters entered into the model are relatively unimportant as they affect the absolute population growth rate but have hardly any effect on the counterfactual which is the metric of relevance (Jitlal, 2017). Therefore, a crucial aspect of PVA modelling is the question of whether or not to include density-dependence as a model parameter (Merrall et al., 2024), rather than which values of adult survival, age of first breeding, breeding success, immature survival and frequency of sabbaticals to select for the focal seabird population. There has therefore been little reason to consider change to the recommended use of demographic data listed by Horswill and Robinson (2015). To save repetition, those data are not copied here, but can be obtained directly from the Horswill and Robinson (2015) report. Long-term studies monitoring breeding success at numerous colonies, and survival rates or return rates of seabirds at a very small sample of UK colonies, continue to list data in the JNCC Seabird Monitoring Programme (SMP) database. In addition, the British Trust for Ornithology (BTO) publishes data on adult survival of seabirds each year gained from the "Retrapping Adults for Survival" programme. The BTO adult survival data are aggregated across colonies and so represent a measure for the entire UK population of each species RAS results | BTO - British Trust for Ornithology. In practice there have been numerous further data points on seabird breeding success for years since 2015, but there have been very few publications since the Horswill and Robinson (2015) report that present new data to the year of publication on adult survival, age of first breeding, immature survival or frequency of sabbaticals in UK seabirds. Incorporating density dependence in seabird population models is potentially important in this context, but the analysis in Merrall et al. (2024) was hampered by the limited number of demographic time series of sufficient length. As such, there would be merit in repeating the analyses in Merrall et al (2024) in a few years, when there would be a larger number of time series of sufficient length for density dependent effects to be tested robustly.

3.9. Examples of HRA assessment of in-combination impacts

Assessed in-combination impacts on seabird populations at SPAs increase as further offshore wind farms are planned to be constructed in a particular area and add impacts to existing totals which are combined additively until a decision is taken to incorporate older impacts into baseline. An area with a particularly high concentration of OWFs as well as seabird SPAs is the southern North Sea. A recent OWF planning application in the southern North Sea for which HRA has been completed was the Sheringham Shoal and Dudgeon Extensions OWF project. The Department for Energy Security & Net Zero published their HRA for this proposal in April 2024 (Department for Energy Security & Net Zero, 2024). It includes in-combination impact assessments that were the subject of ornithology concerns raised by Natural England and the Ex A in relation to impacts on populations of Sandwich tern at North Norfolk Coast SPA, northern gannet, black-legged kittiwake, common guillemot, and razorbill at Flamborough and Filey Coast SPA. We use these examples to illustrate the scale, and uncertainty, of in-combination impact assessed following current SNCB methodology guidance. The following sections are all based on the HRA (Department for Energy Security & Net Zero, 2024).

3.9.1. Sandwich tern at North Norfolk Coast SPA

Potential impacts on Sandwich tern at North Norfolk Coast SPA for which a Likely Significant Effect (LSE) could not be ruled out were collision impacts during the breeding season, as NE agreed that a displacement assessment was not required. An adverse effect on integrity (AEOI) of the SPA feature (Sandwich tern breeding numbers) from the project alone was ruled out but an AEOI could not be ruled out in-combination. The in-combination mortality estimated by NE was between 84.6 and 87.8 birds per year resulting in an increase to baseline mortality of 8.8% to 9.1% (Department for Energy Security & Net Zero, 2024).

3.9.2. Northern gannet at Flamborough and Filey Coast SPA

The predicted annual in-combination mortality arising from displacement of northern gannets from Flamborough and Filey Coast SPA was estimated to be between 55 and 73 adult birds per year based on displacement rates of 60% to 80% and a mortality rate of 1% of displaced birds. That equates to an increase in adult mortality of between 2.54% and 3.36% for that population. The predicted annual in-combination mortality arising from collisions was estimated to be 67 adult birds, increasing baseline mortality by 3.1%. The combined in-combination impacts of displacement and collision was estimated to be between 122.5 and 140.5 adults per year, which increased mortality by between 5.6% and 6.5%. The in-combination counterfactual of the population size was calculated as being 0.75 and the counterfactual growth rate was predicted to be no less than 0.993 (Department for Energy Security & Net Zero, 2024).

3.9.3. Black-legged kittiwake at Flamborough and Filey Coast SPA

Excluding projects for which compensation for impacts on kittiwake had been agreed, the estimated in-combination collision impact was 292.7 adult kittiwakes a year which increased baseline mortality by 1.9%. The median counterfactual growth rate after 35 years would be 0.997 and median counterfactual population size 0.871. That assumes that compensation is effective and so there is no net contribution of mortality from projects providing compensation. Including mortality from those projects resulted in an estimated in-combination collision impact of 394 adult kittiwakes a year, which would increase baseline mortality by 2.6%. In this case the median counterfactual growth rate after 35 years would be 0.996 to 0.995 and median counterfactual population size 0.859 to 0.798 (Department for Energy Security & Net Zero, 2024).

3.9.4. Common guillemot at Flamborough and Filey Coast SPA

Using NE's standard approach, the potential mortality arising from in-combination displacement effects was estimated to be between 112 and 2,608 adults per year, resulting in an increase to baseline mortality of between 1.51% and 35.12%, with NE's best estimate being displacement mortality of 1,498 adults per year (based on 70% displacement and 2% mortality of displaced birds), resulting in an estimated counterfactual reduction in population growth rate of 1.4% and a counterfactual reduction in final population size of 54.3% (Department for Energy Security & Net Zero, 2024).

3.9.5. Razorbill at Flamborough and Filey Coast SPA

Using NE's standard approach, the potential mortality arising from in-combination displacement effects was estimated to be between 21 and 500 adults per year, resulting in an increase to baseline mortality of between 0.50% and 11.76%, with NE's best estimate being displacement mortality of 206 adults per year (based on 70% displacement and 2% mortality of displaced birds), resulting in an estimated counterfactual reduction in population growth rate of 0.6% and a counterfactual reduction in final population size of 22.7% (Department for Energy Security & Net Zero, 2024).

4. Update to uncertainty roadmap for ornithological offshore wind impact assessments

In Searle et al. (2023a) we focused on how environmental variation and structural and parameter uncertainty are recognised, quantified and propagated through the assessment process. The complexities of natural systems and limitations of ecological data collection mean that uncertainty cannot in practice either be perfectly quantified, or reduced to zero. Yet its importance within impact assessment approaches results in a critical need for improvements aimed at both uncertainty quantification and reduction. Here, we revisit the roadmap summarising identified research priorities for improving estimation of, and reducing uncertainties in impact assessments for offshore wind in the UK context. In Table 42 we provide an updated version of the roadmap from Table 1 of Searle et al. (2023a). Updates on the status of scientific progress towards addressing each research priority have been included (Table 42), identify ongoing and planned projects that are identifying and addressing sources of uncertainty to better quantify and reduce uncertainty in assessments. The focus for the review of current and recently completed research projects is on UK-based research, since this is most likely to directly align with, and have direct biological relevance to, the requirements of the UK assessment process. Non-UK research can also have relevance to UK assessments, but it was beyond the scope of this project to review all relevant non-UK research. We have also re-evaluated the research priorities identified in Searle et al. (2023a): we concluded that all of the priorities identified in Searle et al. (2023a) remain relevant, and include three additional research priorities now widely recognised within the scientific community in relation to uncertainty in impact assessments (Table 42, rows 17-19). Note that the assignment of each research category into 'medium' or 'high' contributions to quantifying and reducing uncertainty (Table 42) was established by expert judgement – i.e., the authors' (Searle et al., 2023a) assessment for how much each proposed research priority would improve quantification of uncertainty, and reduce knowledge uncertainty, within the context of the UK assessment process. We do not judge that the scores associated with the research priorities previously identified in Searle et al. (2023a) have changed since publication of that paper. The scores for the three additional research priorities (Table 42, rows 17-19) were also estimated using expert judgement from scientists within the ORJIP AssESs Project team (Searle, Butler, Daunt). "Potential for reducing uncertainty" is used here to refer to the direct reduction of uncertainty (e.g. through the inclusion of new evidence arising from data collection). Other activities, such as sensitivity analysis, are indirectly important in reducing uncertainty (e.g. by allowing for the prioritisation of data collection), but "potential for reducing uncertainty" is used here solely to refer to activities that have potential to directly reduce uncertainty.

The updated roadmap will inform, together with evidence from the sensitivity analysis (WP2) and stakeholder engagement (WP3), the development of recommendations around future research and data collection to better quantify and reduce uncertainty within WP4. These recommendations will be explicitly designed to capture priorities across short-term, medium-term and longer-term timescales.

Table 42. Summary of research priorities for better estimating and reducing uncertainty in seabird offshore wind farm assessments, moving beyond current tools and methodologies. Priorities are grouped into 'medium' and 'high' contributions to a) full quantification of uncertainty, and b) reduction in knowledge uncertainty. Note that the order of priorities within the table broadly follows their relevance to each stage of the assessment process (shown in bold), moving from estimating spatial distributions of birds and apportioning, quantifying displacement and collision impacts, and comparison of impacts via population modelling and PVA. The assignment of each research category into 'medium' or 'high' was done by expert judgement – i.e., the authors' assessment for how much each proposed research priority would improve quantification of uncertainty, and reduce knowledge uncertainty, within the context of the UK assessment process. Updated for ORJIP AssESs Project in December 2024.

