Response to the Examining Authority's First Written Questions (ExQ2) Issued on 23 January 2023 for the Royal Society for the Protection of Birds

Submitted for Deadline 5
6 February 2023

Planning Act 2008 (as amended)

In the matter of:

Application by Awel y Môr Offshore Wind Farm Limited for an Order Granting Development Consent for the Awel y Môr Offshore Wind Farm

Planning Inspectorate Ref: EN010112
RSPB Registration Identification Ref: 20031695

Contents

1. Introduction

1.1. The RSPB's responses to the Examining Authority's Written questions (ExQ2) are set out in the table below.

2. Responses to the Examining Authority's Written Questions

ExQ2	Question to:	Question	RSPB response
2. Biodiversity, Ecology and Natural Environment			
2.2	NRW, DCC, CCBC, RSPB, NWWT	**General** Please advise if you have any issues with the potential mitigation measures in the Schedule of Mitigation [REP2-024] and Marine Licence Principles (REP2-022), and if issues exist, please reference with explanation and evidence to justify.	The current mitigation measures include a minimum blade clearance of the turbine blade of 22m above Mean High Water Springs, the minimum legal requirement (see 2.9. in UK Government Marine Guidance Note.) It has been demonstrated that a greater blade clearance results in lower predicted seabird collision mortalities (Johnstone *et al.,* 2014[1]) and as a consequence most recent offshore wind developments have a greater clearance. For example, for the Hornsea Four project, the Applicant has included a requirement that the clearance of each blade must not be less than 42.43m above LAT (see paragraph 2(2)(c) in Schedule 1, Part 3 (Requirements) of "REP7-039 C.1.1 Draft Development Consent Order (DCO)(Clean)" For the recently submitted Berwick Bank proposed offshore wind farm development, the minimum clearance has been set at 37m above LAT (see Description of Works in Section 36 Consent Application)

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[1] Johnston, A., Cook, A. S., Wright, L. J., Humphreys, E. M., & Burton, N. H. (2014). Modelling flight heights of marine birds to more accurately assess collision risk with offshore wind turbines. *Journal of Applied Ecology*, *51*(1), 31-41.

ExQ2	Question to:	Question	RSPB response
2.11	RSPB	**Offshore – Ornithology** With reference to your Written Representation additional references [REP2-058], please summarise the key issue(s) relevant to the proposed Awel y Môr development.	The additional references were those that we could not provide copies of in time for issuing with our WR. The references are cited in the WR as follows: • Burt, M.L., Mackenzie, M.L., Bradbury, G. and Darke, J. (2022) Investigating effects of shipping on common scoter and red-throated diver distributions in Liverpool Bay SPA. NECR425. Natural England See para 4.8 (Ref no. 59): Red-throated diver displacement effects from offshore windfarms. • Heinänen, S., Ramūnas, Ž., 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 See para 4.8 (Ref no. 58): Red-throated diver displacement effects from offshore windfarms • Lane, J.V. and Hamer, K.C. (2021) Annual adult survival and foraging of gannets at Bass Rock, Scotland: Report to the Ornithology subgroup of the Forth and Tay Regional Advisory Group (FTRAG-O) – October 2021. NnG Document number NNG-LUN-ECF-REP-0002 Rev 0.1

ExQ2	Question to:	Question	RSPB response
			Para 4.24 (ref no. 81): variation in the two-dimensional foraging behaviour of gannets
2.17	NRW, RSPB, Applicant	**Offshore – Ornithology (Collision Risk Modelling)** **For NRW and RSPB** a) Please advise if you have any issues related to collision risk modelling parameters; bird survey data; species data; turbine data; windfarm data; and avoidance rate. If any issues remain, please provide relevant evidence to justify. Could the **Applicant** provide relevant evidence to: b) explain the potential effect on its impact assessment if the collision risk model utilised an avoidance rate for gannet of 98%; and c) explain the potential effect on its impact assessment due to different foraging and behaviour of gannets during the breeding season.	The RSPB has provided details in their Written Representations of the issues around the avoidance rate for gannet used in collision risk modelling and the application of a macro-avoidance correction factor.
2.20	NRW, RSPB	**Offshore – Ornithology** Please comment on the Applicant's response to Written Representations [REP2-002] regarding Highly Pathogenic Avian Influenza (page 212) and Population Viability Analysis for gannet.	The Applicant response (REP2-002) suggests that the number of birds affected by the windfarm will remain as the same proportion of the BDMPS population as prior to the outbreak of HPAI. The contravenes the considerable evidence of inter colony segregation of seabird foraging locations. This evidence demonstrates that choice of foraging hotspots is

ExQ2	Question to:	Question	RSPB response
			determined in part by colony size[2]. This means that any changes in colony population size due to HPAI will lead to changes foraging site selection and potential changes to the numbers of birds using the development footprint and therefore at risk of impact This change will not necessarily be in proportion to BDMPS population change. With regard to the Population Viability Analysis, due to the impact of HPAI on the gannet population, the conclusion that incombination gannet mortality will be less that 1% of the baseline mortality is not robust until more up to date colony counts are carried out, as is planned for 2023.
2.21	RSPB	**Offshore - Ornithology** Please give an update regarding previous disagreement with the Applicant on: a) assessment of no adverse effect on integrity of Liverpool Bay SPA (project alone and in- combination with other plans and projects), for its feature red-throated diver; b) assessment for Manx Shearwater from Copeland Islands SPA, Irish Sea Front SPA, Glannau Aberdaron ac Ynys Enlli/Aberdaron Coast and Bardsey Island SPA, Skomer, Skokholm and the Seas off Pembrokeshire/Sgomer, Sgogwm a Moroedd Penfro SPA;	Since the submission of our Written Representations, Marine Scotland Science has published a review of potential wind farm impacts on Procellariformes[3]. This review highlights the potential for birds, such as Manx shearwaters, to be attracted to, and disorientated by, light, such as those fitted to wind turbines and associated infrastructure. Such attraction and disorientation will substantially increase collision risk and has not been considered at all by the Applicant. The RSPB remains in disagreement with the Applicant in regard to the application of a macro-avoidance

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[2] Bolton, M., Conolly, G., Carroll, M., Wakefield, E. D., & Caldow, R. (2019). A review of the occurrence of inter-colony segregation of seabird foraging areas and the implications for marine environmental impact assessment. *Ibis*, *161*(2), 241-259.

[3] Zoe Deakin, Aonghais Cook, Francis Daunt, Aly McCluskie, Nicola Morley, Emma Witcutt, Lucy Wright and Mark Bolton. (2022) A review to inform the assessment of the risk of collision and displacement in petrels and shearwaters from offshore wind developments in Scotland

ExQ2	Question to:	Question	RSPB response
		c) scoping out of collision impacts for Manx Shearwater at Copeland Islands SPA, Irish Sea Front SPA, Rum SPA, St Kilda SPA, Glannau Aberdaron ac Ynys Enlli/Aberdaron Coast and Bardsey Island SPA, and Skomer, Skokholm and the Seas off Pembrokeshire/Sgomer, Sgogwm a Moroedd Penfro SPA; d) assessment for gannets from Grassholm SPA, Ailsa Craig SPA, Saltee Islands SPA; and e) use of avoidance rates in gannet collision risk modelling If you consider your points have not been resolved in the Applicant's response to Written Representations [REP2-002] please provide relevant evidence and justification.	correction factor to gannet densities prior to collision risk modelling, and our rationale for this position is detailed in our Written Representations. Since the submission of these, NatureScot have produced updated Guidance to Support Offshore Wind Applications. Guidance Note 7, on assessing collision risk, does not recommend the application of this correction factor (The upcoming JNCC report on avoidance rates to use with the Stochastic CRM is highlighted, but this does not deal with the macro-avoidance correction factor).
2.23	RSPB	**Offshore – Ornithology** In your Written Representation [REP1-090], you raise a number of concerns about the methodology used to assess effects on gannets. Please describe, with appropriate supporting evidence, the methods which you think should be used.	For gannet collision risk, the RSPB recommend the use of the stochastic formulation of the Band collision risk model, as updated in 2022 (Caneco, 2022), with both deterministic and stochastic runs using monthly densities derived from site characterisation surveys. This should be run without applying the macro-avoidance correction factor. The model should use the Option 2, that is the basic model using generic flight height distributions[4] Bird parameters should be as follows. Flight speed from Pennycuick, (1997[5]). and Nocturnal Activity

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[4] Johnston, A., Cook, A. S., Wright, L. J., Humphreys, E. M., & Burton, N. H. (2014). Modelling flight heights of marine birds to more accurately assess collision risk with offshore wind turbines. *Journal of Applied Ecology*, *51*(1), 31-41.

[5] Pennycuick, C. 1997. Actual and 'optimum' flight speeds: field data reassessed. Journal of Experimental Biology. 200(17): 2355-2361

ExQ2	Question to:	Question	RSPB response
			Factor from Furness *et al*., (2018)[6]. Avoidance rate should be 98% for the breeding season and 99.2% for the non-breeding season, as evidenced in the Written Representations Displacement should be assessed using the matrix approach (SNCB 2022) with a displacement rate of 70% and mortality of 1 and 3%

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[6] Furness, R.W., Garthe, S., Trinder, M., Matthiopolous, J., Wanless, S., 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.

Bolton, M., Conolly, G., Carroll, M., Wakefield, E. D. and Caldow, R. (2019) A review of the occurrence of inter-colony segregation of seabird foraging areas and the implications for marine environmental impact assessment. *Ibis*, 161(2), pp. 241-259.

Bolton, M., Conolly, G., Carroll, M., Wakefield, E. D. and Caldow, R. (2019) A review of the occurrence of inter-colony segregation of seabird foraging areas and the implications for marine environmental impact assessment. *Ibis*, 161(2), pp. 241-259, which has been published in final form at [redacted]

Running head: seabird foraging segregation

A review of the occurrence of inter-colony segregation of seabird foraging areas and the implications for marine environmental impact assessment

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Understanding the determinants of species' distributions is a fundamental aim in ecology and a prerequisite for conservation, but is particularly challenging in the marine environment. Advances in bio-logging technology have resulted in a rapid increase in studies of seabird movement and distribution in recent years. Multi-colony studies examining effects of intra- and inter-colony competition on distribution have found that several species exhibit inter-colony segregation of foraging areas, rather than overlapping distributions. These findings are timely given the increasing rate of human exploitation of marine resources and the need to make robust assessments of likely impacts of proposed marine developments on biodiversity. Here we review the occurrence of foraging area segregation reported by published tracking studies in relation to the Density-Dependent Hinterland (DDH) model, which predicts that segregation occurs in response to inter-colony competition, itself a function of colony size, distance from the colony and prey distribution. We found that inter-colony foraging area segregation occurred in 79% of 39 studies. The frequency of occurrence was similar across the four seabird orders for which data were available, and included species with both smaller (10 – 100 km) and larger (100 – 1000 km) foraging ranges. Many predictions of the DDH model were met, with examples of segregation in response to high levels of inter-colony competition related to colony size and proximity, and enclosed landform restricting the extent of available habitat. Moreover, as predicted by the DDH model, inter-colony overlap tended to occur where birds aggregated in highly productive areas, often remote from all colonies. The apparent prevalence of inter-colony foraging segregation has important implications for assessment of impacts of marine development on protected seabird colonies. If a development area is accessible from multiple colonies, it may impact those colonies much more asymmetrically than previously supposed. Current impact assessment approaches that do not consider spatial inter-colony segregation will therefore be subject to error. We recommend the collection of tracking data from multiple colonies and modelling of inter-colony interactions to predict colony-specific distributions.

Keywords: central-place foraging, space partition, overlap, aggregation, competition

A fundamental goal in ecology and conservation is to understand the factors that drive patterns of avian distribution and abundance (Sutherland *et al.* 2009, Sutherland *et al.* 2013). Seabirds are more threatened, and their conservation status has deteriorated faster over recent decades, than any other comparable avian group (Croxall *et al.* 2012). During the breeding season, seabirds are central-place foragers, returning periodically to the nest site in order to provision and care for their offspring (Orians and Pearson 1979). In common with other central-place foragers (social insects, bats, pinnipeds, etc.), this constraint radically affects their spatial ecology (Bernstein & Gobbel 1979, Kacelnik 1984). Optimal foraging models commonly assume that animals are adapted to maximise the rate of net energy gain per unit time (Stephens & Krebs 1986). For a central-place forager, the costs of foraging measured in terms of either time or energy, increase with increasing distance from the colony. If prey are uniformly distributed and superabundant (i.e. there is no competition for prey) within the area surrounding the colony, the rate of energy gain and foraging efficiency will be highest close to the colony, where travel costs are lowest. However, if the number of foragers close to the colony is sufficient to reduce the per capita rate of prey capture through local prey depletion (Ashmole 1963), or through interference competition (Lewis *et al.* 2001), the rate and efficiency of energy gains close to the colony will diminish relative to unexploited areas that are more distant. Foraging seabirds do not exhibit territorial defence of areas of sea and may be assumed to follow an ideal free-distribution (Fretwell 1972), whereby the net energy gain is equalised across all individuals. The resulting distribution will represent a gradient of decreasing density of foragers with increasing distance from the colony, reflecting the increasing travel costs associated with foraging at more remote locations. The precise relationship between seabird density and distance from the colony will depend on surrounding coastal morphology, which will determine the extent of marine habitat (and hence competitor dilution) at increasing distance from the colony (Wakefield *et al.* 2017). However, in many situations prey are aggregated in patches rather than being uniformly distributed (Wakefield *et al.* 2009), modifying these theoretical distributions radically.

Ashmole (1963) hypothesised that central place foraging constraints impose an upper limit on colony size through the following mechanism: As a colony grows, increasing intra-specific competition close to the colony forces the use of more distant foraging areas. Mean travel costs will therefore increase, reducing net gains from foraging, until eventually a point is reached where breeding success is so low that colony growth falls to zero. This hypothesis led to the notion of colonies exploiting a “halo” of prey resources in the surrounding waters. Cairns’ (1989) hinterland model of colony foraging areas approached foraging optimality from a different perspective. It suggests that seabirds should only exploit areas of sea that lie closer to their home colony than to any other colony. He reasoned that seabirds should not regularly forage in waters which are closer to another colony, since it would be more efficient to exploit such areas from the closer colony. This would result in adjacent colonies having non-overlapping foraging ranges, bounded by lines of equidistance. Cairns (1989) suggested that in regions of uniform ocean productivity, the size of these hinterlands would determine the size of the associated colony. He found a positive correlation between theoretical hinterland size and colony size for European Shags *Phalacrocorax aristotelis* and Black-legged Kittiwakes *Rissa tridactyla*, but not for Northern Gannets *Morus bassanus* or Atlantic Puffins *Fratercula* *arctica*. There are several potential reasons for the lack of correlation in the latter species, principal of which is that their prey may be more patchily distributed (Weimerskirch 2007, Haury *et al.* 1977). Other reasons could be that some colonies are limited by nest site availability, or they may not be at equilibrium with food availability due to past persecution or unnaturally inflated food resources e.g. from fisheries' discards.

Where neighbouring colonies are separated by less than the combined foraging radius of each, foraging areas can potentially overlap. Although Ashmole (1963) did not explicitly consider how seabirds from neighbouring colonies might interact in areas of potential foraging overlap, his "halo" hypothesis suggests a circular region of seabird usage and prey depletion around each colony. More recent suggestions of seabird foraging distribution have tended to draw upon this image, assuming overlap of circular foraging areas accessible to multiple colonies (e.g. Grecian *et al.* 2012, Thaxter *et al.* 2012). Recent data obtained by tracking seabirds simultaneously from neighbouring colonies reveals that segregation of foraging areas does occur, and may be widespread. For example, a study of Northern Gannets from 12 colonies around Britain and Ireland (Wakefield *et al.* 2013) found that birds from different colonies occupied almost exclusive foraging areas, despite their potential foraging ranges overlapping. However, contrary to Cairns' (1989) hinterland model, boundaries between these areas were not equidistant from adjacent colonies. An alternative model was therefore proposed, termed the Density-Dependent Hinterland (DDH) model (Wakefield *et al*. 2013), which combines elements of both Ashmole's halo model and Cairns' hinterland model. In the DDH model, competition is assumed to be a function of both colony size and distance from the colony. Segregation of foraging areas of two neighbouring colonies will occur if potential competition is high. This is likely to be the case where colonies are close (due both to the imperative for central place foragers to minimise travel costs and the effect of radiative spreading from the colony) and when colonies are relatively large. Conversely, the DDH model predicts that hinterlands may overlap in areas where inter-colony competition is low. For example, this could occur in areas where prey are superabundant, where colonies are small or where they are distant from one another.

Segregated foraging grounds have been demonstrated for a diverse range of other colonial central-place foragers, including not only territorial groups such as ants (Brown & Gordon 2000, Adler & Gordon 2003, Schilder *et al.* 2004), but also species that, like seabirds, are non-territorial away from the colony, such as bats (Dawo *et al.* 2013, August *et al.* 2014, Christie and O'Donnell 2014), seals (Curtice *et al.* 2011, Kirkwood & Arnould 2012, Nordstrom *et al.* 2013, Kuhn *et al.* 2014) and corvids (Griffin & Thomas 2000). However, it is still unclear how widespread the phenomenon is in seabirds and whether the DDH model holds across divergent evolutionary lineages within this group. In part, this reflects the practical difficulties associated with establishing the patterns of space use by seabirds at sea. However, recent reductions in the weight and cost of tracking devices have led to a rapid increase in the number of tracking studies of breeding seabirds. It is therefore opportune to review the occurrence of inter-colony foraging segregation in seabirds. Here we: (i) review the peer-reviewed literature for examples of both the occurrence and absence of intra-specific inter-colony segregation of seabird foraging areas; (ii) assess the frequency of segregation across seabird taxonomic orders; (iii) examine suggested causes of segregation in the light of the DDH model and (iv) consider the implications of the phenomenon for seabird conservation.

OCCURRENCE OF INTRA-SPECIFIC INTER-COLONY SEGREGATION OF SEABIRD FORAGING AREAS

Literature search

Structured, systematic searches of the peer-reviewed scientific literature were carried out to identify publications reporting inter-colony spatial segregation or overlap. To reduce negative reporting bias, searches were also conducted for the absence of segregation. The literature search was focussed on publications documenting multiple colony tracking or colour marking studies, where colonies were separated by less than the combined maximum foraging ranges observed. Keywords were used to search Google Scholar and Web of Science for relevant publications. Combinations of the following keyword search terms were used: “spatial”, “space”, “segregate”, “partition”, “aggregate”, “mix”, “overlap”, “feed”, “forage”, “colony”, “seabird”, “area”, “location”, “inter-colony”, “multiple”, “tracking”, “territory”, “competition”, “bird”, “colour”, “mark” and “home range”. The ‘wildcard’ character (*) was used where appropriate to broaden search results. Web of Science results were filtered using different combinations of keywords until <100 results were returned; the number of results was recorded and results were searched for relevant studies. The number of Google Scholar results for each keyword combination was recorded, as was the number of pages searched. The first 10 pages of results were searched for relevant studies. The literature search was conducted in December 2017.

Multiple publications from the same dataset were considered as a single study. For publications that reported studies of multiple species, the presence or absence of segregation was recorded for each species separately. Study species were classified according to taxonomic order and family, following del Hoyo *et al.* (2014), and species nomenclature follows IOC World Bird Names (Gill & Donsker 2018). Where reported, the breeding stage(s) of the foraging individuals was also recorded. For several species, foraging ranges varied very considerably according to breeding stage. In such cases, we only considered breeding stages during which foraging range exceeded inter-colony spacing. The majority of studies identified involved simultaneous (same year and breeding stage) multi-colony tracking. However, we also included studies were the occurrence or absence of segregation was determined by inclusion of a measure of inter-colony competition (such as distance to neighbouring colonies) in a model of space-use. Such an approach does not require simultaneous (Wakefield *et al.* 2011) or multi-colony (Cecere *et al.* 2015) tracking.

Evidence of foraging area segregation

Many of the studies we reviewed present no formal statistical analysis to determine whether the observed pattern of distribution deviated significantly from a null distribution in which colony distributions overlapped without interaction. Rather, inferences and conclusions about segregation were often drawn from the percentage overlap in areas of distribution, or from visual inspections of tracks or kernel density distributions, but without explicit reference to any expected value. In cases where two colonies were separated by a distance substantially less than the sum of the maximum foraging range of both colonies and no, or negligible, overlap in distributions occurred, conclusions regarding segregation could still be drawn in the absence of any formal statistical analysis.

Some studies reported considerable overlap of foraging areas of birds from multiple colonies (often in locations of high biological productivity, at considerable distance from the breeding locations) which could be regarded as instances of “aggregation”, defined as a higher coincident density of birds from multiple colonies than would be expected from their null distributions. However, it was seldom possible to determine from published information whether the proportions of individuals from different colonies in such areas differed substantially from those predicted by the null distributions. We have therefore not attempted to differentiate instances of aggregation from overlapping null distributions, and distributions were classified as “segregated” or “overlapping” only. However, we recognise that there is considerable potential for the proportion of birds originating from contributing colonies to deviate substantially from those predicted by the null distributions in such cases.

We identified 40 papers that presented information on foraging areas of seabird colonies where ranges of neighbouring colonies could potentially overlap. We did not consider studies that examined foraging areas at sub-colony level only, such as Waggitt *et al.* (2014) and Bogdanova *et al.* (2014). One study used plumage dye marking to determine colony foraging grounds; the remainder used bird-borne tracking devices – either global positioning system (GPS), light-based geolocation (GLS), satellite (PTT), compass loggers or Very High Frequency radio (VHF) tags (Table 1). Some studies considered multiple species and some datasets were covered by several papers. Together they comprised 41 unique studies covering 30 seabird species (Table 1). Foraging area segregation was not a primary focus of all the studies reviewed, and the strength of evidence for conclusions regarding the occurrence or absence of segregation varied. We therefore adopted a tiered approach to the classification of foraging distribution in the studies reviewed (Table 2). The strongest evidence was provided by nine studies that conducted a formal statistical assessment of the occurrence of interactions in space-use by neighbouring colonies. Of these, none found evidence of a positive interaction (i.e. birds from neighbouring colonies aggregating at higher densities that expected); two studies found evidence that distributions overlapped as expected if no inter-colony interaction occurred, and the remaining seven found evidence of negative interactions (segregation). In two (Wakefield *et al.* 2011, Catry *et al.* 2013), segregation was temporally and/or spatially variable, occurring for some colonies and/or breeding stages only.

In a further 30 studies, the authors’ assessment of segregation was based on the extent of overlap calculated as percentage, or by visual inspection of distributions (the latter typically in cases were overlap was entirely absent or extremely low). Inter-colony segregation of foraging areas was judged to occur in 24 studies (temporally and/or spatially variable in eight cases), with overlap occurring in the remaining six studies. In two studies no assessment of the occurrence or absence of segregation was made by the authors. Taken together, 31 (79%)

of the 39 studies where inter-colony segregation was assessed, reported segregated foraging areas, of which 10 related to temporally and/or spatially variable segregation. The proportion of studies reporting segregation was similar for both evidence classes (78% for studies where colony interactions were statistically modelled and 80% for studies based on distribution overlap), which suggests the assessment of segregation is not strongly biased by the methods used.

Occurrence of foraging segregation across species, families and orders.

The occurrence of foraging segregation was reported for 24 of 29 species assessed. There were insufficient data to compare the frequency of occurrence of segregation across families and orders using models that account for phylogenetic non-independence (Grafen 1989, Martins & Hansen 1997). Nonetheless, we found that foraging segregation was widespread and occurred to a similar extent in all four orders, and across the eight families represented. Fig. 1 illustrates the number of studies reporting segregation by seabird order and family. There was evidence of segregation for all five species of Sphenisciforme, for nine of 12 Procellariiforme species, seven of eight Suliforme species (all four sulids studied and three of four phalacrocoracids), and three of four Charadriiforme species (two larids and one of two alcids). The foraging ranges of these species vary from a few tens of km in the cases of shags and cormorants (Sapoznikow & Quintana 2003, Evans *et al.* 2015) to several thousands of km in the case of the albatrosses (Wakefield *et al.* 2011). The distance between colonies for which foraging area segregation has been documented range from as little as 2 km for various species of shag and cormorant (Wanless & Harris 1993, Sapoznikow & Quintana 2003), to several hundred km for Black-browed Albatross *Thalassarche melanophris* (Wakefield *et al.* 2011).

Colony-level foraging distributions which rely on an insufficient sample of tracked individuals will tend to underestimate the extent of the foraging areas (Soanes *et al.* 2013) and hence the extent of overlap between neighbouring colonies, leading in turn to over-estimation of the occurrence and strength of segregation. Few studies have formally tested the sufficiency of their sample to describe colony-level distributions, but those which have, tended to conclude that the level of effort required is greater than that which is commonly achieved (Soanes *et al.* 2013, 2015, but see Lascelles *et al.* 2016). We examined whether differences in sampling effort (individuals tracked per colony) could bias the reported or inferred occurrence of segregation. We found no difference in the mean ($\pm$ sd) number of individuals tracked per colony among studies that showed foraging area segregation (28.0 $\pm$ 35.4 $n = 31$) compared with the remaining studies were no segregation was apparent (20.4 $\pm$ 17.2 $n = 8$, pooled variance t-test on $\log_e$ transformed data $t_{37} = 0.61$, $P = 0.54$). Hence the high occurrence of segregation does not appear to be driven by under-sampling of colony-level distributions and it is unlikely therefore that our review and its conclusions are significantly biased by insufficient tracking effort in the studies considered.

The majority (79%) of studies reviewed provided some evidence of inter-colony segregation of seabird foraging areas, at least at some breeding stages and/or locations, indicating that segregation is a widespread phenomenon. However, non-reporting of studies that fail to demonstrate segregation could lead to publication bias and consequent over-estimation of the frequency of segregation. We attempted to minimise such potential bias by including literature search terms relating to the absence of segregation. Further, the assessment of inter-colony segregation was not a primary objective of many of the studies identified, such that the reported occurrence of segregation is unlikely to have been the primary reason for their publication. Finally, we relied on the analysis and judgement of the authors of the reviewed papers to assess the extent of inter-colony segregation of each study, which was therefore blind to the aims of our review.

This review indicates that inter-colony foraging segregation may be common among seabirds and occurs with similar frequency in all orders for which data are available, and across all scales of foraging movement from tens to thousands of kilometres. This finding might be expected since foraging area segregation has a strong theoretical basis and is predicted to result wherever density dependent inter-colony competition for prey occurs (Wakefield *et al.* 2013).

Drivers of inter-colony foraging area segregation

Optimal foraging theory and the DDH model provide a useful framework for understanding the drivers of seabird foraging distribution and inter-colony segregation.

1. Colony size and location

According to the DDH model, segregation will develop through density-dependent competition-avoidance behaviour. One of the principal drivers of inter-colony competition for prey resources, and hence segregation, is colony size. Several authors have made the link between colony size and foraging range, due to intraspecific competition among colony members (Ashmole 1963, Cairns 1989, Lewis *et al.* 2001, Wakefield *et al.* 2017). However, optimal foraging theory suggests that the density of central place foragers is also a function of distance from the colony, because this determines foraging costs. Hence, both the size and proximity of neighbouring colonies will be important in determining the intensity of potential intra-specific inter-colony competition and therefore segregation.

A number of hypothetical examples illustrate this point: consider two neighbouring colonies that are sufficiently close to have overlapping foraging ranges, surrounded by prey that is uniformly or unpredictably distributed. If colony sizes differ greatly, the DDH model predicts that segregation is likely since foraging profitability of birds from the smaller colony will be higher if they avoid areas with higher numbers of conspecifics from the larger colony. In the vicinity of the larger colony, forager density will be high, leading to higher levels of competition and lower profitability, compared with alternative foraging locations within range of the smaller colony but distant from the larger colony (Fig. 2a). An example comes from Ainley *et al.* (2004), who argued that colony size strongly influenced the foraging distribution of Adélie Penguins *Pygoscelis adeliae* from one large and three small colonies in the Ross Sea, Antarctica. The authors found that foraging grounds of the three small colonies overlapped extensively, but that birds from the small colonies almost never overlapped with the larger colony’s foraging area, despite it being within their potential range. As the breeding season progressed, foraging distance and area increased noticeably, possibly as parents were able to spend longer at sea after the brood-guard stage and/or as a result of shifts in prey distribution or depletion of prey resources close to the colony. As the foraging area of the larger colony increased, the smaller colonies' foraging ranges shifted to avoid the area newly exploited by the larger colony. It is likely that birds from the three small colonies were able to forage communally, but once birds from the large colony intruded, competition was too high and they foraged elsewhere.

Segregation is also likely to occur between two similarly sized neighbouring colonies, if the colonies are sufficiently large: individuals from either colony would achieve higher average profitability by avoiding areas of potential overlap, as competition would be elevated in such areas (Fig. 2b). However, in a final example, if colonies are small, overlap may occur if competition in the shared area is not sufficiently intense to markedly reduce profitability to birds from either colony (Fig. 2c). Evans *et al.* 2015 provide an example from the European Shag, where two colonies of 35 and 96 pairs located c. 4 km apart showed strongly overlapping foraging areas, indicating an absence of inter-colony competition. Note though, that Wanless and Harris (1993) found strong segregation between two colonies of South Georgia Shags *Leucocarbo georgianus* (formerly Blue-eyed Shag *Phalacrocorax atriceps*) 2.5 km apart, numbering just 11 and 32 nesting pairs, showing that colonies perceived to be small may still segregate strongly.

The distances between colonies and their foraging ranges will modify the relationships described above. Where colonies are widely separated relative to their potential foraging ranges, overlap of foraging areas is more likely to occur in areas distant from both colonies. The null density of foragers will be lower further from the colony (due to both the positive relationship between foraging costs and distance, and also radiative spreading with distance) so that net gains are similar to those of more intensely exploited areas. Hence intra-specific competition for prey will be low, and profitability may be affected only marginally by overlap of usage by multiple, distantly located colonies.

2. Coastal morphology and habitat availability

Coastal morphology in the vicinity of breeding colonies may play a large role in determining the extent of marine habitat available and hence levels of competition for resources in those areas (Wakefield *et al.* 2017). Colonies situated on or close to the mainland, or within inlets or bays, have less potential foraging area available to them than those on remote islands surrounded by open sea. Intra-specific competition, and hence the likelihood of segregation, may be greater for colonies with restricted habitat availability. For example, Sapoznikow and Quintana (2003) studied breeding Imperial Cormorants *Phalacrocorax atriceps* and Rock Shags *Phalacrocorax magellanicus* at two neighbouring colonies in the mouth of a bay in Patagonia. They found no overlap between foraging areas used by Imperial Cormorants from the two colonies, despite being separated by just 2.2 km. Imperial Cormorants from the outer colony exclusively exploited open sea areas whilst individuals from the inner colony foraged entirely within the inlet. Rock Shags breeding in the outer colony similarly showed minimal use of the bay, whilst those breeding on the inner islet showed limited use of the outer area (less than expected under a null model of no segregation) and virtually no overlap with the area used by birds from the outer colony.

3. Prey distribution and abundance

Much of the foregoing discussion has assumed a uniform distribution of prey in the waters surrounding breeding colonies. However, the fish, cephalopods, crustaceans, etc. upon which seabirds prey, are patchily distributed. Understanding of the spatial and temporal scales of prey aggregation has important consequences for consideration of inter-colony foraging area segregation. Aggregation is most likely to occur where prey is both superabundant (i.e. is not depleted by foragers to the extent that competition occurs), and temporally persistent (i.e. predictable). Spatio-temporal variation in prey abundance may interact with the distance-dependent foraging costs of central-place foragers. The distance at which prey patches are located from multiple colonies may be an important factor in determining the extent of shared usage. Whilst foraging grounds close to a colony are more likely to be exclusive, at greater distances where competition is generally lower due to higher foraging costs, foraging areas may overlap (Fig. 3). Ramos *et al.* (2013) found that Cory's Shearwaters *Calonectris borealis* from six colonies were substantially segregated throughout most of their foraging areas, but consistently overlapped in high productivity areas along the Canary Current. Similarly, Paredes *et al.* (2014) found that foraging areas of adjacent Black-legged Kittiwake colonies were highly segregated in neritic waters close to the colonies, but overlapped at more remote oceanic locations. These studies suggest that density-dependant competition drives segregation locally, but that temporally stable areas of high productivity located further away are able to support a greater number of predators, causing segregation to break down.

4. Breeding stage

Several studies, all concerning Procellariiformes, reported variation in the extent of foraging area segregation in relation to breeding stage. Segregation was more pronounced during the breeding stage associated with shorter foraging trips: for example, chick-rearing for Black-browed Albatross (Wakefield *et al.* 2011) and incubation for Laysan Albatross *Phoebastria immutabilis* (Young *et al.* 2009). This accords with the prediction from the DDH model that segregation is less likely to occur at the limit of species' foraging ranges where competition is lowest. In addition, intra-specific competition may be higher (i) during the chick-rearing period, because birds must feed not only themselves but also their offspring, (ii) in the post-brood stage, when both adults forage simultaneously (rather than alternately, as during incubation and brooding), resulting in a higher density of foragers, and greater competition.

Segregation at other levels

This review has focussed on segregation among seabird colonies. However, within-colony segregation has also been documented. It is common for sexual segregation to occur among seabirds, often linked to size dimorphism (Catry *et al.* 2006, Phillips *et al.* 2011, Hedd *et al.* 2014, Cleasby *et al.* 2015). For example, Streaked Shearwaters breeding at two colonies in Japan segregate not only by colony but also by sex (Yamamoto *et al.* 2011). Seabirds have also been observed to segregate by age: Fayet *et al.* (2015) found substantial spatial segregation between immature and adult Manx Shearwaters *Puffinus puffinus*, which the authors attributed to differences in experience. Finally, several studies have examined the foraging distribution of birds nesting in different areas of the same colony. Whilst Waggitt *et al.* (2014) found no differences in foraging areas of Northern Gannets nesting in sub-colonies separated by distances of up to several hundred metres, Bogdanova et al. (2014) and Ceia et al. (2015) both found foraging area segregation of European Shag and Cory's Shearwater, respectively, nesting < 2km apart on opposite sides of their breeding islands. In the case of Cory's Shearwater, Ceia *et al.* (2015) reported partially segregated foraging grounds at ranges of up to 200 km. The authors suggested that such segregation could be mediated by directional bias, whereby individuals initiated trips on a bearing consistent with their colony aspect, reinforced by public information transfer between neighbours. These studies raise the question of what constitutes a seabird "colony" and reveal that foraging area segregation can occur at fine spatial scales, and among age classes and genders.

Development of foraging area segregation - information transfer and sociality

Several studies have demonstrated temporally stable individual specialisation in diet and foraging behaviour (see Ceia and Ramos 2015 and Phillips *et al.* 2017 for reviews), which can have fitness consequences (Quinn 2014) and may be spread by information transfer at the colony. It has been hypothesised that information sharing is a benefit of colonial breeding. Ward and Zahavi (1973) suggested that aggregations of birds (breeding colonies and roosts) act as information centres, where individuals gain knowledge about the location of prey. Weimerskirch *et al.* (2010) found that Guanay Cormorants *Phalacrocorax bougainvillii* use social information to select their bearing when departing the colony to forage. Before departure on a foraging trip, the cormorants briefly congregate on the sea to form a raft whose position is continuously adjusted to the bearing of the largest returning columns of cormorants. The departure bearing of birds leaving the raft to forage corresponds to the bearing of the largest groups of returning birds. Grémillet *et al.* (2004) suggested that group foraging behaviour observed in Cape Gannets *Morus capensis* evolved through the benefits of signalling behaviour and increased flight efficiency. They hypothesised that foraging area asymmetry combined with group foraging behaviour foster the development of 'cultural foraging patterns', which are instilled at the colony level through extensive natal colony fidelity (Klages 1994, Votier *et al.* 2011). This may enhance existing competition-avoidance behaviour, thus leading to segregated foraging grounds. On the basis of individual-based models, Wakefield *et al.* (2013) developed this hypothesis, showing how information sharing among birds from the same colony can initiate and maintain segregation of colony-specific foraging areas. They envisaged that unsuccessful or naive birds follow more successful individuals from the colony to prey patches. This allows information on areas that are less profitable, due to the presence of conspecifics from other colonies, to spread through the population. If this occurs across generations, i.e. young birds follow older birds, colony-specific foraging traditions may arise, leading to cultural divergence.

Despite evidence to suggest that sociality may be an important factor contributing to segregated foraging grounds, segregation has also been observed in nocturnal burrowing species such as Leach's Storm-petrel *Oceanodroma leucorhoa*, where visual signalling of foraging success and information transfer is less likely to occur. Pollet *et al.* (2014) found that Leach's Storm-petrels from two colonies in Nova Scotia situated 380 km apart travelled approximately 1 000 km from their colonies to forage and occupied distinctly separate foraging grounds, despite being within range of each other. This suggests that either information sharing and cultural learning of foraging patterns are not required for the development of foraging area segregation, or that information transfer is possible even in nocturnally active burrow nesting species.