	Research priorities & relevant stage of assessment	Contribution to quantifying uncertainty	Contribution to reducing knowledge uncertainty	Update on research activity to address this priority since 2023
1	Data integration from different sources and seasons for better knowledge of year-round distributions to quantify and reduce uncertainty ***Spatial distributions and apportioning***	High	High	ORJIP InTAS (Integration of Tracking and At-sea Survey Data) project has developed new methodologies (building on Blackwell & Matthiopoulos, 2024) for integrating GPS tracking and at-sea survey data to estimate distributions of breeding and non-breeding birds during the breeding season, including improvements to uncertainty quantification in methods for analysing GPS tracking data. OWEC PrePARED (Predators and Prey Around Renewable Energy Developments) project is developing and testing new methodologies for using combinations of boat-based survey data and GPS tracking data to estimate summer distributions of birds (*on-going*)

	Research priorities & relevant stage of assessment	Contribution to quantifying uncertainty	Contribution to reducing knowledge uncertainty	Update on research activity to address this priority since 2023
2	Improving uncertainty quantification in movement models ***Spatial distributions and apportioning***	High	Medium	Various new methodologies are emerging using Continuous Time Movement Modelling (CTMM) approaches to better quantify uncertainty in analyses of GPS tracking data when estimating spatial distributions (e.g., 'ctmm' package in R Software) A range of industry funded work based in the Forth-Tay is funding development of new methods for assessing seabird GPS tracking data in movement models, and in integrating information from a range of bird-based sensors to reduce uncertainty in model outputs (e.g., Bogdanova et al., 2022) The OWEC PrePARED project is developing movement models to incorporate seabird and prey data. It is focusing on developing methods to structure models to better integrate and fit multiple sources of contemporaneous data.

	Research priorities & relevant stage of assessment	Contribution to quantifying uncertainty	Contribution to reducing knowledge uncertainty	Update on research activity to address this priority since 2023
3	Better understanding and quantification of year-round distributions and impacts of displacement to quantify and reduce uncertainty ***Spatial distributions, displacement, and apportioning***	Medium	High	There are an increasing number of studies of year-round marine habitat use in seabirds utilising GLS technology, notably led by the SEATRACK (Seabird Tracking) programme (Norway, Iceland, UK), and more specifically for the UK by the ScotMER Aukestra project. In some cases more reliable GPS tracking methods are being used (although only for larger seabird species such as gannets and gulls) - GPS tracking devices are continually evolving and smaller devices capable of use in smaller species year-round are now available (e.g., for auks and kittiwake), so higher resolution data (i.e. several fixes per day with <10m accuracy) are now accumulating for limited populations and species. Additionally, the OWEC POSEIDON (Planning Offshore Wind Strategic Environmental Impact DecisiONs) project has conducted extensive new at-sea digital surveys of seabirds in UK waters, with resulting maps being made available in 2025 (project led by Natural England).
4	Better understanding and quantification of predator-prey interactions, relationship between prey density and prey availability, impacts of ORDs on prey distributions and availability to quantify and reduce uncertainty ***Spatial distributions, displacement, and collision***	High	High	The OWEC PrePARED (led by Marine Directorate, Scottish Government) and NERC ECOWIND programme (projects ECOWings (Ecosystem Change, Offshore Wind, Net Gain and Seabirds), Pelagio (Physics-to-Ecosystem Level Assessment of Impacts of Offshore Windfarms) and Accelerate (Ecological Implications of Accelerated Seabed Mobility around Windfarms)) are collecting concurrent data on seabirds and prey in relation to Offshore Renewable Developments (ORDs). Resulting inference and data products from these projects (covering the North Sea and Celtic Sea) will become available from 2025 onwards.

	Research priorities & relevant stage of assessment	Contribution to quantifying uncertainty	Contribution to reducing knowledge uncertainty	Update on research activity to address this priority since 2023
5	Estimate link between displacement effects and changes in demographic rates (productivity and survival) to better quantify and reduce uncertainty ***Spatial distributions, displacement, and apportioning***	High	High	This remains a high priority for research, particularly given the need for relatively long-term, consistent studies of individual-level demography concurrent with observations of individual interactions with ORDs. Such work is underway in the Forth-Tay region (e.g., Bogdanova et al., 2022) for four seabird species, yet remains an important research gap more broadly.
6	Effects of displacement on different age classes, e.g., immatures and non-breeders to better quantify and reduce knowledge uncertainty ***Displacement***	Medium	Medium	This remains a significant research gap and source of uncertainty in current assessment methods. Effects of displacement on non-breeders are starting to be addressed in other countries, especially Germany (see Peschko et al. 2020a), and new methods are also being developed to predict non-breeding season impacts of displacement in a few species using Individual-Based Models (e.g., Layton-Matthews et al., 2023a; ORJIP DisNBS project (Effects of displacement from Offshore Renewable Developments in the non-breeding season). NatureScot organised a workshop around evidence relating to displacement in September 2024. As yet, there is very little information on effects of displacement in non-adult birds, representing a considerable knowledge gap and source of ongoing uncertainty.
7	Improve uncertainty quantification within IBMs to better characterise and reduce structural and parameter uncertainty ***Displacement and collision***	High	Medium	The OWEC PrePARED project is developing the IBM SeabORD (Searle et al., 2018), including a new sensitivity analysis for how key model parameters affect impact predictions. This builds on work in the ORJIP QuMR (Quantification of Mortality Rates associated with Displacement) project (Searle et al., 2024), and will be published as part of a peer-reviewed article in 2025.

	Research priorities & relevant stage of assessment	Contribution to quantifying uncertainty	Contribution to reducing knowledge uncertainty	Update on research activity to address this priority since 2023
8	Assess sensitivity of collision risk model outputs to variation in input and structural parameters; understand and quantify covariance between parameters used in collision risk models to better quantify and reduce structural and parameter uncertainty ***Collision***	Medium	Medium	WP2 of this project is investigating sensitivity of collision risk model outputs, and subsequent PVA outputs, to variation and uncertainty in collision risk model parameters.

	Research priorities & relevant stage of assessment	Contribution to quantifying uncertainty	Contribution to reducing knowledge uncertainty	Update on research activity to address this priority since 2023
9	Improve estimates of flight speed and flight height for species to better characterise and reduce parameter uncertainty, quantify influence of environmental conditions to better characterise natural variability, and understand how variation in flight speed and flight height is related to behaviour (e.g., commuting versus foraging) to reduce knowledge uncertainty ***Collision***	Medium	Medium	New evidence for seabird flight speed and height has accumulated relatively quickly in recent years with the application of more GPS tracking studies with improved technology. However, it is not clear that a synthesis of new evidence has been undertaken to allow for improved use of these data within impact assessments. The ScotMER BRAIDS (Bird Responses to Avian Influenza and Developments at Sea) project will deliver new empirical evidence for seabird flight characteristics in the northern North Sea (commenced 2024). OWEC RESCUE (Reducing Seabird Collisions Using Evidence) (Natural England (NE) & BTO) - comprehensively collating seabird flight height data to develop new improved seabird flight height distributions, using appropriate methods for integrating different types of data and sources of uncertainty, and provide associated guidance for use in impact assessments; investigate factors influencing seabird flight heights and collision risk (e.g., weather conditions, time of year, area, and wind farm effects); develop a publicly accessible toolkit to make the outputs (i.e., derivative works associated with the analyses, new flight height distributions and best practice guidance) easily accessible.

	Research priorities & relevant stage of assessment	Contribution to quantifying uncertainty	Contribution to reducing knowledge uncertainty	Update on research activity to address this priority since 2023
10	Improve estimates of avoidance rates and partitioned into micro-, meso- and macro-avoidance to better quantify and reduce structural and parameter uncertainty; improve understanding of the influence of environmental conditions on avoidance to better characterise natural variability; improve understanding of the contribution of model error to predicted collision rates and the implications of this for estimates of avoidance rates ***Collision***	High	High	New evidence for these research priorities is accumulating with the advent of more GPS tracking studies aimed at quantifying seabird responses to operational ORDs (see Lamb et al., 2024 for review). At present, evidence is relatively limited across a few species and populations, however ongoing studies will deliver new evidence, potentially allowing for more strategic analyses for how environmental conditions affect avoidance and displacement behaviours. The ScotMER BRAIDS project will deliver new empirical evidence in relation to this research priority for seabird populations in the northern North Sea (commenced in 2024). ORJIP/OWEC PrediCtOr (Prevalence of Seabird Species and Collision Events in Offshore Wind Farms) project (BTO, The Royal Society for the Protection of Birds (RSPB) & Waardenburg Ecology) will develop a coordinated approach for reducing uncertainty surrounding bird collision risk and improving post-construction monitoring (PCM) at offshore windfarms - Developing and establishing best practice guidance on the use of monitoring technologies, equipment installation offshore and data handling; Providing recommendations for future study designs best suited to monitoring bird flight tracks and collisions in and around offshore wind farms.