IMPLICATIONS FOR ENVIRONMENTAL IMPACT ASSESSMENT

Improved understanding of the extent and causes of seabird foraging segregation is important for marine ecologists who seek to understand the processes responsible for shaping distributions and interactions of marine biota. However, it is also of applied relevance for marine planning and conservation. Globally, the marine environment is subject to increasing anthropogenic demands and developments such as renewable energy generation schemes frequently cover extremely large areas (1000 – 10000 km^2). In many countries, the statutory consent process requires environmental impact assessments (EIA) that quantify likely impacts on marine biodiversity, including mobile species such as seabirds. Since impacts on legally protected breeding colonies are of particular concern, such EIAs must consider the extent of seabird usage, and consequent impacts, of offshore development sites, especially for seabirds from protected breeding colonies. However, because at-sea surveys can rarely assign colony provenance of seabirds surveyed in development areas, and tracking multiple species from all protected colonies within foraging range may be both costly and logistically challenging, evidence regarding the degrees of connectivity of multiple colonies to a given development site is often lacking. Accordingly, in Europe current EIA practice often relies on simplifying assumptions regarding the distribution of foraging seabirds, such as species-level generic foraging ranges, assuming non-interacting spatial overlap of birds from adjacent colonies (Douse & Tyler 2014). However, if space use of a proposed development area is exclusive to a single colony, impacts will also fall exclusively, exerting a larger impact on the affected colony, whilst excluded colonies will bear no impact. Current EIA practice of apportioning impacts assuming overlapping foraging distributions will therefore be subject to errors of unquantified magnitude (of both over- and under-estimation) in cases where segregation occurs. The apparently high prevalence of inter-colony foraging segregation indicated by this review suggests that such errors may be widespread.

The DDH model allows us to consider which colonies may be most affected by error in EIAs that are introduced by the assumption of shared space use. Perhaps most notably, larger colonies are predicted to competitively exclude smaller neighbouring colonies, thus making larger colonies more likely to show sole use of a foraging area. Since statutory protection is usually afforded to larger colonies, there is a risk that current EIA practice will tend to under-estimate impacts on protected colonies, whilst over-estimating impacts on smaller, unprotected colonies. Conversely, seabirds are most likely to show overlapping foraging areas at the limit of the foraging range where forager densities and competition are lowest. Current EIA practice may therefore be least prone to error in situations where developments occur toward the limit of species' foraging ranges, and also where prey is abundant. However, the studies reviewed here and elsewhere (e.g. Thaxter *et al.* 2010) show that there is often considerable intra-specific inter-colony variation in foraging range such that, in the absence of empirical, site-specific data, the application of generic species-level foraging radii is prone to considerable error.

The studies reviewed here deal solely with the central-place foraging behaviour of breeding seabirds. It is not known to what extent foraging area segregation also applies to non-breeding adults and immatures during the breeding season. Many non-breeding adults and immatures attend the nesting colonies during the breeding season, and although they have greater flexibility regarding the timing of commuting, they nonetheless behave as central place foragers, so will be subject to similar, though not identical, costs and benefits as breeding adults. Due to the difficulty of tracking non-breeding adults and immatures there are currently extremely few empirical data on the marine distribution of these groups (though see Votier *et al.* (2017) for a recent example).

The DDH model predicts that in areas of high prey abundance, such as upwelling or frontal zones, seabirds from multiple colonies may aggregate. If a marine development is situated in such an area, the usage by birds from multiple colonies might lead to impacts on birds from numerous colonies, even at considerable distance from the development. Engineering considerations may favour location of offshore structures, such as windfarms, in shallow waters overlying banks, which are generally productive areas and likely to be a focus of seabird aggregation. Douse and Tyler (2014) recognised that the use of generic foraging ranges may underestimate the geographic extent of impacts, since birds may travel exceptionally long distances to forage in highly productive areas (Dean *et al.* 2015). Therefore, even in cases where impacts are shared among multiple colonies, the simple distance-decay relationships used in EIAs may underestimate the impacts on colonies using highly profitable, if distant, foraging areas. Such considerations may be particularly important for species that show a dual foraging strategy, alternating short trips that permit frequent chick provisioning, with longer trips to more productive areas for self-maintenance (e.g. Shoji *et al.* 2015).

The findings of this review indicate that over- or under-estimation of impacts on individual colonies when using approaches based on simplifying assumptions typically employed in EIAs will be the rule rather than the exception. Furthermore, offshore developments such as arrays of wind turbines, typically cover very considerable areas. If such developments lead to avoidance of such areas by seabirds (Desholm & Kahlert 2005) this indirect form of habitat loss may result in increased competition, and hence segregation, in the surrounding areas used by displaced birds. Under such circumstances, the cumulative effects of multiple adjacent developments will be extremely difficult to predict.

CONCLUSIONS AND FUTURE RESEARCH

This review has examined spatial segregation in seabirds and discussed potential implications of the phenomenon when apportioning impacts of marine developments to particular seabird colonies, particularly those protected by legal designations. The studies reviewed suggest that inter-colony segregation of foraging areas may be widespread across seabird taxa and spatial scales and will arise wherever intra-specific inter-colony competition for prey is sufficiently intense. The spatial and temporal extent of segregation is somewhat variable, even within species. Such variability is likely driven by variation in both the distribution of prey, the size of neighbouring colonies and the distances between colonies. Competition may be absent or of minor importance in circumstances where colony sizes are well below their natural carrying capacity due to anthropogenic impacts (bycatch, predation by invasive species, harvest for human consumption, pollution, etc.). However, seabird declines of recent decades in areas of northwest Europe are generally considered to result from food limitation (Frederiksen *et al.* 2006, Frederiksen *et al.* 2007, 2013, Louzao *et al.* 2015), so prey are unlikely to be superabundant, suggesting that segregation should occur in this region. Historically, harvesting of seabirds for human consumption and lower human exploitation of seabird prey, may have resulted in seabird population sizes falling below prey carrying capacity, leading to lower inter-colony competition and segregation than currently. However, if segregation is mediated by cultural processes (Wakefield *et al.* 2013), there may be some lag in the onset of segregation in response to environmental change as populations become food-limited. It is unclear how long such a lag might continue, but it is unlikely that many seabird populations in this region are in equilibrium with prey availability.

Little information is currently available regarding the incidence of segregation among non-breeding and immature birds associated with different colonies, as tracking studies are usually carried out on breeding adults (but see Camphuysen 2011, Votier *et al.* 2011, Sherley *et al.* 2017). Nor is it clear the extent to which breeding adults from a given colony may segregate at sea from other groups of conspecifics that may be associated with the same colony during the breeding season (e.g. failed breeders, immature birds, etc.), though see Votier *et al.* (2017). This is potentially an important aspect to understand as impacts of marine developments on future breeders may have substantial consequences for population dynamics and, ultimately, colony fate (Sherley *et al.* 2017). Though not a focus of this review, there is a strong suggestion that segregation at the sub-colony level also occurs, but it is not clear what factors cause some sub-colonies to show segregation in some cases (Ceia *et al.* 2015) but not others (Waggitt *et al.* 2014). This review has shown that the strength of segregation may change during the course of the breeding season (e.g. Ainley *et al.* 2004, Yamamoto *et al.* 2011) and there is also a suggestion that segregation can occur outside the breeding season (e.g. Thiebot *et al.* 2011, Fort *et al.* 2012, Ratcliffe *et al.* 2014). Greater understanding of foraging area segregation outside the breeding season will require the development of safe, low cost, long term attachment methods for high precision tags.

The assumption of non-interacting, overlapping colony foraging distributions that underpins the current, widely-used approaches to apportionment of the potential impacts of marine developments to seabird colonies in the UK, appears unrealistic in many situations. Segregation of seabird foraging areas appears commonplace and consequently the distribution of impacts among colonies will differ from the predictions of existing models: fewer colonies are likely to be impacted, but to a higher degree. Whilst we have discussed a variety of such factors that may influence the extent of foraging segregation, with reference to examples from the literature, given the current state of knowledge it is it not possible to reliably determine the extent of colony segregation, or the absence of segregation, for any particular marine location.

For most of the studies reviewed here, the authors' assessment of inter-colony foraging area interactions was not based on inclusion of a measure of inter-colony competition in a space-use model, but rather on a somewhat subjective judgement based on the percentage overlap, or by visual inspection of colony distributions, but without reference to a defined null (i.e. overlapping) distribution. In cases where segregation was complete, statistical analysis may be redundant, but in order to identify effects of inter-colony competition on space use in an unbiased manner, a modelling approach incorporating a measure of inter-colony competition is required. Whilst we recognise that identification of inter-colony interactions was not a primary focus of many of the studies we reviewed here, we would urge authors of future multi-colony seabird foraging distribution studies to include a statistically robust assessment of the extent and direction of potential inter-colony interactions, which account for accessibility and prey availability wherever possible. In addition, we strongly suggest that the assessment of future offshore developments should require the simultaneous collection of tracking data from a representative sample of birds from colonies likely to be affected. The collection and analysis of such data will represent a valuable contribution to improving our understanding of the factors that shape colony foraging distribution and segregation.

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Table 1. Occurrence of inter-colony segregation of foraging areas of seabirds. Breeding stage: PL = pre-laying, incubation = Inc, chick-rearing = CR; Evidence: S = statistical test, O = assessment of overlap, N = No assessment.

Species	Common name	Order	Area	Breeding stage	Method	Evidence	Distribution	Reference
Pygoscelis papua	Gentoo Penguin	Sphenisciformes	Falkland Islands	CR	GPS	O	Segregation	(Masello *et al.* 2010)
Pygoscelis adeliae	Adelie Penguin	Sphenisciformes	Ross Sea, Antarctica	CR	VHF	O	Variable segregation	(Ainley *et al.* 2004)
Eudyptes chrysocome	Southern Rockhopper Penguin	Sphenisciformes	Falkland Islands	CR	GPS	O	Segregation	(Masello *et al.* 2010)
Eudyptes chrysolophus	Macaroni Penguin	Sphenisciformes	South Georgia	CR	PTT	O	Variable segregation	Trathan *et al.* (2006)
Spheniscus magellanicus	Magellanic Penguin	Sphenisciformes	Patagonia, Argentina	CR	PTT	N	Not assessed	(Boersma *et al.* 2009, Wilson *et al.* 2005)
Spheniscus magellanicus	Magellanic Penguin	Sphenisciformes	Falkland Islands	CR	GPS	O	Segregation	(Masello *et al.* 2010)
Oceanodroma leucorhoa	Leach's Storm-petrel	Procellariiformes	Nova Scotia, Canada	IN	GLS	O	Segregation	(Pollet *et al.* 2014)
Phoebastria immutabilis	Laysan albatross	Procellariiformes	Pacific Ocean, Hawaii	IN, CR	GLS	O	Variable segregation	(Young *et al.* 2009)
Phoebastria irrorata	Waved Albatross	Procellariiformes	Galapagos, Ecuador	IN, CR	GPS	O	Variable segregation	(Awkerman *et al.* 2014)
Phoebetria fusca	Sooty Albatross	Procellariiformes	South Atlantic, SW Indian Ocean	IN, CR	GPS & PTT	O	Overlap	(Schoombie *et al.* 2017)
Thalassarche melanophris	Black-browed Albatross	Procellariiformes	Kerguelen	CR	Colour mark	O	Variable segregation	(Weimerskirch *et al.* 1988)
Thalassarche melanophris	Black-browed Albatross	Procellariiformes	Falkland Islands	CR	PTT	O	Segregation	(Huin 2002)
Thalassarche melanophris	Black-browed Albatross	Procellariiformes	Southern Ocean	IN, CR	PTT	S	Variable segregation	(Wakefield *et al.* 2011)
Thalassarche melanophris	Black-browed Albatross	Procellariiformes	Falkland Islands	CR	GPS & GLS	S	Variable segregation	(Catry *et al.* 2013)
Macronectes giganteus	Southern Giant Petrel	Procellariiformes	South Atlantic	IN, CR	GPS	O	Segregation	(Quintana *et al.* 2010)
Pterodroma cookii	Cook’s petrel	Procellariiformes	New Zealand	CR	GLS	O	Segregation	(Rayner *et al.* 2008)
Puffinus tenuirostris	Short-tailed Shearwater	Procellariiformes	Tasmania/SE Australia	CR	PTT & GLS	O	Overlap	(Raymond *et al.* 2010)

Puffinus tenuirostris	Short-tailed Shearwater	Procellariiformes	Bass Strait, SE Australia	CR	GPS & GLS	O	Overlap	(Berlincourt and Arnould 2015)
Calonectris leucomelas	Streaked Shearwater	Procellariiformes	Japan	PL, IN	GLS	O	Variable segregation	(Yamamoto *et al.* 2011)
Calonectris diomedea	Scopoli's Shearwater	Procellariiformes	Tunisia and Italy	IN, CR	GPS	S	Segregation	(Cecere *et al.* 2015)
Calonectris diomedea	Scopoli's Shearwater	Procellariiformes	Mallorca, Menorca, Collumbretes	IN, CR	GPS	O	Segregation	(Genovart *et al.* 2018)
Calonectris borealis	Cory’s Shearwater	Procellariiformes	North Atlantic Ocean	IN, CR	GPS & compass loggers	O	Variable segregation	(Paiva *et al.* 2010)
Calonectris borealis	Cory’s Shearwater	Procellariiformes	North Atlantic Ocean	CR	GPS & PTT	O	Variable segregation	(Ramos *et al.* 2013)
Puffinus puffinus	Manx Shearwater	Procellariiformes	Britain and Ireland	IN, CR	GPS	O	Overlap[1]	(Dean *et al.* 2012, Dean *et al.* 2015)
Morus bassanus	Northern Gannet	Suliformes	Britain and Ireland	CR	GPS & PTT	S	Segregation	(Wakefield *et al.* 2013)
Morus capensis	Cape Gannet	Suliformes	South Africa	CR	GPS	S	Segregation[2]	(Grémillet *et al.* 2004, Grémillet *et al.* 2008)
Morus serrator	Australasian Gannet	Suliformes	Bass Strait, SE Australia	IN	GPS	O	Segregation	(Angel *et al.* 2016)
Sula variegata	Peruvian Booby	Suliformes	Northern Peru	CR	GPS	O	Segregation	(Zavalaga *et al.* 2010a, Zavalaga *et al.* 2010b)
Phalacrocorax magellanicus	Rock Shag	Suliformes	Patagonia, Argentina	IN, CR	VHF	O	Segregation	(Sapoznikow and Quintana 2003)
Leucocarbo atriceps	Imperial Cormorant	Suliformes	Patagonia, Argentina	IN, CR	VHF	O	Segregation	(Sapoznikow and Quintana 2003)
Phalacrocorax aristotelis	European Shag	Suliformes	Isles of Scilly, United Kingdom	IN, CR	GPS	O	Overlap	(Evans *et al.* 2015)
Phalacrocorax aristotelis	European Shag	Suliformes	Britain and Ireland	IN, CR	GPS	S	Overlap	(Wakefield *et al.* 2017)
Leucocarbo georgianus[3]	South Georgia Shag[3]	Suliformes	South Georgia	CR	VHF	O	Segregation	(Wanless and Harris 1993)
Rissa tridactyla	Black-legged Kittiwake	Charadriiformes	Prince William Sound, Alaska	CR	VHF	O	Segregation	(Ainley *et al.* 2003)
Rissa tridactyla	Black-legged Kittiwake	Charadriiformes	Pribilof Islands, Bering Sea	CR	GPS	O	Segregation	(Paredes *et al.* 2012, Paredes *et al.* 2014)
Rissa tridactyla	Black-legged Kittiwake	Charadriiformes	North Sea, NE England	CR	GPS	O	Overlap	(Redfern and Bevan 2014)
Rissa tridactyla	Black-legged Kittiwake	Charadriiformes	Britain and Ireland	IN, CR	GPS	S	Segregation	(Wakefield *et al.* 2017)

Larus fuscus	Lesser Black-backed Gull	Charadriiformes	German coast	IN	GPS	O	Segregation	(Corman *et al.* 2016)
Ptychoramphus aleuticus	Cassin's Auklet	Charadriiformes	Channel Islands, California	IN, CR	VHF	N	Not assessed	(Adams *et al.* 2004)
Alca torda	Razorbill	Charadriiformes	Britain and Ireland	IN, CR	GPS	S	Overlap	(Wakefield *et al.* 2017)
Uria algae	Common Guillemot	Charadriiformes	Britain and Ireland	IN, CR	GPS	S	Segregation	(Wakefield *et al.* 2017)

[1] On short trips (most frequent during chick-rearing) little overlap occurred as foraging ranges were generally less than inter-colony distance for most colonies

[2] Segregation not assessed in Grémillet *et al*. 2008 who studied colonies in South Africa and Namibia, but reported for same South African colonies studied by Grémillet *et al*. 2004.

[3] Formerly known as Blue-eyed Shag *Phalacrocorax atriceps*

Table 2. Number of studies where seabird inter-colony distributions were assessed as overlapping, segregated, or variably segregated, according to the strength of evidence used for the assessment.

Evidence type	**Inter-colony distribution**	**Number of studies**
Formal statistical assessment of inter-colony effect (9 studies)	Overlap	2
	Segregation	5
	Variable segregation	2
Author judgement, based on percentage overlap or visual inspection of colony-level distributions (30 studies)	Overlap	6
	Segregation	16
	Variable segregation	8
No assessment made (2 studies)	No assessment	2

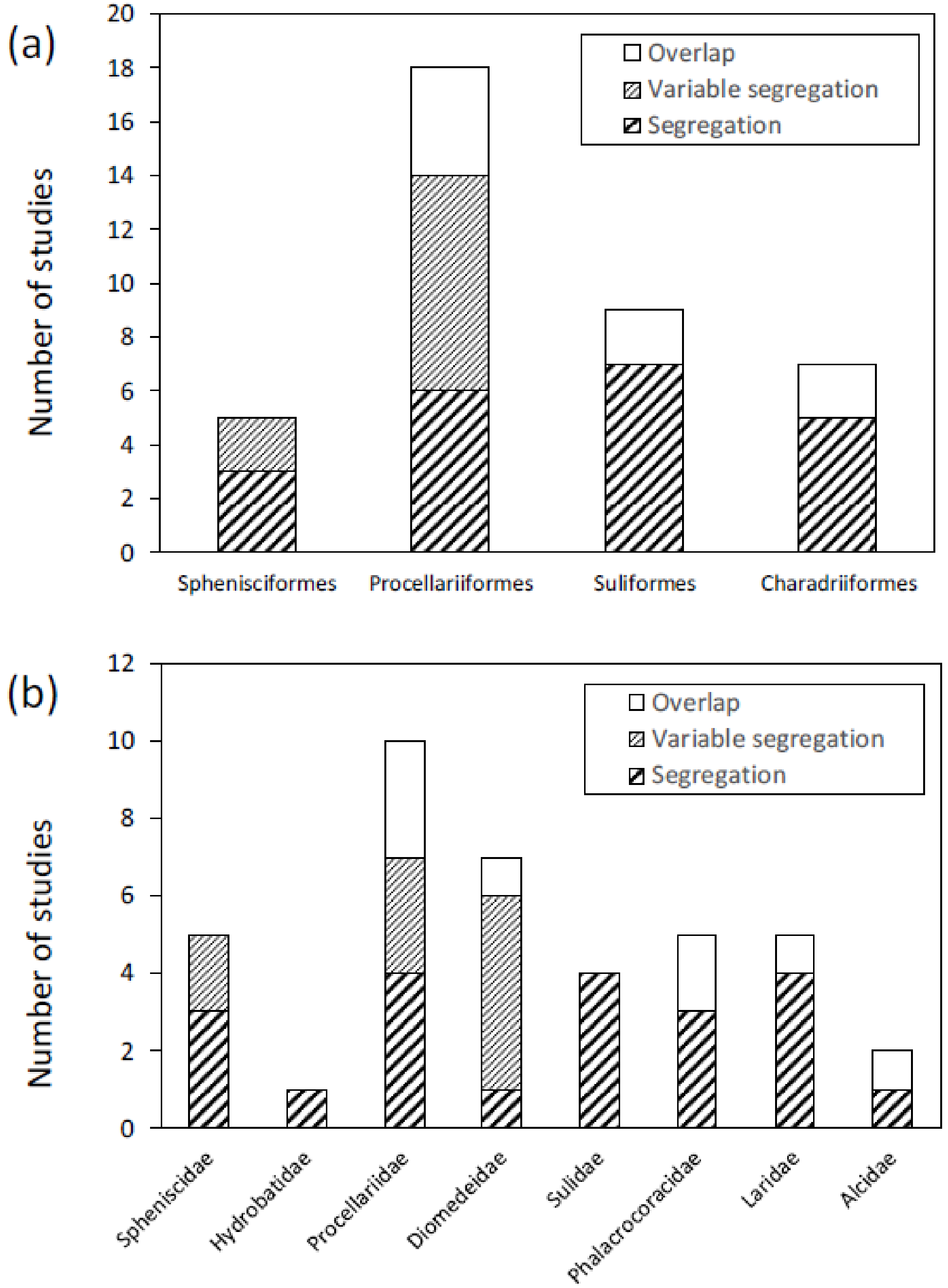
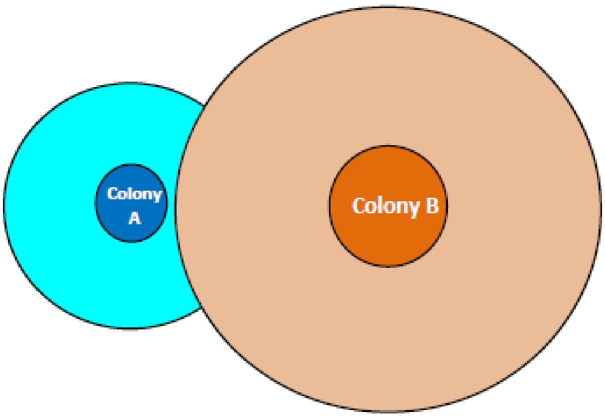
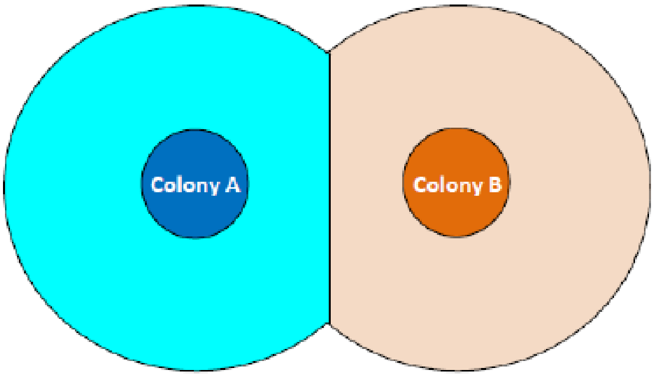


Figure 1. Occurrence of inter-colony foraging area segregation in seabirds by order (a) and family (b).

(a) Colonies of greatly differing size



(b) Similarly large colonies



(c) Small colonies

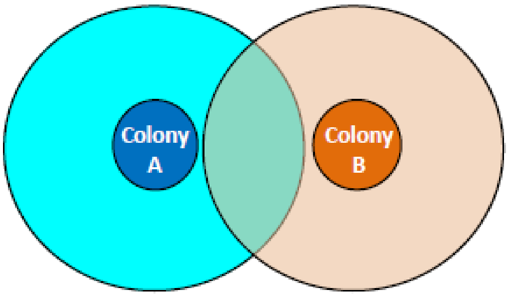


Figure 2. Colony-specific distribution patterns as a function of colony size. Segregation is likely to occur in the vicinity of large colonies where forager density is high (a and b), but least likely where colonies are small and prey availability less likely to be affected by density-dependent competition (c).

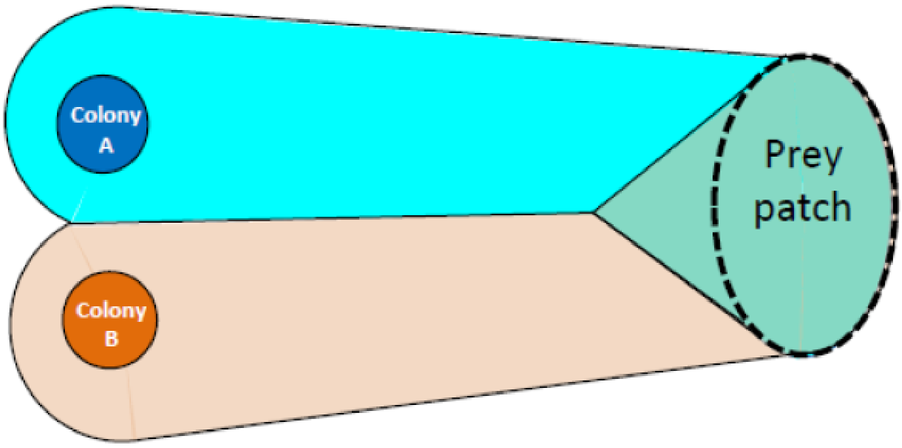


Figure 3. Close to the adjacent colonies, foraging grounds are segregated due to density-dependent competition. However, at greater distances foraging grounds may overlap, especially in areas of predictably high prey density, where effective competition is lower.

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Nocturnal flight activity of northern gannets *Morus bassanus* and implications for modelling collision risk at offshore wind farms

Keywords: offshore wind farm, seabird, collision risk modelling, data logger, nocturnal flight activity

ABSTRACT

Assessing the potential impacts of proposed offshore wind farm developments on seabird populations requires estimation of nocturnal flight activity of seabirds for input into collision risk models. One of the seabirds considered most at risk from collision with offshore wind turbines is the northern gannet *Morus bassanus*. The recommended correction for gannet nocturnal flight activity is currently a highly precautionary value. Here we use data from tracking studies to derive evidence-based correction factors for nocturnal flight activity of adult gannets during the breeding and nonbreeding seasons, and of immature gannets during the summer prospecting phase. Flight and diving activity of gannets was minimal during the night, astronomical and nautical twilight, for adults during the breeding season and nonbreeding season, and for immatures. Some flight activity occurred during the short period of civil twilight, but on average at about half the level seen during the day. Based on evidence from numerous tracking studies, we recommend that precautionary values of the nocturnal (sunset to sunrise) flight activity factor for estimating collision risk should be 8% of daytime flight activity during the breeding season and 3% of daytime flight activity during the nonbreeding season. Use of these evidence-based correction factors will improve the accuracy, and reduce the uncertainty of collision risk models, providing a more reliable assessment of the impacts of offshore wind farms on gannets.

1. Introduction

One of the key environmental issues facing developers of offshore wind farms in Environmental Impact Assessments is the impact that turbines may have on seabird populations as a consequence of mortality of birds that collide with rotating blades (Garthe and Hüppop, 2004; Furness et al., 2013). Bird collision mortality can be estimated using the Band collision risk model (Band, 2012). However, this requires an estimate of nocturnal flight activity as one of the model inputs. Seabird surveys at proposed offshore wind farm sites do not record the numbers of birds flying through the area at night, as visual (boat-based counts) or photographic (aerial) surveys are only practical during daylight hours. It is, therefore, necessary to use a correction factor, relative to daytime data, to allow for nocturnal flight activity of seabirds. Garthe and Hüppop (2004) assigned nocturnal flight activity scores to seabird species in five categories (scores of 1 to 5), based on existing limited evidence, their own judgement, and that of a panel of experts. They indicated that a score of 1 represented '*hardly any flight activity at night*' while a score of 5 represented '*much flight activity at night*'. These scores simply indicated that bird species that scored higher were likely to show more nocturnal flight activity than bird species that scored lower on the scale. Nevertheless Band (2012) advocated an arbitrary but precautionary translation of the Garthe and Hüppop (2004) scores for collision risk modelling as follows:

- 1=0% of daytime flight activity,
- 2=25% of daytime flight activity,
- 3=50% of daytime flight activity,
- 4=75% of daytime flight activity,
- 5=100% of daytime flight activity.

It is important to note that these suggested percentages were not based on evidence. It is also clear from Garthe and Hüppop (2004) that many of the scores for other seabird sensitivity metrics that they assigned were categorical rather than linear. Explicit examples are their scoring of population size; 1=>3million, 2=1-3 million, 3=500,000-1 million, 4=100,000-500,000, 5=<100,000, and their scoring of flight altitude where scores 1 and 2 were based on median flight heights on a non-linear scale but scores 3 to 5 were based on 90th percentile flight heights on that scale.

One of the seabird species that appears most vulnerable to collision mortality at offshore wind farms is the northern gannet *Morus bassanus* (hereafter gannet) (Furness et al., 2013). The impact of collision mortality on gannet populations has been one of the primary concerns of recent planning applications for offshore wind farms. Band model calculations estimate a cumulative total of 2,561 gannets per year may be killed by collisions at constructed and consented offshore wind farms in the United Kingdom sector of the North Sea (MacArthur Green 2018). In relation to the Habitat Regulations Assessment (HRA) component of the planning application for Hornsea Two offshore wind farm, Natural England (2015) were unable to conclude beyond all reasonable scientific doubt that the estimated cumulative collision total for offshore wind farms would not have an adverse effect on the integrity of the Flamborough and Filey Coast proposed Special Protection Area (FFC pSPA) gannet population. In relation to East Anglia THREE offshore wind farm, The Planning Inspectorate (2017) stated "two key HRA matters were the focus of the Examination: The effect of the proposed development in combination with other offshore wind farms on the kittiwake and gannet features of the FFC pSPA". Therefore, estimated collision mortality of gannets has the potential to stop the considerable further development of offshore wind farms planned for the North Sea (The Crown Estate, 2018).

Garthe and Hüppop (2004) assigned a nocturnal flight activity score of two for gannets, based on evidence from Garthe et al. (1999, 2000, 2003) and Hamer et al. (2000), and this was converted to 25% of the daytime level by Band (2012). During mid-summer, the correction for nocturnal flight activity makes only a small difference to estimated numbers of collisions, since the night is short in mid-summer (Fig. 1). However, in winter the effect is larger: because the night is about twice as long as day during winter, the Band (2012) model estimates an additional 0.5 collisions at night for each collision during the day for an offshore wind farm located in the southern North Sea. Most offshore wind farms in Europe are in the southern North Sea. Gannet numbers in that region are low in summer and peak strongly during November (Stone et al., 1995, Furness et al., 2018), so the influence of nocturnal correction is likely to be close to the 0.5 nocturnal collisions per daytime collision. This means that an evidence-based correction for this parameter would be important in improving confidence in the estimated cumulative impact of collisions at offshore wind farms on gannet populations, especially where the cumulative total is close to a level that could result in consenting risk for further offshore wind farm developments.

Garthe and Hüppop (2004) did not provide an explicit definition of day and night. Collision risk modelling using the Band model defines day as sunrise to sunset and night as sunset to sunrise, with the estimation of the times of sunrise and sunset derived from Forsythe et al. (1995). However, that definition of night contrasts with the official concept of 'twilight' and 'night'. 'Civil twilight' is defined as from sunset to the sun falling 6° below the horizon and in the morning from when the sun reaches 6° below the horizon until sunrise. 'Nautical twilight' is defined as the sun being between 6° and 12° below the horizon. 'Astronomical twilight' is defined as the sun being between 12° and 18° below the horizon, and 'night' is from then until the sun has risen back to 18° below the horizon. In the regions where gannets over-winter, the transition through twilight can be rapid. However, in summer, there may be no official 'night' at all, because astronomical twilight persists if the sun never falls more than 18° below the horizon (Fig. 1). This suggests that a more subtle definition of 'day' and 'night' is required than that used in Band (2012) to take account of the considerable variation in light levels between sunset and sunrise in summer, and especially at higher latitudes. Since gannets are visual predators (Garthe et al., 2000, 2003, Lewis et al. 2002), it is likely that flight activity is determined by the minimum light levels to allow foraging, commuting or migrating. Cleasby et al. (2015a) noted that gannet dives tend to be shallower close to sunrise and sunset, which supports the argument that diving at twilight is limited by the birds' ability to see their prey.

There are now many data sets showing flight activity levels of gannets at different times of day, both for breeding birds and for birds during the migration period and in winter. In this paper we assess the available evidence in order to provide evidence-based corrections for nocturnal flight activity of gannets for use in Band model collision assessments. This will give more accurate results than estimates based on the conversion of scores assigned by Garthe and Hüppop (2004). Here we consider data from throughout the range of the gannet. However, we focus on deriving appropriate corrections for use in examining impacts on gannets in the North Sea, the region with by far the largest number of constructed and proposed offshore wind farms (The Crown Estate, 2018).

2. Methods

We carried out a literature search, focused on Web of Knowledge and Google Scholar but also searching ‘grey literature’ (such as consultant reports and SNCB guidance documents) to find data on daytime and nocturnal flight activity of gannets. GLS logger data (from Garthe et al., 2012) were used in order to identify variation in flight behaviour according to the time of day. We considered activity data divided into ‘day’, ‘civil twilight’, ‘nautical twilight’, ‘astronomical twilight’, and ‘night’ (Fig. 1). We used Time and Date (2018) to extract timings of sunrise, sunset, civil, nautical, astronomical twilight and night appropriate for the location and date of each study.

We considered data derived from tags deployments: a) data from breeding gannets incorporated into Garthe and Hüppop (2004), b) data from breeding gannets collected since Garthe and Hüppop (2004), c) geolocator (GLS) data from gannets during the non-breeding season, d) data from tags on immature gannets (Jeglinski et al. unpublished data). Flight activity is frequently referred to as the percentage of each hour spent in flight. Several different types of tag have been used to infer at sea behaviours of seabirds. Travel speed of birds at sea derived from GPS tracking can be assigned to resting on the sea or to flying if there is a clearly bimodal distribution of travel speeds, with the faster mode representing flight (Gremillet et al., 2004). Few studies have used accelerometer data from tags, but these can aid interpretation of behaviour of birds (Warwick-Evans et al., 2015). Geolocator tags that have a salt-water switch provide accurate data for gannets because birds are either in the water (switch on) or flying (switch off), as it can be assumed that gannets are not on land during the nonbreeding season when away from the colony (Garthe et al., 2012). Some loggers record diving activity but not flight activity. Since gannets only dive from the air, and not from the sea surface, diving activity implies flight activity.

Details of tag types deployed in different studies and numbers of data sets obtained are summarised in Table 1. Garthe et al. (1999) deployed GPS loggers on adult gannets at Hermaness, Shetland. Their loggers provided data on feeding events and on flight activity throughout the 24-hour period, but the study was limited to a sample of just three individuals tracked for a few days in mid-July 1997. Garthe et al. (2000) deployed time-depth loggers on adult gannets at Funk Island, Canada. Hamer et al. (2000) deployed satellite PTTs on chick-rearing gannets at the Bass Rock, Scotland. Garthe et al. (2003) deployed loggers on chick-rearing adult gannets at Funk Island, Canada recording diving activity and flight activity. Hamer et al. (2007) presented data from satellite tracking or GPS loggers over three breeding seasons for a total of 53 gannets breeding on the Bass Rock. RSPB deployed satellite PTTs and GPS tags on gannets breeding at Bempton during three breeding seasons (2010, 2011 and 2012), obtaining tracks of chick-rearing birds and some tracks of post-breeding dispersal and migration. Garthe et al. (2014) reported on diving activity of breeding birds from Bonaventure, Canada. Warwick-Evans et al. (2015) presented data on the diurnal pattern of plunge dives by gannets breeding at Alderney, Channel Islands. Garthe et al. (2017) reported on plunge diving activity of breeding birds from Helgoland, Germany. Garthe et al. (2012) deployed geolocator loggers on breeding adult gannets on the Bass Rock in 2002, 2003, and 2008, and presented data on flight and resting behaviour of those birds.

Although we were able to access published data on flight activity of adult gannets, no data on flight activity of immature gannets have been published. We therefore include new data on immature gannets. Jeglinski (unpublished data) deployed 7 GPS GSM tags (Pathtrack Ltd. Leeds, UK) on 2-3 year old immature gannets on the Bass Rock between the 3rd and the 11th of July 2016. These tags were equipped with a dynamic algorithm that adjusts the GPS fix rate to the battery voltage, and the GPS fix rate was programmed to 5 minute intervals but in reality GPS fixes were taken every 10 ± 5.4 minutes. Based on the location and timestamp of each position, the data were categorized into four categories (dawn, day, dusk, night) using the function crepuscule (R package maptools). The function uses algorithms provided by the National Oceanic & Atmospheric Administration (NOAA) to implement flexibility for various formal definitions of times of dawn and dusk. The definition of dawn and dusk was based on a solar angle of <6^{o} below the horizon so represents civil twilight. The speed between successive GPS locations was calculated for each individual and each period. Data were divided into two categories based on the thresholds defined by Bennison et al. (2017) and Wakefield et al. (2013): <3.5 km/hr (likely corresponds to resting e.g. drifting on the sea surface), >3.5 km/hr which may indicate some flight activity as this speed is unlikely to occur due to drift alone, although erroneous high speeds can occasionally occur as a result of inaccuracies in GPS logger location estimates (S. Garthe unpubl. data). We overlay the GPS locations at night with an ocean shapefile based on ocean coastlines at a scale of 10 m (Natural Earth, 2017) to identify if immatures spend the night at sea or on land. All locations were positioned at sea, so we concluded that immatures do not sleep on land, which is a pre-requisite for assigning behaviours as either flying or resting on the sea.

Although differences in methodology among studies might make comparisons of activity budgets between studies difficult, in this paper we only make comparisons within studies, comparing between periods of the day, and so the methodology used in defining flight activity is identical between the relevant periods of the day that we compare.

3. Results

3.1 Breeding adults

Garthe et al. (1999, 2000, 2003, 2014), Hamer et al. (2000, 2007), and Warwick-Evans et al. (2015, 2017) reported that breeding gannet flight activity was negligible during nautical twilight, astronomical twilight and night, and was much lower during civil twilight than during the day. This was also inferred by Langston et al. (2013). The numerous studies of breeding gannets at different colonies and deploying a range of different types of tag show consistent results. Flight (Fig. 2a-e) and diving activity (Fig. 3a-c) of gannets occurred throughout the daylight period, sometimes with a slight tendency to peak just after sunrise and to a lesser extent just before sunset. Flight and diving activity were lower immediately after sunset and immediately before sunrise than during the day, and fell to negligible levels shortly after sunset. Flight activity remained at negligible levels through the night until shortly before sunrise (Fig. 2a-e). When averaged over the period from sunset to sunrise, flight activity of breeding adults averaged 7.1% of the daytime level across six studies, while diving by breeding adults averaged 2.9% of the daytime level across seven studies (Table 1).

3.2 Immatures

Tracked immature gannets moved at an average speed of 1.4 ± 1.21 (SE) km/h during nautical and astronomical twilight and night. Based on the threshold metric, 95.3% of the time periods were defined as birds resting on the water (Electronic Supplement Table S1). During dawn the average speed was 3.67 ± 6.44 km/h, which is higher than at night, so suggests some flight activity. During dusk the average speed was 8.87 ± 14.81 km/h, also higher than at night and suggesting some flight activity. Based on the threshold metric, on average 83.3% and 65.8% of dawn and dusk periods were identified as birds resting on the water. With a typical flight speed of 25 km/hr (e.g. Hamer et al., 2000), birds would have to spend less than 25% of a period in flight even during the few periods when flight appears likely, in order to result in a mean speed of movement of only 4 to 5 km/hr, which was the average maximum speed recorded across the sample of birds during night in the few intervals when some flight activity was indicated (Electronic Supplement Table S2). During the day, 61.7% of time intervals were classified as birds resting on the sea surface. However, during the day, immatures moved with an average speed of 11.48 ± 17.14 km/hr, reaching maximum speeds between 55 and 87 km/hr.

Even though the dominant activity during night was resting, all birds apparently spend a very small proportion of the night in flight. The data show that immature flight activity, based on flight speed, was higher during civil twilight than at night, with birds generally flying faster (and spending almost double the proportion of time periods during which there seemed to be some flight activity) during dusk than during dawn. Overall, the data show low flight activity by immatures between sunset and sunrise, though possibly slightly more than seen among breeding adults.

3.3 Adults in the non-breeding season

Flight activity of adult gannets during the non-breeding season between sunset and sunrise averaged 2.5% of daytime level during autumn and 1.9% of daytime level during winter (Garthe et al., 2012). Data for the subset of birds migrating through the North Sea in autumn, which may be the most appropriate in relation to collision risk at North Sea offshore wind farms, are presented in Figure 2d. During peak autumn migration (mid-late October) adult gannets from the Bass Rock that were going to remain in the North Sea or Channel overwinter spent on average 31.9% of daylight time in flight, 2% of civil and nautical twilight combined in flight and 0% of astronomical twilight and night time in flight. Considering the nocturnal period (sunset to sunrise) flight activity of adult gannets in the North Sea that remained in the area overwinter averaged 1% of the daytime level, whereas flight activity of adult gannets that migrated to winter off southern Europe or west Africa averaged 3.8% of the daytime level (Table 1).

Data for flight activity during winter of birds remaining in the southern North Sea during winter are presented in Figure 2e. In winter (December), birds spent hardly any of the night or astronomical twilight in flight, with a mean of 0.2% of the night spent flying. Garthe et al. (2012) showed that during daylight hours, birds spent more time flying in autumn than in winter, and birds that were migrating to West Africa spent more time flying during the day (40% of daylight hours) than birds that wintered in UK waters (30% of daylight hours flying). In winter, birds spent on average 26% of daylight hours in flight (Garthe et al., 2012). However, birds wintering in the southern North Sea flew more than birds wintering in west Africa, so that the daytime baseline level of flight activity against which nocturnal activity is compared needs to be considered on a regional basis to ensure a like-for-like comparison. Birds in the southern North Sea in December flew 31% of daylight hours. By comparison, they flew 4.5% of civil twilight and nautical twilight periods, and 0.2% of astronomical twilight and night periods. Flight activity between sunset and sunrise averaged 0.75% of the time, compared with 31% of the time between sunrise and sunset. Flight between sunset and sunrise therefore averaged 2.4% of the rate during the day (Table 1).

4. Discussion

Multiple studies of breeding gannets in multiple years at colonies in Scotland, Germany, Canada and the Channel Islands, and one of immature gannets, all show extremely low levels of flight activity of gannets at night, and no plunge diving at night (Table 1). Data from birds in the non-breeding season show flight activity between sunset and sunrise is consistently around 1% to 4% of the amount recorded during daytime. Data from breeding adults show flight activity between sunset and sunrise averaging 7% of the daytime level, with diving activity between sunset and sunrise averaging 3% of the daytime level (Table 1). Higher nocturnal flight activity of breeding adults than of adults during the non-breeding season could possibly reflect high energy demands of breeding, forcing birds to extend foraging effort, but may simply result from the fact that almost all nocturnal flight activity occurs during civil twilight. In summer, civil twilight lasts longer than in winter (Fig. 1), and there may be no astronomical twilight or night. In winter, night represents a much greater proportion of the period from sunset to sunrise (Fig. 1), so lack of flight activity at night reduces the average level of flight activity between sunset and sunrise during winter compared to summer.

The logger data from non-breeding adult gannets are robust as they are from a large sample size over several winters (Garthe et al., 2012), and can be disaggregated by region and by migration extent of individuals. The low level of flight activity at night is consistent with the understanding of gannet natural history; as visual hunters gannets will not be able to locate fish on which to plunge-dive during hours of darkness (Lewis et al. 2002, Cleasby et al. 2015a), and in the non-breeding season will not need to fly at night to return to nest sites. Gannet migrations are very slow compared to migrations of other seabird species (Garthe et al., 2012; Fifield et al., 2014) and so birds are not under any pressures to migrate during the night.

It is unclear whether differences in estimates of nocturnal flight activity among studies of breeding adults represent differences in behaviour of birds from different colonies or simply reflect chance variation. The largest estimates of the amount of nocturnal flight activity by breeding adults tended to come from the studies based on the smallest sample sizes; a weighted average based on the number of birds in each study would reduce this estimate by about 50%. However, it is also possible that ecological conditions affect amount of flight activity by breeding gannets. The highest estimate of flight activity between sunset and sunrise (20.9% of the daytime level) was from Garthe et al. (1999) who studied just three birds at a colony in Shetland. At that latitude in summer there is a long period of civil twilight, and no astronomical twilight or night. Birds at higher latitude colonies (such as Shetland) might show relatively more nocturnal flight because civil twilight represents a greater part of the period from sunset to sunrise than further south.

Diving activity (and therefore foraging rather than commuting flight) was even less frequent between sunset and sunrise than flight activity. Given that collision risk is higher when gannets are foraging rather than when they are commuting (Cleasby et al., 2015b), the low amount of foraging flight during the twilight period will further reduce collision risk at that time of day compared to flight during the day. This suggests that a case could also be made for using a lower flight height distribution for the few birds still flying during twilight compared with that used for gannets flying during the day.

Gannet flight activity differs considerably between daytime, civil twilight, and darker periods (nautical twilight, astronomical twilight and night). Thus there would be merit in developing a more nuanced Band model taking account of the activity patterns in these different periods. This, additionally, would account for the very different durations of twilight and night at different times of year. However, in the short term, predictions from Band modelling could be improved by adopting the evidence-based values for flight activity during the nocturnal period (i.e. from sunset to sunrise) in the current Band model.

Based on the average percentage of daytime flight activity that was observed between sunset and sunrise, we recommend that precautionary values of the nocturnal activity factor used with the Band model for estimating collisions should be 8% during the breeding period and 3% during the nonbreeding period. This would not require the Band model to be altered to add separate calculations for twilight periods, although that might be a longer term objective. These values are strongly founded on evidence, and are more appropriate than the 25% value currently suggested by Band (2012) which was not evidence-based. Furthermore, we consider that these evidence-based estimates remain precautionary because they use the unweighted average across studies, and a weighted average accounting for sample sizes would reduce the estimate for breeding birds further. 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 *Rissa tridactyla*, great black-backed gull *Larus marinus* and lesser black-backed gull *L. fuscus*. Evidence-based estimates for those species would also help to reduce uncertainty in environmental impact assessments for offshore wind farms.

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Table 1. Summary of flight and plunge-diving activity between sunset and sunrise in relation to daytime levels

Reference	Season	Colony	Number of tags and tag type	Hours of data on foraging trips	Flight activity from sunset to sunrise as % of flight activity during day	Number of plunge dives recorded	% of plunge dives occurring between sunset and sunrise
Garthe et al., 1999	Breeding	Hermaness	3 stomach temperature	215	(20.9%)	32	(0%)
Garthe et al., 2000	Breeding	Funk Island	11 depth & wing-beat		-	336	4.3%
Hamer et al., 2000	Breeding	Bass Rock	17 Argos satellite PTTs	>2000	0%	-	0
Garthe et al., 2003	Breeding	Funk Island	16 depth & temperature	330	6%	315	0.5%
Hamer et al., 2007	Breeding	Bass Rock	53 Argos PTTs, or GPS tags	>5000	0%	-	-
Langston et al., 2013	Breeding	Bempton	42 Argos satellite PTTs	-	0%	-	-
Garthe et al., 2014	Breeding	Bonaventure	5-7 GPS & altimeter	-	-	731	15%
Warwick-Evans et al., 2015	Breeding	Alderney	9 GPS & accelerometer	678	-	1236	0%
Warwick-Evans et al., 2017	Breeding	Alderney	9 GPS & accelerometer	678	15.7%	-	-
Garthe et al., 2017	Breeding	Helgoland	14 GPS & depth	-	-	2557	0.7%
Garthe et al., 2012	Autumn	North Sea	59 geolocator & temperature	-	2.5%	-	-
Garthe et al., 2012	Winter	North Sea to Africa	50 geolocator & temperature	-	1.9%	-	-
This analysis	Autumn	Remaining in North Sea	4 geolocator & temperature	35 days	1%	-	-
This analysis	Autumn	Migrating to Africa	8 geolocator & temperature	71 days	3.8%	-	-
This analysis	Breeding	Bass Rock, immatures	7 geolocator & temperature	4 days	Very low	-	-
This analysis	Winter	Birds in North Sea	4 geolocator & temperature	-	2.4%	-	-

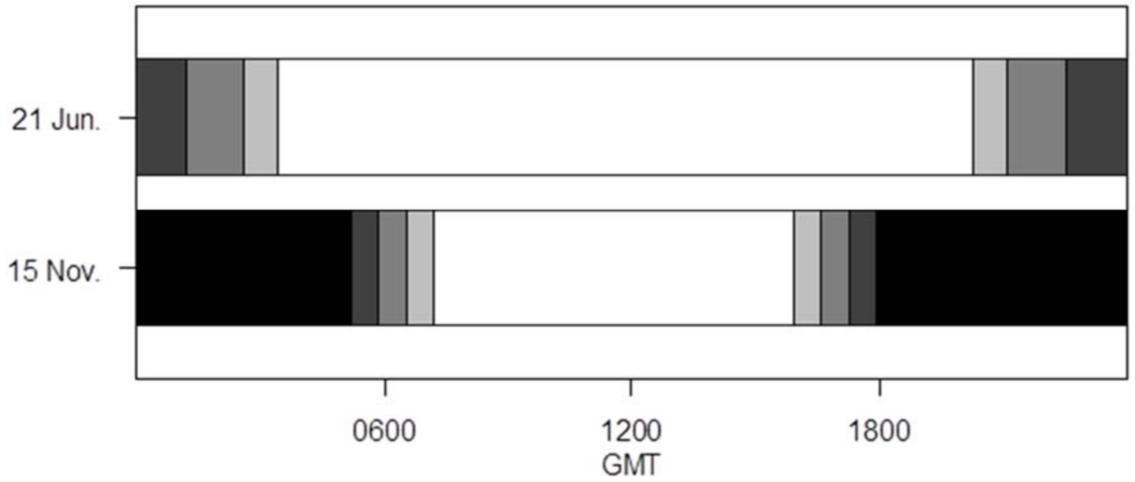

Fig. 1. Illustration of the differences in duration of night (black), astronomical twilight (dark grey), nautical twilight (medium grey), civil twilight (light grey), and day (white) at midsummer (21 June) and during peak migration of gannets through the southern North Sea (November) at 52°30’N 2°30’E, a typical location for a southern North Sea offshore wind farm. Times in GMT.

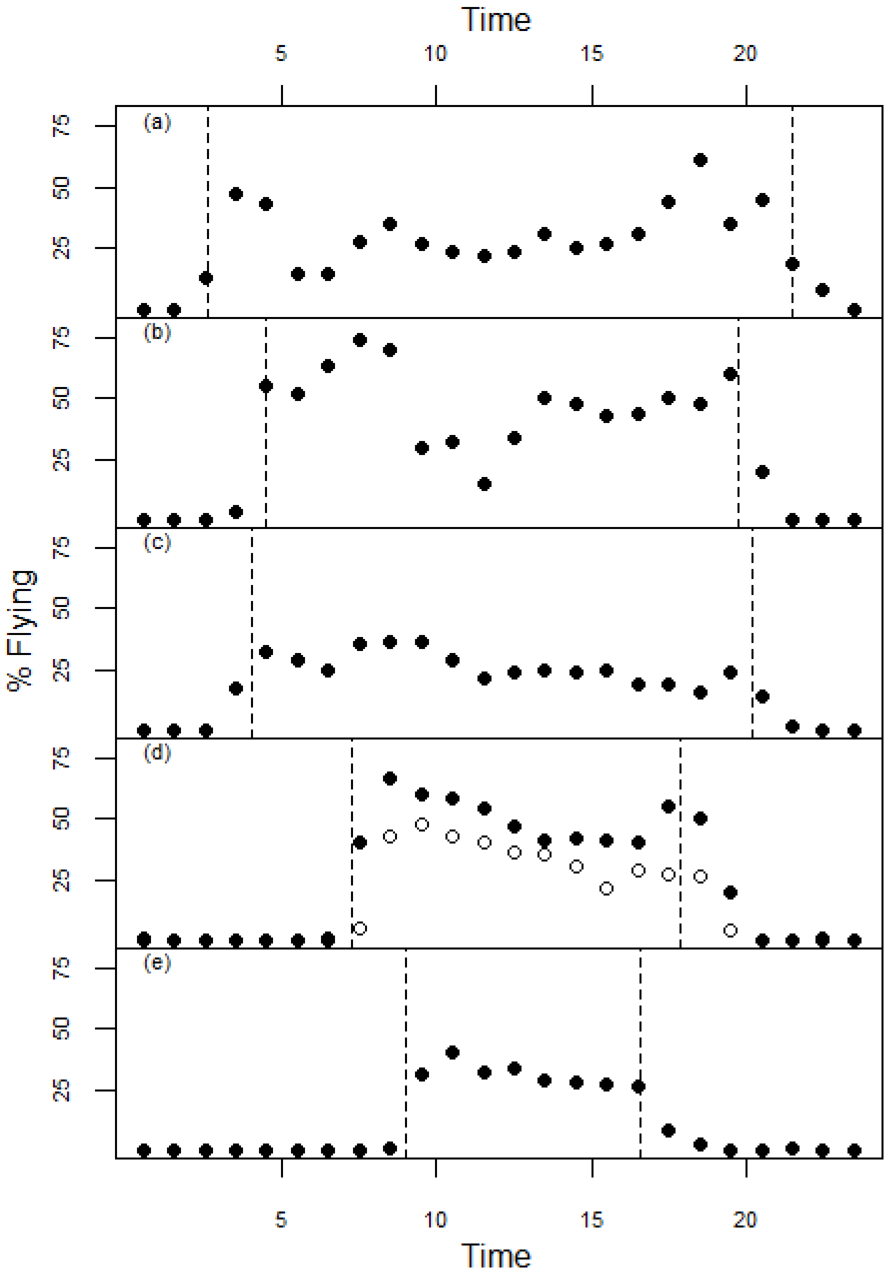

Fig. 2. Flight activity of gannets in relation to hour of the day a) in mid-July 1997 at Hermaness, Shetland (Garthe et al., 1999); b) of chick-rearing gannets from Funk Island, Canada (Garthe et al., 2003); c) of breeding adult gannets at Alderney, Channel Islands in June 2013 (Warwick-Evans et al., 2017); d) in autumn migration (mid-late October) of adult gannets that overwinter in the North Sea (open symbols) or were migrating to overwinter in west Africa (solid symbols) (Garthe et al., 2012); e) in December by adult gannets from Bass Rock that

remained in the southern North Sea through winter (Garthe et al., 2012). Dashed lines show times of sunrise and sunset.

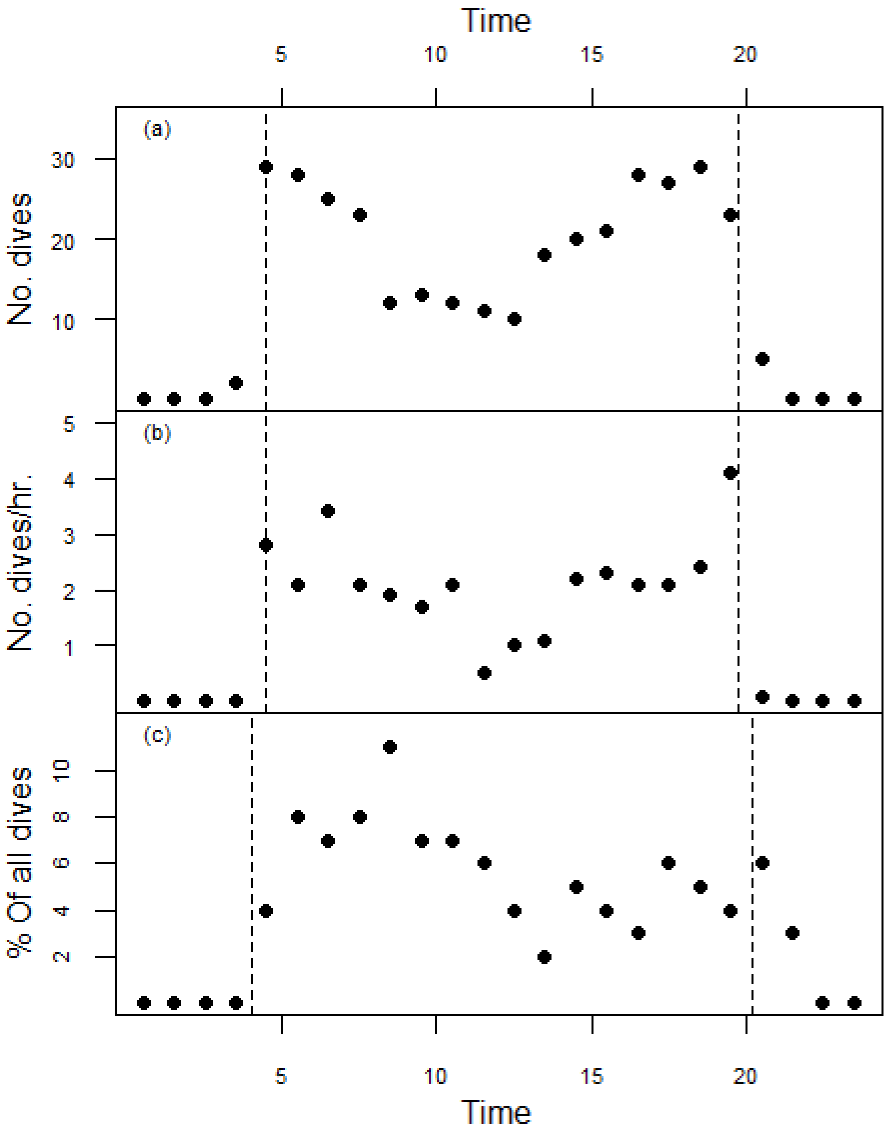

Fig. 3. Diving activity of chick-rearing gannets from a) Funk Island, Canada at different times of day (Garthe et al. (2000)). b) Funk Island, Canada at different times of day (Garthe et al. (2003)). c) Bonaventure Island, Canada at different times of day (Garthe et al. (2014)). Dashed lines show times of sunrise and sunset.

Electronic Supplement. Table S1: Summary of time budgets of seven immature gannets based on categories: speed <3.5 km/h defined as birds resting on the water, speed >3.5 km/h defined as birds possibly spending part of that period in flight

Tag	Period	Periods when speed indicates birds were resting on sea surface (%)	Periods where some flight may have occurred (%)
14904	Day	68.4	31.6
14904	Dusk	68.2	31.8
14904	Night	95.7	4.3
14904	Dawn	84.1	15.9
14905	Day	59.6	40.4
14905	Dusk	63.4	36.6
14905	Night	93.0	7.0
14905	Dawn	80.6	19.4
14907	Day	38.1	61.9
14907	Dusk	58.3	41.7
14907	Night	89.4	10.6
14907	Dawn	81.0	19.0
14908	Day	66.6	33.4
14908	Dusk	79.3	20.7
14908	Night	97.5	2.5
14908	Dawn	77.9	12.1
14909	Day	65.5	34.5
14909	Dusk	60.5	39.5
14909	Night	98.1	1.9
14909	Dawn	89.0	11.0
14910	Day	73.9	26.1
14910	Dusk	65.0	35.0
14910	Night	99.1	0.9
14910	Dawn	98.8	1.2
14912	Day	60.0	40.0
14912	Dusk	66.2	33.8
14912	Night	94.2	5.8
14912	Dawn	71.5	28.5

Electronic Supplement. Table S2. Summary statistics of GPS sampling interval, distance between subsequent locations and speed for the seven immature birds tracked in July 2016.

Tag	period of day	time between locations (mean ± sd, min)	distance (mean ± sd, km)	speed (mean ± sd, km/h)	speed (max, km/h)
14904	night	6.31 ± 2.48	0.16 ± 0.20	0.92 ± 1.25	10.23
	dawn	7.60 ± 2.56	0.53 ± 0.79	5.28 ± 9.72	35.95
	day	7.08 ± 2.70	1.31 ± 3.50	9.16 ± 16.56	77.27
	dusk	6.71 ± 2.41	0.93 ± 1.54	9.62 ± 15.64	43.87
14905	night	10.13 ± 3.72	0.26 ± 0.39	1.79 ± 2.74	26.33
	dawn	10.65 ± 4.95	0.74 ± 1.67	4.32 ± 10.06	36.80
	day	11.06 ± 4.80	2.23 ± 3.5	13.22 ± 19.18	87.07
	dusk	9.68 ± 3.68	1.30 ± 1.99	8.72 ± 12.72	36.98
14907	night	11.91 ± 4.66	0.39 ± 0.30	1.95 ± 1.16	6.94
	dawn	10.66 ± 4.97	0.37 ± 0.29	2.41 ± 1.34	5.25
	day	12.86 ± 10.72	3.26 ± 5.21	15.47 ± 17.88	63.69
	dusk	10.53 ± 5.10	1.17 ± 2.78	5.37 ± 10.90	59.17
14908	night	7.11 ± 6.44	0.11 ± 0.08	1.22 ± 0.80	5.73
	dawn	6.24 ± 2.64	0.61 ± 1.12	6.25 ± 11.64	45.49
	day	7.07 ± 11.08	1.04 ± 1.67	11.08 ± 17.18	73.50
	dusk	6.25 ± 4.12	0.66 ± 1.51	8.02 ± 15.61	62.37
14909	night	13.11 ± 2.75	0.18 ± 0.10	0.87 ± 0.54	2.44
	dawn	13.19 ± 2.99	0.80 ± 2.22	3.44 ± 8.83	42.94
	day	12.39 ± 4.36	2.04 ± 3.15	10.51 ± 15.91	55.57
	dusk	13.27 ± 2.66	2.58 ± 3.98	12.99 ± 19.46	56.21
14910	night	9.76 ± 4.00	0.26 ± 0.22	1.28 ± 0.89	3.27
	dawn	9.91 ± 3.87	0.22 ± 0.09	1.59 ± 0.83	2.97
	day	10.94 ± 3.82	1.44 ± 3.00	7.47 ± 14.66	80.45
	dusk	9.88 ± 4.05	1.48 ± 1.98	10.52 ± 14.81	44.16
14912	night	11.77 ± 2.74	0.26 ± 0.22	1.40 ± 1.11	5.37
	dawn	11.52 ± 2.6	0.46 ± 0.50	2.43 ± 2.64	14.79
	day	11.68 ± 3.51	2.53 ± 3.67	13.48 ± 18.57	77.23
	dusk	11.12 ± 2.77	1.38 ± 3.34	6.88 ± 14.53	58.60

Joint SNCB[1] 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

January 2017 (updated January 2022 to include reference to the Joint SNCB Interim Advice on the Treatment of Displacement for Red-Throated Diver)

Summary of input requirements for displacement assessment

Inputs required:

- Full details of survey techniques.
- Site-based density estimates to include birds on water and in flight.
- Proportions of different age classes of birds (where possible).
- Monthly population estimates presented for minimum two years[2] pre-consent monitoring.
- Raw count data to be included in report appendices.
- Counts to be assessed as mean seasonal peaks[3] (averaged over the years of survey).
- Population estimates for the development footprint and also for the development footprint plus a standard displacement buffer. Buffer of 2km for all species with the exception of divers and sea ducks where a 4km displacement buffer is recommended and red-throated diver where a 10km[4] buffer is recommended.
- Full details of the development (with worst case and typical scenarios) including size of development footprint alone and size plus appropriate outer buffer – usually 2km[5]. (Abundance estimates will be required for site with and without buffer zone).

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[1] SNCB – Statutory Nature Conservation Bodies in this case comprising Natural Resources Wales (NRW), Department of Agriculture, Environment and Rural Affairs / Northern Ireland Environment Agency (DAERA/NIEA), Natural England (NE), Scottish Natural Heritage (SNH) and Joint Nature Conservation Committee (JNCC)

[2] Lower level of data provision may be agreed in some cases (e.g. 18 months ensuring 2 breeding season periods covered if other baseline data available).

[3] Mean seasonal peaks – the mean of the peak counts for each season assessed. If season is April – July and monthly counts of 338, 720, 418 and 552 are recorded the season peak is 720. If three repeat seasons are assessed and the peak counts from the three seasons are 720, 979 and 501 the mean seasonal peak value is the mean of these three counts i.e. 733.

[4] Joint SNCB Interim Advice On The Treatment Of Displacement For Red-Throated Diver (2022).

[5] 2km for most species, 4km for sensitive species (e.g. divers and seaducks) with the exception of red-throated diver (see Joint SNCB Interim Advice On The Treatment Of Displacement For Red-Throated Diver (2022)).

Advice on the treatment of displacement for red-throated diver

Specific advice on the treatment of displacement for red-throated diver is provided within an annex[6] to this main advice note.

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[6] Joint SNCB Interim Advice On The Treatment Of Displacement For Red-Throated Diver (2022)

Summary of data treatment for displacement assessment

Data manipulation and assessment criteria:

- A ‘power analysis’ should be used to identify the probability of being able to detect specified levels of change in abundance associated with varying survey effort. Surveys should provide complete seasonal coverage.
- Any count adjustment and correction to be fully documented (e.g. for availability bias, distance sampling effects).
- Species to be assessed should be selected based on sensitivity scores and local observation or empirical data.
- Breeding season[7] assessment to be done against an appropriate regional population scale, as agreed with SNCBs (but likely to cover total colony counts[8] within mean-max foraging range[9]).
- Non-breeding season assessment done against appropriate population scale (e.g. Furness 2015), as agreed with SNCBs.
- Use published indices of disturbance (e.g. Furness *et al*. 2013) to assign a range of displacement levels for each species individually. The SNCBs note that further evidence is emerging that may confirm or suggest modifications to these scores and likely displacement levels (e.g. Wade *et al. 2016).*
- 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).
- Use above two metrics to compile a ‘Matrix Approach’ table (i.e. representing proportions of birds potentially displaced/dying as a result of OWF development). Table should be presented from 0-100%, in 10% increments for displacement levels. Percentage increments for mortality should also be presented between 0-100%, but including smaller increments at lower values (e.g. 0%, 1%, 2%, 5%, 10%, 20%…..). At this time impacts to breeding success, although plausible are not being considered, unless site specific information exists. The approach here assesses mortality of full grown individuals connected to the development site.
- Impacts to be assessed for a minimum of two seasons (i.e. breeding and non-breeding season). For some species more than two seasons may be appropriate (e.g. based on post-breeding dispersal periods for auks or migration seasons defined for species in Furness 2015), on discussion with SNCBs.
- Seasonal impacts should be summed across seasons. While acknowledged that this could result in birds being assessed in more than one season, and thus double counted, the precautionary approach is required in absence of empirical information on seasonal turnover on development sites.
- Displacement impacts and collision impacts will be added together for assessment of total impacts. This is acknowledged to involve some degree of double counting, but is adopted as a precautionary approach in the absence, at present, of being able to distinguish between birds which might be subject to collision and those that may be displaced.

[7] Potentially suitable seasons/periodicity can be found in Furness (2015), but can vary by location so should also be agreed with SNCBs.
[8] JNCC Seabird Monitoring Programme a good source of most recent UK colony count data.
[9] See Thaxter *et al.* (2012), although more recent tracking data to be used, in discussion with SNCBs, if more up-to-date.

1. Aim of document

This interim displacement advice note replaces an earlier NE and JNCC joint advice note from 2012 (NE and JNCC 2012). It updates the previous note to take account of potential areas of disparity in approaches that have arisen in casework since the original note was issued. It also follows on from a Displacement Workshop (6-7 May 2015), run by JNCC and the Marine Renewables Ornithology Group (MROG) and funded by The Crown Estate, which sought to make progress towards developing a more refined best practice approach to assessing displacement impacts.

Following recommendations made at the workshop, it was agreed that this Joint SNCB interim displacement advice note would contribute towards achieving one of the recommendations (i.e. the creation of a short-term SNCB advice position). This document is intended to address critical areas of clarification and SNCB positioning. It will not attempt to cover (or make progress towards) the more complex issues of displacement assessment at this time. Nor will it cover the expert elicitation recommendation that came out of the displacement workshop, as it was agreed at a meeting of the SNCBs in June 2015 that this could more realistically be produced against a medium-term objective, in a further round of SNCB guidance.

SNCB advice and positioning on displacement assessment methods and approaches will be an iterative process, with at least three stages expected (see Displacement Workshop report 'Next Steps' section, for more details).

The key changes to this document since the earlier advice note are:

- A clearer definition of displacement and barrier terms.
- Further clarity on the application of the 'Matrix Approach'.
- Further clarity on the use of sensitivity scores in relation to the 'Matrix Approach' (based on evidence obtained since the original NE and JNCC advice note (NE and JNCC 2012)).

In addition, this interim advice note aims to provide:

- Advice on how to present information to enable comparable and transparent assessment of the magnitude and potential impacts of seabird displacement from OWFs.
- A method to enable displacement impacts to be compared and potentially combined across multiple sites/projects/activities, with an eye to improving Cumulative Impact Assessment (CIA) approaches for this impact.

Future revision of this advice note is anticipated when new empirical evidence of displacement levels and associated population-level impacts (e.g. changes to productivity or mortality levels) becomes available. Currently our recommendations are aimed at capturing the full range of potential impacts, while encouraging developers to present any species-specific evidence to further refine this as part of both Habitat Regulations Assessment (HRA) and Environmental Impact Assessment (EIA) processes. It is anticipated we will be able to narrow down predicted range of impacts as more results from post-consent monitoring and other studies are produced.

2. Background

Individual species react differently to the construction, operation and decommissioning of OWFs (and other offshore developments). Several species groups display avoidance of operational OWFs. However, for all development types during operation, construction and decommissioning, activities such as towing, pile driving or presence of maintenance/service vessels in the vicinity may cause disturbance (Fox and Petersen 2006; Krijgsveld *et al*. 2011; Vanermen *et al*. 2014). Displacement (see definitions below) can pose a potential ecological threat to seabirds as it can result in habitat loss, in the form of foraging or rafting areas. For adaptive species this may not be a problem, but for less adaptive or constrained species/individuals (e.g. during breeding season) this may result in ecological and/or population level consequences.

3. Definitions of disturbance, displacement, and barrier effects

Disturbance

Disturbance exists when a bird’s normal pattern of activity is interrupted by an anthropogenic activity. Birds using a given area of sea for a range of activities e.g. feeding, resting, commuting etc. may be disturbed by the occurrence of human activities or artifacts in or near those areas. Birds may choose to avoid such sources of disturbance (e.g. by swimming or flying away during the disturbance event to continue their activity elsewhere) and may not return until sometime later. The duration of return times coupled with the frequency of disturbing events, may combine to result in longer term and potentially continual reductions of numbers in an area of impact (i.e. displacement) which may be partial or total.

Displacement

In relation to offshore wind farm development, Furness *et al*. (2013) define displacement as ‘a reduced number of birds occurring within or immediately adjacent to an offshore wind farm’. Displacement, as an effect, may occur both in the area of the disturbance or development and to some distance beyond it – known as a ‘buffer’ (e.g. Mendel *et al*. 2014). The degree of displacement, both in terms of length of time and proportion of the original source population affected, may vary seasonally and between species. We define **displacement as affecting birds present both in the air and on the water**. This is in contrast to the definition in Cook *et al*. (2014) which included only birds on the water as capable of being displaced (birds in flight which were deterred from entering the wind farm are considered to form the component of ‘macro-avoidance’), but while these birds are not at risk of collision they are potentially at risk of impacts arising from their displacement from wind farm areas. Birds that would have previously passed through the footprint of the disturbance area to a more distant feeding, resting or nesting area, but now choose either to stop short or detour around the location are said to be affected by barrier impacts (see below).

Barrier

A barrier is a physical factor that limits the migration, or free movement of individuals or populations, thus requiring them to divert from their intended path in order to reach their original destination. This effect is expected to increase the energy expenditure of birds if they have to fly around the area in question in order to reach their goal. Birds experiencing barrier effects are typically in flight, but not necessarily always so. For the purposes of this description, however, **we interpret barrier effects to mean applying to birds in flight**. Barrier effects are more likely to result in individual/population level impacts, if they occur during the breeding season (and at colonies close to an OWF). Individuals may repeatedly deviate from their normal foraging trajectories at this crucial stage in their annual cycle. Individuals are less constrained during the non-breeding season (i.e. no longer central-placed foragers). Therefore, increases to overall flight costs due to barrier effects while on migration are likely to be very small (Topping and Petersen 2011).

A key distinction between barrier and displacement is that birds experiencing barrier effects typically travel longer distances (i.e. to some point beyond the OWF) and did not intend to forage/utilise the OWF site itself, but some area beyond it. However, it is hard to define where an individual may have intended to travel to, even using tracking data. Therefore, in this advice note we do not provide specific recommendations on the treatment/assessment of barrier effects. As and when technological advances allow for quantitative distinction between these two effects, it may be possible to separate these two impacts within future Displacement Assessment Frameworks (DAFs).

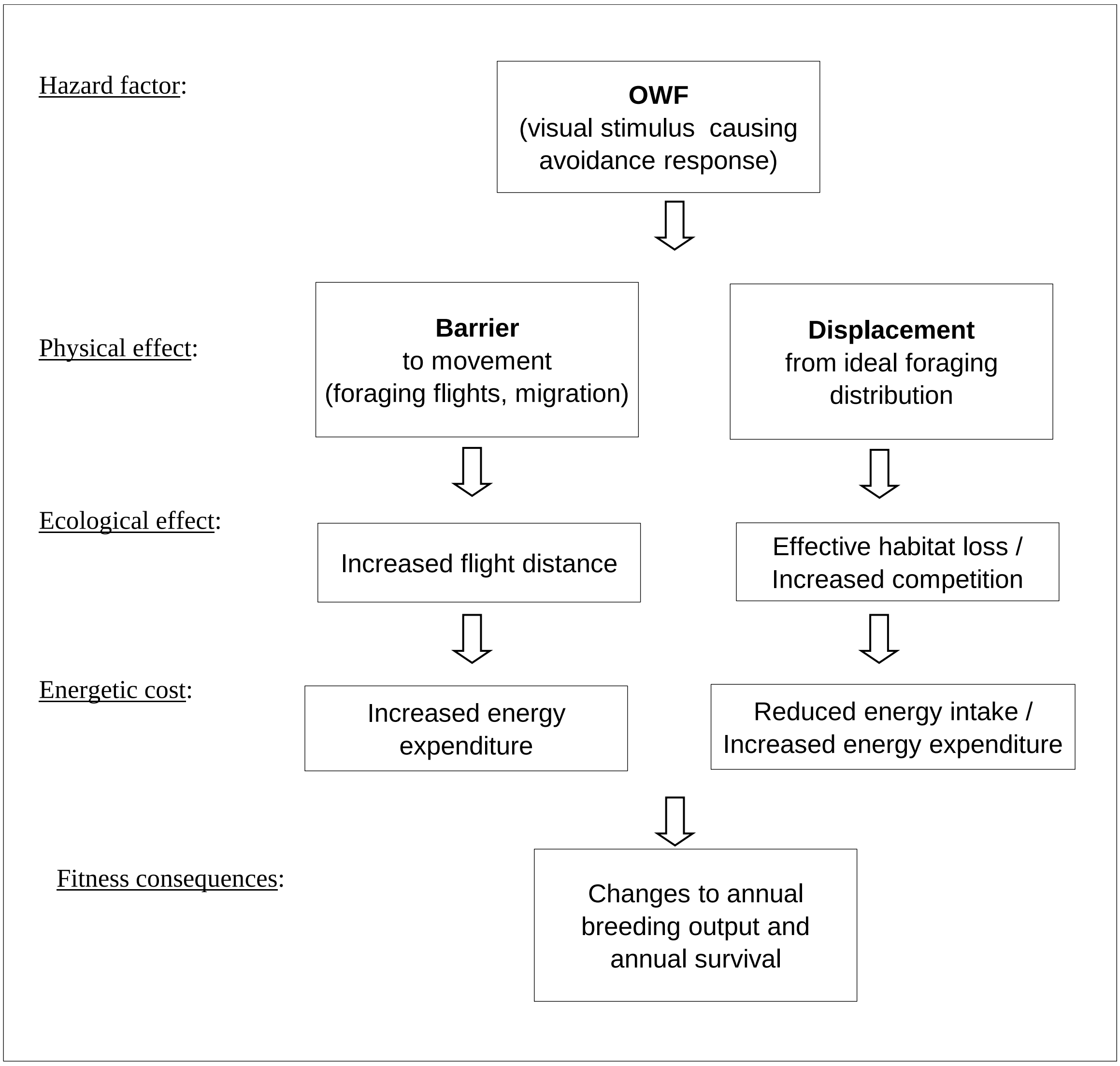
Figure 1. Barrier and displacement effects illustrated (adapted from Petersen *et al*. 2006).