	Research priorities & relevant stage of assessment	Contribution to quantifying uncertainty	Contribution to reducing knowledge uncertainty	Update on research activity to address this priority since 2023
11	Improve estimates for abundance, productivity, adult and immature survival, carryover effects, and inter-colony movements (including uncertainty in rates) to better quantify and reduce parameter uncertainty ***PVA***	High	High	New evidence for inter-colony movements is being developed for black-legged kittiwakes in the ORJIP MetaKitti (Modelling of Kittiwake Metapopulation Dynamics) project (led by Glasgow University). Similarly, the OWEC MOTUS (Remote Tracking of Seabirds at Sea) project (initiated in 2024) will develop a data collection approach and associated modelling framework for estimating inter-colony movements in this species. However, improved estimates for immature survival and carryover effects remains an important, outstanding research gap in most seabird species.
12	Empirical estimation of correlation in demographic rates and influence of environmental stochasticity to better characterise natural variability and improve quantification of structural and parameter uncertainty ***PVA***	Medium	High	Some work in this area has been completed in recent years (Horswill et al., 2023, Layton-Matthews et al., 2023b). However, it remains an important research gap for most seabird species, and a significant source of uncertainty in current assessment methods. WP2 of this project will also partly address this, by evaluating sensitivity of assessment outputs in relation to assumptions regarding correlation, and in relation to distinguishing between variability and uncertainty.

	Research priorities & relevant stage of assessment	Contribution to quantifying uncertainty	Contribution to reducing knowledge uncertainty	Update on research activity to address this priority since 2023
13	Understand relationship between demographic rates and prey availability to better quantify and reduce knowledge uncertainty; improve estimates for interactions between demographic rates and climate and other environmental variables to include in population forecasts to better characterise natural variability ***PVA***	High	High	Ongoing work in the NERC ECOWIND programme will provide new evidence and data products for how variation in seabird prey species is linked to demographic rates (ECOWINGS - ECOWind project), with evidence expected to be published in 2025 and 2026. Other projects in the ECOWIND programme are also considering how environmental variability (especially related to oceanographic variables) influences seabird movements, and potentially demographic rates (expected to deliver in 20256/2026). Some recent publications have advanced understanding for how seabird prey shapes demographic rates (primarily breeding success), particularly in relation to prey abundance and effects of forage fish fisheries (Searle et al. 2023b).
14	Integrated population modelling and model fitting methods to better quantify structural and parameter uncertainty by using all available abundance data to inform estimation of demographic rates; improved models of observation error for abundance estimates to support this ***PVA***	Medium	Medium	The OWEC ProcBe (Procellariiform Behaviour & Demographics) project, led by JNCC, is investigating how seabird species (storm petrel, Manx shearwater) interact with OWFS, to improve demographic rate and population modelling approaches.

	Research priorities & relevant stage of assessment	Contribution to quantifying uncertainty	Contribution to reducing knowledge uncertainty	Update on research activity to address this priority since 2023
15	Sensitivity analyses for PVAs to help prioritise efforts to reduce structural and parameter uncertainty ***PVA***	Medium	Medium	WP2 of this project will be addressing this in relation to parameter uncertainty (for selected tools and species) and considering elements of structural uncertainty.
16	Better understanding and quantification of density dependent processes in populations to reduce knowledge uncertainty ***PVA***	Medium	Medium	This remains an important research priority for seabirds in general. Some new evidence has arisen very recently, with a study investigating forms of density dependence in a range of UK seabird species and subsequent consequences of including density dependence within PVAs for resulting inference (Merrall et al. 2024)
Emerging priorities since publication of Searle et al. (2023):				
17	Improved quantification of seabird diet outside of chick-rearing, especially during non-breeding season, and ability to switch between different prey species as availability changes; associated variability in demographic rates in relation to diet and prey availability ***Spatial distributions, PVA***	High	Medium	The ScotMER Aukestra project (led by UKCEH) is collating data and estimating adult diet for several species of seabirds outside of the chick-rearing period.

	Research priorities & relevant stage of assessment	Contribution to quantifying uncertainty	Contribution to reducing knowledge uncertainty	Update on research activity to address this priority since 2023
18	Impact of external shocks (extreme weather, marine heatwaves, disease outbreaks) on the abundance, distribution, behaviour and demographics of seabirds to better quantify and reduce structural and parameter uncertainty ***All components of assessment***	High	High	The ScotMER BRAIDS project (led by RSPB) and the NERC ECOWIND programme projects are quantifying impacts of climate and external shocks (BRAIDS – HPAI; ECOWIND – climate, marine heatwaves, extreme weather) on seabird behaviour and demography, for both current and future projections.
19	Quantification of the rates of 'turnover' of seabirds observed at sea, including estimates for foraging site fidelity, to reduce structural and parameter uncertainty ***Displacement, Collision, Apportioning***	High	Medium	The ORJIP QuMR project (led by UKCEH) provides a set of recommendations for addressing this research priority (Searle et al. final project report to ORJIP, *in press*). Work within the ScotMER BRAIDS project (led by UKCEH) will also provide estimates for rates of at-sea turnover in seabird space use in 4 seabird species using data from south-east Scotland (due 2025/2026). Site fidelity in seabird space use in four seabird species has recently been estimated by Regan et al., (2024).

5. Conclusions

We have (Section 2) reviewed current SNCB guidance in order to determine the quantitative approaches and tools that are commonly used in ornithological assessments within the UK context. Models and tools are used to quantify annual level of mortality from collision and displacement, for species for which these impact mechanisms are of concern, to apportion these impacts to protected populations, and to evaluate the longer-term consequences of impact through Population Viability Analysis (PVA). Mortality from collision is typically quantified using either deterministic or stochastic implementations of the Band Model, whilst mortality from displacement is frequently quantified via the Displacement Matrix but can also be quantified using individual-based models (e.g. SeabORD). Apportioning approaches vary between the breeding and non-breeding season, whilst PVAs can either be based on density dependent or density-independent models.

The approaches and tools that are used within assessments require inputs, and these, in turn, require information derived from empirical data. Some of the inputs relate to design characteristics of turbines and windfarms, but the remaining inputs to collision and displacement models relate to seabird biology and to the form and magnitude of seabird-windfarm interactions. These biological inputs can represent complex processes that are challenging to observe empirically, and may involve considerable uncertainty. Crucially, uncertainties can include structural uncertainties (e.g. around the structure of the model) and uncertainties regarding the most appropriate sources of information to use (e.g. around evaluation of uncertainty), as well as directly quantified and reported levels of uncertainty. Where uncertainty was been quantified and reported this can take a range of different forms, including standard errors, standard deviations, ranges, confidence levels and sets of bootstrap samples. We have (Section 3) provided a detailed review of the available evidence on parameter values, and associated uncertainties, for each of the biological parameters involved in collision and displacement risks models. This has included, where relevant, reviewing the different underlying source of information that could be used to inform the values of each of these parameters. We have also highlighted challenges in relating to empirical information to the parameters used within the models (e.g. in relation to the interpretation of the avoidance rate parameter used in collision risk models).

Searle et al. (2021, 2023a) provided a strategic review of the sources of uncertainty in the ornithological assessment process including inherent variability, uncertainty in parameters and inputs, uncertainty around model structures and assumptions, and uncertainties arising from broader issues including field-specific terminology, language barriers and decision-making. The review noted that the extent to which uncertainty and variability is quantified within tools of the assessment process and their inputs varies substantially between the different tools and provided a roadmap towards both improving uncertainty quantification and reducing uncertainty across the assessment process. They highlighted specific priorities of work required to deliver these reductions and improvements. However, as research and evidence on ecological impacts of offshore renewables are developing rapidly (as a result of project-level monitoring, strategic research programmes such as ORJIP, OWEC, ECOWIND, and ScotMER, and emerging evidence from other countries) it has become necessary to update the review and prioritisation exercise to identify new knowledge gaps and recognise the ones that have now been addressed or are currently being targeted by ongoing research, so in Section 4 we have updated this roadmap to reflect developments since the publication of Searle et al. (2023a). This update has identified some additional emerging knowledge gaps (e.g. around HPAI), and has reviewed research activity that is currently underway in relation to each of the knowledge gaps outlined in Searle et al. (2023a).

The outputs from this work package will feed directly into the remaining work within the AssESs Project. The identification of tools (Section 2), sources of evidence for the values of particular parameters and quantified levels of uncertainty in these parameters (Section 3), will be used in determining the remit of the sensitivity analyses within WP2. The outputs from both WPs 1 and 2 will then feed into the stakeholder engagement activities around uncertainty and precaution within WP3. This stakeholder engagement will, in turn, feed into the development of recommendations around future changes to assessment approaches (WP4). The updated roadmap (Section 4) and sensitivity analysis results (WP2) will, in parallel, feed into the development of recommendations around future research wit.

References

Afan, I., Navarro, J., Grémillet, D., Coll, M. and Forero, M.G. (2019). Maiden voyage into death: are fisheries affecting seabird juvenile survival during the first days at sea? Royal Society Open Science 6: 181151.

Alerstam, T., Rosen, M., Bäckman, J., Ericson, P.G.P. and Hellgren, O. (2007). Flight speeds among bird species: allometric and phylogenetic effects. PLoS Biology 5: e197.

Allen, S., Banks, A.N., Caldow, R.W.G., Frayling, T., Kershaw, M. and Rowell, H. (2019). Developments in understanding of red-throated diver responses to offshore wind farms in marine Special Protected Areas. Pp. 573-586 In: Humphreys, J. and Clark, R.W.E. (Eds.) Marine Protected Areas: Science, Policy and Management. Elsevier, Amsterdam.

Amélineau, F., Péron, C., Lescroël, A., Authier, M., Provost, P. and Grémillet, D. (2014). Windscape and tortuosity shape the flight costs of northern gannets. Journal of Experimental Biology 217: 876-885.

APEM (2015). London Array additional analysis. APEM report to London Array Ltd.

APEM (2021). Final ornithological monitoring report for London Array offshore windfarm – 2021. APEM report to London Array Ltd.

APEM (2022a). Awel y Mor Offshore Wind Farm Category 6: Environmental Statement Volume 4, Annex 4.1: Offshore Ornithology Baseline Characterisation Report. Revision B. RWE Renewables UK, Swindon.

APEM (2022b). Hornsea Project Four: Gannet displacement and mortality evidence review. https://infrastructure.planninginspectorate.gov.uk/wp-content/ipc/uploads/projects/EN010098/EN010098-001144-Hornsea%20Project%20Four%20-%20Other-%20G2.9%20Gannet%20Displacement%20and%20Mortality%20Evidence%20Review.pdf

Band, W. (2012). Using a Collision Risk Model to Assess Bird Collision Risks for Offshore Wind Farms. Report to The Crown Estate.

Barrett, R.T., Anker-Nilssen, T., Gabrielsen, G.W. and Chapdelaine, G. (2002). Food consumption by seabirds in Norwegian waters. ICES Journal of Marine Science 59: 43-57.

Birt, V.L., Birt, T.P., Goulet, D., Cairns, D.K. and Montevecchi, W.A. (1987). Ashmole's halo: direct evidence for prey depletion by a seabird. Marine Ecology Progress Series 40: 205-208.

Black, J., Cook, A.S.C.P. and Anderson, O.R. (2019). Better estimates of collision mortality to black-legged kittiwakes at offshore wind farms. JNCC Report 644. JNCC, Peterborough.

Blackwell, P. and Matthiopoulos, J. (2024) Joint inference for telemetry and spatial survey data. Ecology. e4457. ISSN 0012-9658 https://doi.org/10.1002/ecy.4457

Bodey, T.W., Cleasby, I.R., Bell, F., Parr, N., Schultz, A., Votier, S.C. and Bearhop, S. (2017). A phylogenetically controlled meta-analysis of biologging device effects on birds: Deleterious effects and a call for more standardized reporting of study data. Methods in Ecology and Evolution 9: 946-955.

Boersch-Supan, P.H., Brighton, C.H., Thaxter, C.B. and Cook, A.S.C.P. (2024). Natural body size variation in seabirds provides a fundamental challenge for flight height determination by single-camera photogrammetry: a comment on Humphries et al. (2023). Marine Biology 171: 122.

Bogdanova, M.I., Wischnewski, S., Cleasby, I., Whyte, K., Regan, C., Gunn, C., Newell, M., Benninghaus, E., Langlois Lopez, S., Quintin, M., Witcutt, E., Kinchin-Smith, D., Holmes, E., Fox, D., Searle, K., Butler, A., Jones, E., McCluskie, Al. & Daunt, F. (2022) Seabird GPS tracking on the Isle of May, St Abbs Head and Fowlsheugh in 2021 in relation to offshore wind farms in the Forth/Tay region. Contract report to Neart na Gaoithe Offshore Wind Limited and SSE Renewables.

Borkenhagen, K., Corman, A.M. and Garthe, S. (2018). Estimating flight heights of seabirds using optical rangefinders and GPS dataloggers: a methodological comparison. Marine Biology 165: 17.

Buckingham L, Bogdanova MI, Green JA, Dunn RE and others (2022) Interspecific variation in non-breeding aggregation: a multi-colony tracking study of two sympatric seabirds. Mar Ecol Prog Ser 684:181-197. https://doi.org/10.3354/meps13960

Burnell, D., Perkins, A.J., Newton, S.F., Bolton, M., Tierney, T.D. and Dunn, T.E. (2023). Seabirds Count: A census of breeding seabirds in Britain and Ireland (2015-2021). Lynx Nature Books, Barcelona.

Burt, M.L., Mackenzie, M.L., Bradbury, G. and Darke, J. (2017). Investigating effects of shipping on common scoter and red-throated diver distributions in Liverpool Bay SPA. CREEM and WWT, report to Natural England.

Butler, A., Carroll, M., Searle, K., Bolton, M., Waggitt, J., Evans, P., Rehfisch, M., Goddard, B., Brewer, M., Burthe, S. and Daunt, F. (2020). Attributing seabirds at sea to appropriate breeding colonies and populations (CR/2015/18). Scottish Marine and Freshwater Science Vol 11 No 8, 140pp. DOI: 10.7489/2006-1

Carbon Trust, (2024). AppSaS – Apportioning seabirds seen-at-sea: WP3 - Method evaluation. Project Report, Offshore Renewables Joint Industry Programme (ORJIP) for Offshore Wind. https://ctprodstorageaccountp.blob.core.windows.net/prod-drupal-files/2024-03/ORJIP_AppSaS_WP3_Final.pdf

Carroll, M.J., Bolton, M., Owen, E., Anderson, G.Q.A., Mackley, E.K., Dunn, E.K. and Furness, R.W. (2017). Kittiwake breeding success in the southern North Sea correlates with prior sandeel fishing mortality. Aquatic Conservation: Marine and Freshwater Ecosystems 27: 1164-1175.

Cleasby, I.R., Wakefield, E.D., Bearhop, S., Bodey, T.W., Votier, S.C. and Hamer, K.C. (2015),.Three-dimensional tracking of a wide-ranging marine predator: flight heights and vulnerability to offshore wind farms. J Appl Ecol, 52: 1474-1482. https://doi.org/10.1111/1365-2664.12529

Collins, P.M., Green, J.A., Elliott, K.H., Shaw, P.J.A., Chivers, L., Hatch, S.A. and Halsey, L.G. (2020). Coping with the commute: behavioural responses to wind conditions in a foraging seabird. Journal of Avian Biology 51: e02057.

Cook, A.S.C.P. (2021). Additional analysis to inform SNCB recommendations regarding collision risk modelling: Report to Natural England. BTO Research Report 739. BTO, Thetford.

Cook, A.S.C.P., Humphreys, E.M., Masden, E.A. and Burton, N.H.K. (2014). The Avoidance Rates of Collision Between Birds and Offshore Turbines. Marine Scotland, Edinburgh.

Cook, A.S.C.P., Ward, R.M., Hansen, W.S. and Larsen, L. (2018). Estimating seabird flight height using LiDAR. Scottish Marine and Freshwater Science 9: 14.

Corman, A.M. and Garthe, S. (2014). What flight heights tell us about foraging and potential conflicts with wind farms: a case study in lesser black-backed gulls (Larus fuscus). Journal of Ornithology 155: 1037-1043.

Costa, R.A., Sá, S., Pereira, A.T., Angelo, A.R., Vaqueiro, J., Ferreira, M. and Eira, C. (2020). Prevalence of entanglements of seabirds in marine debris. Marine Pollution Bulletin 161: 111746.

Dagys, M. (2001). Anthropogenic effects on populations of breeding seabirds in Britain and Ireland: a ring recovery analysis. PhD thesis, University of Manchester.

Daunt, F., Benvenuti, S., Harris, M.P., Dall'Antonia, L., Elston, D.A. and Wanless, S. (2002). Foraging strategies of the black-legged kittiwake Rissa tridactyla at a North Sea colony: evidence for a maximum foraging range. Marine Ecology Progress Series 245: 239-247.

Department for Energy Security & Net Zero (2024). Habitats Regulations Assessment for an Application Under the Planning Act 2008 Sheringham Shoal and Dudgeon Extensions Offshore Wind Farm Projects Regulation 63, 64 and 68 of the Conservation of Habitats and Species Regulations 2017, Regulation 28, 29 and 36 of the Conservation of Offshore Marine Habitats and Species Regulations 2017 and Regulation 125 of the Marine and Coastal Access Act 2009.

Dierschke, V., Furness, R.W. and Garthe, S. (2016). Seabirds and offshore wind farms in European waters: avoidance and attraction. Biological Conservation 202: 59-68.

Dierschke, V., Furness, R.W., Gray, C.E., Petersen, I.K., Schmutz, J., Žydelis, R. and Daunt, F. (2017). Possible behavioural, energetic and demographic effects of displacement of red-throated divers. JNCC Report 605. JNCC, Peterborough.

Elliott, K.H., Chivers, L.S., Bessey, L., Gaston, A.J., Hatch, S.A., Kato, A., Osborne, O., Ropert-Coudert, Y., Speakman, J.R. and Hare, J.F. (2014). Windscapes shape seabird instantaneous energy costs but adult behavior buffers impact on offspring. Movement Ecology 2: 17.

Equinor (2022a). Sheringham Shoal and Dudgeon Offshore Wind Farm Extension Project Environmental Statement Chapter 11: Offshore Ornithology. PINS Document reference EN010109-000229- 6.1.11.