SNCB advice section – barrier and displacement effects

It is recognised that a proportion of the birds recorded in wind farm areas may be transiting through the site (and therefore potentially affected by barrier effects, rather than displacement from the wind farm area) and that this is more likely to be the case for flying birds. However, at present we do not have enough evidence to separate these impacts out and apportion to the two groups. Therefore it is assumed that total numbers of birds on site (flying and on water) are subject to displacement impacts. However, as remote tracking of seabirds continues to expand our knowledge on seabird behavior it may be possible to provide further information on the relative impacts of both issues – this position will be kept under review.

4. Data recording and presentation

In order to address displacement impacts for offshore wind developments, developers should present the following minimum level of data collected in the manner described in guidance documents elsewhere (see Appendix 1). That information should include:

- Full details of survey techniques (platform, transects, temporal and spatial extent of surveys) and how density estimates (and derived abundance estimates) have been calculated.
- Details of if/how density estimates have been corrected to account for availability bias and detection probabilities.
- Accurate information on size of OWF area plus appropriate buffer area calculations.
- Total abundance estimates of birds on water and in flight (and summed). This should be presented separately for the OWF site plus appropriate buffer area, with the extent of buffer area clearly indicated (see Section 6).
- Age or age-class of birds (where this can be determined).

SNCBs recommend **assessing impacts of displacement based on the overall mean seasonal peak numbers of birds (averaged over the years of survey)** in the development footprint and appropriate buffer (see Section 6 on defining appropriate buffer zones). This is a combined estimate of the **number of birds on the water** (corrected for survey coverage and distance analysis/diving species availability bias, if appropriate) and of the **number of birds in flight** (corrected for survey coverage). Methods for estimating birds at sea, both on the water and in flight, have advanced dramatically in recent years. However, standard methodologies for correcting for diving species availability bias are still in development. Hence, decisions made with regards to these components of input data (both for Collision Risk Models (CRM) and displacement) should be discussed and agreed with SNCBs at the time.

Where possible, the ratio of detected age classes should be reported. Age class ratios may differ seasonally and regionally, and ratios obtained from on-site survey data are preferred (if of sufficient quality). Where site specific data on age class ratios are not available there may be other sources of evidence that can be used such as other offshore datasets, colony studies of age ratios or ratios from stable age structures generated from population models. While separation of age classes is not directly used in the 'Matrix Approach' (the matrix should include abundance figures that relate to all birds in the project area, across all age classes), it can be crucial for later stages in the assessment process (e.g. when applying appropriate biologically relevant population scales and making assessments of population-level impacts).

SNCBs advise that at least two full years of monthly survey data should be collected pre-construction. This should be considered the bare minimum for assessment purposes. However, a more appropriate approach is to initially conduct a power analysis to confirm how many years survey data are required to adequately characterise any potential changes to bird abundances (on a species-by-species basis) in response to future OWF development. The number of years survey effort is likely to vary between species, site, and data collection method (e.g. digital aerial versus boat-based observers). Ideally, survey programmes should commence at the beginning of a clearly defined biological season, such that the period of survey will provide complete seasonal coverage in terms of data collection (without the need to combine incomplete data for seasons across different years, when calculating mean seasonal peak abundance estimates).

Data should be provided in a format that allows the calculation of **mean seasonal peak population estimates based on several years data**. For example, for a species with a breeding season from April to July, this requires the average of the peak count between April and July in year one, and the peak count between April and July in a second year. This may require the counts to originate from different months in the two years (e.g. May in the first year and June in the second year). In practice this requires consistent monthly abundance estimates for each year of survey. This allows for year-to-year variation in the precise time (and magnitude) of peak abundance estimates to be taken into account in arriving at a mean peak population estimate. To allow recalculation of values, best practice requires presentation of monthly values in summary and full data from all surveys in an appendix to any report.

5. Selection of species for displacement assessment

Sensitivity to displacement differs considerably between seabird species. To focus impact assessment, SNCBs recommend that consideration is given to each species observed within a development site and informed by:

i) Species presence at the development site (or development sites in the case of in-combination assessments).

ii) Susceptibility to disturbance and habitat specialisation scores for species found in Scottish waters (Furness *et al.* 2013), and the expanded list for wider UK waters (Bradbury *et al.* 2014), covering additional species not previously included in Furness *et al.* (2013).

Furness *et al*. (2013) assessed seabird species occurring in Scottish waters by; 1) scoring species for sensitivity to disturbance by wind farm structures, ship and helicopter traffic, and 2) the degree of habitat specialisation. These two metrics together give an indication of which species are expected to be most susceptible to displacement impacts. The same scoring system and scores were used by Bradbury *et al*. (2014), although they expanded the species list to account for additional species that occur in English waters. Reference to these values will help developers and SNCBs determine the most relevant species for assessment at the site-specific level.

SNCB advice section – screening species for displacement assessment

It is recognised that, regardless of these scores, 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.* 2014; Petersen *et al.* 2006; Mendel *et al.* 2014). **The priority species for assessment of displacement effects will typically be diver and sea duck species, guillemot, razorbill, puffin and gannet.**

As a general guide, any species scoring 3 or more under either category (‘*Disturbance Susceptibility*’ or ‘*Habitat Specialization’*) in Table 1, and which is present in the OWF site or buffer should be progressed to the matrix stage unless there is strong empirical evidence to the contrary. Gannet, with a score of 2, is an obvious exception to this general guide as there are empirical studies demonstrating they are sensitive to displacement and barrier effects (Krijgsveld *et al*. 2011, Vanermen *et al.* 2013). The scores for this species have been revised in a recent publication by Wade *et al. (2016.)*.

Table 1. 'Disturbance Sensitivity' and 'Habitat Specialization' scores from Bradbury *et al*. (2014) (expanded from Furness *et al.* 2013). No 'real' value is implied by these scores, although species with higher scores are considered more sensitive to displacement. (Grey content = species with scores of 3 or higher in either category).

Species	**Scientific name**	**Disturbance Susceptibility**	**Habitat Specialization**
Common scoter$	*Melanitta nigra*	5	4
Red-throated diver$	*Gavia stellata*	5	4
Black-throated diver$	*Gavia arctica*	5	4
White-billed diver$	*Gavia adamsii*	5	4
Velvet scoter$	*Melanitta fusca*	5	3
Great northern diver$	*Gavia immer*	5	3
Greater scaup$	*Aythya marila*	4	4
Common goldeneye$	*Bucephala clangula*	4	4
Goosander$	*Mergus merganser*	4	4
Great cormorant†	*Phalcrocoax carbo*	4	3
Common eider$	*Somateria mollisima*	3	4
Long-tailed duck$	*Clangula hymalis*	3	4
Red-breasted merganser$	*Mergus serrator*	3	4
Great-crested grebe	*Podiceps cristatus*	3	4
Slavonian Grebe	*Podiceps auritus*	3	4
Black guillemot*	*Cepphus grylle*	3	4
Shag	*Phalacrocorax aristoltelis*	3	3
Common guillemot	*Uria aalge*	3	3
Razorbill	*Alca torda*	3	3
Little tern	*Sternula albifrons*	2	4
Sabine's gull*	*Xena sabini*	2	3
Black tern	*Childonias niger*	2	3
Sandwich tern	*Sterna sandivicensis*	2	3
Roseate tern	*Sterna dougalii*	2	3
Arctic tern	*Sterna paradisaea*	2	3
Atlantic puffin	*Fratecula arctica*	2	3
Mediterranean gull*	*Larus melanocephalus*	2	2
Common gull*	*Larus canus*	2	2
Great black-backed gull*	*Larus marinus*	2	2
Black-legged kittiwake*	*Rissa tridactyla*	2	2
Little auk	*Alle alle*	2	2
Northern gannet&*	*Morus bassanas*	2	1
Lesser black-backed gull*	*Larus fuscus*	2	1
Herring gull*	*Larus argentatus*	2	1
Iceland gull*	*Larus glaucoides*	2	1
Glaucous gull*	*Larus hyperboreus*	2	1

Species	Scientific name	Disturbance Susceptibility	Habitat Specialization
Black-headed gull*	*Chroicocephalus ridibundus*	1	3
Grey phalarope	*Phalaropus fulicarius*	1	2
Red-necked phalarope	*Phalaropus lobatus*	1	2
Pomarine skua	*Stercorarius pomarinus*	1	2
Arctic skua	*Stercorarius parasiticus*	1	2
Great skua	*Stercorarius skua*	1	2
Long-tailed skua	*Stercorarius longicaudus*	1	2
Northern fulmar	*Fulmaris glacialis*	1	1
Cory's shearwater	*Calonectris diomedea*	1	1
Great shearwater	*Puffinus gravis*	1	1
Sooty shearwater	*Puffinus griseus*	1	1
Manx shearwater	*Puffinus puffinus*	1	1
Balearic shearwater	*Puffinus mauretanicus*	1	1
Wilson's storm petrel	*Oveanites oceanites*	1	1
European storm petrel	*Hydrobates pelagicus*	1	1
Leach's storm petrel	*Oceanodroma leucorhoa*	1	1

[&] Species to be progressed to 'Matrix Approach' regardless of scores, due to more recent empirical data (see main text for references).

† Species not usually to be progressed to 'Matrix Approach', due to more recent empirical data demonstrating frequent attraction to OWFs (see main text for references).

* Species where some age class differentiation is expected in survey counts.

[$] Species where buffer distance for assessment would be 4 km (2 km being the default for others).

In previous SNCB advice on displacement assessment (NE and JNCC 2012), a 1% threshold of regional population scales was given as a guide for species to be taken forward to quantitative displacement assessment, with the exception of those species with a significant element of turnover (i.e. passage migrants, which might be undercounted). This is no longer recommended as a suitable guide due to the potential for species to be screened out of predictive displacement impact assessments at an individual project level, which might otherwise have been flagged as an issue at the CIA level.

There is an issue with how to appropriately treat species that are more likely to be encountered in development areas as passage migrants (i.e. likely to be transiting through the area and where there may be a high degree of turnover of individuals at a particular site). For these types of species (e.g. skuas and shearwaters) it might be predicted that, as individuals are using the development area only briefly and rarely, they might be more realistically examined solely from the perspective of barrier effects. However, as there is no standardised method for examining barrier effects (albeit some developers have developed useful passage migrant models to predict impacts, largely for collision, on these types of species) we recommend that if turnover is thought to be an issue for a given species at a particular site, this be considered on a site-by-site basis.

6. Displacement buffers

Seabirds showing avoidance reactions to OWF areas may not only be displaced from the footprint itself, but may also be displaced (possibly to a lesser degree) from the surrounding area (or buffer zone). This additional area must be considered, alongside the OWF site footprint, and included in any displacement assessment.

SNCBs recommend for most species a **standard displacement buffer of 2 km** with the exception of the species groups of divers and sea ducks. Divers and sea ducks have been assessed as being the most sensitive species groups to offshore development and associated boat and helicopter traffic. **Therefore for divers and sea ducks a 4 km displacement buffer** is recommended. This is based on evidence of displacement distances which extend beyond 2km for those species groups (e.g. Percival 2010; Kaiser 2002; Percival 2014; Petersen *et al*. 2006; Fox & Petersen 2006; Petersen *et al.* 2013). **For red-throated diver, a 10km displacement buffer** is recommended in line with the Joint SNCB Interim Advice On The Treatment Of Displacement For Red-Throated Diver (2022).

The SNCBs acknowledge that the evidence for displacement effects leading to reduced densities post-construction beyond 2km from operational wind farms in these sensitive species is mixed but note that there is some evidence of displacement effects up to at least 3km (Percival 2010), and even up to 13km (Petersen *et al.* 2014). Extrapolation of the evidence from Percival (2010) suggests an effect that may radiate out to 5.5km before post-construction densities match those pre-construction. While this is an extrapolation, this effect is considerably less than the extent of significant reductions in diver density reported around Horns Rev (Petersen *et al.* 2013). SNCBs acknowledge that in reality there is likely to be a gradient in the reduction of density with increasing distance from OWF site, but the evidence regarding the slope of this gradient beyond 2km is limited. Until further evidence is gathered, it is recommended that a standard displacement level (%) is applied out to 4km for these more sensitive species groups.

SNCB advice section – use of buffer zones for Offshore Wind Farms

All species taken forward to the matrix stage of displacement assessment should be assessed against impacts to development site plus appropriate buffer. For most species the buffer should be 2km outside the OWF footprint. Exceptions for more sensitive species (i.e. divers and sea ducks) require a 4km buffer zone be applied. In both cases no gradient of impact of displacement level should be applied to the buffer zone, as there is not sufficient evidence to underpin any such gradient application on a species-by-species basis. However, as displacement levels in some instances may exceed 4km, the SNCBs feel this flat application of displacement level across the OWF site plus buffer is sufficiently precautionary. For red-throated diver a 10km buffer zone and gradient should be applied as per the Joint SNCB Interim Advice On The Treatment Of Displacement For Red-Throated Diver (2022).

7. Displacement levels

There is a small but increasing evidence-base on species-specific displacement levels from post-construction monitoring of OWFs. However, at present the published evidence remains sparse and often contradictory. SNCBs consequently need to ensure adequate precaution while at the same time taking due account of emerging evidence. Therefore, developers are encouraged to seek and present emerging sources of empirical evidence to provide support for their displacement assessment.

In the face of limited empirical evidence regarding the percentage of individuals likely to be displaced from an OWF footprint and buffer, SNCBs recommend that the full range of potential displacement (from 0% to 100% of the mean seasonal peak bird numbers observed pre-construction) is presented within a ‘Matrix Approach’ (see Section 12 for further details). The values should be presented in 10%

intervals. Matrix tables should be presented with and without appropriate buffer data included, to allow for future changes in understanding regarding buffer zones and effects.

Presentation of 0-100% displacement levels in a matrix is a necessary step for all species taken forward to this stage of the assessment, in the face of current levels of uncertainty. However, it may be appropriate to highlight particular sections within the matrix where displacement levels are most likely to fall (i.e. through interpretation of the 'Disturbance Susceptibility' scores and/or reliable empirical data for a given species). Sufficient evidence should be presented to support selection of any highlighted area within the matrix on a species-by-species basis. Moreover, presentation of the full range of figures should not be interpreted as an indication that the SNCBs will inevitably focus their attention and formulate their advice on the most precautionary scenario.

The use of the collected age class data does not occur at the matrix stage, where the total number of full-grown birds is used. Later stages of the process may use the age data to refine what the impacts to sub-sets of the development site population will be.

8. Translating 'Disturbance Susceptibility' scores into displacement levels for 'Matrix Approach'

The 'Disturbance Susceptibility' scores from ship and helicopter traffic (and to a lesser extent OWF) in Bradbury *et al.* (2014) (Table 1) give a possible indication of potential displacement levels that may be exhibited by each species. Without any additional evidence it is assumed that the scores give a crude, but useful, approximation of the levels of displacement that may be experienced by seabirds and can be used to inform the most likely range of displacement for a given species). However, the SNCBs would note that further evidence is emerging that may confirm or suggest future modification to these scores and likely displacement levels (e.g. Wade *et al. 2016*).

SNCB advice section – translating 'Disturbance Susceptibility' scores

The SNCBs intend to use 'Disturbance Susceptibility' scores as a general guide to appropriate displacement levels on a species-by-species basis, rather than to prescriptively read across to particular levels of displacement. That said, for those species lacking in empirical data on likely displacement levels resulting from OWF construction, there is potential utility in using the scores in order to maintain consistency of approach across different developments (where appropriate). For example, for auk species the SNCBs would typically advise a displacement level of 30-70% (Guillemot and Razorbill have a 'Disturbance Susceptibility' score of 3). For diver species a displacement level of 90-100% is likely to be advised (red-throated diver has a 'Disturbance Susceptibility' score of 5 and empirical studies report high levels of displacement). Some species with 'Disturbance Susceptibility' scores of 1 (e.g. northern fulmar) may not be displaced or hardly displaced. If assessment of these species is recommended in a particular case, usually a displacement level of 10% or less is assumed.

9. Displacement impacts - adult mortality and productivity

Displaced individuals, and other individuals with which displaced birds subsequently interact and compete, may experience fitness consequences (i.e. changes to their likelihood of survival and level of reproductive output). 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 (e.g. Burton *et al*. 2006; Durell *et al*. 2001, 2000). This would result in an increase in the energetic cost of average foraging bouts and consequently to a change in daily energy and time budgets (McDonald *et al.* 2012; Searle *et al.* 2014). During the breeding season this in turn could lead to reduced chick provisioning rates and therefore reduced reproductive success. Young birds fledging at lower weights may also have reduced survival. The increased stress on adult birds that are provisioning chicks means they may end the breeding season in poorer condition than they otherwise would have. This might be expected to have consequences on adult survival during the rest of the year, particularly over winter.

However, there is a lack of empirical evidence on the consequence of displacement to seabirds, in terms of both their mortality and productivity. For other types of birds, e.g. waders, it has been established that displaced individuals are more likely to die than other individuals (Burton *et al.* 2006). Behaviour-based computer simulation models of waders, geese and sea ducks have also demonstrated that displacement can, through changes to the energy budgets of individuals, lead to changes to mortality levels (Pettifor *et al.* 2000; West *et al.* 2003; Kaiser *et al.* 2002). However, Topping and Petersen's model showed no such effects on wintering divers (Topping and Petersen 2011). Searle *et al.* (2014) have recently developed a simulation model that predicts changes to seabird productivity and adult survival arising from simulated displacement and barrier effects associated with OWFs in the Forth & Tay regions of Scotland. However, whether an impact on demographic rates is predicted by such models is highly dependent upon the particulars of the case being modeled and no simple generalities can be drawn.

It seems probable that the fitness consequences of displacement (in terms of productivity and mortality) might vary between stages of the annual life cycle. However, once again, empirical data on this is lacking. Until supporting data can be collected this is considered theoretically plausible but unproven.

SNCB advice section – productivity impacts not assessed

Due to the large degree of uncertainty regarding the impact of displacement on different components of seabird demography (for example, impacts on chick survival arising from displacement effects experienced by adult birds) the SNCBs currently advise that only **mortality of individuals displaced from the development site (plus buffer)** be considered in the 'Matrix Approach' at this time.

10. Selecting appropriate mortality levels for the 'Matrix Approach'

As highlighted in Section 9, Searle *et al.* (2014) demonstrated through simulation modelling, that displacement and barrier effects could impact both breeding season productivity and adult mortality throughout the year. However, as this model operated at an individual-based and colony level, it is not possible to directly translate percentages (of productivity and mortality) from this study into useful application with the 'Matrix Approach' as the latter is based on site-based abundance estimates.

Bird species showing limited flexibility in habitat use will be expected to experience greater fitness consequences from displacement compared to those species that are more generalised (at least in non-marine habitats e.g. Colles *et al*. 2009; Duraes *et al*. 2013). Therefore, the scores of species-specific 'Habitat Specialisation' (Table 1) can be used to provide an indication of the relative scale of mortality arising from displacement for each species. Species considered less flexible in their habitat use, are likely to be more vulnerable to displacement from favoured habitats. A high score for specialisation would therefore be expected to indicate a higher level of potential mortality.

Although it appears to be a sound principle, there is very little, if any, evidence connecting 'Habitat Specialization' scores (Bradbury *et al.* 2014) of individual species with potential mortality levels as a consequence of displacement. Therefore the SNCBs do not advise a standardised translation of these scores across to mortality percentages within the matrix. **It is recommended that the presentation of 0-100% mortality of displaced birds for all species taken forward to the matrix stage. Once again, this should be presented in 10% increments.** However, in acknowledgement that for some less constrained species (e.g. shearwaters) the level of both adult mortality and reduced productivity resulting from displacement are likely to be in the lower range (i.e. 1-10%) it is appropriate to have a finer gradation of percentage mortality impacts at the lower range of the scale (see Table 3).

While the SNCBs do not recommend a direct translation of the 'Habitat Specialisation' score into a specific mortality level, this information is still useful, when combined with expert opinion, as to the likely range of possible mortality impacts resulting from particular levels of displacement.

Finally, it is important to recognise and (qualitatively) account for the quality of habitat being lost at an OWF site and its importance relative to alternative available habitat, which displaced birds may reasonably utilise instead. Expert opinion on mortality levels should take account of site-specific characteristics in coming to a judgement on likely mortality levels. In future it is hoped that, with more empirical evidence linking displacement levels to mortality/productivity consequences, a more quantitative approach can be developed.

SNCB advice section – mortality and productivity

At present the 'Matrix Approach' should only be applied, in relation to **predicted adult mortality levels for birds present on the site (plus buffer)** for each defined season. In other words, a separate productivity matrix is not required at this time. However, this is something which may be revised in subsequent advice should suitable methods be developed along with an improved evidence-base. Appropriate **mortality levels** should be selected based on **expert opinion and in discussion with SNCBs**. The selected mortality levels should be appropriately precautionary, given it is currently intended to (qualitatively) address the potential population level impacts of displacement on both mortality and productivity combined.

As with displacement levels, **mortality levels should be presented for the full range of 0-100%**. However, for mortality the assessment should be presented at **10% increments, as well as 1% increments from 0-5%**, with expert opinion focusing in on highlighting likely potential ranges within this complete range.

11. Seasonality

In addition to the complexity introduced by the uncertainty over likely impacts to different demographic parameters (i.e. mortality versus productivity), there is also the potential for displacement levels and impacts to vary according to season. Given there is currently no empirical evidence on the impacts of displacement to seabirds, the SNCBs do not view it as appropriate at this time to apply varying mortality levels by season. This is because the theoretical arguments, as highlighted in previous sections, regarding breeding versus non-breeding season impacts, could be made in either direction. Therefore, the SNCBs recommend that, for the time being, seasonality in the assessment process, in terms of predicted impacts, should be treated consistently. However, the same need not apply to the treatment of varying abundance estimates for the OWF site (plus buffer) by season.

SNCBs recommend that mean seasonal peak abundance be used to produce, as a minimum, two seasonal matrices (breeding and non-breeding season). However, for a number of species there may be evidence to support an additional breakdown of the non-breeding period to account for periods when distribution, activity or population mix are distinctly different (for example post-breeding aggregations of some auk and sea duck species associated with flightless periods, migration periods etc.). Furness (2015) provides a guide to suggested seasonal divisions for a range of species based on evidence for distribution and abundance of species in UK offshore waters at different times of the year.

The ecology of several species supports a need to consider additional seasons (e.g. the post-breeding season) as a distinct period in their annual cycle, during which the impact of displacement may differ from other periods. A lack of empirical evidence requires that the full range of potential mortality (0 – 100%) be presented (albeit with a selected likely range of percentages being highlighted, according to the sensitivity score proxies, for example).

The predicted mortality levels should be summed across seasons. SNCBs acknowledge that this is a precautionary approach, as it is clearly possible that the same bird may be assessed more than once. However, since a large proportion of the birds present in the non-breeding season are often predicted to be different individuals from those present in the breeding season, assessing against different populations for each season is justified. The relevant SNCB should be contacted for advice on the appropriate population scale to use for each season. Therefore, in apportioning impacts back to SPA colonies (e.g. for HRA), only a small number of mortalities in the non-breeding season will be attributed to a particular colony decreasing the likelihood that these will be the same individuals that were assessed during the breeding season. Similarly, in assessing displacement impacts at a wider population scale (e.g. in EIA), it is assumed that individuals present in the project area in the breeding season will be dispersed over a much larger area during the non-breeding season. This reduces the probability that individuals present at the project site at that time will be the same individuals present in the breeding season. Methods that do not consider mortality impacts on populations across all seasons may result in potential impacts being underestimated.

SNCB advice section – seasonality and summing across seasons

The ‘Matrix Approach’ should be applied to a minimum of two seasons (breeding and non-breeding season) using mean seasonal peak abundance estimates for the OWF site (plus buffer). Where appropriate, additional matrix tables should be created for other discrete seasons (e.g. post breeding and migration periods for relevant species). However, decisions regarding how to treat seasonality in any displacement assessment should be made on a site and species-specific basis, in discussion with SNCBs.

When a multi-season assessment is taking place, the predicted mortalities from these various tables should be summed across seasons, **where the relevant geographical range and population scale remains the same or where the assessment involves apportioning back to an SPA colony**. However, an alternative approach for EIA may have to be taken where the appropriate population scale varies with each season. In these instances, the assessment of potential impacts may need to be undertaken against the most appropriate population scale, for each season in turn, although the default position is to assess the summed annual mortality against the largest population scale in the annual cycle for EIA.

12. 'Matrix Approach'

Data on predicted displacement of seabirds from an OWF site should be presented in the form of a gridded matrix table (or tables) as shown below (Table 3). While presenting the full range of potential displacement and mortality impacts, SNCBs encourage developers to indicate their interpretation of the most likely displacement levels and mortality scenarios by highlighting a range of cells within the matrix, and simultaneously to provide sufficient empirical/modelling evidence to support any highlighted subset of cells.

SNCBs also advise that a range of displacement values are taken through to the assessment of population impacts and not a single figure. The range of population impacts can then also be presented as a matrix so that those levels of displacement which might exceed a particular level of population impact can be easily identified and evaluated. But if only a single figure can be taken forward, this in most cases should be the more precautionary of the sub-set selected (e.g. 20% displaced, 50% mortality, in the below example).

Table 3. Example of Matrix Approach. Cell entries present the estimated number of birds of a given species predicted to be at risk of adult mortality following displacement during a particular season given; i) the seasonal mean peak population within the impacted area (5,000 individuals in this example) ii) the proportion of those birds assumed to be displaced from the impact area; and iii) the assumed proportion of those birds deemed to be at risk of adult mortality as a result of displacement. Cells which are considered, in the light of empirical evidence, to represent the more realistic scenarios can be colour-coded with increasing intensity (shades of green in this instance).

Species **(season)**	**Mortality Level** **(% of displaced birds that die)**													
		0%	1%	2%	3%	4%	5%	10%	15%	20%	30%	50%	80%	100%
Displacement Level **(% of all birds on site)**	0%	0	0	0	0	0	0	0	0	0	0	0	0	0
	10%	0	5	10	15	20	25	50	75	100	150	250	400	500
	20%	0	10	20	30	40	50	100	150	200	300	500	800	1000
	30%	0	15	30	45	60	75	150	225	300	450	750	1200	1500
	40%	0	20	40	60	80	100	200	300	400	600	1000	1600	2000
	50%	0	25	50	75	100	125	250	375	500	750	1250	2000	2500
	60%	0	30	60	90	120	150	300	450	600	900	1500	2400	3000
	70%	0	35	70	105	140	175	350	525	700	1050	1750	2800	3500
	80%	0	40	80	120	160	200	400	600	800	1200	2000	3200	4000
	90%	0	45	90	135	180	225	450	675	900	1350	2250	3600	4500
	100%	0	50	100	150	200	250	500	750	1000	1500	2500	4000	5000

Note: This matrix table would need to be replicated for each screened-in species, each season, and for the OWF site with and without buffer zones included (in terms of total abundance estimates).

In order to determine whether the figures presented in tables (e.g. Table 3 above) are likely to lead to population level effects (i.e. changes to population abundance) it will be necessary to determine which reference population scale(s) (or BDMPS) it is appropriate to relate these predicted displacement impacts to. This will vary between EIA and HRA processes as well as sites and seasons and may range from the breeding population of a species at a single designated site to a north-west European biogeographic migratory or wintering population of a species, possibly even wider. Note that in the case of HRA, where displacement effects take place within areas that are known to be used or likely to be used by birds associated with particular SPAs, assessment of the overall figures must be made at the scale of the populations of each of those individual SPAs (apportioned where necessary between SPAs). The relevant SNCB should be contacted for advice on the appropriate population scale for a given season. For project proposals in English, Irish or Welsh waters the respective SNCBs recommend consideration should be given to the Natural England and JNCC advice on Habitats Regulations Assessment (HRA) screening for seabirds in the breeding season (NE & JNCC 2013) and the non-breeding season populations of seabirds in UK waters report by Furness (2015), when considering appropriate population scales for a given season, for an HRA. For project proposals in Scottish waters, advice should be sought from Scottish Natural Heritage (SNH) on the appropriate population scale to use for each season.

Therefore, unless one particular population scale can be identified as being the only one appropriate to consider for a particular species/season/site combination, the numbers presented in the tables outlined above are thereafter considered in the context of a range of possible reference populations (but see separate guidance on these elements).

13. Combining collision impacts and displacement impacts

The number of birds at risk of reduced individual fitness (i.e. mortality and productivity losses) as a result of displacement is based on the numbers of birds present within a development area and buffer both on the water and in flight. Assessment of the number of birds at risk of mortality as a result of collisions (e.g. with wind turbines) is based on the number of birds present within a development area that are in flight only. The mortality impacts estimated from CRM are assumed to be in addition to any mortality caused by displacement impacts. Productivity impacts due to displacement would be a further addition (but this is not currently quantitatively accounted for under existing methods/advice).

Therefore, at present, the SNCBs regard the **two impacts (collision and displacement) as additive and advise that they should be summed**. In summing the predicted mortalities that arise via these two mechanisms, there is a risk of some degree of double counting as a bird that collides with a turbine cannot be displaced and vice versa. Thus, it is acknowledged that this simplistic approach will therefore incorporate a degree of precaution. The level of precaution is difficult to gauge, but will be highest when the number of birds recorded flying at turbine height (and therefore the predicted number of collisions) is greatest.

SNCBs are seeking further evidence from ongoing and proposed studies into avoidance rates that will help clarify the relationship between collision risk, displacement and so called ‘macro’ avoidance. A recent review of avoidance rates has been completed by the BTO on behalf of Marine Scotland (Cook *et al.* 2014). At some point in the future it is possible that SNCB advice may revisit this additive approach, in light of more advanced techniques for discriminating between birds in flight and birds on the water (in terms of pre-construction abundance data) and between barrier, macro-avoidance and displacement effects.

14. Cumulative impact assessment for displacement

While there is currently no established standardised method for undertaking a CIA process for displacement (or for collision), the **SNCBs recommend that a similar approach be taken to additively combining multiple project’s displacement impacts, to that undertaken for a single project.** In other words, for projects undertaking a CIA for displacement across multiple projects, provided density information and OWF site footprint data (plus appropriate buffer zones) are available, it should be feasible to standardise displacement assessment approaches across even historic projects. Ideally, historic projects will have conducted a displacement assessment along similar lines to those laid out in this interim displacement advice note. However, it is recognised that there are likely to be discrepancies, in terms of variation in displacement levels used for different species, as well as likely mortality levels, and seasons presented, etc.

Several North Sea developers have now undertaken cumulative and in-combination displacement impact assessments for a range of species. Moreover, they have also applied a method to calculate predicted displacement impacts for historic projects that did not present displacement figures for particular species – See:

http://infrastructure.planninginspectorate.gov.uk/wp-content/ipc/uploads/projects/EN010051/2.%20Post-Submission/Representations/ExA%20Questions/20-11-2014%20-%20ExA%20Second%20Written%20Questions/Forewind%20-%20Final%20HRA%20In-combination%20ornithology%20tables.pdf

Use of such methods (or refinement of displacement assessments from historic projects required to feed into CIA for future OWF development applications) should be done in consultation with the SNCBs. Finally, it is not within the scope of this displacement advice note to address all aspects of cumulative assessment. Guidance is available to assist with this elsewhere (King *et al*. 2009).

SNCB advice section – assessing cumulative displacement impacts

In broad terms, displacement impacts from different OWF development sites (plus appropriate buffer zones) should be considered cumulatively (i.e. additively). Any differences in assumptions about species sensitivity to displacement or habitat flexibility between individual project sites should be clearly identified, explained and agreed with SNCBs prior to further analysis. All areas should be assumed to be at carrying capacity, unless there is specific evidence to the contrary. Where displacement assessments may have varied between historic and more recent projects, efforts should be made to standardise approaches. If necessary historic assessments and matrices should be revisited to re-analyse site-based abundance data and bring it into line with current thinking on likely displacement levels, mortality rates, seasons and buffer zones for relevant species.

15. Future development of a 'Displacement Assessment Framework '(DAF)

Several areas of displacement (and barrier) impact assessment remain problematic and there is a need for further investigation and gathering of empirical evidence to support decisions. Nearly all aspects of the assessment of displacement and barrier impacts would benefit from robust and rigorous post-consent monitoring.

The SNCBs recognise that, in several areas, the current document outlines an approach that incorporates high levels of uncertainty. As a consequence aspects of the advised method may be somewhat precautionary (although this does depend on the selection of appropriate displacement and mortality levels within the matrix tables).

Displacement assessment methods are an area of active interest for industry, SNCBs and regulators and needs to be reflected in post-consent monitoring where displacement effects remain uncertain. This joint SNCB interim displacement advice note will be reviewed and updated when new information or approaches are brought to light.

As captured in recommendations from a recent Displacement Workshop (May 2015) organised by JNCC and the MROG, this joint SNCB advice note is intended to address only a short-term gap in advice provision and standardisation of DAF methods within the OWF industry sector. It is anticipated that further steps, with regards to both medium and long-term displacement method development and advice, will follow the publication of this note. Recommendations from the Displacement Workshop are currently being progressed through MROG and SNCB discussions with industry. It is anticipated that further displacement advice revisions may be produced by the SNCBs jointly in the next year.

This advice note was prepared by the Marine Industry Group for ornithology (MIG-Birds), with contributions from Joint Nature Conservation Committee, Natural England, Natural Resources Wales, Northern Ireland Environment Agency and Scottish Natural Heritage

JNCC
Joint Nature Conservation Committee

Cyfoeth Naturiol Cymru
Natural Resources Wales

NATURAL
ENGLAND

Scottish Natural Heritage
Dualchas Nàdair na h-Alba
All of nature for all of Scotland
Nàdar air fad airson Alba air fad

NIEA Northern Ireland Environment Agency
www.daera-ni.gov.uk

An Agency within the Department of
Agriculture, Environment and Rural Affairs
www.daera-ni.gov.uk

INVESTORS IN PEOPLE

References:

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Appendix 1: - Links to guidance on associated topics.

SNH Guidance

Recommendations for the presentation and content of interim marine bird, mammal and basking shark survey reports for marine renewable energy developments. **Available at** <http://www.snh.gov.uk/docs/A1325759.pdf > Accessed 23 March 2016.

Guidance on Methods for Monitoring Bird Populations at Onshore Wind Farms. Available at <http://www.snh.gov.uk/docs/C205417.pdf> Accessed 23 March 2016.

The Crown Estate Guidance

Guide to an onshore wind farm. Available at [redacted] Accessed 23 March 2016

Towards Standardised seabirds at-sea census techniques in connection with environmental impact assessments for offshore wind farms in the UK. Available at [redacted] Accessed 23 March 2016

COWRIE reports

Available at [redacted] Accessed 23 March 2016

RSPB Information

Offshore wind farms and birds : Round 3 zones . Available at [redacted] Accessed 23 March 2016

SOSS Projects

Available at [redacted] Accessed 23 March 2016

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Modelling flight heights of marine birds to more accurately assess collision risk with offshore wind turbines

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Summary

1. The number of offshore wind farms is rapidly increasing as they are a critical part of many countries' renewable energy strategies. Quantifying the likely impacts of these developments on wildlife is a fundamental part of the impact assessments required in many regions before permission for developments is granted. A key concern related to wind turbines is the risk of birds colliding with turbine blades. We present a novel method to generate species-specific flight height distributions which can be used to improve the assessment of collision risk by better reflecting the proportion of in-flight populations at risk of collision.

2. Data describing the flight heights of birds from surveys of 32 potential offshore wind farm development sites were combined to estimate continuous distributions for 25 marine bird species. Observations of flying birds assigned to discrete height categories were treated as observations from independent multinomial distributions with a shared underlying continuous distribution. This analysis enables calculation of the uncertainty around the estimates of the proportion of the in-flight population at risk and consideration of different turbine designs.

3. The mean r^2 for model fit across species was 0·85, and for seven of the species, good independent model validation (80% of independent observations within 95% confidence intervals) provides some confidence for use of the results at alternative sites.

4. All species exhibited positively skewed flight height distributions. These results demonstrate that under the conditions in which the data were collected, raising hub height and using fewer, larger turbines are effective measures for reducing collision risk.

5. *Synthesis and applications*. The methods presented here for modelling continuous flight height distributions provide measures of uncertainty and enable comparison of collision risk between different turbine designs. This approach will improve the accuracy of impact assessments and provide estimates of uncertainty, allowing better evidence to inform decision-making.