Equinor (2022b). Sheringham Shoal and Dudgeon Offshore Wind Farm Extension Projects Environmental Statement Vol. 3. Appendix 11.1 Offshore Ornithology Technical Report. PINS Document reference EN010109-000424-6.3.11.1.

Evans, T.J., Young, R.C., Watson, H., Olsson, O. and Åkesson, S. (2020). Effects of back-mounted biologgers on condition, diving and flight performance in a breeding seabird. Journal of Avian Biology 2020: e02509.

Fijn, R.C., Krijgsveld, K.L., Poot, M.J.M. and Dirksen, S. (2015). Bird movements at rotor height measured continuously with vertical radar at a Dutch offshore wind farm. Ibis 157: 558-566.

Fijn, R.C. and Collier, M.C. (2022). Distribution and flight heights of Sandwich terns *Thalasseus sandvicensis* during different behaviours near wind farms in the Netherlands. Bird Study 69: 53-58.

Fijn, R.C. and Gyimesi, A. (2018). Behaviour related flight speeds of Sandwich terns and their implications for wind farm collision rate modelling and impact assessment. Environmental Impact Assessment Review 71: 12-16.

Fijn, R.C., Thaxter, C.B., Aarts, G., Adema, J., Middelveld, R.P. and van Bemmelen, R.S.A. (2022). Relative effects of static and dynamic abiotic conditions on foraging behaviour in breeding Sandwich terns. Marine Ecology Progress Series 692: 137-150.

Forster, J., Macleod, K. and Scott, M. (2024). Natural body size variation in seabirds provides a fundamental challenge for flight height determination by single-camera photogrammetry Response. Marine Biology 171: 120.

Furness, R.W. (2015). Non-breeding season populations of seabirds in UK waters: Population sizes for Biologically Defined Minimum Population Scales (BDMPS). Natural England Commissioned Reports, Number 164. Natural England.

Furness, R.W., Wade, H. and Masden, E.A. (2013). Assessing vulnerability of marine bird populations to offshore wind farms. Journal of Environmental Management 119: 56-66.

Furness, R.W., Garthe, S., Trinder, M., Matthiopoulos, J., Wanless, S. and Jeglinski, J. (2018). Nocturnal flight activity of northern gannets Morus bassanus and implications for modelling collision risk at offshore wind farms. Environmental Impact Assessment Review 73: 1-6.

Galbraith, H., Russell, S. and Furness, R.W. (1981). Movements and mortality of Isle of May shags as shown by ringing recoveries. Ringing & Migration 3: 181-189.

Garthe, S. and Hüppop, O. (2004). Scaling possible adverse effects of marine wind farms on seabirds: developing and applying a vulnerability index. Journal of Applied Ecology 41: 724-734.

Garthe, S., Grémillet, D. and Furness, R.W. (1999). At-sea activity and foraging efficiency in chick-rearing northern gannets (Sula bassana): a case study in Shetland. Marine Ecology Progress Series 185: 93-99.

Garthe, S., Benvenuti, S. and Montevecchi, W.A. (2000). Pursuit plunging by northern gannets (Sula bassana) feeding on capelin (Mallotus villosus). Proceedings of the Royal Society B 267: 1717-1722.

Garthe, S., Benvenuti, S. and Montevecchi, W.A. (2003). Temporal patterns of foraging activities of northern gannets, Morus bassanus, in the northwest Atlantic Ocean. Canadian Journal of Zoology 81: 453-461.

Garthe, S., Montevecchi, W.A. and Davoren, G.K. (2007). Flight destinations and foraging behaviour of northern gannets (Sula bassana) preying on a small forage fish in a low-Arctic ecosystem. Deep-Sea Research II 54: 311-320.

Garthe, S., Ludynia, K., Hüppop, O., Kubetzki, U., Meraz, J.F. and Furness, R.W. (2012). Energy budgets reveal equal benefits of varied migration strategies in northern gannets. Marine Biology 159: 1907-1915.

Garthe, S., Guse, N., Montevecchi, W.A., Rail, J-F. and Grégoire, F. (2014). The daily catch: flight altitude and diving behavior of northern gannets feeding on Atlantic mackerel. Journal of Sea Research 85: 456-462.

Garthe, S., Markones, N. and Corman, A.M. (2017). Possible impacts of offshore wind farms on seabirds: a pilot study in northern gannets in the southern North Sea. Journal of Ornithology 158: 345-349.

Götmark, F. (1980). Foraging flights of kittiwakes – some functional aspects. Vår Fågelvärld 39: 65-74.

Greig, S.A., Coulson, J.C. and Monaghan, P. (1983). Age-related differences in foraging success in the herring gull (Larus argentatus). Animal Behaviour 31: 1237-1243.

Hamer, K.C., Phillips, R.A., Wanless, S., Harris, M.P. and Wood, A.G. (2000). Foraging ranges, diets and feeding locations of gannets in the North Sea: evidence from satellite telemetry. Marine Ecology Progress Series 200: 257-264.

Hamer, K.C., Humphreys, E.M., Garthe, S., Hennicke, J., Peters, G., Grémillet, D., Phillips, R.A., Harris, M.P. and Wanless, S. (2007). Annual variation in diets, feeding locations and foraging behaviour of gannets in the North Sea: flexibility, consistency and constraint. Marine Ecology Progress Series 338: 295-305.

Heggøy, O. (2013). "Short-term effects of data loggers on behaviour and physiology of two species of seabirds : The black-legged kittiwake Rissa tridactyla and the common guillemot Uria aalge." Masters Thesis. Norwegian University of Science and Technology.

Heggøy, O., Christensen-Dalsgaard, S., Ranke, P.S., Chastel, O. and Bech, C. (2015). GPS-loggers influence behaviour and physiology in the black-legged kittiwake Rissa tridactyla. Marine Ecology Progress Series 521: 237-248.

Heinänen, S. and Skov, H. (2018). Offshore Wind Farm Eneco Luchterduinen Ecological Monitoring of Seabirds T3 (Final) Report. IfAö and DHI.

Heinänen, S., Žydelis, R., Dorsch, M., Nehls, G., Kleinschmidt, B., Quillfeldt, P. and Morkūnas, J. (2016). Distribution modelling of red-throated diver based on aerial digital surveys and hydrodynamics. International Workshop on red-throated divers, Hamburg, 24-25 November 2016.

Heinänen, S., Žydelis, R., Kleinschmidt, B., Dorsch, M., Burger, C., Morkūnas, J., Quillfeldt, P. and Nehls, G. (2020). Satellite telemetry and digital aerial surveys show strong displacement of red-throated divers (Gavia stellata) from offshore wind farms. Marine Environmental Research 160: 104989.

Horswill, C. and Robinson, R.A. (2015). Review of seabird demographic rates and density dependence. JNCC Report No. 552. JNCC, Peterborough.

Horswill, C., Warwick-Evans, V., Esmonde, N. P. G., Reid, N., Kirk, H., Siddiqi-Davies, K. R., Josey, S. A., & Wood, M. J. (2023). Interpopulation differences and temporal synchrony in rates of adult survival between two seabird colonies that differ in population size and distance to foraging grounds. Ecology and Evolution, 13, e10455. https://doi.org/10.1002/ece3.10455

Humphries, G.R.W., Fail, T., Watson, M., Houghton, W., Peters-Grundy, R., Scott, M., Thomson, R., Keogan, K. and Webb, A. (2023). Aerial photogrammetry of seabirds from digital aerial video images using relative change in size to estimate flight height. Marine Biology 170: 18.

ICES (2023). Herring Assessment Working Group for the area south of 62oN (HAWG). ICES Scientific Reports 5: 23. ICES, Copenhagen.

IFC, EBRD and KfW (2023). Post-construction bird and bat fatality monitoring for onshore wind energy facilities in emerging market countries. Good practice handbook and decision support tool. IFC, EBRD and KfW, Washington, London and Frankfurt.

Jarrett, D., Cook, A.S.C.P., Woodward, I., Ross, K., Horswill, C., Dadam, D. and Humphreys, E.M. (2018). Short-term behavioural responses of wintering waterbirds to marine activity (CR/2015/17). Scottish Marine and Freshwater Science 9:7.

Jitlal, M. (2017). Testing and validating metrics of change produced by population viability analysis (PVA). Scottish Marine and Freshwater Science 8: 23.

JNCC (2024). Review of data used to calculate avoidance rates for collision risk modelling of seabirds 2023. Review of data used to calculate avoidance rates for collision risk modelling of seabirds | JNCC Resource Hub

JNCC, Natural England, Natural Resources Wales, NatureScot (2024). Joint advice note from the statutory nature conservation bodies (SNCBs) regarding bird collision risk modelling for offshore wind developments. JNCC, Peterborough.

Johnston, A., Cook, A.S.C.P., Wright, L.J., Humphreys, E.M. and Burton, N.H.K. (2014). Modelling flight heights of marine birds to more accurately assess collision risk with offshore wind turbines. Journal of Applied Ecology 51: 31-41, and associated corrigendum.