Key-words: collision risk, Environmental Impact Assessment, flight behaviour, multinomial distribution, offshore wind farm, pre-construction survey, seabirds, wind turbine

Introduction

Offshore wind energy forms a significant part of international efforts to reduce reliance on fossil fuels. Much of the initial development of offshore wind capacity has occurred in Europe where there is a binding agreement for 20% of energy consumed to come from renewable sources by 2020 (Directive 2009/28/EC), a target which requires a substantial contribution from offshore wind farms (European Commission 2008). Elsewhere, the offshore wind industry is expected to experience significant growth in key markets, such as the United States and China (Snyder & Kaiser 2009; Da *et al.* 2011).

There are concerns about the potential for offshore wind farms (OWFs) to negatively impact wildlife including fish, marine mammals and birds (e.g. Wahlberg & Westerberg 2005; Drewitt & Langston 2006; Gilles, Scheidat & Siebert 2009) through effects such as noise pollution, displacement or direct collision. However, estimating the impacts of OWFs on species and populations is often difficult and imprecise. The estimates are

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important in many regions for permission to develop OWFs and some projects have recently been cancelled or delayed, at a substantial financial cost, due to the impacts predicted for birds (e.g. DECC 2012; Gill 2012). As the industry expands globally, improving the evidence base and reducing the uncertainty surrounding these assessments (Hill & Arnold 2012) will enable more informed decisions to be made about OWFs, benefiting both the renewable industry and statutory national conservation advisors and regulators.

There has been much research into the potential impacts of wind farms on bird populations, in particular the risk of collision with turbines (e.g. Desholm & Kahlert 2005). Marine birds may be particularly sensitive to increases in adult mortality, as they are typically long-lived with low annual productivity (Boyd, Wanless & Camphuysen 2006). Estimates of the number of potential bird collisions with turbines reflect both the abundance of a species in the area concerned and flight behaviour, making some species more likely to collide than others (e.g. Lucas *et al.* 2008; Furness, Wade & Masden 2013). Models have been developed which estimate species-specific collision risk, accounting for characteristics including body length, wing span, flight speed and level of nocturnal activity (e.g. Band, Madders & Whitfield 2007; Band 2012). One key aspect of flight behaviour which contributes to estimates of collisions is the height at which birds fly (Chamberlain *et al.* 2006; Stumpf *et al.* 2011; Furness, Wade & Masden 2013). However, knowledge about the flight height distributions of birds is limited, and the precision of estimates is often not quantified.

To assess the impacts of proposed OWFs, ornithological surveys are carried out to estimate the abundance of species within an area, during which observed birds are usually assigned to a series of height bands (Camphuysen *et al.* 2004). These bands are often delineated by the upper and lower limits of the rotor-swept area of the turbines proposed for the site. This method of estimating the proportion at risk has a number of limitations. The proportion of birds flying between the upper and lower limits is defined here as the proportion flying 'at risk height'. However, as the rotor-swept area is circular, collision risk is not evenly distributed within this band. The greatest risk occurs where the horizontal width of the rotor-swept area is greatest (Fig. 1). Moreover, this overlaps with the central hub, the point at which the chance of being hit by a moving blade is the greatest. Additionally, by assigning birds to fixed height bands, the uncertainty surrounding estimates of the proportion of birds at risk is not calculated, making it hard to determine the precision of estimated collision rates (Cook *et al.* 2012).

We combine pre-construction monitoring data collected from OWF sites across Europe to estimate continuous flight height distributions for a range of marine birds to better estimate the proportion of birds at risk of collision. This distribution makes it possible to consider how different turbine designs and heterogeneous collision risk within the rotor-swept area may affect collision rate estimates.

Materials and methods

DATA COLLATION

We collated estimates of the flight heights of seabirds at sea from pre-construction surveys at OWF sites, by reviewing data contained in published impact assessments, technical reports and peer-reviewed publications and by contacting developers directly (Cook *et al.* 2012). In total, we obtained information for 25 species from 32 sites in the UK and Europe (Fig. 2 and see Table S1 in Supporting Information). In each of these studies, flying birds

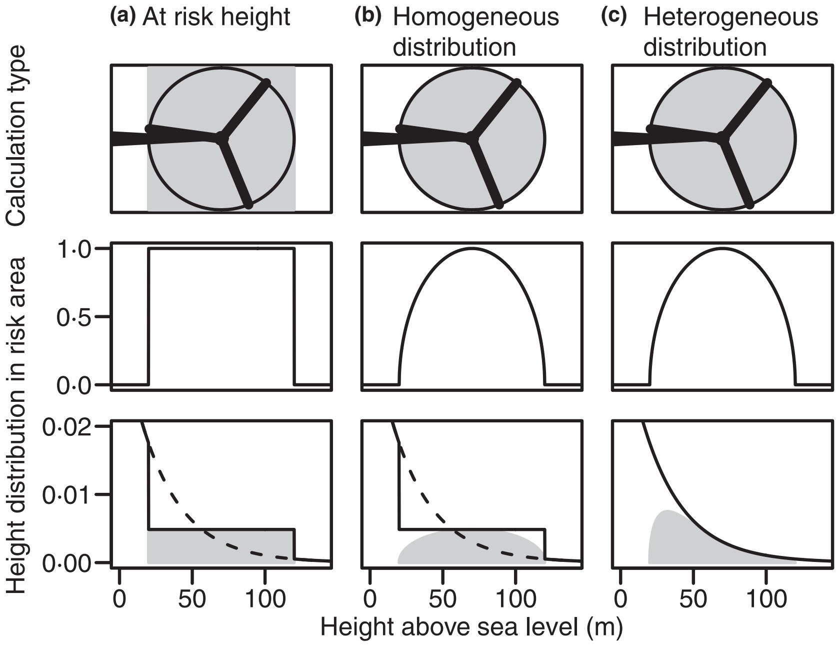

Fig. 1. Diagram representing three methods of calculating the proportion of the population at risk. (a) The proportion at risk height; (b) the proportion within the rotor-swept area assuming a homogeneous distribution within the risk heights; and (c) the proportion within the rotor-swept area assuming a heterogeneous distribution. The grey-shaded areas in the first row represent the areas which are used for each calculation. The second row represents the proportion of birds at each height which are in the risk area. The third row is a hypothetical flight height distribution and the grey-shaded part of this graph represents the estimated proportion of the population at risk. For (a) and (b), the homogeneous distribution is shown with a solid line, and the true heterogeneous distribution with a dotted line.

were assigned to one of several height bands. However, height bands varied between sites as they were typically chosen to reflect the proposed turbine design and to make use of fixed structures as reference points, for example the height of a ship's mast.

The majority of data sets ($N = 27$) were boat surveys, conducted by trained observers following standard industry protocol (Camphuysen *et al.* 2004). Data were limited to those collected during 'snapshot' counts of airborne birds, which excluded those birds following the survey vessel. Of the remaining data sets, three came from shore-based observations of birds at OWF sites close to shore (see Table S1). These followed a similar protocol (see Rothery, Newton & Little 2009) with trained observers assigning birds to height bands defined using fixed objects of known height. Lastly, two remaining data sets came from trained observers positioned on offshore platforms (e.g. Krijgsveld *et al.* 2011). In these studies, birds were assigned to height bands using trigonometry based on estimates of the distance and angle between the observer and the bird.

STATISTICAL METHODS

Continuous distributions of flight heights were estimated for each species, assuming the same distribution across all sites. These distributions were fitted with a flexible curve, not constrained to any specific distributional form. Details of the approach taken are described below.

The number of birds flying at different heights (N_h) was modelled with a cubic spline on the log scale with six knots (Wood 2006 p. 124). Splines are nonparametric, so unconstrained in the shapes they fit, and can be unimodal, bimodal or more complex. This flexibility is useful in fitting to data that may not conform to standard distributional forms. The number of knots defines the degree of flexibility, and six knots was chosen empirically by considering the degree of flexibility required to model bird flight height behaviour. The locations of the knots, k, were set at evenly spaced quantiles of the mid-points of the height categories across all sites, so that more knots were placed where the data were of a higher resolution. The equation for the cubic spline was given by:

$$log(N_h) = \beta * Z \qquad \text{eqn 1}$$

where β is a vector of six coefficients which are estimated in the model fitting process, Z is a matrix of a polynomial function of differences between each height and each of the six knot locations and N_h is the estimated relative number of birds flying at height h (which were based on 1 m categories in this analysis).

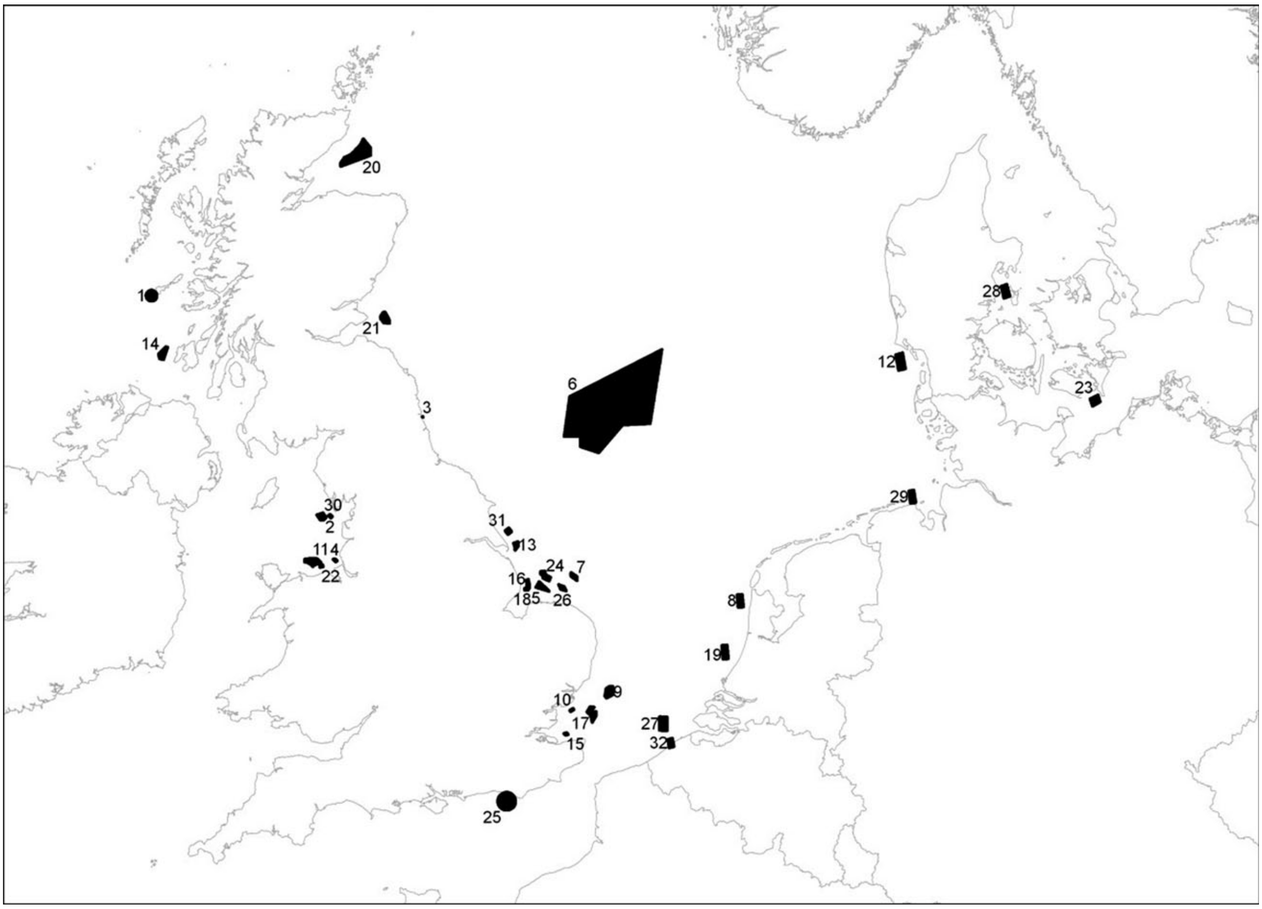

Fig. 2. Location and extent of 32 sites from which bird flight height data were available. These sites include areas of both constructed and proposed offshore wind farms; all data were collected during pre-construction surveys. Site names are: 1 Argyll Array, 2 Barrow, 3 Blyth, 4 Burbo Bank, 5 Docking Shoal, 6 Dogger Bank, 7 Dudgeon, 8 Egmond ann Zee, 9 Greater Gabbard, 10 Gunfleet Sands, 11 Gwynt y Mor, 12 Horns Rev, 13 Humber Gateway, 14 Islay, 15 Kentish Flats, 16 Lincs, 17 London Array, 18 Lynn & Inner Dowsing, 19 Meetpost Noordwijk, 20 Moray Firth, 21 Neart na Gaoithe, 22 North Hoyle, 23 Nysted, 24 Race Bank, 25 Rampion, 26 Sheringham Shoal, 27 Thorntonbank, 28 Tuno Knob, 29 Wangerooge, 30 West of Duddon Sands, 31 Westermost Rough, 32 Zeebrugge.

This spline was fitted to the categorical height data using the following procedure. The number of birds within each categorical height band at each site was assumed to have a multinomial distribution, so each flying bird had a given probability of being in each of the height bands, and the total probability for all height bands combined was one. The likelihood was therefore the product of a multinomial likelihood at each site (or on the log scale the sum of a multinomial likelihood at each site), which assumes the data from each site are independent. The log likelihood was therefore defined as:

$$\ln(\mathcal{L}(\beta|x,k)) = \sum_s \sum_j x_{s,j} \cdot \ln\left[\int_{h=j1}^{j2} N_h dh\right] \qquad \text{eqn 2}$$

where x represents the data, k is a vector of the knot locations, $x_{s,j}$ is the observed number of birds at site s in height band j, and $j1$ and $j2$ are the lower and upper limits of height band j. To fit the spline to the data, this log likelihood was maximized across all sites s and height bands j, using the function 'nlm' in R (R Development Core Team 2012).

Maximising the log likelihood produced estimates of β, which when inserted into eqn 1 described a continuous spline which was the best fit to all the categorical data for each species. The fitted spline provided an estimated number of birds in each height category, N_h, which were standardized *post hoc* to represent the proportion of birds flying in a given 1 m height category (p_h), between 0 and 300 m above sea level. We did not model above 300 m for two reasons: marine birds rarely fly at heights of >300 m (Spear & Ainley 1997; Garthe & Hüppop 2004) and it is hard for observers to accurately record heights over 300 m (Camphuysen *et al.* 2004).

Bootstrapping was carried out to estimate confidence intervals around this maximum likelihood estimate of the flight height distribution. Using the site as the bootstrap unit, 200 bootstrap samples were produced, with a balanced design, such that each site appeared 200 times across all bootstraps. The β coefficients were estimated for each bootstrap sample, by maximizing the log likelihood as above, and 95% confidence intervals for the flight height distribution were calculated from these bootstrapped estimates.

MODEL VALIDATION

To test for an effect of survey method, we examined with a linear model whether the residuals significantly differed by survey method (i.e. boat survey, offshore platform, shore-based count) and also examined interactions between height band and survey method. No effect of survey method was detected ($P > 0{\cdot}9$ for the survey variable and the interaction).

To check the model fit, we correlated the observed proportion of birds in each height category at each site with the modelled proportion of birds expected in each height category. This correlation was weighted by the number of birds at each site, so that sites with more birds contributed more to the correlation coefficient.

For a more independent model validation, each site was removed from the analysis in turn, to produce jackknifed samples, and the estimation and bootstrap procedure were carried out on the rest of the data set. Two hundred bootstraps were conducted on each jackknifed sample, and for each bootstrap estimate of the proportion in each category, 10000 random realisations of height category observations were produced, based on the total number of birds at a site. These were combined to produce a distribution of expected numbers in the category, incorporating uncertainty about the estimate, and random variation in observed numbers, given a fixed proportion. The 95% limits of the expected numbers were taken from the 2·5th and 97·5th quantiles of all 2 million estimates for each category (10000 random realizations × 200 bootstraps). The 95% limits of these distributions were then compared to the observed numbers from the removed site. This process was repeated for each jackknifed sample. If the results can be confidently applied to new sites, we would expect 95% of the observed proportions from the removed sites to lie within the modelled 95% confidence intervals.

Analysing the data in this way assumes that each flying bird observed is independent and therefore that no birds are observed in groups. Although this is not accurate for many species of marine bird, this assumption was necessary as the data did not contain information about group size. Violation of this assumption may be revealed by model predictions having a poor fit to removed sites. This analysis method also assumes that birds are correctly assigned to height categories. In practice, there is likely to be some error associated with assigning birds to height categories by human observers (Pearce-Higgins *et al.* 2009), but categorical measurements will reduce this error, particularly where height categories reflect physical structures. An additional assumption of combining data from several sites in this way is that the flight height distribution is the same at each site and during each survey. Although there are many factors which impact flight height distributions, for example time of year, time of day and wind speed, the data available precluded consideration of these factors.

Estimated proportions of the in-flight populations at risk of collision and associated 95% confidence intervals were calculated for turbines with a 100 m rotor sweep diameter and a hub 70 m above sea level (typical for turbines currently being installed). For each of the 200 bootstraps, we calculated the proportion of the in-flight population estimated to be flying: (a) within the upper and lower risk heights; and within the circular rotor-swept area assuming (b) a homogeneous distribution of birds or (c) a heterogeneous distribution of birds taken from the flight height distribution (Fig. 1). The estimated proportion of the population at risk and the lower and upper 95% confidence intervals were the 50th, 2.5th and 97.5th quantiles of the 200 bootstrap estimates, respectively.

TURBINE DESIGN

We considered two aspects of turbine design: hub height and turbine diameter. To examine the impact of hub height, we calculated the proportion of the heterogeneously distributed in-flight population within the rotor-swept area for 100-m diameter turbines with varying hub heights located 55–110 m above sea level. To examine the impact of turbine diameter, we selected three turbine designs currently deployed and arranged them in homogeneous 20-km arrays, each with a 30 MW total capacity. The outputs of the three turbine designs were 2, 3 and 5 MW, and the diameter of the rotor-swept areas was 80, 90 and 126 m, respectively. The number of turbines required to generate 30 MW output were therefore 15, 10 and 6 for the three arrays, respectively. Given the fixed total array size (20 km), there was great interturbine distance for the array with larger turbines. To remove the effect of height in the comparison of different designs, the hub heights of each turbine were set such that the lower limit of the rotor-swept area was 20 m above sea level. For each of the 30 MW arrays, we calculated the proportion of the heterogeneously distributed in-flight population estimated to fly in the rotor-swept area across the entire array.

Results

MODEL VALIDATION

Correlations between the observed and modelled proportion of flying birds within each height category indicated a good fit of the modelled spline to the data for most species (Fig. 3), with the mean correlation within species r^2=0·85 (Table 1). Common eider *Somateria mollissima* had particularly poor fit with r^2=0·20, as the differences between sites seemed particularly marked (see Fig. S1, Supporting Information). However, these differences led to larger confidence intervals (Fig. 3), and consequently the proportion of observations from removed sites within the modelled 95% confidence intervals was relatively high for common eider (Table 1). Auks and terns had good model fit with average r^2=0·94 and r^2=0·90, respectively. Application to removed sites was less good, with an average percentage of observations within 95% confidence intervals of 86% and 67%, for auks and terns, respec-

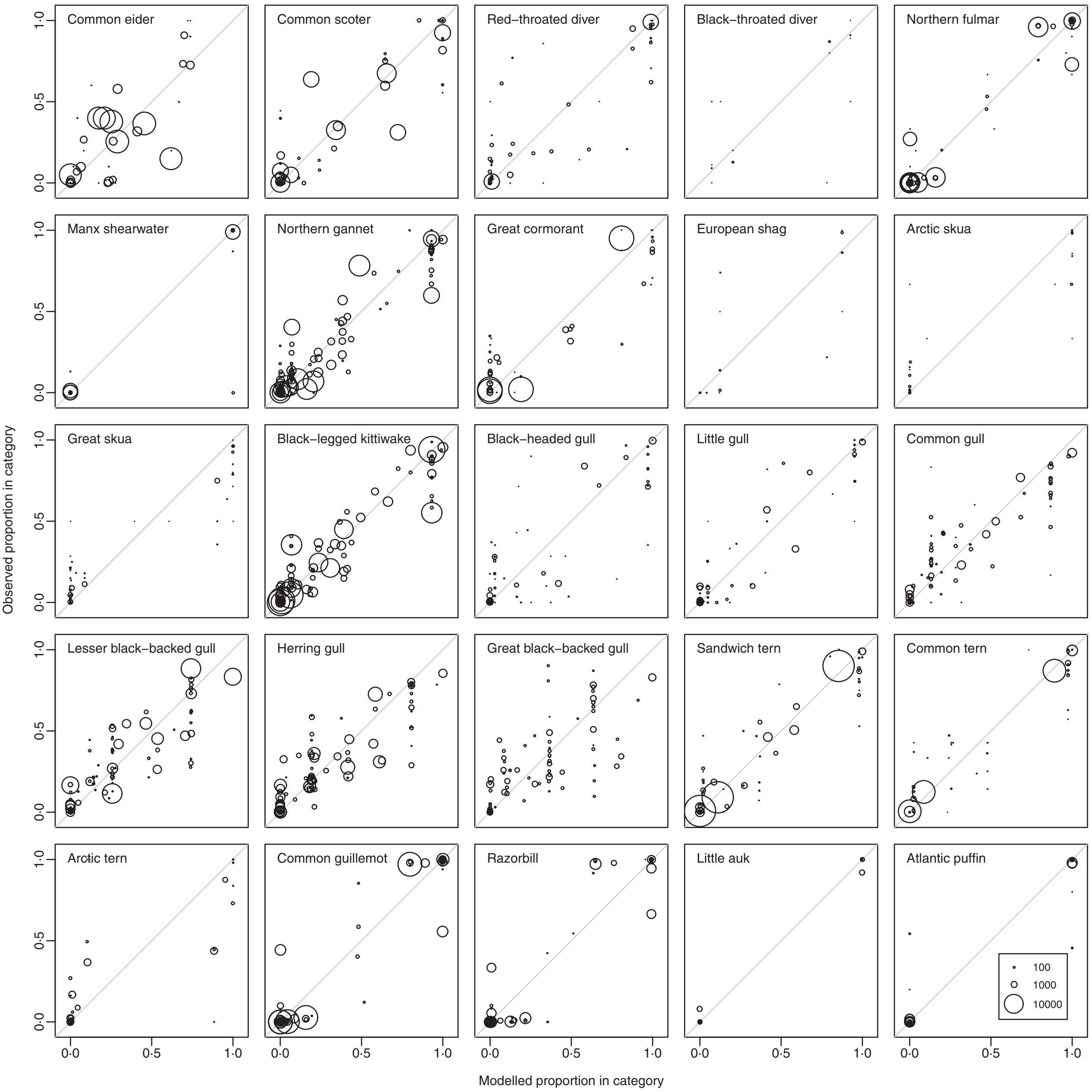

Fig. 3. Modelled and observed proportion of birds in each height category at each site. The relative area of the circle represents the total number of individuals of that species seen at the site. The grey line represents the line of equality (modelled and observed proportions are equal), and well-fitting models will therefore have most points near this line.

Table 1. Correlations and validation statistics for the models for each species. The correlation of model fit is the correlation between the observed and predicted (point estimate) proportions, weighted by the number of individuals of that species observed at the site. The model validation is the percentage of independent observations within the 95% confidence intervals. The proportions of birds estimated to be at risk of collision with a turbine 20–120 m above sea level and associated 95% confidence intervals are presented using three calculation methods

Species	Number of sites	Number of sites >1% of birds	Number of birds	Weighted correlation of model fit (r^2)	Model validation (%)	Proportion of birds at risk height (95% confidence interval)	Proportion of birds within rotor-swept area (95% confidence interval)	
							Homogeneous distribution	Heterogeneous distribution
Common eider *Somateria mollissima*	11	6	34513	0·203	82	0·262 (0·003, 0·683)	0·206 (0·002, 0·537)	0·162 (0·001, 0·411)
Common scoter *Melanitta nigra*	18	8	30847	0·748	55	0·001 (0·000, 0·026)	0·001 (0·000, 0·021)	0·000 (0·000, 0·006)
Red-throated diver *Gavia stellata*	18	9	9686	0·943	64	0·010 (0·003, 0·096)	0·008 (0·002, 0·075)	0·002 (0·001, 0·036)
Black-throated diver *Gavia arctica*	6	4	126	0·901	93	0·073 (0·000, 0·397)	0·058 (0·000, 0·312)	0·024 (0·000, 0·221)
Northern fulmar *Fulmarus glacialis*	22	10	29168	0·931	88	0·002 (0·000, 0·061)	0·001 (0·000, 0·048)	0·000 (0·000, 0·018)
Manx shearwater *Puffinus puffinus*	10	3	6801	0·970	79	0·000 (0·000, 0·000)	0·000 (0·000, 0·000)	0·000 (0·000, 0·000)
Northern gannet *Morus bassanus*	27	14	44851	0·810	45	0·070 (0·021, 0·130)	0·055 (0·016, 0·102)	0·020 (0·005, 0·039)
Great cormorant *Phalacrocorax carbo*	14	6	20227	0·922	58	0·001 (0·000, 0·107)	0·001 (0·000, 0·084)	0·000 (0·000, 0·031)
European shag *Phalacrocorax aristotelis*	4	4	233	0·812	71	0·125 (0·020, 0·704)	0·098 (0·016, 0·553)	0·031 (0·004, 0·272)
Arctic skua *Stercorarius parasiticus*	12	6	331	0·916	71	0·000 (0·000, 0·000)	0·000 (0·000, 0·000)	0·000 (0·000, 0·000)
Great skua *Stercorarius skua*	12	7	1202	0·958	46	0·000 (0·000, 0·013)	0·000 (0·000, 0·010)	0·000 (0·000, 0·004)
Black-legged kittiwake *Rissa tridactyla*	24	13	62939	0·874	44	0·068 (0·035, 0·116)	0·053 (0·028, 0·091)	0·019 (0·010, 0·034)
Black-headed gull *Chroicocephalus ridibundus*	16	9	4436	0·843	59	0·029 (0·000, 0·127)	0·022 (0·000, 0·100)	0·007 (0·000, 0·031)
Little gull *Hydrocoloeus minutus*	17	9	3907	0·880	72	0·048 (0·017, 0·080)	0·038 (0·013, 0·063)	0·012 (0·004, 0·021)
Common gull *Larus canus*	20	14	10190	0·908	62	0·132 (0·083, 0·340)	0·104 (0·065, 0·267)	0·040 (0·024, 0·113)
Lesser black-backed gull *Larus fuscus*	23	10	35045	0·808	55	0·258 (0·118, 0·481)	0·203 (0·093, 0·378)	0·080 (0·034, 0·165)
Herring gull *Larus argentatus*	20	14	25253	0·731	39	0·193 (0·130, 0·354)	0·151 (0·102, 0·278)	0·060 (0·039, 0·119)
Great black-backed gull *Larus marinus*	19	16	8911	0·635	42	0·365 (0·200, 0·520)	0·287 (0·157, 0·409)	0·122 (0·062, 0·185)
Sandwich tern *Sterna sandvicensis*	19	6	33982	0·988	59	0·020 (0·014, 0·124)	0·016 (0·011, 0·097)	0·004 (0·003, 0·030)
Common tern *Sterna hirundo*	18	5	19329	0·996	77	0·025 (0·024, 0·095)	0·020 (0·019, 0·074)	0·006 (0·006, 0·026)
Arctic tern *Sterna paradisaea*	9	6	2571	0·704	64	0·000 (0·000, 0·000)	0·000 (0·000, 0·000)	0·000 (0·000, 0·000)
Common guillemot *Uria aalge*	22	9	36256	0·912	87	0·001 (0·000, 0·081)	0·001 (0·000, 0·063)	0·000 (0·000, 0·023)
Razorbill *Alca torda*	19	10	13172	0·857	89	0·008 (0·000, 1.000)	0·006 (0·000, 0·785)	0·002 (0·000, 0·986)
Little auk *Alle alle*	4	2	1287	0·992	82	0·000 (0·000, 0·000)	0·000 (0·000, 0·000)	0·000 (0·000, 0·000)
Atlantic puffin *Fratercula arctica*	8	6	5979	0·990	86	0·000 (0·000, 0·002)	0·000 (0·000, 0·001)	0·000 (0·000, 0·000)

tively. With auks, particularly, the amount of information available to inform the distribution was small, as many height bands had all or none of the observations (Fig. 3). Gulls had a much greater range of observed proportions (Fig. 3) and fairly good model fit (average r^2=0·81). Application of the modelled proportions to removed sites was poor, with an average of removed observations within 95% confidence intervals of 53%, possibly reflecting the more aggregated behaviour of gulls.

For none of the 25 species were more than 95% of observations from removed sites within the modelled 95% confidence intervals, for only one species was the figure over 90%, and for a further six species, the figure was at least 80% (Table 1). Five species had very poor validation with <50% of observations from removed sites within modelled 95% confidence intervals. This validation revealed that for some species, a high proportion of independent sites conformed to the modelled distributions, but many species had large variation between sites. This may reflect violation of other assumptions, such as independence of observations.

SPECIES FLIGHT HEIGHTS

The modelled distributions of flight heights indicated that for all species of birds considered, the majority of flights

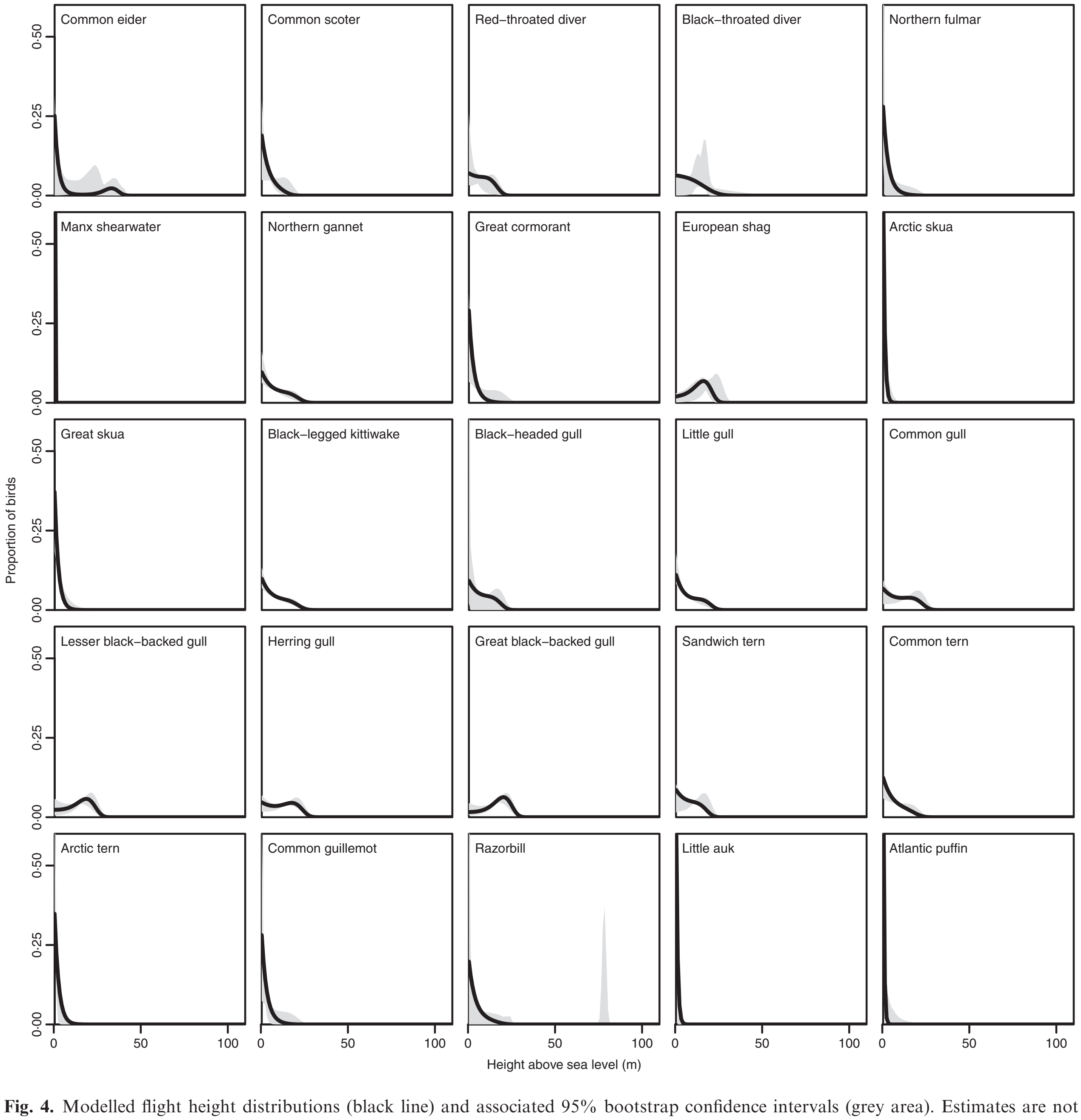

Fig. 4. Modelled flight height distributions (black line) and associated 95% bootstrap confidence intervals (grey area). Estimates are not always in the centre of the confidence limits, because the confidence limits are nonparametric, and proportions are calculated for each bootstrap.

were within 20 m of the sea surface (Fig. 4 and see Appendix S1 in Supporting Information). For several species, confidence intervals revealed a potential secondary peak in flight activity at greater heights (Fig. 4). Flight height distributions were most strongly weighted near the sea surface for Arctic skua *Stercorarius parasiticus*, Manx shearwater *Puffinus puffinus*, little auk *Alle alle* and Atlantic puffin *Fratercula arctica* (Fig. 4). The least skewed modelled distributions were for several of the gull species.

PROPORTION AT RISK

Across species, the proportion within the rotor-swept area from the heterogeneous distribution was on average 26% of the proportion flying at risk height and 33% of the homogenous distribution within the rotor-swept area (Fig. 1, Table 1). However, there was considerable inter-species variability in these figures, and those species with greater proportions flying at risk heights generally had less of a reduction in the proportion at risk when considering the heterogeneous distribution.

TURBINE DESIGN

As hub height increased, the proportion of birds estimated to be at risk of collision declined (see Fig. S2 in Supporting Information). Increasing turbine diameter led to a lower proportion of the in-flight population at risk of collision for most species (Fig. 5). Averaging across all 25 species in the analysis, the proportion of the population at risk of collision in the entire 20-km array was 0·16% with 2 MW turbines, halving to 0·08% with 5 MW turbines. This pattern holds within species; the proportion at risk across the array declined by 29% when the array changed from 2 to 3 MW turbines and by a further 29% when the array changed to 5 MW turbines.

Discussion

Estimating the number of birds likely to collide with turbines is a key part of the impact assessment process for OWFs and requires an understanding of the height at which birds fly. Currently, birds are assigned to site-specific height bands (often determined by a single turbine design) during pre-construction ornithological surveys (Camphuysen *et al.* 2004). This method of estimating the number of birds flying at risk height has three significant drawbacks: (i) It is only possible to consider collision risk with reference to the height bands recorded. Consequently, collision risk for alternate turbine designs cannot be assessed. (ii) It is not possible to account for interactions between a species flight height distribution and the properties of the rotor-swept area. (iii) Estimating uncertainty is difficult, which is vital for understanding the confidence surrounding the estimated impacts. By using a novel approach to combine data collected across multiple sites, we produced continuous flight height distributions that enable all three of these issues to be addressed.

IMPLICATIONS FOR COLLISION RISK AND MANAGEMENT

Our models are consistent with other studies demonstrating that the majority of marine birds have a positively skewed distribution of flight heights and many birds

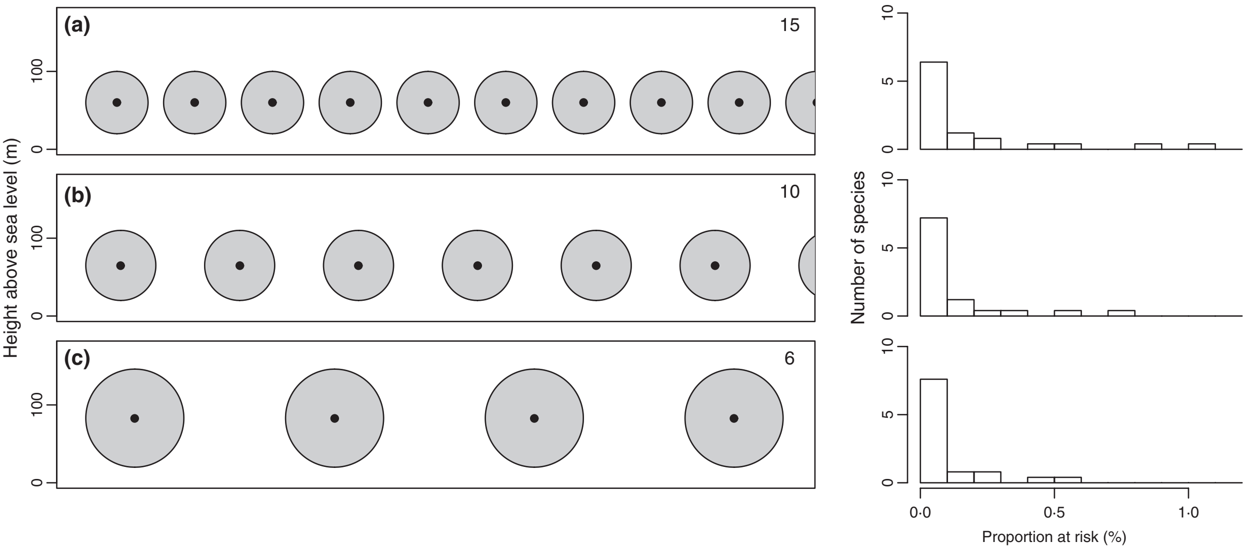

Fig. 5. Left-hand column is a schematic diagram of the rotor-swept area of a section of three 20-km-wide turbine arrays, each with a homogeneous set of turbines which produces 30 MW of electricity. The spaces between the turbines reflect relative spacing, but are not to the scale of the turbines. The number in the top right-hand corner of each turbine diagram indicates the number of turbines required to generate 30 MW of electricity. The right-hand column shows a histogram of the estimated percentage of each species at risk for the entire turbine array.

therefore fly within 20 m of the sea surface (e.g. Krijgsveld *et al.* 2011). Consequently, the proportion of birds within the rotor-swept area of the turbine was substantially lower when considering a heterogeneous rather than a homogeneous distribution within the risk heights. Existing methodologies assume the latter scenario, potentially resulting in an overestimate of the number of birds exposed to the risk of collision.

These results demonstrate that, for the conditions under which these data were collected, the use of higher hubs and larger turbines can be an effective mitigation measure with which to reduce the risk of collision in marine birds. While the total surface area of the turbine rotors remained similar across the three arrays we considered, by increasing rotor diameter, fewer turbines were required, interturbine distances increased and the mean hub height of the turbines was increased. As a consequence, by using turbines with a diameter of 126 m rather than 80 m, the proportion of in-flight populations at risk was on average halved across all species. However, mitigation by use of larger turbines or higher hubs must also take into account the greater altitudes used by migrating birds (Newton 2010; Krijgsveld *et al.* 2011), which may experience an increased collision risk as a result of the use of larger turbines.

The methods presented here to estimate flight height distributions may be of particular value for rare species, for which individual surveys may have small sample sizes and which may be at greater risk of population-level impacts from collisions. This method may also be applied to other situations where knowledge of species flight distributions is needed to inform collision risk, for example construction of power lines (Janss 2000; Martin & Shaw 2010) or onshore wind farms (Lucas *et al.* 2008).

The use of the figures presented here in collision risk models may be appropriate for species which demonstrate consistent distributions across sites and have good validation to independent sites. However, even for species with good validation, good practice should corroborate the figures presented here by comparison of the modelled distributions to site-specific data, as there may be some sites which have very different flight height patterns. It should also be noted that accurate outputs from collision risk models require accurate estimates of all the parameters in the model and associated estimates of uncertainty. Avoidance rates, if derived empirically from observed mortality rates, require an estimation of predicted mortality rates usually with a collision risk model. Birds which are flying in the lower part of the risk height band are at lower risk of collision due to the circular shape of the rotor-swept area. When using a homogeneous distribution, this is encompassed in the apparent 'avoidance' rates derived, however, when using the heterogeneous distribution, this is encompassed in the flight height distribution. There is therefore a need to generate accurate estimates of avoidance that better reflect actual bird avoidance behaviour.

DATA LIMITATIONS AND MODEL ASSUMPTIONS

While our results represent a substantial improvement on the estimates currently used in assessing the proportion of birds at risk of collision, there are nonetheless limitations associated with the data and the underlying model assumptions. It is important to note that most of these assumptions are inherent in the existing approach as well.

Two key assumptions are that heights have been estimated accurately and that birds are not attracted to or displaced by the survey vessel. As no data were available on group size, the model assumes that each bird was an independent observation. Consequently, flocking behaviour will lead to pseudoreplication, and in our model validation, we would expect more observations from removed sites to be outside the confidence limits. Membership of a group may boost foraging success in gulls (Gotmark, Winkler & Andersson 1986), possibly explaining the low proportion of independent observations within the confidence limits for gulls.

Individual birds may alter their flight height behaviour according to weather conditions, time of day, foraging strategy and whether commuting, migrating or foraging (Garthe & Hüppop 2004; Shamoun-Baranes *et al.* 2006; Blew *et al.* 2008; Newton 2010; Krijgsveld *et al.* 2011; Stumpf *et al.* 2011; Wright *et al.* 2012). However, as most data were collected as part of boat surveys, practicalities associated with observer safety and the detectability of birds limited the data collection to periods of daylight, with moderate winds and good visibility (Camphuysen *et al.* 2004; Hyrenbach *et al.* 2007). Evidence about variation in flight behaviour during different conditions is therefore limited. However, many of our study species are considered less likely to forage during the night than during the day (e.g. Daunt *et al.* 2002; Garthe & Hüppop 2004). Birds may avoid areas of heavy wind and rain or spend more time at or under the water surface in these conditions (Pinder 1989; Velando, Ortega-Ruano & Freire 1999), although Procellariiformes (such as northern fulmar *Fulmarus glacialis* and Manx shearwater *Puffinus puffinus*) may have higher flight altitudes during strong winds (Spear & Ainley 1997). Consequently, the absence of data collected during poor weather may bias estimates of the proportion of birds at risk, both when using the modelled distributions and existing methods. Data were also summarized across the year as a whole, again reflecting how they are currently used. Consequently, our data may include observations of migrating birds. During migration, birds are likely to fly at greater altitudes than when foraging or commuting between sites (Garthe & Hüppop 2004; Blew *et al.* 2008; Newton 2010; Krijgsveld *et al.* 2011; Wright *et al.* 2012). If the data do include migrating birds, this variation is likely to be captured by the estimates of precision surrounding our modelled distributions.

Considering these limitations, caution is required when using the presented results to estimate impact, and in general, a precautionary approach is necessary when assessing the potential impacts of developments on wildlife (Sanderson & Petersen 2002). As additional data become available, it will be possible to refine the outputs generated using our approach, increasing its value to the OWF industry by improving the accuracy of the estimates of collision risk.

ALTERNATIVE METHODS FOR ESTIMATING FLIGHT HEIGHT

A key concern about the use of visual observations to estimate flight altitudes is that the data will be negatively biased as recording birds at higher altitudes is difficult. Alternatives for assessing the flight heights of seabirds include tagging, high-definition imagery and radar. Tagging data may overcome some bias associated with weather conditions and diurnal behaviour (Bridge *et al.* 2011; Stumpf *et al.* 2011; Klaassen *et al.* 2012), but offers a restrictive sample size and is not suitable for all species (Burger & Shaffer 2008). High-definition digital imagery is increasingly common in aerial surveys of OWFs (Buckland *et al.* 2012), but data are hard to use on a species-specific basis (Mellor & Maher 2008; Hexter 2009). Radar may positively bias estimates of flight altitudes as low-flying birds are under-recorded due to reflections from the sea surface (Hüppop *et al.* 2006) and species-specific information is sparse (Schmaljohann *et al.* 2008). Consequently, migrants which may fly above 1000 m are included in data sets (Hüppop *et al.* 2006; Krijgsveld *et al.* 2011), positively biasing estimates of flight height. Studies using radar and visual observations suggest that seabird movements occur at lower altitudes, while observations at higher altitudes are migrating passerines or waders (Blew *et al.* 2008; Krijgsveld *et al.* 2011). These comparative studies suggest that the risk of overestimating flight heights of seabirds using radar data may exceed the risk of underestimating altitudes using visual observations. Underestimating seabird flight heights may underestimate the proportion of birds at risk of collision, which should be considered in all uses of visual observations to assess the proportion of birds at risk of collision.

CONCLUSIONS

Accurately estimating the collision risk is a step towards a better understanding of the potential impacts on birds of the rapidly expanding offshore wind energy industry. The standard assessment of the proportion of the in-flight population of birds occurring at a collision risk height is static and can only be used in the height categories in which the data were recorded and also measures the proportion of birds at risk height, overestimating those in the rotor-swept area. Continuous flight height distributions generated by the presented modelling approach enable different turbine designs to be considered, and for some species, the results can be applied with reasonable confidence to novel sites which have a similar use by birds to the sites in this study. Results demonstrate that increasing turbine height or diameter may be a good ways of reducing the risk of collision for many marine birds. This method provides a significant advance in estimating the collision risk of birds with wind turbines and opens up avenues for further refinement of these estimates.

Acknowledgements

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Supporting Information

Additional Supporting Information may be found in the online version of this article.

Fig. S1. Observed and modelled proportions of birds in each height category by species.

Fig. S2. Modelled estimates of the proportion of the population at risk for a 100-m diameter turbine at varying heights.

Table S1. Original sources for flight height data.

Appendix S1. Large graphs of species flight height distributions.

Guidance Note 7: Guidance to support Offshore Wind Applications: Marine Ornithology - Advice for assessing collision risk of marine birds

[redacted]

Published: 2023
Version 1: January 2023

This document is part of a series of guidance notes available on Advice on marine renewables development - marine ornithology.

This guidance note sets out our advice on collision risk modelling. Please see Guidance Note 1 - Guidance to support Offshore Wind Applications: Marine Ornithology which provides the context within which this guidance note is provided. Check the log of updates in guidance note 1 to ensure the most current version is being followed. We expect each and every developer to adhere to this guidance, including the recommended parameters to provide consistent and comparable results for all Scottish projects to be assessed as an individual project and cumulatively. This guidance note should be used in conjunction with the Cumulative Effects Framework (CEF).

Please note this guidance was written prior to the development of a NatureScot position on how to account for the ongoing Highly Pathogenic Avian Influenza (HPAI) mortality event within an impact assessment. We are currently developing this advice and we will provide updates and guidance as they become available. In the interim please contact marineenergy@nature.scot should you have any specific queries.

Contents

1. Introduction

Collision risk modelling (CRM) is the approach used to quantify the risk of collision with offshore wind farm turbines. This part of the assessment process for offshore wind farms has been the focus of continual research and development in the last decade. In addition to the Statutory Nature Conservation Bodies (SNCB's) joint guidance note on avoidance rates, this guidance note specifically sets out NatureScot's recommendations for good practice impact assessment using CRM and signposts the most recent resources we expect to be used within the impact assessments for Scottish casework.

We acknowledge that there is ongoing research on this topic and that this note will therefore be subject to updates. Check the log of updates in our guidance note 1 to ensure the most current version is being followed.

This guidance note covers all seabirds except Procellariforms. Should you be required to undertake an assessment on procellariforms we advise you to contact NatureScot. Migratory birds (other than seabirds) are assessed differently, using a migratory CRM – a report on which is being prepared by Marine Scotland.

2. Collision Risk Models, Options and Scenarios

NatureScot recommends the use of the stochastic collision risk model (sCRM) developed by Masden, E. 2015. Developing an avian collision risk model to incorporate variability and uncertainty. Scottish Marine and Freshwater Science. 6(14) and expanded into the sCRM tool (MacGregor et al. 2018), commissioned by Marine Scotland.

We advise the use of:

> **the 2022 update to the sCRM tool shiny app (Caneco 2022)**. This update should also be used to run deterministic output (with seed specified to enable repeatability). We require that outputs for both stochastic and deterministic CRM are presented using this tool.

The sCRM outputs are provided for the basic Band (options 1 and 2) and extended (option 3) models. We note there are discrepancies between the original offshore Band (2012) descriptions of options and those used in the sCRM. These are described by the sCRM user manual as:

- Option 1 uses the 'basic' model and uses a simple proportion of birds within the rotor reach / collision height also known as proportion at collision height (PCH), applied uniformly over the rotor (the proportion is entered into the tool either using generic or site-specific data).
- Option 2 uses the 'basic' model but uses the flight height distributions entered by the user (and therefore can be derived from site-specific or generic datasets). However, this is an addition to option 1 as it uses Q2r (proportion of birds flying within at risk height band) instead of PCH; where Q2r is a calculated PCH from a flight height distribution.
- Option 3 uses the 'extended' model and uses the flight height distributions entered by the user (and therefore can be derived from site-specific or generic datasets). This is a further addition to option 2 as it includes a flux value within the rotor swept area, based on the flight height distribution.

  'Option 4' from the original offshore Band model has not been included in either Masden 2015 or the sCRM tool but is the equivalent of using the extended model with site-specific flight height distributions.

We note that the extended model can use either site-specific or generic datasets. 'Option 3' is typically referred to as the use of the extended model with the generic dataset Johnston, A., Cook, A.S.C.P., Wright, L.J., Humphreys, E.M., Burton N.H.K. 2014. Modelling flight heights of marine birds to more accurately assess collision risk with offshore wind turbines – Corrigendum. *Journal of Applied Ecology*. 51L 1126-1130. The site specific data can be presented, but generic data based assessment **must be** presented too.

There will be a number of windfarm design scenarios considered for the application. The terms ‘most likely scenario’ and ‘worst case scenario’ refer to the project design envelope and the identification of what is likely to be built out and what would be the worst case scenario from the range of turbine design options provided within the project design envelope. Agreement should be reached with NatureScot and Marine Scotland (MSLOT) on what MLS and WCS scenarios are to be used in advance of final modelling being undertaken, noting these may not be the same for all species.

NatureScot advise that as a minimum the following four scenarios should be modelled and results presented for each CRM species:

1. Most likely scenario (MLS) - Option 2 (using the generic flight height dataset)
2. Most likely scenario (MLS) - Option 3 (using the generic flight height dataset)
3. Worst case scenario (WCS) - Option 2 (using the generic flight height dataset)
4. Worst case scenario (WCS) - Option 3 (using the generic flight height dataset)
5. We require that both breeding and non-breeding season totals for all CRM species, as well as the annual total, are calculated from the monthly outputs (see section 7 and 8). See below for presentation guidance (sections 7 and 8).

3. Biological parameters

We advise the values presented in Appendix 1 are used as biological parameters for the sCRM tool. BTO Birdfacts are currently the main reference source for biological (bird) parameters used in the collision models, i.e. body length and wing span. The default values for these parameters are presented in the sCRM tool.

Nocturnal activity scores remain the same as those used in the Scottish Round 3 / STW applications (Garthe & Hüppop, 2004), **other than for gannet**, where values derived from the review of tagged birds (Furness et al. 2018) should be used. Any deviation from these values will require consultation with NatureScot, with any additional data sources presented including clear reference to which definition of nocturnal is being used, noting that our preference is for nautical twilight to be used.

At present we continue to rely on published data for flight speeds (i.e. Pennycuick 1997; Alerstam et al. 2007), however we recognise more recent studies are contributing to new evidence. Any changes to flight speed would require changes to any avoidance rates used. Until more recent studies are published and agreed the default values for each species should be used.

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(4): 724-734.

Furness, R.W., Garthe, S., Trinder, M., Matthiopolous, J., Wanless, S., 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.

Pennycuick, C. 1997. Actual and ‘optimum’ flight speeds: field data reassessed. Journal of Experimental Biology. 200(17): 2355-2361

Alerstam, T., Rosén, M., Bäckman, J., Ericson, P.G.P., Hellgren, O. 2007. Flight Speeds among Bird Species: Allometric and Phylogenetic Effects. PLOS Biol. 5(8): e197.

4. Densities

Monthly density estimates of flying birds should be derived from site characterisation surveys (as described in Guidance note 2: Baseline Characterisation Surveys and Reporting). The sCRM tool provides three approaches for estimating the variability for monthly density data. We advise that 1000 samples from a distribution of mean densities (e.g. from a bootstrapped sample) is used.

Where stochastic models have been used we require a clear statement as to which variability approach has been chosen and should the first or second approach be used, this will require justification. The bootstrapped data should be provided to enable the modelling to be re-run and the outputs checked.

5. Flight height

Johnston et al. (2014) Modelling of flight heights of marine birds to more accurately assess collision risk with offshore wind turbines. Journal of Applied Ecology (with associated corrigendum) currently remains the recommended reference for generic flight heights and is the default within the sCRM tool. We acknowledge uncertainty remains around best practice for flight height data collection methods, primarily due to absence of agreed validation of techniques. Further discussion and agreement on use of flight height data derived from the site-specific surveys is required for use in either options 2 or 3. If site-specific flight height data are to be presented for context we expect a full description of method, accuracy, precision and comparison with Johnston et al. (2014), with explanation of any differences to inform discussions with NatureScot.

We note that use of site-specific flight height for sCRM requires recalculations of avoidance rates. Our assessment will be based on the use the generic flight height data.

6. Avoidance rates

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. However, an update to this is due out imminently, which will provide estimates calculated specifically for the stochastic CRM (Ozsanlev-Harris et al. *in prep*). This incorporates collision data from all suitable terrestrial, coastal and marine offshore wind farms. Once these are publicly available we will update this guidance note accordingly. Option 3 requires the use of site specific avoidance rates to be calculated, if this is to be undertaken NatureScot require notification of this intention.

For species that have not previously been considered in offshore assessments, i.e. great skua and Arctic skua, we will be recommending that the ‘All Gull’ rates should be used (refer to appendix 1). Any deviations from this advice will require clear justification and evidence.

7. Presentation of CRM - Breeding Season

Our Guidance Note 9: Seasonal periods for birds in the Scottish Marine Environment should be used for determining which months to include for each species’ breeding season. A summary of the CRM results should be presented in the main Ornithology chapter of the EIA Report and HRA. We provide an example of a summary table in Appendix 2. The breeding season total should be calculated from the monthly totals. The table (Appendix 2) should be utilised for each of the four scenarios (MLS – option 2 and 3, WCS – options 2 and 3) and include the stochastic and deterministic outputs for each collision risk species.

We expect the monthly outputs (as per the sCRM tool) for a deterministic model to be presented within a technical ornithology appendix in the EIAR/HRA. The seed run for the deterministic models should be clearly stated along with input parameters.

We advise that the pdf produced by the Shiny App tool should be presented as an Appendix. Presentation of results should be accompanied by input values used, to enable NatureScot to rerun the model, if we deem it necessary to do so. The seed run should be clearly stated in the outputs to allow exactly the same model to be run again.

8. Presentation of CRM - non breeding season

We advise that the CRM should be presented for the non-breeding season for all species at collision risk, these periods should be defined using our Guidance Note 9: Seasonal periods for birds in the Scottish Marine Environment.

The non-breeding assessment should be presented, as per the breeding season guidance (above – section 7) – however, as described in our Guidance Note 10: Advice for apportioning impacts to breeding colonies these collisions should be considered in the context of the regional populations as defined by the 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 Report: 164.

9. Migratory collision risk assessments

An updated review of migratory routes and vulnerabilities across the UK is currently being prepared on behalf of Marine Scotland and The Crown Estate. This work also includes development of a stochastic migration CRM tool (known as mCRM) to enable quantitative assessment of risks to migratory SPA species including swans, geese, divers, seaduck and raptors. The updated review and its associated mCRM tool should be used if available within assessment timescales.

If the updated review is not available within assessment timescales, our interim position is that potential collision risk to migratory species should be assessed qualitatively with reference to the survey results and the existing strategic level report WWT and MacArthur Green. 2014. Strategic assessment of collision risk of Scottish offshore wind farms to migrating birds. Scottish Marine and Freshwater Science Report: 5(12). We note the 2014 review does not include mapping of migratory fronts for Slavonian grebe, goosander, eider, red-breasted merganser or goldeneye, which should still be considered, using other available research. Furthermore, the three diver species were included but were treated as seabirds (with migration bands around the coast): we do not support this approach and wider migratory pathways will be considered in the current review.

10. Future updates to this guidance note

This guidance note will be reviewed and updated as new evidence becomes available, i.e. following publication of relevant research projects, including but not limited to:

- Publication of JNCC review of Avoidance Rates (Ozsanlev-Harris et al. *in prep*)
- Post Consent Monitoring
- The Cumulative Effects Framework
- The Migratory Collision Risk work
- The joint SNCB guidance note

Appendix 1. Interim recommended parameters by species

Species	**AR**	**Flight Speed (m/s)** [1]	**NAF**[2]	**Body length (m)** [3]	**Wingspan (m)**[4]	**Flight Type**	**% of flights upwind**
Northern gannet* (All gulls rate)	0.992	14.9	8 % 1.32	0.94	1.72	Gliding	50

Species	AR	Flight Speed (m/s) [1]	NAF[2]	Body length (m) [3]	Wingspan (m)[4]	Flight Type	% of flights upwind
Black-legged Kittiwake (All gulls rate)	0.992	13.1	25-50% 2-3	0.39	1.08	Flapping	50
Lesser Black-backed Gull (Large gulls rate)	0.994	13.1	25-50% 2-3	0.58	1.42	Flapping	50
Herring gull (Large gulls rate)	0.994	12.8	25-50% 2-3	0.6)	1.44	Flapping	50
Great Black-backed Gull (Large gulls rate)	0.994	13.7	25-50% 2-3	0.71	1.58	Flapping	50
Sandwich tern (All gulls and terns rate)	0.990	10.3	Consult NatureScot	0.38	1	Flapping	50
Common gull, Black-headed gull (small gulls rate)	0.995	Consult NatureScot	Consult NatureScot	Consult NatureScot	Consult NatureScot	Flapping	50
Other marine species (All gulls and terns rate)	0.990	Consult NatureScot	Consult NatureScot	Consult NatureScot	Consult NatureScot	Consult NatureScot Consult NS	Consult NatureScot

Table 1 – interim recommended parameters for the Basic Band model – Option 1 or 2 (Band 2012)

[1] All flight speeds from Alerstam (1997) except for Gannet from Pennycuick (1987) and Sandwich Tern from Fijn and Gyimesi (2018)
[2] All based on Garthe & Hüppop (2004) other than Gannet which is from Furness et al (2018)
[3] All named species from Snow & Perrins (1987)
[4] All named species from Snow & Perrins (1987)

Species	AR	Flight Speed (m/s)[5]	NAF[6]	Body length(m) [7]	Wingspan (m)[8]	Flight Type	% of flights upwind
Northern gannet* (All gulls rate)	0.993 (±0.0003)	14.9 (0)	0.08 +-0.10	0.94 (0.0325)	1.72 (0.0375)	Gliding	50
Black-legged Kittiwake (All gulls rate)	0.993 (±0.0003)	13.1 (0.40)	Consult NatureScot	0.39 (0.005)	1.08 (0.0625)	Flapping	50
Lesser Black-backed Gull (Large gulls rate)	0.994 (±0.0004)	13.1 (1.90)	Consult NatureScot	0.58 (0.03)	1.42 (0.0375)	Flapping	50
Herring gull (Large gulls rate)	0.994 (±0.0004)	12.8 (1.80)	Consult NatureScot	0.6 (0.0225)	1.44 (0.03)	Flapping	50
Great Black-backed Gull (Large gulls rate)	0.994 (±0.0004)	13.7 (1.20)	Consult NatureScot	0.71 (0.035)	1.58 (0.0375)	Flapping	50
Sandwich tern (All gulls and terns rate)	0.991 (±0.0004)	10.3 (3.4)	Consult NatureScot	0.38 (0.005)	1 (0.04)	Flapping	50
Common Gull, Black-headed Gull (small gulls rate)	0.995 (±0.0002)	Consult NatureScot	Consult NatureScot	Consult NatureScot	Consult NatureScot	Flapping	50

Species	AR	Flight Speed (m/s)[5]	NAF[6]	Body length(m) [7]	Wingspan (m)[8]	Flight Type	% of flights upwind
Other marine species (All gulls and terns rate)	0.991 (±0.0004)	Consult NatureScot	Consult NatureScot	Consult NatureScot	Consult NatureScot	Consult NatureScot	Consult NatureScot

Table 2 – interim recommended summary data for the stochastic CRM model (McGregor et al 2018)

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[5] All flight speeds from Alerstam (1997) except for Gannet from Pennycuick (1987) and Sandwich Tern from Fijn and Gyimesi (2018)
[6] All based on Garthe & Hüppop (2004) other than Gannet which is from Furness et al (2018)
[7] All named species from Snow & Perrins (1987)
[8] All named species from Snow & Perrins (1987)

Appendix 2: Example of a scenario summary table for collision risk outputs.

This should be used for the four scenarios being presented in the main ornithology chapter, the example here is for Most Likely Scenario (MLS). The full list of input parameters, outputs (as detailed in this note) should be included within an annex of the technical appendix.

	MLS – Option 2 – Generic Flight Height				MLS – Option 3 – Site Specific Height			
Species	**Avoidance Rate**	**Breeding Season**	**Non-Breeding Season**	**Annual Total**	**Avoidance Rate**	**Breeding Season**	**Non-Breeding Season**	**Annual Total**
e.g. Black-legged kittiwake	-	Deterministic value	-	-	-	-	-	-
e.g. Black-legged kittiwake	-	Stochastic mean (range)	-	-	-	-	-	-

The table above is an example of how to present outputs in the form of a template table - noting that there are blank cells for relevant values to be inserted as part of an individual project specific impact assessment.

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ACTUAL AND ‘OPTIMUM’ FLIGHT SPEEDS: FIELD DATA REASSESSED

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Summary

Previously published field observations of the air speeds of 36 species of birds, all observed by the same method (ornithodolite), were compared with estimates of the corresponding minimum power speeds, calculated with a default body drag coefficient of 0.1. This value, which was derived from recent wind tunnel studies, represents a downward revision from default values previously used and leads, in turn, to an upward revision of estimated minimum power speeds. The mean observed air speeds are now distributed around the minimum power speed, rather than in between the speeds for minimum power and maximum range, as they were before. Although the field data do not represent migration, examination of the marginal effects of small changes of speed, on power and lift:drag ratio, indicates that flying at the maximum range speed on migration may not represent an ‘optimal’ or even a practical strategy and that cruising speeds may be limited by the muscle power available or by aerobic capacity. Caution in constructing ‘optimisation’ theories is indicated.

Key words: bird, flight, speed, measured optimum.

Introduction

The mechanical power available from a bird’s flight muscles is limited, and this in turn places both lower and upper limits on the speed at which it can fly horizontally. Similar limits are part of the everyday experience of pilots of both fixed-wing and rotary-wing aircraft. They result from elementary physical principles, which apply to all heavier-than-air flying machines or animals that support their weight aerodynamically. For any such animal or machine, the curve of power *versus* speed for horizontal flight passes through a minimum at an air speed which is commonly called the ‘minimum power speed’ (V_{mp}). There is also a higher air speed, the ‘maximum range speed’ (V_{mr}) at which the ratio of speed to power is a maximum, and therefore so is the (air) distance flown per unit work done. Alerstam and Lindström (1990) and Hedenström and Alerstam (1995*a*) have considered what choice of air speed would minimise either energy consumption or flight time on a multi-stage migratory flight and claimed that the bird’s choice between various alternative ‘currencies’ could be distinguished by observing actual cruising speeds. Such an approach assumes that the characteristic air speeds V_{mp} and V_{mr} can be accurately calculated for the bird under observation, that there are no unknown implications of flying at one speed rather than another, and that the bird is free to select whatever speed produces optimum results, as defined by the theorist. These are bold assumptions.

One cannot observe directly in the field whether or not a bird is flying at V_{mp}, at V_{mr} or at some other point on the power curve. The field observer can only measure the bird’s absolute air speed. Calculating an estimate of V_{mp} or V_{mr}, with which to compare the observed air speed, is a separate operation. The estimate is no better than the theory underlying it, or the values measured or assumed for any variables required in the calculation. This paper is a reassessment of previously published field observations by this author of the air speeds at which 36 species of birds were observed flying (Pennycuick, 1982*b*, 1987, 1990; Pennycuick and de Santo, 1989). These are not speed measurements drawn from anywhere in the literature, but a homogeneous set in which all the field data were obtained by the same method (ornithodolite) and were subject to the same assumptions and sources of error. They are compared with estimates of the minimum power speeds, which were also all obtained in the same way, from Program 1A of Pennycuick (1989). Although estimates of V_{mp} and V_{mr} were published along with the original field data, evidence from recent wind tunnel studies (Pennycuick *et al.* 1996) indicates that these earlier estimates were too low, because the default values used for the body drag coefficient were too high. Upward revision of the calculated speeds calls for a reinterpretation of the speeds at which the birds were observed flying, and this in turn highlights the hazards of extrapolating from a physical theory to predict the speeds at which birds ‘should’ fly in different circumstances.

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Materials and methods

Observed speeds

The ornithodolite, described by Pennycuick (1982*a*), is an optical instrument which records a series of three-dimensional, timed points along a bird's track. The horizontal and vertical distances between each pair of successive points are calculated, and divided by the time difference to get the average horizontal and vertical ground speeds. The air speed (rather than the ground speed) is required for comparison with calculated speeds, and this has to be obtained from the ground speed by vector subtraction of the wind vector (wind speed and direction). Wind measurement is always a significant source of error in field observations of flight speeds, as the wind cannot be measured at the exact time and place where the bird is flying. It was measured immediately after each observation from an electrical anemometer with speed and direction sensors mounted on a pole near the observing position, with a remote readout on the ornithodolite. The ornithodolite is limited to ranges between approximately 50 and 150 m and, while this severely limits the circumstances in which it can be used, it also ensures that the wind sensor is not far from the bird. Care is still needed to ensure that neither the bird nor the wind sensor is affected by upwind obstructions such as trees or buildings, which would modify the wind. All the species were observed in sustained, level, flapping or flap-gliding flight. Speed observations were excluded if the bird was seen to be searching for food, landing, taking off or performing other transient manoeuvres. Most of the observations were of birds flying between feeding areas and nests or roosts in their breeding or wintering areas, covering distances long enough to allow the bird to settle down in steady flight, i.e. a few hundred metres to a few kilometres. The ornithodolite is not suited to measuring very short flights (tens of metres), and these would be excluded anyway, as being inherently unsteady. At the other extreme, some of the birds were observed moving about in migration staging areas, but were believed to be moving relatively short distances when actually observed, and were not thought to be engaged in non-stop migratory flights of hundreds of kilometres.

Calculated speeds

The minimum power speed V_{mp} was estimated from the formula:

$$V_{\mathrm{mp}} = [0.807k^{1/4}(m\boldsymbol{g})^{1/2}]/[\rho^{1/2}b^{1/2}(S_{\mathrm{b}}C_{\mathrm{Db}})^{1/4}]\,, \qquad (1)$$

whose derivation was explained by Pennycuick (1975). Estimates are required for each of the seven variables on the right-hand side of equation 1, three of which are morphological, two environmental and two aerodynamic. It is an inherent weakness of field studies that the primary morphological variables, the mass (m) and the wing span (b), have to be estimated from the means of measurements from samples of specimens, in contrast to wind tunnel studies, where these measurements can be determined for the particular individual under observation. The body frontal area (S_{b}) is a secondary morphological variable, estimated from the mass (Pennycuick *et al.* 1988). The two environmental variables are the acceleration due to gravity ($\boldsymbol{g}$), for which the standard value 9.81 m s^{-2} was used, and the air density (ρ), which was recorded together with each speed observation. All observations in this data set were made near sea level, permitting the air density to be set to a constant value of 1.23 kg m^{-3}.

The two aerodynamic variables are the induced power factor (k) and the drag coefficient of the body (C_{Db}). New evidence on likely values for these variables is the reason for this re-evaluation of the field observations. Program 1A, as published by Pennycuick (1989), calculates V_{mp} (amongst other results) for a specified bird, using a fixed default value of k=1.2, and a default value for C_{Db} between 0.25 (for large birds) and 0.40 (for small ones). These values were derived from wind tunnel measurements of the drag of frozen bird bodies, from which the wings had been removed, but were recognised as anomalous, being far higher than the values usually associated with streamlined bodies (Pennycuick *et al.* 1988; Tucker, 1990). Recently, wind tunnel observations of a teal (*Anas crecca*) and a thrush nightingale (*Luscinia luscinia*) by Pennycuick *et al.* (1996) revealed a well-defined minimum in the graph of wingbeat frequency *versus* air speed, and evidence was presented that this minimum-frequency speed was identical with the minimum power speed. In both birds, the measured minimum frequency speeds were approximately 50 % higher than the V_{mp} estimates from equation 1, using the above default values for k and C_{Db}. The discrepancy could not be resolved by revising the value of k, even far outside the range of values considered possible, but it was resolved by retaining k=1.2 and assigning a value near 0.08 to C_{Db} for both birds, i.e. reducing the earlier estimates by a factor between 3 and 5. The provisional recommendation was to use a default value of C_{Db}=0.1, recognising that the true value might be as low as 0.05 in birds with well-streamlined bodies, but might also be above 0.1 in species with drag-enhancing characteristics, such as trailing legs or prominent heads.

Results

Revising the value assumed for C_{Db} has no effect on the observed speed (V_{obs}), but alters the estimate of the minimum power speed (V_{mp}), in the sense that a lower body drag estimate leads to a higher estimate for V_{mp}, and hence to a lower estimate for the ratio $V_{\mathrm{obs}}/V_{\mathrm{mp}}$. In Table 1, the values of the morphological variables and the observed mean speed (V_{obs}) are listed for each species, together with two estimates for the minimum power speed. V_{mp1} is the estimate obtained by using the 'old' default procedure, which gives a value for C_{Db} in the range 0.25–0.40, depending on the size of the bird, as above. V_{mp2} is a higher value, obtained by using a fixed value of C_{Db}=0.1, regardless of the size of the bird. In Fig. 1, the two ratios $V_{\mathrm{obs}}/V_{\mathrm{mp1}}$ and $V_{\mathrm{obs}}/V_{\mathrm{mp2}}$ are shown as open and filled circles respectively. The species in both Table 1 and Fig. 1 have been arranged in descending order of the ratio $V_{\mathrm{obs}}/V_{\mathrm{mp2}}$.

Table 1. *Morphological data and observed and calculated speeds*

Species	Body mass (kg)	Wing span (m)	Wing area (m^2)	V_{obs} ($m\,s^{-1}$)	V_{mp1} ($m\,s^{-1}$)	V_{mp2} ($m\,s^{-1}$)
1 *Tachycineta bicolor*	0.0201	0.320	0.0133	11.3	4.8	6.8
2 *Oceanites oceanicus*	0.035	0.396	0.0215	10.4	5.2	7.3
3 *Stercorarius parasiticus*	0.390	1.05	0.117	13.3	7.3	10.1
4 *Colaptes auratus*	0.132	0.510	0.0478	12.7	7.1	10.1
5 *Fratercula arctica*	0.398	0.549	0.0369	17.6	10.2	14.0
6 *Rissa tridactyla*	0.387	0.965	0.101	13.1	7.6	10.5
7 *Pachyptila desolata*	0.155	0.635	0.0469	11.1	6.7	9.5
8 *Uria aalge*	0.950	0.707	0.0544	19.1	12.5	16.5
9 *Catharacta skua*	1.35	1.37	0.214	14.9	10.3	13.3
10 *Daption capensis*	0.418	0.875	0.0773	12.3	8.3	11.3
11 *Alca torda*	0.620	0.661	0.0462	16.0	11.0	14.8
12 *Fulmarus glacialis*	0.815	1.13	0.124	13.0	9.3	12.4
13 *Rynchops niger*	0.300	0.990	0.0888	9.9	6.9	9.5
14 *Falco sparverius*	0.090	0.502	0.0344	9.1	6.3	8.9
15 *Larus marinus*	1.55	1.65	0.285	13.0	9.9	12.7
16 *Macronectes giganteus/M. halli*	3.24	1.98	0.326	15.2	11.8	14.9
17 *Phalacrocorax auritus*	1.41	1.16	0.179	14.5	11.3	14.7
18 *Sula bassanus*	3.01	1.85	0.262	14.9	11.9	15.0
19 *Larus atricilla*	0.325	1.03	0.106	9.5	6.9	9.6
20 *Sterna maxima*	0.470	1.15	0.108	10.0	7.5	10.2
21 *Diomedea melanophris*	3.08	2.19	0.354	13.3	11.1	13.9
22 *Eudocimus albus*	0.900	0.951	0.160	12.9	10.6	14.0
23 *Casmerodius albus*	0.874	1.34	0.222	10.6	8.8	11.7
24 *Phalacrocorax aristotelis*	1.81	1.04	0.158	15.4	13.2	16.9
25 *Diomedea exulans*	8.55	3.01	0.583	15.0	13.3	16.7
26 *Egretta caerulea*	0.340	0.980	0.134	8.8	7.2	10.0
27 *Fregata magnificens*	1.47	2.29	0.408	9.3	8.2	10.6
28 *Ajaia ajaja*	1.30	1.25	0.226	11.9	10.6	13.8
29 *Cathartes aura*	1.55	1.75	0.442	10.6	9.6	12.4
30 *Larus argentatus*	0.950	1.36	0.203	9.9	9.0	11.9
31 *Pandion haliaetus*	1.49	1.59	0.300	10.6	9.9	12.8
32 *Ardea occidentalis*	2.50	1.91	0.493	11.0	11.0	13.9
33 *Pelecanus occidentalis*	3.39	2.26	0.450	10.1	11.2	14.1
34 *Ardea herodias*	1.92	1.76	0.419	9.4	10.4	13.2
35 *Haliaeetus leucocephalus*	4.68	2.24	0.756	11.2	12.6	15.8
36 *Coragyps atratus*	2.08	1.38	0.327	10.8	12.1	15.4

V_{obs}, observed mean speed; V_{mp1}, minimum power speed calculated assuming C_{Db}=0.25–0.40 depending on size of bird; V_{mp2}, minimum power speed calculated assuming C_{Db}=0.1; C_{Db}, drag coefficient of the body.

Points that fall on the solid vertical line in Fig. 1 mean that the observed air speed was equal to the appropriate estimate of V_{mp}; that is V_{mp1} for the open circles, and V_{mp2} for the filled circles. Further to the right are two vertical dashed lines representing speeds of 1.45V_{mp} and 1.70V_{mp}. The calculated maximum range speeds (V_{mr}) from Program 1A of Pennycuick (1989) fall between these dashed lines for all species in the sample.