Johnston, D.T., Thaxter, C.B., Boersch-Supan, P.H., Davies, J.G., Clewley, G.D., Green, R.M.W., Shamoun-Baranes, J., Cook, A.S.C.P., Burton, N.H.K. and Humphreys, E.M. (2023). Flight heights obtained from GPS versus altimeters influence estimates of collision risk with offshore wind turbines in lesser black-backed gulls Larus fuscus. Movement Ecology 11: 66.

Jones et al (2024). Software reference github page - https://github.com/thecarbontrust/ANBS

Kogure, Y., Sato, K., Watanuki, Y, Wanless, S. and Daunt. F. (2016). European shags optimize their flight behavior according to wind conditions. Journal of Experimental Biology 219: 311-318.

Kotzerka, J., Garthe, S. and Hatch, S.A. (2010). GPS tracking devices reveal foraging strategies of black-legged kittiwakes. Journal of Ornithology 151: 459-467.

Krijgsveld, K.L., Fijn, R.C., Japink, M., van Horssen, P.W., Heunks, C., Collier, M.P., Poot, M.J.M., Beuker, D. and Dirksen, S. (2011). Effect Studies Offshore Wind Farm Egmond aan Zee. Final Report on Fluxes, Flight Altitudes and Behaviour of Flying Birds. Bureau Waardenburg Report No. 10-219, NZW-Report R231T1 flux&flight. Bureau Waardenburg, Culmeborg.

Lamb J, J Gulka, E Adams, A Cook, KA. Williams. (2024). A synthetic analysis of post-construction displacement and attraction of marine birds at offshore wind energy installations. Environmental Impact Assessment Review, Volume 108, 2024, 107611, ISSN 0195-9255, https://doi.org/10.1016/j.eiar.2024.107611.

Lane, J.V., Spracklen, D.V. and Hamer, K.C. (2019). Effects of windscape on three-dimensional foraging behaviour in a wide-ranging marine predator, the northern gannet. Marine Ecology Progress Series 628: 183-193.

Langston, R.H.W., Teuten, E. and Butler, A. (2013). Foraging ranges of northern gannets Morus bassanus in relation to proposed offshore wind farms in the UK. Report to DECC; reference DECC URN:13D/306.

Layton-Matthews et al. (2023a). Layton-Matthews, K., Buckingham, L., Critchley, E. J., Nilsson, A. L. K., Ollus, V. M. S., Ballesteros, M., Christensen-Dalsgaard, S., Dehnhard, N., Fauchald, P., Hanssen, F., Helberg, M., Masden, E., May, R. F., Sandvik, H., Tarroux, A., & Reiertsen, T. K. (2023a). Development of a Cumulative Impact Assessment tool for birds in Norwegian Offshore Waters: Trollvind OWF as a case study. https://brage.nina.no/nina-xmlui/handle/11250/3073159

Layton-Matthews et al. (2023b). Layton-Matthews, K., Reiertsen, T. K., Erikstad, K.-E., Anker-Nilssen, T., Daunt, F., Wanless, S., Barrett, R. T., Newell, M. A., & Harris, M. P. (2023b). Consequences of cross-season demographic correlations for population viability. Ecology and Evolution, 13, e10312. https://doi.org/10.1002/ece3.10312

Leemans, J.J., van Bemmelen, R.S.A., Middelveld, R.P., Kraal, J., Bravo Rebolledo, E.L., Beuker, D., Kuiper, K. and Gyimesi, A. (2022). Bird fluxes, flight- and avoidance behaviour of birds in offshore wind farm Luchterduinen. Bureau Waardenburg Report No. 22-078. Bureau Waardenburg, Culemborg.

Leopold, M.F. (2018). Common guillemots and offshore wind farms: an ecological discussion of statistical analyses conducted by Alain F. Zuur (WOZEP Birds-1). Wageningen University & Research report C093/18. Wageningen Marine Research, Den Helder.

Leopold, M.F., van Bemmelen, R.S.A. and Zuur, A.F. (2013). Responses of local birds to the offshore wind farms PAWP and OWEZ off the Dutch mainland coast. IMARES Report No. C151/12. IMARES, Wageningen.

Lindegren, M., van Deurs, M., MacKenzie, B.R., Clausen, L.W., Christensen, A. and Rindorf, A. (2018). Productivity and recovery of forage fish under climate change and fishing: North Sea sandeel as a case study. Fisheries Oceanography 27: 212-221.

Masden, E.A. (2015). Developing an avian collision risk model to incorporate variability and uncertainty. Scottish Marine and Freshwater Science 6: 14.

Masden, E.A., Cook, A.S.C.P., McCluskie, A., Bouten, W., Burton, N.H.K. and Thaxter, C.B. (2021). When speed matters: the importance of flight speed in an avian collision risk model. Environmental Impact Assessment Review 90: 106622.

Masden, E. and Cook, A. (2016). Avian Collision Risk Models for Wind Energy Impact Assessments. Environmental Impact Assessment Review, 56, 43-49. https://doi.org/10.1016/j.eiar.2015.09.001

Mateos-Rodríguez, M. and Bruderer, B. (2012). Flight speeds of migrating seabirds in the Strait of Gibraltar and their relation to wind. Journal of Ornithology 153: 881-889.

McGregor, R., King, S., Donovan, C., Caneco, B. and Webb, A. (2018). A stochastic Collision Risk Model for seabirds in flight. Report No. HC0010-400-001, Marine Scotland Science.

Mendel, B., Kotzerka, J., Sommerfeld, J., Schwemmer, H., Sonntag, N. and Garthe, S. (2014). Effects of the alpha ventus offshore test site on distribution patterns, behaviour and flight heights of seabirds. pp. 95-110 In: Ecological Research at the Offshore Wind Farm alpha ventus. Springer Fachmedien, Wiesbaden.

Mendel, B., Schwemmer, P., Peschko, V., Mūller, S., Schwemmer, H., Mercker, M. and Garthe, S. (2019). Operational offshore wind farms and associated ship traffic cause profound changes in distribution patterns of loons (Gavia spp.). Journal of Environmental Management 231: 429-438.

Merrall, E., Green, J.A., Robinson, L.A., Butler, A., Wood, M.J., Newell, M.A., Black, J., Daunt, F. and Horswill, C. (2024). Incorporating density-dependent regulation into impact assessments for seabirds. Journal of Applied Ecology doi:10.1111/1365-2664.14750.

NatureScot (SNH) (2018) Interim guidance on apportioning impacts from marine renewable development to breeding seabird populations in SPAs. https://www.nature.scot/doc/interim-guidance-apportioning-impacts-marine-renewable-developments-breeding-seabird-populations [Accessed 14/11/2025]

Natural England (2022). Natural England interim advice on updated Collision Risk Modelling parameters (July 2022).

NatureScot (2023a). Guidance Note 1: Guidance to support Offshore Wind Applications: Marine Ornithology – Overview. NatureScot, Inverness. Advice on marine renewables development | NatureScot

NatureScot (2023b). Guidance Note 2: Guidance to support Offshore Wind Applications: Advice for Marine Ornithology Baseline Characterisation Surveys and Reporting. NatureScot, Inverness. Advice on marine renewables development | NatureScot

NatureScot (2023c). Guidance Note 3: Guidance to support Offshore Wind Applications: Marine Birds – Identifying theoretical connectivity with breeding site Special Protection Areas using breeding season foraging ranges. NatureScot, Inverness. Advice on marine renewables development | NatureScot

NatureScot (2023d). Guidance Note 4: Guidance to support Offshore Wind Applications: Ornithology – Determining Connectivity of Marine Birds with Marine Special Protection Areas and Breeding Seabirds from Colony SPAs in the Non-Breeding Season. NatureScot, Inverness. Advice on marine renewables development | NatureScot

NatureScot (2023e). Guidance Note 5: Guidance to support Offshore Wind Applications: Recommendations for marine bird population estimates. NatureScot, Inverness. Advice on marine renewables development | NatureScot

NatureScot (2023f). Guidance Note 6: Guidance to support Offshore Wind Applications: Marine Ornithology Impact Pathways for Offshore Wind Developments. NatureScot, Inverness. Advice on marine renewables development | NatureScot

NatureScot (2023g). Guidance Note 7: Guidance to support Offshore Wind Applications: Marine Ornithology – Advice for assessing collision risk of marine birds. NatureScot, Inverness. Advice on marine renewables development | NatureScot

NatureScot (2023h). Guidance Note 8: Guidance to support Offshore Wind Applications: Marine Ornithology Advice for assessing the distributional responses, displacement and barrier effects of Marine birds. NatureScot, Inverness. Advice on marine renewables development | NatureScot

NatureScot (2023i). Guidance Note 9: Guidance to support Offshore Wind Applications: Marine Ornithology Advice for Seasonal Definitions for Birds in the Scottish Marine Environment. NatureScot, Inverness. Advice on marine renewables development | NatureScot

NatureScot (2023j). Guidance Note 10: Guidance to support Offshore Wind Applications: Marine Ornithology Advice for apportioning impacts to breeding colonies. NatureScot, Inverness. Advice on marine renewables development | NatureScot

NatureScot (2023k). Guidance Note 11: Guidance to support Offshore Wind Applications: Marine Ornithology – Recommendations for Seabird Population Viability Analysis (PVA). NatureScot, Inverness. Advice on marine renewables development | NatureScot

Nelson, E., Caryl, F. and Vallejo, G. (2015). Analysis of marine ecology monitoring plan data – Robin Rigg offshore wind farm. Operational year five technical report – Ornithological monitoring. Natural Power Ltd Report No. 1101321 to E.ON Climate and Renewables Ltd.

Orsted (2021). Hornsea Project Four: Environmental Statement (ES) PINS Document Reference: A5.5.1. Volume A5, Annex 5.1: Offshore and Intertidal Ornithology.

Ozsanlav-Harris, L., Inger, R. and Sherley, R. (2023). Review of data used to calculate avoidance rates for collision risk modelling of seabirds. JNCC Report 732. JNCC, Peterborough. https://hub.jncc.gov.uk/assets/de5903fe-81c5-4a37-a5bc-387cf704924d

Parker et al. (2022a). Parker, J., Banks, A., Fawcett, A., Axelsson, M., Rowell, H., Allen, S., Ludgate, C., Humphrey, O., Baker, A. and Copley, V. 2022a. Offshore Wind Marine Environmental Assessments: Best Practice Advice for Evidence and Data Standards. Phase 1: Expectations for pre-application baseline data for designated nature conservation and landscape receptors to support offshore wind applications. Natural England, Worcester. Version 1.1.

Parker et al. (2022b). Parker, J., Banks, A., Brown, E. and Copley, V. 2022b. Offshore Wind Marine Environmental Assessments: Best Practice Advice for Evidence and Data Standards. Phase II: Expectations for pre-application engagement and best practice advice for the evidence plan process. Natural England, Worcester. Version 1.1.

Parker et al. (2022c). Parker, J., Fawcett, A., Banks, A., Rowson, T., Allen, S., Rowell, H., Harwood, A., Ludgate, C., Humphrey, O., Axelsson, M., Baker, A. and Copley, V. 2022c. Offshore Wind Marine Environmental Assessments: Best Practice Advice for Evidence and Data Standards. Phase III: Expectations for data analysis and presentation at examination for offshore wind applications. Natural England, Worcester. Version 1.2.

Parker et al. (2022d). Parker, J., Fawcett, A., Rowson, T., Allen, S., Hodgkiss, R., Harwood, A., Caldow, R., Ludgate, C., Humphrey, O., and Copley, V. 2022d. Offshore Wind Marine Environmental Assessments: Best Practice Advice for Evidence and Data Standards. Phase IV: Expectations for monitoring and environmental requirements at the post-consent phase. Natural England, Worcester. Version 1.0.

Pavat, D., Harker, A.J., Humphries, G., Keogan, K., Webb, A. and Macleod, K. (2023). Consideration of avoidance behaviour of northern gannet (Morus bassanus) in collision risk modelling for offshore wind farm impact assessments. Natural England Commissioned Report 512. Natural England.

Pennington, M., Osborn, K., Harvey, P., Riddington, R., Okill, D., Ellis, P. and Heubeck, M. (2004). The Birds of Shetland. Christopher Helm, London.

Pennycuick, C.J. (1987). Flight of auks (Alcidae) and other northern seabirds compared with southern procellariiformes – ornithodolite observations. Journal of Experimental Biology 128: 335-347.

Pennycuick, C.J. (1997). Actual and "optimum" flight speeds: field data reassessed. Journal of Experimental Biology 200: 2355-2361.

Pennycuick, C.J., Fast, P.L.F., Ballerstädt, N. and Rattenborg, N. (2012). The effect of an external transmitter on the drag coefficient of a bird's body, and hence on migration range, and energy reserves after migration. Journal of Ornithology 153: 633-644.

Pennycuick, C.J., åkesson, S. and Hedenström, A. (2013). Air speeds of migrating birds observed by ornithodolite and compared with predictions from flight theory. Journal of the Royal Society Interface 10 (86) 20130419.

Percival, S. (2010). Kentish Flats offshore wind farm: diver surveys 2009-10. Report to Vattenfall.

Percival, S. (2013). Thanet offshore wind farm. Ornithological monitoring 2012-13 final report. Report to Thanet Offshore Wind Ltd. Ecology Consulting, Durham.

Percival, S. (2014). Kentish Flats offshore wind farm: diver surveys 2011-12 and 2012-13. Report to Vattenfall.

Perrow, M.R., Skeate, E.R. and Gilroy, J.J. (2011). Visual tracking from a rigid-hulled inflatable boat to determine foraging movements of breeding terns. Journal of Field Ornithology 82: 68-79.

Peschko (2020a). Peschko, V., Mendel, B., Mūller, S., Markones, N., Mercker, M. and Garthe, S. 2020a. Effects of offshore windfarms on seabird abundance: strong effects in spring and in the breeding season. Marine Environmental Research 162: 105157.

Peschko (2020b). Peschko, V., Mercker, M. and Garthe, S. 2020b. Telemetry reveals strong effects of offshore wind farms on behaviour and habitat use of common guillemots (Uria aalge) during the breeding season. Marine Biology 167: 118.

Peschko, V., Mendel, B., Mercker, M., Dierschke, J. and Garthe, S. (2021). Northern gannets (Morus bassanus) are strongly affected by operating offshore wind farms during the breeding season. Journal of Environmental Management 279: 111509.

Peschko, V., Schwemmer, H., Mercker, M., Markones, N., Borkenhagen, K. and Garthe, S. (2024). Cumulative effects of offshore wind farms on common guillemots (Uria aalge) in the southern North Sea – climate versus biodiversity? Biodiversity and Conservation 33: 949-970.

Petersen, I.K., Christensen, T.K., Kahlert, J., Desholm, M. and Fox, A.D. (2006). Final results of bird studies at the offshore wind farms at Nysted and Horns Rev, Denmark. NERI Report, commissioned by DONG energy and Vattenfall A/S.

Petersen, I.K., Nielsen, R.D. and Mackenzie, M.L. (2014). Post-construction evaluation of bird abundances and distributions in the Horns Rev 2 offshore wind farm area, 2011 and 2012. Report to DONG Energy. Aarhus University, Aarhus.

Pollock, C.J., Johnston, D.T., Boersch-Supan, P.H., Thaxter, C.B., Humphreys, E.M., O'Hanlon, N.J., Clewley, G.D., Weston, E.D., Shamoun-Baranes, J. and Cook, A.S.C.P. (2024). Avoidance and attraction responses of kittiwakes to three offshore wind farms in the North Sea. Marine Biology 171: 217.

Regan, C.E., Bogdanova, M.I., Newell, M. C Gunn, S Wanless, MP. Harris, S Langlois Lopez, E Benninghaus, M Bolton, F Daunt & KR. Searle. (2024). Seabirds show foraging site and route fidelity but demonstrate flexibility in response to local information. Mov Ecol 12, 46 (2024). https://doi.org/10.1186/s40462-024-00467-9

Redfern, C and Bevan, R (2014). A comparison of foraging behaviour in the North Sea by Black-legged Kittiwakes *Rissa tridactyla* from an inland and a maritime colony. Bird Study 61, 17-28. https://doi.org/10.1080/00063657.2013.874977

Rehfisch, M., Barrett, Z., Brown, L., Buisson, R., Perez-Dominguez, R. and Clough, S. (2014). Assessing northern gannet avoidance of offshore wind farms. APEM, Cambridge.

Riotte-Lambert, L. and Weimerskirch, H. (2013). Do naïve juvenile seabirds forage differently from adults? Proceedings of the Royal Society B 280: 20131434.

Saraux, C., Sydeman, W.J., Piatt, J.F., Anker-Nilssen, T., Hentati-Sundberg, J. et al. (2020). Seabird-induced natural mortality of forage fishes varies with fish abundance: evidence from five ecosystems. Fish and Fisheries 22: 262-279.

Schaub, T., Millon, A., De Zutter, C., Buij, R., Chadoeuf, J., Lee, S.M., Mionnet, A. and Klaassen, R.H.G. (2023). How to improve the accuracy of height data from bird tracking devices? An assessment of high-frequency GPS tracking and barometric altimetry in field conditions. Animal Biotelemetry 11: 31.

Schneider, S.R., Kramer, S.H., Bernstein, S.B., Terrill, S.B., Ainley, D.G. and Matzner, S. (2024). Autonomous thermal tracking reveals spatiotemporal patterns of seabird activity relevant to interactions with floating offshore wind facilities. Frontiers in Marine Science 11: 1346758.

Searle, K. R., Mobbs, D. C., Butler, A., Furness, R. W., Trinder, M. N., and Daunt, F. (2018) Finding out the Fate of Displaced Birds. Scottish Marine and Freshwater Science Vol 9 No 8, 149pp.

Searle, K.R., Mobbs, D., Daunt, F. and Butler, A. (2019). A population viability analysis modelling tool for seabird species. CEH report to Natural England. Natural England Commissioned Report NECR 274.

Searle, K.R., Jones, E.L., Trinder, M., McGregor, R., Donovan, C., Cook, A., Daunt, F., Humphries, L., Masden, E., McCluskie, A. and Butler, A. (2021). Correct treatment of uncertainty in ornithological assessments. JNCC Report No. 677. JNCC, Peterborough.