Effect of revised body drag estimates

Fig. 1 shows that, if the estimate of V_{mp} is based on the 'old' estimates of body drag (open circles), most of the field observations fall between the estimates for V_{mp} and V_{mr}. This unsurprising result was noted in the original papers describing the field observations and was taken as evidence that the predicted values for V_{mp} and V_{mr} were not seriously in error, even though the drag measurements, on which they were based, were known to be anomalously high. This position became untenable following the wind tunnel observations on the teal and thrush nightingale described by Pennycuick *et al.* (1996), as the calculated minimum power speeds were below the lowest speeds at which either bird would fly. The filled circles in Fig. 1 show the effect of recalculating the minimum power speeds of the birds in the field sample, using a lower value of C_{Db}=0.1. This value is reconcilable with the wind tunnel observations, but may be an underestimate for some species in the sample, which have poorly streamlined bodies. The result of increasing the estimates of V_{mp} and V_{mr} is that

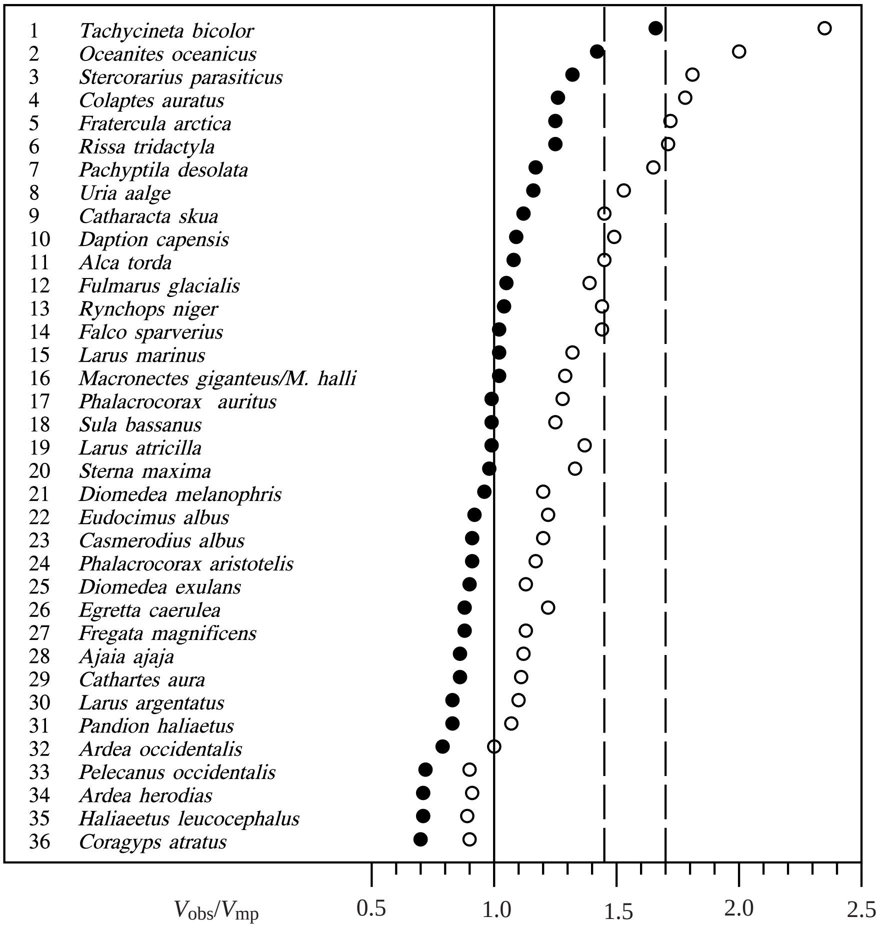

Fig. 1. Ratio of observed air speed V_{obs} to calculated minimum power speed V_{mp} for 36 species, from data in Table 1. Open circles: minimum power speed calculated according to default values of body drag coefficient given in Pennycuick (1989). Filled circles: minimum power speed recalculated according to reduced body drag estimates from Pennycuick *et al.* (1996).

the observed speeds are now centred around the new estimate of V_{mp}, rather than falling between V_{mp} and V_{mr}.

Trend with body mass

Fig. 2 shows that the position of a species in Fig. 1 is correlated with its mass. The smaller species tend to be at the top of the list, that is, with the largest values of the ratio V_{obs}/V_{mp2}, while the larger species are at the bottom. Fig. 3 expresses this in a different way. The ratio V_{obs}/V_{mp2} is well above 1 in the smallest species, but below 1 in some of the larger ones. This ratio can be above 1 for one or both of two different and independent reasons: (1) the bird actually was flying faster than its minimum power speed, and/or (2) the estimate of the minimum power speed is too low. Different reasons may apply to different species, and the differences may be size-related.

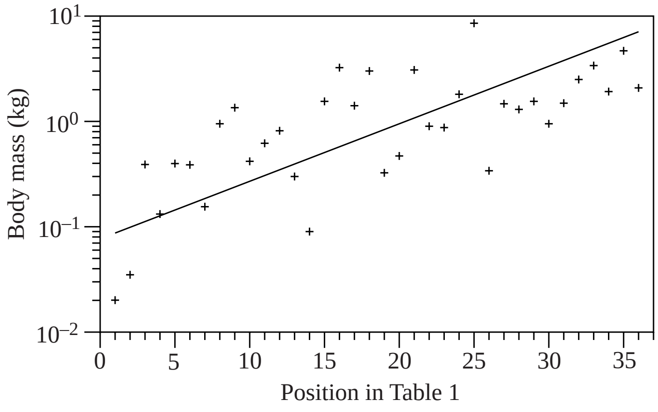

Fig. 2. Linear–logarithmic plot with standard major axis line of body mass *versus* position in Table 1, for the 36 species in the table (r=0.692).

Discussion

Species apparently flying faster than V_{mp}

There is a well known scale effect that makes it easier for small than for large species to extract enough power from their

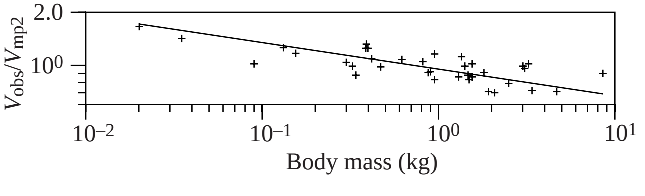

Fig. 3. Double logarithmic plot of the ratio of observed air speed V_{obs} to calculated minimum power speed V_{mp2} (revised drag assumptions), *versus* body mass, for the 36 species in Table 1, with standard major axis line (r=–0.743).

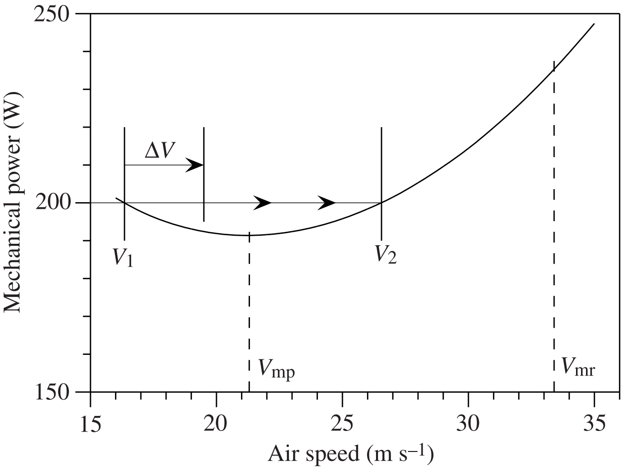

Fig. 4. Curve of mechanical power *versus* air speed at sea level for the whooper swan of Table 2. A steady power output of 200 W from the flight muscles corresponds to a stable equilibrium at speed V_2 or an unstable equilibrium at speed V_1 (see text for explanation). Note that the zeroes are suppressed on both *x* and *y* scales.

flight muscles to fly level at any characteristic speed such as V_{mr} or V_{mp} (Pennycuick, 1989). This is no doubt part of the reason for the trends shown in Figs 2 and 3, and for the isolated position of the two smallest species at the top of the table. These are the tree swallow *Tachycineta bicolor* and Wilson's storm petrel *Oceanites oceanicus*, both showing values of $V_{\mathrm{obs}}/V_{\mathrm{mp2}}$ which suggest that they were flying nearer the maximum range speed rather than the minimum power speed (Fig. 1, filled circles). Under the old assumptions of body drag, the estimates for V_{mr} for these two species would be lower, so that the observed speeds would be much faster than V_{mr} (open circles).

The swallows were passing through a migration staging area, Assateague Island on the east coast of the USA, where they were spending much time feeding, but might also have been moving intermittently further south. The storm petrels were departing from their nesting area on Bird Island, South Georgia, and were believed to be on foraging flights which might take them a few tens or possibly hundreds of kilometres out to sea (Pennycuick, 1982*b*). Either of these situations might give the birds an incentive to maximise their range by flying at or near V_{mr}, but there is no obvious incentive to fly faster than V_{mr}. The observed speeds would be anomalously high under the old estimates of body drag, but intelligible under the new, lower estimates. The next species in order, *Stercorarius parasiticus*, is a chasing predator which may be adapted for high speed in level flight. This is followed by a very heterogeneous set of species, which appeared to be flying near V_{mr} under the old assumptions, but nearer V_{mp} under the new ones.

Species apparently flying slower than V_{mp}

Looking now at the bottom of Table 1, the last four species were apparently flying at only approximately $0.7V_{\mathrm{mp}}$ under the new assumptions and $0.9V_{\mathrm{mp}}$ under the old ones. This is even more unlikely than flying faster than V_{mr}, for the reason shown in Fig. 4, which is a calculated curve of power *versus* air speed for a particular whooper swan (*Cygnus cygnus*), calculated using Program 1A of Pennycuick (1989). The values used in the calculation are given in Table 2. If we suppose that the swan's flight muscles can produce 200 W (for example) of mechanical power, then the swan would have sufficient power to fly below V_{mp}, at the speed marked V_1. Suppose some disturbance, such as a gust, causes it to speed up by a small amount ΔV, while the power output from its muscles remains unchanged. The power required (curve) decreases, so the bird is now exerting more power than is needed to maintain equilibrium at the new speed. It therefore continues to accelerate, until it reaches V_2, where the power required is the same as at the original speed V_1. Equilibrium is possible with the same power output (200 W) at either V_1 or V_2, but the equilibrium at V_2 is stable, whereas that at V_1 is unstable. Steady flight at V_1 is possible, but requires continuous control inputs to overcome the tendency to accelerate. Birds are therefore not expected to fly slower than V_{mp} on foraging or migratory flights. They may do so when searching for, or attempting to catch, slow-moving prey or in other special circumstances such as song flights (Hedenström, 1995; Hedenström and Alerstam, 1995*b*). According to Thomas (1993), spreading the tail lowers V_{mp}, so that the bird is able to fly more slowly without speed instability. Birds seen to be searching for food or making low-speed manoeuvres were excluded from the field data.

Table 2. *Measurements of a female whooper swan (*Cygnus cygnus*), and values assumed for other variables, to calculate the curves of Figs 4 and 5 from Program 1A of Pennycuick (1989)*

Empty body mass (kg)	10.0
Fat load (kg)	1.5
Wing span (m)	2.26
Wing area (m^2)	0.589
Aspect ratio	8.67
Profile power ratio	0.969
Induced drag factor	1.2
Body drag coefficient	0.1
Air density ($kg\,m^{-3}$)	1.23

The profile power ratio is $8.4/R$, where R is the aspect ratio.

Birds that proceed by flap-gliding rather than steady flapping flight are not necessarily subject to speed instability at speeds below V_{mp}. A flap-gliding bird slows down during the gliding phase, possibly to some speed well below V_{mp}. During the flapping phase, it speeds up again, but only until it stops flapping, which it may do before it reaches V_{mp}. In that case, the speed oscillates over a range that never reaches V_{mp}. Birds that normally flap-glide on foraging or positioning flights, such as the American black vulture *Coragyps atratus* (at the bottom of Table 1), may therefore maintain an average speed below V_{mp}. Some other species low down in Table 1, such as the two large herons, *Ardea occidentalis* and *A. herodias*, normally flap steadily in horizontal flight. In their case, it is unlikely that they would be flying slower than V_{mp}, for the reason given above, and more likely that the estimate of V_{mp} is too high. This would be the case if their body drag coefficients were higher than was assumed when calculating V_{mp}. In fact, the old estimates of C_{Db}, 0.25 or higher, might be correct in the case of large herons, which have long legs that trail behind in flight and also prominent heads that retract upwards and are not fully faired in flight. As the open circles show, the observed speeds were near the old estimates of V_{mp} for these aerodynamically 'dirty' species (Fig. 1).

'Optimum' speed selection – marginal costs and benefits

Whilst it is true that the greatest range (relative to the air) is obtained by flying at the maximum range speed (V_{mr}), it does not necessarily follow that flying at that speed represents an optimal or even a practical strategy for a migrating bird. The upper curve in Fig. 5 is the same curve of mechanical power *versus* air speed as in Fig. 4, and below it is the corresponding curve of effective lift:drag ratio (also from Program 1 of Pennycuick, 1989), which is proportional to the distance flown per unit work done. The flat maximum in the curve of lift:drag ratio indicates that the value of V_{mr} is poorly defined, meaning that, in the vicinity of V_{mr}, substantial changes of speed, either way, make very little difference to the effective lift:drag ratio. Moreover, the exact value of V_{mr} depends on the shape of the upper (power) curve, which is poorly known. A small change in the curvature of this region of the power curve would have a large effect on the value of V_{mr}, but very little effect on the maximum effective lift:drag ratio. In contrast, small changes of speed in the vicinity of V_{mr} require large changes of power, because that part of the power curve bends ever more steeply upwards.

The bird's practical options in the selection of speed depend on the *marginal* changes of lift:drag ratio and power resulting from small changes of speed. By definition, the power required to fly is least at the minimum power speed (V_{mp}), but since the power curve is almost horizontal in the region of V_{mp}, the power required is almost the same if the bird elects to fly a little faster (or slower) than V_{mp}. For example, the marginal increase in power required to fly at $1.1V_{mp}$, rather than actually at V_{mp}, is only 0.8 %. As this represents a 10 % increase in speed, the ratio of speed to power increases by 9 %, and so also does the effective lift:drag ratio, which determines the range. Further equal increments of speed result in progressively larger increments of power, because the power curve bends upwards. The curve of effective lift:drag ratio, however, bends downwards and peaks at V_{mr}. The marginal gain in range declines, until eventually the curve becomes horizontal at V_{mr}. To get the last 1 % to the maximum value of the effective lift:drag ratio, the bird has to accelerate from 30.5 to 33.4 m s^{-1}, which requires a 7.7 % increase of power.

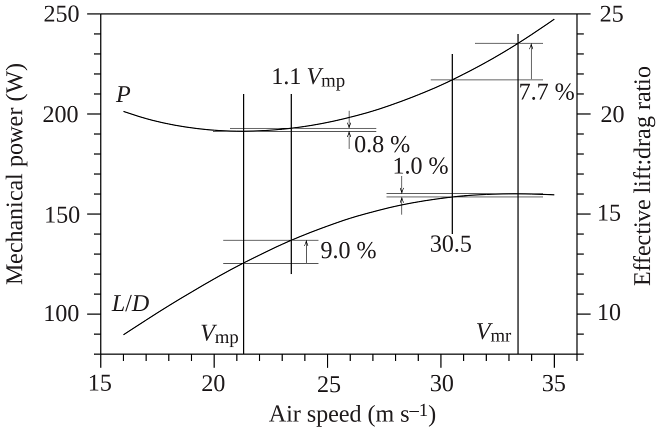

Fig. 5. The same power curve as in Fig. 4 (*P*), with a corresponding curve of effective lift:drag ratio (*L*/*D*), illustrating the marginal effects (given as percentages) of small changes of speed (see text explanation).

One would expect any bird whose immediate need is to make some progress over the ground to fly somewhat faster than V_{mp}, because this results in a substantial saving of energy and shortening of the journey time at the cost of a very small increase in power. However, the marginal benefit from further increases of speed diminishes to a barely perceptible level long before the mathematical 'optimum' speed (V_{mr}) is reached, while the marginal increase in the power required becomes ever larger. Birds that migrate long distances over hostile terrain have the strongest incentive to fly in whatever manner maximises their range, but even these may not fly actually at V_{mr}. In the example of Fig. 5, the swan needs 7.7 % more power to get the last 1 % of range, and that translates (roughly) into 7.7 % more muscle, which in turn requires an enlarged respiratory and circulatory system with 7.7 % greater aerobic capacity. It may be energetically cheaper to fly slower than V_{mr}, at a slightly lower effective lift:drag ratio, because this will permit a substantial reduction in the weight of muscles and supporting systems required, so eliminating the energetic cost of transporting that extra weight and the metabolic cost of maintaining the extra tissues. The reverse might be true if part of the flight muscles can be consumed as supplementary fuel in the course of long flights, as suggested by Pennycuick (1975).

Physiological complications

Sustained cruising flight (other than soaring) requires the bird to have sufficient aerobic capacity to supply the flight muscles with fuel and oxygen, and to remove heat, at a fast enough rate to maintain equilibrium. This is a separate requirement from mechanical muscle power, discussed above. The speeds at which some birds can fly may be limited by aerobic capacity, rather than by muscle power. As an extreme example, there is evidence that some large galliform birds, whose muscle power is sufficient for explosive take-off and rapid acceleration under predator attack, are actually incapable of cruising aerobically at any speed (Pennycuick *et al.* 1994). In terms of added mass or metabolic maintenance costs, little or nothing is known about the implications of providing increased aerobic capacity. This would have to be rectified, before any meaningful theory of 'optimisation', involving these variables, could be formulated.

Response to wind

It has commonly been observed that birds increase their air speeds when flying against a head wind (Alerstam, 1990). This is easily understood if the bird's objective is to make progress relative to the ground rather than the air (Pennycuick, 1975). However, if a bird's objective were simply to remain airborne and making progress over the ground were unimportant, then the selection of air speed would not be affected by wind. For example, a swift flying at night, and unable to feed, might be unconcerned where it is carried by the wind, and in that case it would be free to select an air speed near V_{mp}, where the power is minimal, regardless of the wind strength. The observation that a particular population of migrating or foraging birds increases air speed when flying against a head wind, and slows down with a tail wind, indicates only that position relative to the ground is significant for the birds. It does not indicate anything about the whereabouts of the chosen air speed relative to the power curve.

Conclusion

The power curve for a particular bird, to the extent that it can be reliably calculated, is a convenient summary of the physics of horizontal flight. The two characteristic air speeds V_{mp} and V_{mr} are properties of the power curve that do not in themselves define an 'optimum strategy' for a migrating bird. As noted above, limitations of either muscle power or aerobic capacity may limit the cruising speed to a value below V_{mr}. Such complications would have to be fully understood before it would be possible to build a further layer of theory purporting to predict the speeds at which birds 'should' fly, in order to achieve objectives which may appear more clearly defined from the viewpoint of the theorist than from that of the bird. As to the speeds at which birds actually do fly, the present comparison of air speeds observed in the field with the best currently available estimates of V_{mp} suggests that most of the species in the sample habitually fly at speeds near V_{mp}, at least on short journeys. A longer-range method of speed measurement, such as tracking radar, would be needed to determine whether the same conclusion applies to birds engaged on long, non-stop migratory flights.

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A review to inform the assessment of the risk of collision and displacement in petrels and shearwaters from offshore wind developments in Scotland

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Scottish Government
Riaghaltas na h-Alba

A review to inform the assessment of the risk of collision and displacement in petrels and shearwaters from offshore wind developments in Scotland

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Contents

1 Executive summary

- Scottish Government published the Sectoral Marine Plan for Offshore Wind in 2020, setting out sustainable plan options for the continued development of commercial-scale offshore wind energy in Scotland, as a key contribution to achieving the target of net-zero greenhouse gas emissions by 2045. In January 2022, Crown Estate Scotland announced the lease option agreements for 17 new projects within 14 Plan Option Areas, principally on the eastern and northern coasts.

- Scotland's seas and coastlines are home to a rich diversity of marine life, including internationally important colonies of seabirds, many protected under Scottish, UK and international designations. The need to ensure that future offshore developments do not adversely impact on protected sites and species is embedded within the Scottish Government's National Marine Plan, and potential impacts to marine life and other users of the sea are required to be assessed as part of planning, consenting and licencing processes.

- Several frameworks, methods and tools have been developed in recent years to facilitate the assessment of the likely impacts of offshore wind farm developments on seabirds, and these require data inputs on a variety of parameters relating to species morphology, ecology, behaviour and distribution.

- This key information has not been collated for a group of seabird species for which Scotland holds some of the largest colonies in UK, Europe and globally; namely the Manx Shearwater *Puffinus puffinus*, Leach's Storm-petrel *Hydrobates leucorhous*, and the European Storm-petrel *Hydrobates pelagicus*. These species are active nocturnally, and there is evidence to suggest they are sensitive to light attraction ("phototaxis"), which could render them especially vulnerable to negative impacts from offshore windfarms, for example, if attracted to the rotor-swept area by lights on the turbines that are required for navigation purposes. We also consider, in less detail, two further species from the same taxonomic group, namely Northern Fulmar *Fulmarus glacialis* and Sooty Shearwater *Ardenna grisea*.

- Low fecundity rates and a relatively protracted time to reach maturity (3–6 years) for these species, means seemingly small impacts on survival rates can have large impacts on population viability, making them particularly vulnerable to lethal impacts of wind farm development.

- We reviewed the published peer-reviewed and grey literature for information on the 24 key parameters/data groups required to assess the vulnerability of these species to potential impacts of offshore wind farms and associated structures and activities.

- We compiled a library of more than 1000 scientific papers, reports and other publications, from which we extracted all relevant information to assist in the implementation of methods and tools to quantify the likely population-level impacts of sites leased in the Sectoral Marine Plan Option areas. We highlight critical data gaps that currently prevent a reliable assessment of population-level impacts on protected colonies of these three species.

- Good quality data from within Scotland exist for ten of the key parameters/data groups for all three main species (Manx Shearwater and the two storm-petrel species), and for just three parameters for the other two species. Data collected from elsewhere, or from closely related species, are available for 21 key parameters for all three main species. Less information is available for Leach's Storm-petrel in Scotland than for the Manx Shearwater or European Storm-petrel.

- The evidence needs that were highlighted as being most important for the three focal species were to improve understanding of: (i) biases in detectability of birds at sea; (ii) flight height and speed (and their variation); (iii) avoidance behaviour; (iv) light attraction and (v) foraging ranges from breeding colonies.

- There is a need for experimental validation of potential biases in aerial survey methods, including detectability, identification and diel variation. Detectability could be tested by carrying out targeted digital aerial surveys or vessel-based surveys with an experimental approach, using either tagged model "decoys" or tagged free-roaming birds, though achieving adequate sample sizes of the latter may be challenging.

- Estimates of flight parameters such as speed and height can be gained from tracking data, but acquiring accurate estimates is difficult, even with high resolution data. Where possible, "instantaneous" flight speeds from GPS tags, based on Doppler-shift information derived from the movement of the tag relative to the movement of the satellites, will be more accurate than that derived from distance covered between successive fixes. Constraints on device size/weight suitable for use on storm-petrels limit the range of tracking devices that can be deployed on these species.

- Assessment of macro-avoidance of windfarm development can be achieved by comparing marine distributions of seabird pre- and post-construction. In light of the limited tracking of the three focal species in Scotland to date, we recommend further tracking studies from key colonies to better understand the pre-construction movements and distribution of these species. Such tracking studies should continue as construction occurs and after it is completed, to inform understanding of avoidance behaviour. Such work will also increase understanding of drivers of marine distribution and foraging ranges.

- Crucially, we found that there is currently a lack of evidence on which to judge the existence and strength of light attraction in these species. It is clear from the evidence base that all three focal species may become disorientated by powerful light. This typically occurs in foggy conditions and particularly affects recently fledged young, who may still have under-developed visual capabilities. Under such circumstances, birds may circle a light source for many hours, until succumbing to dehydration or exhaustion. In the context of assessment of the likelihood of collision with turbine blades, the probability of collision is vastly increased, since a bird may pass through the rotor swept area many times. Attraction to or disorientation by light can also be considered a form of displacement, for example if birds are drawn away from foraging areas or behaviours.

- A further compounding factor is the extent to which birds are drawn from a distance to the lights on turbine towers, or whether such attraction is very local (i.e. “micro-scale attraction”). Whilst there are many documented cases in the literature of seabirds dazzled by lighthouses, ships’ lights, gas flares from oil platforms, etc., the distances from which birds may be attracted are unknown. This is a critical distinction. If birds are attracted to bright light sources from considerable distance (i.e. hundreds of metres to kilometres) the potential for adverse impacts from collision is greatly increased, as the number of birds attracted scales as the square of the range from which they are drawn. Taken together, the effect of disorientation, causing birds to circle for many hours and increasing the number of passes through the rotor-swept area, and the potential for birds to be attracted from an area covering tens of square km, would render current methodologies of assessing impacts unreliable.

- We recommend urgent studies to quantify the distance over which flight paths of these species may be influenced by bright light sources, to examine the age class of individuals most likely to be affected, and to assess whether the wavelength and pattern of illumination (flashing vs constant) may affect the level of attraction or disorientation. Such studies will require the novel application of tracking technology (e.g. use of thermal video imaging, radar, VHF and/or GPS tags). The most appropriate approach for each species will depend on device size/weight constraints and logistic constraints of particular breeding locations. We make recommendations as to how such studies may be conducted, suggest suitable locations, and highlight potential challenges.

- We detail several options for mitigation of potential impacts, such as altering the wavelength or pattern of illumination of navigation lights on turbines and associated structures. We discuss the current technical and legislative constraints to such modifications.

2 Introduction

Scottish Government's Energy Strategy (Dec 2017) set a 2030 target for 50% of the energy for Scotland's heat, transport, and electricity consumption to be supplied by renewable sources, and the Climate Change (Emissions Reduction Targets) (Scotland) Act 2019 sets a target of net-zero greenhouse gas emissions by 2045. These targets will require the development of extensive areas for marine renewable energy extraction. In October 2020, the Scottish Government published the Sectoral Marine Plan for Offshore Wind (SMP; Scottish Government, 2020), which identified the most sustainable plan options for the future development of commercial-scale offshore wind energy in Scotland. With six operational offshore wind farms in Scotland, and a further eight having received consent, in January 2022 the Crown Estate Scotland announced the results of a further leasing round ('ScotWind'). Seventeen proposed projects within 14 of the 15 Plan Options now have lease option agreements, covering over 7,000 km^2 of seabed in Scotland, with a total generating capacity of nearly 25 GW. These projects are predominantly off the north and east coasts of mainland Scotland, in waters beyond the 12 nm territorial limit (Figure 1; Crown Estate Scotland, 2022).Unleased areas, including Plan Option NE1, underwent a Clearing process which closed on 10 May 2022, and resulting Option Agreements from the Clearing process will be signed in Autumn 2022. In February 2022, Marine Scotland published an Initial Plan Framework for a Sectoral Marine Plan for Innovation and Targeted Oil and Gas Decarbonisation (INTOG; Scottish Government, 2022). This provides potential areas for future seabed leasing for small scale innovation and offshore wind farms specifically for the purpose of providing low carbon electricity to power oil and gas installations and help to decarbonise the sector. These areas are all in the north and east of Scotland, therefore potential new offshore wind farm projects could come forward in these locations in the future.

The need to ensure that future offshore developments do not adversely impact Scotland's internationally important marine environment is embedded in the aims of both the National Marine Plan (NMP; Scottish Government, 2015) and SMP. In addition, Habitat Regulation Appraisals and Strategic Environmental Assessments require evaluation of potential impacts of proposed marine developments on marine biodiversity, and on other marine users. The SMP, which assessed for up to 10 GW maximum installed capacity at a national level, identified the key risk factors of development in all Plan Option regions include "risks to bird species, including collision risk and displacement, as well as potential impacts to birds on migratory pathways".

A number of frameworks and tools have been developed to assess the vulnerability and sensitivity of seabird species to marine pressures (e.g. Williams et al., 1995, Furness and Wade, 2012, Furness et al., 2013, Bradbury et al., 2014, Certain et al., 2015, Rogerson et al., 2021), to quantify risk of collision with offshore turbines (Band et al., 2007, McGregor et al., 2018), to apportion impacts of marine development to particular breeding populations (NatureScot, 2018) and to estimate survival consequences for seabirds of displacement from former foraging areas (Searle et al., 2018, Searle et al., 2019). Implementing these frameworks and tools requires key information about the distribution, ecology, morphology, behaviour, and population status of the species of interest. For example, knowledge of species' population sizes and marine distributions is required to assess overlap with licensing areas for offshore wind farms, and the density of birds within these areas of overlap; factors such as flight height and levels of nocturnal activity will influence a species' vulnerability to collision, and an understanding of a species' behaviour is important for determining the probability and scale of avoidance of an offshore development, and therefore the levels of displacement or barrier effects. In addition, seabird species that are nocturnally active, such as Manx Shearwaters, European and Leach's Storm-petrels are potentially vulnerable to attraction to artificial lighting (Rodríguez et al., 2019) such as that associated with offshore wind structures and related shipping. Structures associated with offshore wind farms in UK waters are required to display illumination to meet the lighting requirements of the Air Navigation Order 2009 (CAA, 2016), the Northern Lighthouse Board, and the Maritime and Coastguard Agency. Attraction to illumination associated with offshore wind farms may affect the risk of collision for these nocturnally active procellariiform seabird species. This key information is not currently synthesised for Procellariiformes in Scotland and, given the rapidly increasing activity relating to offshore wind farms in the country, there is a clear and urgent need to do so. The current ScotWind leasing round, and potential future rounds, are likely to include areas to the north and west of Scotland not previously developed, and in closer proximity to the breeding colonies and foraging areas of procellariform seabirds.

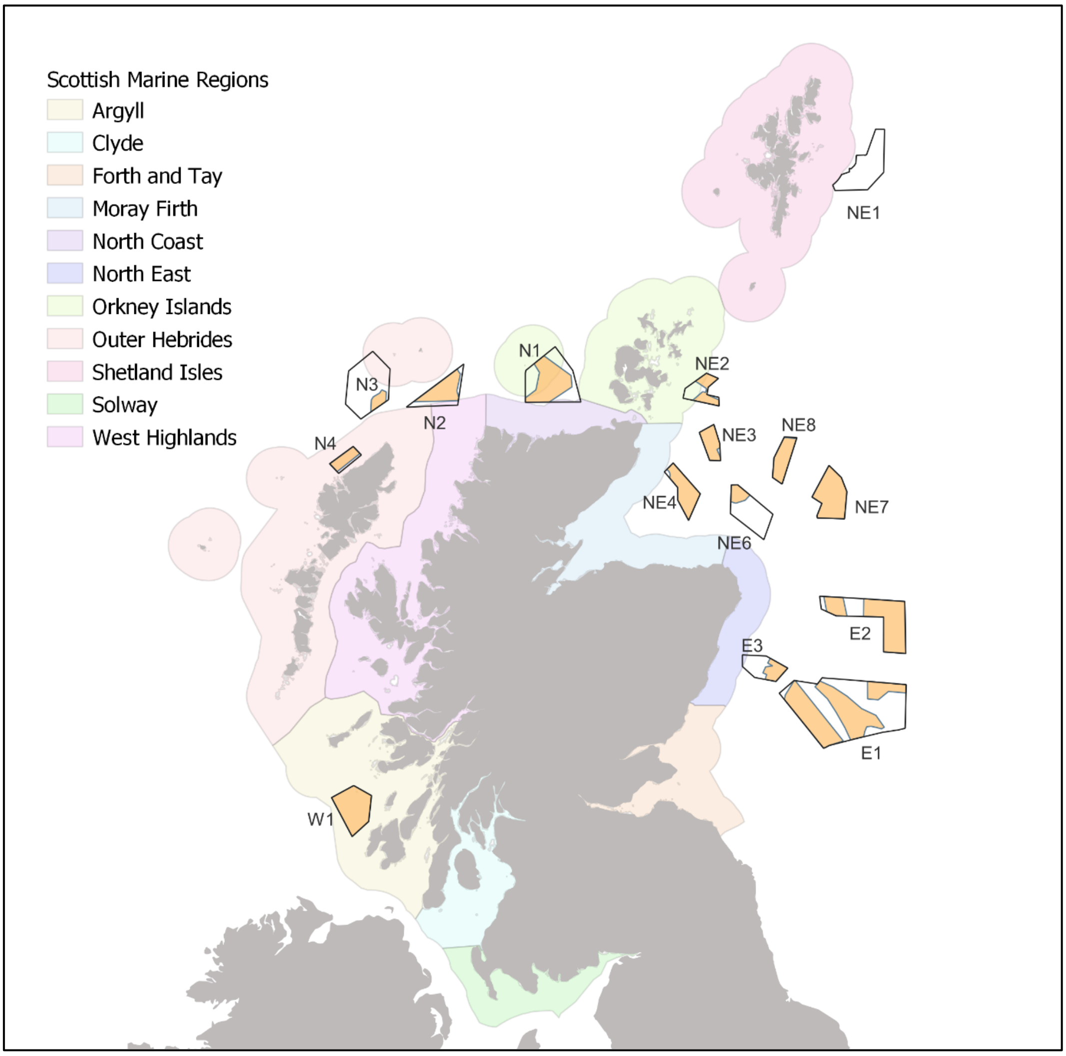

Figure 1. Scottish Marine Regions and Sectoral Marine Plan Options (black polygons). Orange areas within Plan Option polygons indicate sites awarded lease option agreements in the ScotWind leasing round. (Contains information from the Scottish Government (Marine Scotland) licensed under the Open Government Licence v3.0 and from Crown Estate Scotland under Crown Copyright.)

Here we present a review of the published literature to collate and synthesise the existing evidence base for the assessment of the impacts of offshore wind farms and associated activities on three focal species: Manx Shearwater *Puffinus puffinus,* European Storm-petrel *Hydrobates pelagicus* and Leach's Storm-petrel *Hydrobates leucorhous*. We have included less detailed accounts for two additional procellariiform species: Northern Fulmar *Fulmaris glacialis* and Sooty Shearwater *Ardenna grisea*. We identify critical gaps in existing knowledge, outline the challenges to filling data gaps, and make recommendations for possible approaches for improving the existing evidence base. Preliminary findings were shared with subject experts, and through two online workshop we clarified the state of existing knowledge and critical knowledge gaps, especially in relation to the attraction of focal species to artificial lighting of offshore renewable structures and support vessels. We include particular reference to Scotland's Sectoral Marine Plan Options, the specific risks posed to nocturnally active petrels and shearwaters by artificial lighting, and how light attraction may influence assessment of other risks (e.g. collision). We outline potential mitigation methods.

3 Methods

We conducted a systematic literature search to compile, for each focal species, a summary of the current knowledge of: (i) distribution; (ii) population status and abundance; (iii) demography; (iv) foraging ecology and (v) pressures on populations. Our review makes particular reference to attributes that are of critical importance in assessing the vulnerability of these species to potential impacts of offshore wind turbines and associated infrastructure and activities, including: (i) collision risk; (ii) displacement and barrier effects; and (iii) attraction to artificial light. Given the potential differences in the distributions and behaviours of a species during the breeding and non-breeding periods, at different stages of the breeding season and for different age classes, we considered attributes and risks separately for different groups and times of year. While we primarily focus on studies conducted within Scotland, relevant information collected elsewhere, and on closely related species, is also included. We used a set of search terms (listed in Appendix X) to search Google Scholar and Web of Science to identify relevant peer-reviewed and grey literature. We examined the first 100 publications produced by each search and also examined the reference list of publications identified by the literature search that were deemed relevant to the topic. We also made use of the extensive personal libraries and subject knowledge of the authors, noting any publications not identified by the systematic literature review.

We compiled a set of all input parameters required to populate the various methods, models and tools that are currently used for assessment of impacts of offshore wind farms on seabirds, specifically: (i) Collison Risk Models (Band et al., 2007, McGregor et al., 2018); (ii) the NatureScot Apportioning method (NatureScot, 2018); (iii) the Marine Scotland Apportioning Tool (Butler et al., 2020); (iv) the Fate of Displaced Birds Tool (SeaBORD; Searle et al., 2018); (v) displacement matrices (SNCBs, 2017) and (vi) the Natural England PVA Modelling Tool (Searle et al., 2019). We cross-referenced the publications identified by the literature search with the set of input parameters and catalogued the data sources accordingly, noting whether studies had been conducted in Scotland or elsewhere. Input parameters for which no information was identified by the literature review were considered to represent data gaps.

4 Species accounts

4.1 General ecology

The five seabird species that are the subject of this review all belong to the Order Procellariiformes, which includes the most pelagic of all seabirds. The three species which are the primary focus (Manx Shearwater, European Storm-petrel and Leach’s Storm-petrel) are small-bodied and relative immobile on land, which renders them vulnerable to predation, so breeding colonies are restricted to islands that are often very remote, where the birds nest out of sight in burrows and cavities, and only attend the colony at night. The breeding season is divided into four distinct stages. During the pre-laying stage birds occupy and defend nest crevices, attract a mate and copulate. Females typically spend a protracted period (c. one week) at sea prior to laying (the so-called “pre-laying exodus”) to acquire the nutrients needed to produce the single large egg. Once laid, the egg is incubated for around six weeks (the “incubation period”), until hatching. The newly hatched chick is incapable of full thermoregulation and parents take turns brooding the chick at the nest whilst the other feeds at sea (“brooding period”). Once chicks acquire the ability to thermoregulate independently, both adults spend the day feeding at sea, returning during darkness at intervals of 1 to 7 days to feed the chick (the “post-brooding” period). Although the storm-petrels are little bigger than a sparrow, and shearwaters the size of a pigeon, these species may cover thousands of kilometres at sea on a single foraging trip and the maximum range of a trip may lie hundreds of kilometres from the colony.