Searle et al. (2023a). Searle KR, S H O'Brien, E L Jones, A S C P Cook, M N Trinder, R M McGregor, C Donovan, A McCluskie, F Daunt & A Butler. 2023a. A framework for improving treatment of uncertainty in offshore wind assessments for protected marine birds. ICES Journal of Marine Science, fsad025, https://doi.org/10.1093/icesjms/fsad025

Searle et al. (2023b). Searle, KR, CE. Regan, MR. Perrow, A Butler, A Rindorf, MP. Harris, MA. Newell, S Wanless & F Daunt. 2023b. Effects of a fishery closure and prey abundance on seabird diet and breeding success: Implications for strategic fisheries management and seabird conservation. Biological Conservation: 281: 109990, ISSN 0006-3207, https://doi.org/10.1016/j.biocon.2023.109990.

Searle, KR, F Daunt, C. Pollock & A. Butler. (2024). Quantifying Mortality Rates of Displaced Birds: Summary Report to ORJIP. In press.

Skov, H., Leonhard, S.B., Heinänen, S., Zydelis, R., Jensen, N.E., Durinck, J., Johansen, T.W., Jensen, B.P., Hansen, B.L., Piper, W. and Grøn, P.N. (2012). Horns Rev 2 monitoring 2010-2012. Migrating birds. Orbicon, DHI, Marine Observers and Biola. Report to DONG Energy.

Skov, H., Heinänen, S., Norman, T., Ward, R., Méndez-Roldán, S. and Ellis, I. (2018). ORJIP Bird Collision and Avoidance Study. Final Report – April 2018. The Carbon Trust, United Kingdom.

SNCB (2014). Joint response from the Statutory Nature Conservation Bodies to Marine Scotland Science avoidance rate review, 2014.

SNCB (2017). Joint SNCB interim displacement advice note. Advice on how to present assessment information on the extent and potential consequences of seabird displacement from offshore wind farm (OWF) developments. https://hub.jncc.gov.uk/assets/9aecb87c-80c5-4cfb-9102-39f0228dcc9a

SNCB (2022a). Joint SNCB interim displacement advice note. Advice on how to present assessment information on the extent and potential consequences of seabird displacement from offshore wind farm (OWF) developments. https://hub.jncc.gov.uk/assets/9aecb87c-80c5-4cfb-9102-39f0228dcc9a

SNCB (2022b). Joint SNCB interim advice on the treatment of displacement for red-throated divers. https://hub.jncc.gov.uk/assets/9aecb87c-80c5-4cfb-9102-39f0228dcc9a

Snow, D.W. and Perrins, C.M. (Eds.) (1998). The Birds of the Western Palearctic, concise edition, Volume 1 Non-passerines. Oxford University Press, Oxford.

SSE Renewables (2022). Berwick Bank Wind Farm Offshore Environmental Impact Assessment Appendix 11.1: Baseline Ornithology Technical Report.

Symons, S.C. and Diamond, A.W. (2019). Short-term tracking tag attachment disrupts chick provisioning by Atlantic puffins Fratercula arctica and razorbills Alca torda. Bird Study 66: 53-63.

Tjørnløv, R.S., Skov, H., Armitage, M., Barker, M., Jørgensen, J.B., Mortensen, L.O., Thomas, K. and Uhrenholdt, T. (2023). Resolving key uncertainties of seabird flight and avoidance behaviours at offshore wind farms. Final report for the study period 2020-2021. Vattenfall, Aberdeen. European Offshore Wind Deployment Centre - Vattenfall

Trinder, M., O'Brien, S.H. and Deimel, J. (2024). A new method for quantifying redistribution of seabirds within operational offshore wind farms finds no evidence of within-wind farm displacement. Frontiers in Marine Science 11: 1235061.

Vallejo, G.C., Grellier, K., Nelson, E.J., McGregor, R.M., Canning, S.J., Caryl, F.M. and McLean, N. (2017). Responses of two marine top predators to an offshore wind farm. Ecology and Evolution 2017: 1-11.

van Bemmelen, R.S.A., Leemans, J.J., Collier, M.P., Green, R.M.W., Middelveld, R.P., Thaxter, C.B. and Fijn, R.C. (2024). Avoidance of offshore wind farms by Sandwich terns increases with turbine density. Ornithological Applications 126: duad055.

van Erp, J.A., Sage, E., Bouten, W., van Loon, E., Camphuysen, C.J. and Shamoun-Baranes, J. (2023). Thermal soaring over the North Sea and implications for wind farm interactions. Marine Ecology Progress Serries 723: 185-200.

van Erp, J.A., van Loon, E.E., De Grove, J., Bradaric, M. and Shamoun-Baranes, J. (2024). A framework for post-processing bird tracks from automated tracking radar systems. Methods in Ecology and Evolution 15: 130-143.

Vandenabeele, S.P., Shepard, E.L., Grogan, A. and Wilson, R.P. (2012). When three percent may not be three per cent; device-equipped seabirds experience variable flight constraints. Marine Biology 159: 1-14.

Vanermen, N., Onkelinx, T., Courtens, W., van de Walle, M., Verstraete, H. and Steinen, E.W.M. (2015). Seabird avoidance and attraction at an offshore wind farm in the Belgian part of the North Sea. Hydrobiologia 756: 51-61.

Vanermen, N., Courtens, W., van de Walle, M., Verstraete, H. and Steinen, E.W.M. (2016). Seabird monitoring at offshore wind farms in the Belgian part of the North Sea. Updated results for the Bligh Bank and first results for the Thorntonbank. Instituut voor Natuur en Bosonderzoek.

Vanermen, N., Wouter, C., van de Walle, M., Verstraete, H. and Steinen, E.W.M. (2021). Belgian seabird displacement monitoring program. Macro-avoidance of GPS-tagged lesser black-backed gulls and potential habituation of auks and gannets. In: Memoirs on the Marine Environment. Environmental Impacts of Offshore Wind Farms in the Belgian part of the North Sea.

Vanermen, N., Coutens, W. van de Walle, M., Verstraete, H. and Stienen, E. (2023). Seabirds and offshore wind farms – displacement monitoring 2.0. Chapter 5 in Degraer, S., Brabant, R., Rumes, B. and Vigin, L. (Eds.) Environmental Impacts of Offshore Wind Farms in the Belgian Part of the North Sea: Progressive Insights in Changing Species Distribution Patterns Informing Marine Management. Memoirs on the Marine Environment. Royal Belgian Institute of Natural Sciences, Brussels.

Vilela, R., Burger, C., Diederichs, A., Nehls, G., Bachl, F., Szostek, L., Freund, A., Braasch, A., Bellebaum, J., Beckers, B. and Piper, W. (2020). Divers (Gavia spp.) in the German North Sea: changes in abundance and effects of offshore wind farms. A study into diver abundance and distribution based on aerial survey data in the German North Sea. Report to Bundesverband der Windparkbetreiber Offshore e.V.

Wakefield, E.D., Owen, E., Baer, J., Carroll, M.J., Daunt, F., Dodd, S.G., Green, J.A., Guilford, T., Mavor, R.A., Miller, P.I., Newell, M.A., Newton, S.F., Robertson, G.S., Shoji, A., Soanes, L.M., Votier, S.C., Wanless, S. and Bolton, M. (2017), Breeding density, fine-scale tracking, and large-scale modeling reveal the regional distribution of four seabird species. Ecol Appl, 27: 2074-2091. https://doi.org/10.1002/eap.1591

Wakeling, J.M. and Hodgson, J. (1992). Optimisation of the flight speed of the little, common and Sandwich tern. Journal of Experimental Biology 169: 261-266.

Warwick-Evans, V., Atkinson, P.W., Gauvain, R.D., Robinson, L.A., Arnould, J.P.Y. and Green, J.A. (2015). Time-in-area represents foraging activity in a wide-ranging pelagic forager. Marine Ecology Progress Series 527: 233-246.

Warwick-Evans, V., Atkinson, P.W., Walkington, I. and Green, J.A. (2018). Predicting the impacts of windfarms on seabirds: an individual based model. Journal of Applied Ecology 55: 503-515.

Webb, A., Irwin, C., Mackenzie, M., Scott-Hayward, L., Caneco, B. and Donovan, C. (2016). Lincs Wind Farm: Third annual post-construction aerial ornithological monitoring report. HiDef Aerial Surveying Ltd report to Centrica Renewable Energy Ltd. Report CREL REF: LN-E-EV-013-0006-400013-007.

Weber, S.B., Richardson, A.J., Brown, J. and Broderick, A.C. (2021). Direct evidence of a prey depletion “halo” surrounding a pelagic predator colony. Proceedings of the National Academy of Science 118: e2101325118.

Welcker, J. and Nehls, G. (2016). Displacement of seabirds by an offshore wind farm in the North Sea. Marine Ecology Progress Series 554: 173-182.

Zydelis, R., Heinänen, S., Dorsch, M., Nehls, G., Kleinschmidt, B., Quillfeldt, P. and Morkūnas, J. (2016). High mobility of red-throated divers revealed by satellite telemetry. International Workshop on red-throated divers, Hamburg, 24-25 November 2016.

carbontrust.com

+44 (0) 20 7170 7000

The Carbon Trust is a company limited by guarantee and registered in England and Wales under Company number 04190230 with its Registered Office at: Level 5, Arbor, 255 Blackfriars Road, London SE1 9AX, UK.

Published in the UK: 2025