The eggs and chicks have very protracted development, requiring nearly four months from egg laying to fledging. For European Storm-petrels, within a particular colony the timing of breeding may be poorly synchronised among individuals, such that there will be birds engaged in breeding activities for more than six months of the year. The combination of large foraging ranges and very protracted breeding seasons means that birds will be exposed to risks from marine activities over a wider geographic area, and for a longer period of the year, than many other seabird species. All three focal species are trans-equatorial migrants, and immature birds do not return to the breeding colonies for several years, presumably remaining within, or close to, the wintering grounds in the South Atlantic. Storm-petrels and shearwaters do not reach breeding age for 3–6 years which, coupled with low fecundity, means that adults need to maintain a high annual survival rate for population stability, and any adverse impact on adult survival will have large, and long-term, consequences for population size.

While most of the above also applies to the two additional species included in this review, Northern Fulmars, and sometimes Sooty Shearwaters, will nest on mainland coasts as well as islands. Unlike the other species reviewed here, the Northern Fulmar is a surface nester, will attend its nest both during the day and at night, and is present in Scottish waters during both its breeding and non-breeding periods.

4.1.1 Note on spatial and temporal distribution

Information on the distribution of seabirds at sea is available from boat and aerial surveys, and more recently from tracking birds during the breeding season, and during their migration and non-breeding seasons. Both methods suffer from various drawbacks: aerial and vessel-based surveys tend to under-represent storm-petrel occurrence/density due to the birds’ small size and dark colouration, and at-sea surveys are unable to determine the provenance, age or breeding stage of birds encountered at sea. The European Seabirds At Sea (ESAS) database used to produce maps of marine distribution of seabird (e.g. Stone et al., 1994, Waggitt et al., 2020) includes data collected over many decades and may not reliably reflect current marine distribution of seabirds, due to changing seabird population sizes, breeding colony distributions and prey distribution in recent decades. Tracking studies usually provide data on a relatively small number of birds from a given colony in a particular year and GPS tracking is almost exclusively restricted to the breeding season as birds must be recaptured to retrieve the tags and download the data. Therefore, tracking studies tend to provide a snapshot of the movements of a small number of birds within a restricted period. This snapshot may not always be representative of the foraging movements of the wider colony, particularly as the tagging itself may influence the behaviour of the birds (e.g.Gillies et al., 2020). Geolocators (GLS) can be used to track year-round movements but at a much lower resolution (two locations per day) and, again, the tags must be retrieved to download the data. The necessity of tag retrieval means that tracking studies for these species focus almost exclusively on breeding adults that return reliably to the colony, but tracking of juveniles and immature birds, which may not return to their natal colony for several years, presents methodological challenges.

4.1.2 Note on population status and abundance

The most complete population estimates available for seabirds breeding in Scotland are currently from Mitchell et al. (2004), which summarises the results of Seabird 2000, a census of Britain and Ireland's breeding seabirds, conducted between 1998 and 2002. The results of the latest census, Seabirds Count conducted between…, are due to be published in 2023 and will provide an important update on Scotland's seabird population status and trends.

4.2 Manx Shearwater

4.2.1 Spatial and temporal distribution in Scotland

4.2.1.1 Breeding distribution

The UK and Ireland's breeding colonies of Manx Shearwater are all located on islands, almost entirely on the western fringes, with additional colonies on the Irish Sea coasts (Figure 1). Very small numbers (< 10 pairs) formerly bred in Orkney (Cramp et al., 1974) and Shetland (Mitchell et al., 2004) but recent confirmation of breeding in these archipelagos is lacking. The species is present in Scottish waters between April and October, with nests occupied between April and August (Harris, 1966b, Guilford et al., 2009, Waggitt et al., 2020).

4.2.1.2 Marine distribution

4.2.1.2.1 Derived from at-sea surveys

Observations from offshore surveys in north-west European waters between 1979 and 1993 are collated in Stone et al. (1994) and show that, during the breeding season, the highest concentrations of birds are around the largest colonies of Rum (west Scotland), and Skomer and Skokholm (south-west Wales). During April, the highest densities were in the inshore waters of west Scotland. Feeding birds were observed in the offshore waters of west Scotland in May and June but occurred in inshore waters later in the season, during chick-rearing, when high densities of birds were present. This change in distribution is likely explained by the longer foraging trips during incubation when shifts average six days (Harris, 1966b), compared with chick-rearing, when each parent visits the nest at least every two days (Thompson, 1987). The species was observed around North Rona and Sula Sgeir between March and August and in low densities around Shetland, Orkney and the North Sea between May and August, remaining in the North Sea into September. Sightings in the North Sea were restricted to areas off the north-east coasts of Scotland and England, and especially the Moray Firth, Firth of Forth and Flamborough Head. The Manx Shearwater is a designated feature of the Outer Firth of Forth and St Andrews Bay Complex SPAfrom the ESAS database, that were collected between 1979 and 2006, showed distributions of Manx Shearwaters during the period May–October very similar to those described above, and during the period October–November low numbers were recorded, with small concentrations in the Celtic and Irish Seas and to the south-east of Orkney (Kober et al., 2009).

Waggitt et al. (2020) produced monthly distribution maps for seabirds in the north-east Atlantic based on distribution models using data collated from aerial and vessel surveys between 1980 and 2018. These maps show similar Manx Shearwater distributions in Scottish waters to Stone et al. (1994) and Kober et al (2009), with the highest densities to the west of Scotland, and densities increasing through the breeding season, peaking in August.

Projected distributions based on foraging range and colony sizes also indicate a Manx Shearwater hotspot to the west of Scotland (Critchley et al., 2018).

4.2.1.2.2 Derived from tracking data

Manx Shearwaters have been tracked using GPS tags from breeding colonies on Rum in Scotland, Copeland in Northern Ireland, High Island and Great Blasket in south-west Ireland, Lundy in the Bristol Channel, and Skomer and Skokholm in south-west Wales (Guilford et al., 2008, Dean et al., 2010, Freeman et al., 2012, Dean et al., 2013, Dean et al., 2015, Padget et al., 2019, Critchley et al., 2020, Kane et al., 2020). Prior to laying, females undertake a “pre-laying exodus”, during which they may travel a considerable distance to productive feeding grounds (e.g. at the edge of the continental shelf) which are not regularly visited when birds are attending eggs or chicks (Dean, 2012). During the breeding season birds from all colonies (except Lundy and Skokholm, where fewer birds have been tracked to date) visited Scottish waters. Birds tracked from Rum moved extensively through the North and West SMP regions (Dean et al., 2015, Padget et al., 2019), those from High Island and Great Blasket moved extensively through the West SMP region and into the North SMP region (Wischnewski et al., 2019, Kane et al., 2020), and birds from Copeland used the West SMP region extensively (Dean et al., 2015, Padget et al., 2019). The Solway and Clyde Scottish Marine Regions (SMRs) were used by birds from multiple colonies at all stages of the breeding season, although the extent of use varied between years (Dean et al., 2013, Dean et al., 2015).

In September, Manx Shearwaters depart in a south-westerly direction from the colonies to their wintering areas off South America, and arrive back from a westerly direction between March and May (Guilford et al., 2009). Fledglings depart on migration almost immediately after fledging and take a similar but more direct migration route than breeding adults (Wynn et al., 2021).

Few data exist on the distribution of immature Manx Shearwaters in Scottish waters. Immatures were tracked from Skomer in June–July 2013 and 2014 and, while adult movements extended into Scottish waters, immatures remained significantly further south, and there was little overlap between the core use areas of adults and immatures (Fayet et al., 2015).

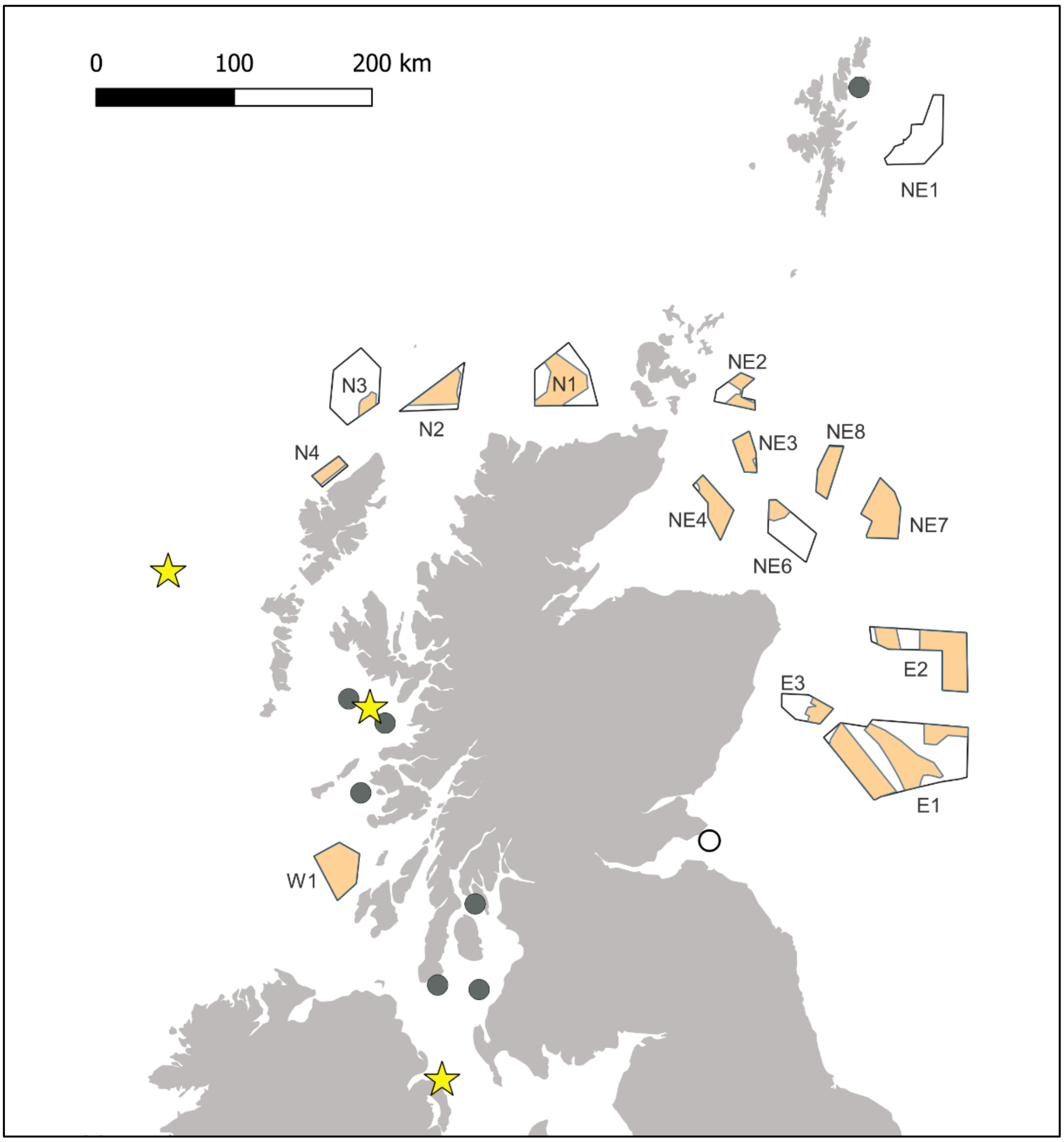

Figure 2. Manx Shearwater colonies (Mitchell et al., 2004) in relation to Plan Options. Yellow stars represent SPAs with Manx Shearwater as a designated feature. Grey circles are non-SPA colonies. White circles are colonies identified since Seabird 2000. Orange polygons within Plan Options indicate sites awarded lease option agreements in the ScotWind leasing round. Note that breeding colonies in Ireland and Wales (not shown) may also be at risk of impacts from developments in Scottish waters. (Contains information from the Scottish Government (Marine Scotland) licensed under the Open Government Licence v3.0 and from Crown Estate Scotland under Crown Copyright.)

4.2.2 Population status and abundance

4.2.2.1 Breeding

The most recent global population estimate for Manx Shearwaters is 338,000–411,000 pairs, with 305,000–374,000 of these breeding in Britain and Ireland. Scotland holds internationally important numbers of the species, with an estimated 126,545 (95% CLs 112,285–141,701) breeding pairs (Table 1), representing 38% of the population in Britain and Ireland (Mitchell et al., 2004). The breeding colony on Rum was estimated as 120,000 (107,000–134,000) breeding pairs in 2001 (Mitchell et al., 2004), making it the largest single-island colony in the world at the time. A more recent survey conducted in 2021, using slightly different survey methods, estimated the colony size as 288,894 (226,010 – 403,915) pairs (Inger et al., 2022). It is not clear to what extent the difference in survey estimate represents a genuine increase in population size since estimates of nesting density were similar in both surveys. The apparent increase in population size in 2021 resulted almost entirely from an increase in the size of the area considered to be suitable for nesting. The colonies at both Rum and St Kilda qualify as internationally important (Mitchell et al., 2004).

Table 1 Numbers of breeding Manx Shearwater Apparently Occupied Sites (AOS) in Scotland 1999–2002 (Mitchell et al., 2004). Only colonies designated as SPAs are listed individually

Administrative area	SPA colonies	AOS	95% LCL	95% UCL	Colonies counted	Colonies not counted
Shetland	*Non-SPA*	7	7	7	1	
Western Isles	St Kilda	4,803	3,593	5,909	1[1]	
Lochaber	Rum	120,000	107,000	134,000	1	
	Non-SPA	252	202	302	3	
Argyll & Bute	*Non-SPA*	1,483	1,483	1,483	2	1 possible
Cunninghame	*Non-SPA*	Present?			0	1
Kyle & Carrick	*Non-SPA*	Present?			0	1
Total		**126,545**	**112,285**	**141,701**	**8**	**3**

[1] Four islands of St Kilda treated as one colony.

4.2.2.2 Non-breeding

Skov et al. (1994) estimate that immatures could comprise around 25% of the entire population of Manx Shearwaters and Guilford et al. (2008) estimate that up to half the population could be non-breeders, in any given year (i.e. including birds of breeding age that were unpaired). An estimated 200,000 individuals are present in Scottish waters on passage (Furness and Wade, 2012).

4.2.3 Productivity and survival

4.2.3.1 Age at first breeding

Age at first breeding is not known for Manx Shearwaters breeding in Scotland but studies at the Pembrokeshire colonies suggest that some will breed from three years old, but that most do not breed before the age of five (Harris, 1966a).

4.2.3.2 Productivity

In Scotland, productivity data are collected on Rum, Canna and Sanda, and the average productivity between 1986 and 2018 was 0.62 chicks fledged per AOS (JNCC, 2021b), with no indication of a temporal trend over this period. Productivity as high as 0.97 has been recorded on Sanda (Mavor et al., 2006).

More experienced breeders may have higher productivity than first-time breeders. Hatching success on Skokholm between 1973–76 was lower in newly formed pairs (66.2%) than established pairs (79.2%; Brooke, 1978).

Manx Shearwater burrows can be susceptible to flooding and high rainfall during incubation has been shown to reduce hatching success on Rum and Canna (Thompson and Furness, 1991).

4.2.3.3 Survival

The average annual adult survival rate on Skokholm was estimated to be 0.902 between 1963 and 1968, ranging from 0.794 to 0.965 (Harris, 1966a, Perrins et al., 1973). Mean adult annual survival on Skomer was 0.87 between 1978 and 2018 (Zbijewska et al., 2020). Data from Rum (Hallival, Askival and Trollaval) from 1994–2014 and Sanda Island from 2000–2005 indicate a mean (± SD) annual adult survival rate of 0.93 ± 0.03 (Horswill et al., 2016).

Estimating survival for non-breeders is challenging due to the low re-encounter rates, but Harris (1966a) estimated that pre-breeders frequenting the Skokholm colony, from the age of four years old, have an annual survival rate of around 0.8. On Skokholm, average survival rates of young Manx Shearwater from fledging to returning to the colony as pre-breeders were estimated as 26.7% for 1964–69 (Perrins et al., 1973) and 21.8% for 1967–71 (Brooke, 1977), but these are likely to be underestimates as not all surviving birds will be recaptured, and Perrins (2014) suggests the actual survival rate is around 28–37%. The percentage of birds surviving to breed will be lower since pre-breeders spend more time on the surface at the colony and are at higher risk of predation than breeding birds (Perrins et al., 1973). There is some indication that chicks fledging later in the season have lower survival rates (Perrins, 1966).

4.2.4 Foraging ecology

The Manx Shearwater is a pursuit-plunger (Brown et al., 1978), and birds studied at Skomer had an average dive depth of 9.6 m, and a maximum depth of 55 m, with dives lasting 13.5 seconds on average and a maximum of 46 seconds, with no differences between the sexes (Shoji et al., 2016). For birds tracked from Skomer, diving occurred during the day and peaked in the evening (Shoji et al., 2016), but nocturnal foraging was observed from tracking of birds from High Island, Ireland (Kane et al., 2020). The species displays a dual foraging strategy during the breeding season, undertaking a combination of short trips for chick provisioning and long trips for self-provisioning (Shoji et al., 2015, Wischnewski et al., 2019). Since dual foraging can lead to bimodality in foraging ranges, it should be considered when using foraging range data to assess the risks posed by wind farm developments.

Currently, the standard foraging ranges for Manx Shearwater used in offshore wind casework come from Woodward et al. (2019), who collated the available data to give a maximum foraging range of 2890 km, a mean maximum of 1346.8 ± 1018.7 km, and a mean of 36.1 ± 88.7 km. Foraging trip durations and distances vary between years, colonies, breeding stage and the sexes (Thompson, 1987, Gray and Hamer, 2001, Guilford et al., 2008, Dean et al., 2010, Dean, 2012, Dean et al., 2013, Dean et al., 2015, Wischnewski et al., 2019). Published tracking data from Scotland is limited to 75 trips by 20 chick-rearing birds on Rum, which each lasted one day and had a median total trip distance of 184 km (interquartile range 128–274 km) and median maximum distance from the colony of 35 (29–73) km (Dean et al., 2015). The distances recorded by birds from Rum were shorter than those of chick-rearing birds from other colonies tracked simultaneously, the longest of which were undertaken by birds from Skomer, which covered a total of 297 (203–581) km during trips of one to two days, and reached a median maximum distance of 86 (61–134) km from the colony (Dean et al., 2015). Manx Shearwaters tend to travel further during incubation than chick-rearing, with incubating birds from Skomer undertaking trips of 8 (7–11) days duration, covering total distances of 1,517 (925–2,117) km and reaching a maximum distance from the colony of 254 (176–295) km (Dean et al., 2015). In some years, pre-laying females from Skomer have been found to make long trips to or beyond the continental slope, up to 727 km from the colony (Dean, 2012). Birds tracked from colonies in Ireland had foraging ranges of up to 1,109 km (Wischnewski et al., 2019).

Of the 528 trips from four colonies (Rum, Copeland, Skomer and Lundy) studied by Dean et al. (2015), almost all were largely restricted to the waters over the continental shelf, with only six trips extending beyond the shelf edge. Birds foraged in areas near their respective colonies, with little overlap between colonies, but individuals from all colonies also travelled to a more distant shared foraging area at the highly productive Irish Sea Front and nearby stratified waters of the Western Irish Sea (Dean et al., 2015). This productive area is approximately 375 km from Rum and was visited on 60% of the long-distance trips made from the colony (Dean et al., 2015). There is some evidence that the foraging areas of birds from the Pembrokeshire colonies may have shifted northwards since the 1950s (Guilford et al., 2008).

Manx Shearwaters gather in dense flocks on the sea in the vicinity of breeding colonies from late afternoon, before coming ashore after nightfall. This so-called “rafting” behaviour was studied using radio telemetry of chick-reading adults at Rum, Bardsey and Skomer to inform designation of colony extension marine protected areas (McSorley et al., 2008, Wilson et al., 2009). The maximum extent of rafts (95% kernel contours of raft locations) varied according to colony: 4, 6 and 9 km respectively for Skomer, Rum and Bardsey. Locations of radio-tagged individuals in rafts were estimated by triangulation from adjacent coasts, though some potential raft locations were unobservable due to the lack of direct line-of-sight from tracking locations. It would be instructive to analyse the large body of tracking data collected using GPS tags in recent years to improve our understanding of rafting behaviour. Dean et al. (2013) found that GPS tagged birds tended to roost on the sea within 20 km of the colony prior to landfall and that they resumed roosting on the sea adjacent to the colony after their visit.

4.2.4.1 Diet

Limited dietary analysis has been conducted for Manx Shearwaters. Thompson (1987) collected 104 samples on Rum in the 1980s and found that 65.8% contained fish (clupeids, sandeels and one sheppy Argentine), 47.4% squid (Ommastrephidae, Cranchiidae, Gonatidae, Onychoteuthidae and Mastigoteuthidae) and 7.9% crustaceans, although these were always associated with fish remains and were believed to be eaten by the fish rather than the shearwater directly. Most fish were likely around 5 cm or longer, and the largest intact fish was an 8 cm long sprat (Thompson, 1987). The data suggest that squid were more important before egg-laying and that the diet during chick-rearing was primarily fish, despite squid likely being over-represented due to their digestion-resistant beaks (Thompson, 1987). Ommastrephidae squid migrate to the surface at night and the other squid families identified, plus the sheppy Argentine, are all bioluminescent, suggesting that shearwaters were feeding at night prior to chick-rearing (Thompson, 1987). All diet samples observed by Brooke (1990) on Skomer contained fish, many of them clupeids around 15 cm long.

4.3 European Storm-petrel

4.3.1 Spatial and temporal distribution in Scotland

4.3.1.1 Breeding distribution

The UK and Ireland's breeding populations of European Storm-petrel are located on rat-free islands around the northern and western coasts (Figure 3; Mitchell et al., 2004, De León et al., 2006). The species is present in Scottish waters between May and October, with nests occupied between May and September, although small numbers may be present in nests in April and October (Davis, 1957a, Waggitt et al., 2020). Breeding was confirmed for the first time on the Isle of May (Firth of Forth) in 2021 ([redacted]).

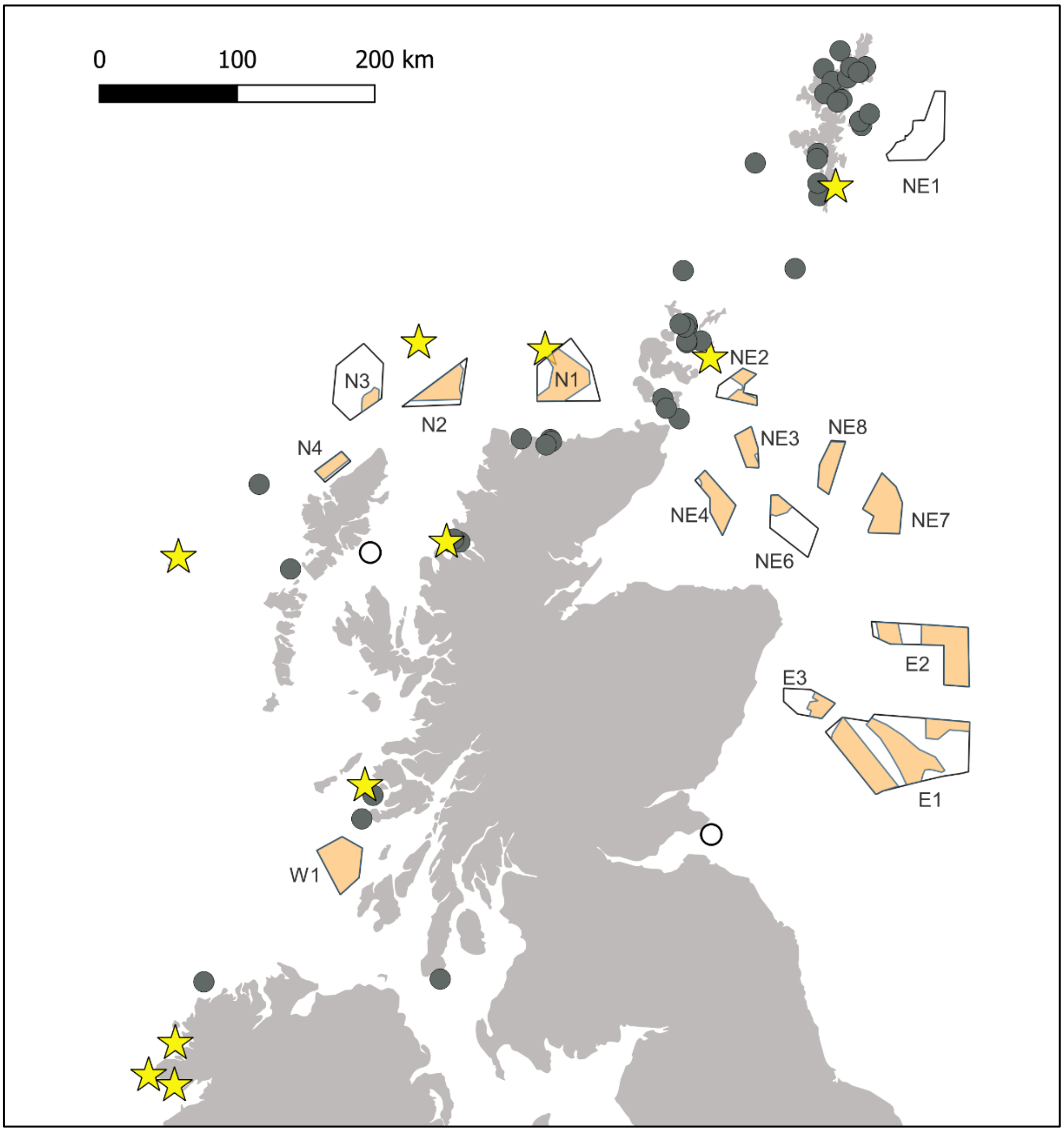

Figure 3. European Storm-petrel colonies in relation to Plan Option areas. Yellow stars represent SPAs with European Storm-petrel as a designated feature. Grey circles are non-SPA colonies surveyed in the Seabird 2000 census (Mitchell et al., 2004). White circles are colonies identified since Seabird 2000. Orange polygons within Plan Options indicate sites awarded lease option agreements in the ScotWind leasing round. Note that breeding colonies in Ireland and Wales (not shown) may also be at risk of impacts from developments in Scottish waters. (Contains information from the Scottish Government (Marine Scotland) licensed under the Open Government Licence v3.0 and from Crown Estate Scotland under Crown Copyright.)

4.3.1.2 Marine distribution

4.3.1.2.1 Derived from at-sea surveys

The ESAS data for 1979 to 1994 show that during May and June European Storm-petrels were mainly found over the outer shelf and shelf break to the north-west of Scotland, with low densities near Fair Isle, Shetland and the west coast of Scotland (Stone et al., 1995). In July and August the species was widespread to the north of Scotland and over the continental shelf to the west of Scotland, with densities highest at the shelf edge (Stone et al., 1995).

There were also low densities around Orkney, Shetland and the north-western North Sea (Stone et al., 1995). Between September and November densities were low, but birds were observed on the continental shelf west of Scotland, along the north-east Scotland coasts, and north-east of Orkney and Shetland (Stone et al., 1995). There were no sightings in Scottish waters between December and April (Stone et al., 1995).

Poisson kriging of ESAS data collected between 1979 and 2006 showed distributions of European Storm-petrels during the breading season (June–October) broadly similar to those described above, with high densities widely distributed over the outer shelf and a large area of high density situated around 100 km north of Lewis (Kober et al. 2009).

Monthly distribution maps based on data collated from aerial and vessel surveys between 1980 and 2018 suggest a similar distribution, with European Storm-petrels present in all SMP regions and peak densities along the continental shelf, from north-east of Shetland to south-west of Ireland (Waggitt et al., 2020). Peak densities are in August, and birds are still present in October (Waggitt et al., 2020).

4.3.1.2.2 Derived from tracking data

Few tracking data exist for European Storm-petrels in the Atlantic. Bolton (2021) tracked breeding adults from the largest UK colony on Mousa, Shetland, during incubation and chick-rearing between 2014 and 2017 using GPS tags. Rather than travelling to the shelf edge, all birds used an area to the south of Shetland, moving extensively over the north-east SMP region (Bolton, 2021). A further nine chick-rearing birds GPS-tracked from Mousa in 2018 travelled in a similar direction but remained closer to the colony than birds tracked in previous years (Z. Deakin, unpublished data). The broad area of use across all years is also identified in the distribution maps produced from at-sea survey data by Waggitt et al. (2020), and broadly indicated as an area of moderate usage in Kober et al. (2009). None of the birds tracked from Mousa travelled to the west or north of Shetland, suggesting that the high concentrations of birds at the continental shelf edge are from the large colonies in the Faroes, although may also include large numbers of non-breeding birds.

Contracted by Marine Scotland, the RSPB tracked 19 breeding European Storm-petrels from Lunga, Treshnish Isles, in 2021. All birds remained on the continental shelf, moving extensively through the West SMP region, with one bird travelling 198 km from the colony to the shelf edge (RSPB unpublished data).

To date there are no published tracking studies of the migration pathways and wintering areas of European Storm-petrels breeding at Scottish colonies, although eight individuals breeding in Shetland were successfully tracked using GLS tags for 9–10 months, from the chick-rearing period in 2016 to the onset of the following breeding season in 2017 (RSPB unpublished data). A further 20 GLS tags were deployed on European Storm-petrels breeding on Lunga, Treshnish Isles in 2021 for retrieval in 2022. Limited information from ringing recoveries of birds found dead indicates the wintering areas are located off southern Africa (Marchant et al., 2002).

4.3.2 Population status and abundance

4.3.2.1 Breeding

The global population of European Storm-petrels is thought to be in the region of 1.5 million individuals (Brooke, 2004), although estimates are far from accurate and the global population trend is unknown. Data from the Seabird 2000 census suggest that Scotland holds 83% of Britain’s 25,710 (95% CLs 21,043-33,517) breeding pairs of European Storm-petrels (Table 2; Mitchell et al., 2004), including the largest colony on the island of Mousa, Shetland (Mitchell et al., 2004). There were more than 50 known colonies in Scotland during Seabird 2000, but only three (Mousa, Treshnish Isles and Priest Island) were estimated to hold more than 1,000 Apparently Occupied Sites (AOS), with both Mousa and Treshnish holding internationally important numbers (Mitchell et al., 2004). Resurvey of the Mousa colony in 2008 identified a 118% increase in the population since 1996 (Bolton et al., 2010), but this growth had slowed by the most recent survey in 2015 when the population was estimated at 10,778 (95% CLs 8,857-13,207) AOS (Bolton et al., 2017). A 2018 resurvey of the Treshnish Isles of Fladda, Lunga and Sgeir a’ Chaisteil estimated the population to be 8,664 AOS, representing a 109% increase since the Seabird 2000 survey in 1996 (Ward, 2018). The population at Priest Island in the Summer Isles was estimated at 3,584 ± 437 individuals in 2012, having declined by 50% since the Seabird 2000 estimate in 1999 (Insley et al., 2014), but a 2019 survey estimated 4,640 AOS, a 5% increase since Seabird 2000 (JNCC, 2021a). Resurvey of North Rona, Western Isles, in 2009 resulted in an estimate of 371 AOS, suggesting the population had remained stable since the Seabird 2000 survey in 2001 (Murray et al., 2010).

Table 2 Numbers of breeding European Storm-petrel Apparently Occupied Sites (AOS) in Scotland 1999–2002 (Mitchell et al, 2004). Only colonies designated as SPAs are listed individually. LCL and UCL are lower confidence limit and upper confidence limit, respectively.

Administrative area	Colony	AOS	95% LCL	95% UCL	Colonies counted
Shetland Islands	Mousa	6,800	4,800	8,800	1
	Non-SPA	703	678	729	25
Orkney Islands	Auskerry	994	372	3,196	1
	Sule Skerry & Sule Stack	309	309	309	1
	Non-SPA	567	429	750	12
Sutherland	*Non-SPA*	449	339	594	4
Ross & Cromarty	Priest Island	4,400	3,300	6,100	1
	Non-SPA	66	50	88	3
Argyll & Bute	Treshnish Isles	5,040	5,040	5,040	1
	Non-SPA	208	208	208	3
Western Isles	St Kilda	1,121	825	2,242	1
	North Rona	368	335	413	1
	Sula Sgeir	9	9	9	1
	Non-SPA	335	303	377	4
Total		**21,370**	**16,997**	**28,855**	**59**

4.3.2.2 Non-breeding

An estimated 100,000 individuals are present in Scottish waters on passage (Furness and Wade, 2012).

4.3.3 Productivity and survival

4.3.3.1 Age at first breeding

Most immatures are thought to begin prospecting at colonies in their second year and begin breeding in their third year, with almost all birds recruited into the breeding population by year five (Okill and Bolton, 2005).

4.3.3.2 Productivity

On Mousa, Shetland, breeding success from laying to fledging was 0.76 in nest boxes and 0.50 in natural nest sites in 1993; a difference that was not statistically significant (Bolton, 1996). Productivity on Skokholm in the 1950s and 60s was estimated at 0.59 and 0.49 young fledged per breeding pair by Davis (1957b) and Scott (1970), respectively. This estimate from Scott (1970) does not include data from 1967, when breeding success was only 0.27, apparently due to loss of chicks by starvation during a period of poor food availability. Mean (± SE) productivity on Skokholm for 2014–20 was 0.58 ± 0.04 (range 0.45–0.74), and was 0.80 in 2021, although the number of monitored nests is small (Brown and Eagle, 2022). Breeding success on Enez Kreiz, Brittany in 1999 was 0.53 young fledged per egg laid (Cadiou, 2001), and 0.62 at Biarritz, south-west France in 1974-79 (Hémery, 1980).

Breeding success in a population of the Mediterranean subspecies, *Hydrobates pelagicus melitensis*, in 1993–2006 had a mean (± SE) of 0.53 ± 0.05 but was lower in birds less than four years old (0.22 ± 0.07) (Sanz-Aguilar et al., 2009). Higher infestations of the tick *Ornithodoros maritimus* have been found to reduce the body condition and survival of chicks in the Mediterranean Storm-petrel (Sanz-Aguilar et al., 2020).

4.3.3.3 Survival

Scott (1970) estimated that the annual survival rate of European Storm-petrels on Skokholm in the 1960s was 0.88-0.91. The mean survival estimate (± SD) for European Storm-petrels based on the BTO's Retrapping for Adult Survival (RAS) scheme is 0.79 ± 0.04, using data obtained without using tape-lures from Eilean Hoan, Sanda Island, Priest Island and Lunga between 1996 and 2015 (Horswill et al., 2016). A previous analysis using all BTO ring recoveries from 1967–97 estimated annual survival at 0.858 ± 0.016 (SE) (Dagys, 2001), and a similar analysis for 2001–12 suggests that survival across Britain and Ireland slowly increased during this period, staying above 0.90 from 2008–12 (Insley et al., 2014). On Priest Island, survival was 0.92 ± 0.08 (SE) in 2001–02 and 0.96 ± 0.08 in 2002–03, but was lower in 2003–12, with an overall annual survival of 0.80 during this period, and a low of 0.61 ± 0.10 in 2012 (Insley et al., 2014). Ringing data on Skomer for 2006-17 suggest an annual survival rate of 0.88 for breeding adults and 0.59 for transients (Zbijewska et al., 2020), although estimates for transient, pre-breeding birds that prospect multiple colonies before recruiting are likely to be underestimates. Survival estimates for European Storm-petrels at Aketx Islet in the Bay of Biscay for 1990–2006 ranged from 0.82–0.89, depending on the model used (Zabala et al., 2011).

Very small numbers of European Storm-petrels are ringed as chicks and immature survival rates are not currently available. Scott (1970) estimated that annual mortality between birds first returning to the colony and recruiting into the breeding population might be approximately 10-15%.

Sanz-Aguilar et al. (2008) found reduced survival in Mediterranean storm-petrels following their first breeding attempt or an unsuccessful breeding attempt.

4.3.4 Foraging ecology

European Storm-petrels are surface-feeders, although can make shallow dives up to 5 m (Flood et al., 2009, Albores-Barajas et al., 2011), and may target areas where prey is brought to the surface by upwellings and internal waves (Scott et al., 2013). Although, for the most part, European Storm-petrels are highly pelagic during the day (Bolton, 2021), they are known to forage inshore at night and occasionally during daylight (Stegeman, 1990, Koerts, 1992, D'Elbee and Hemery, 1997, Thomas et al., 2006, Poot, 2008, Flood et al., 2009, Albores-Barajas et al., 2011).

The continental shelf edge has been highlighted as having high concentrations of foraging European Storm-petrels (Stone et al., 1995, Waggitt et al., 2020), but breeding adults tracked from Mousa, Shetland Islands, all foraged in the shallow shelf waters to the south of the colony (Bolton, 2021). Birds from Mousa had foraging trips lasting one to three days during incubation, one day during brooding and one to two days during the post-brood stage (Bolton, 2021). Overall, the median total distance travelled was 391 km (maximum 958 km) and the median maximum range from the colony was 159 km (maximum 397 km), with no statistically significant difference between breeding stages (Bolton, 2021).

Nineteen European Storm-petrels tracked from Lunga, Treshnish Isles in 2021 (RSPB unpublished data) similarly foraged over shallow shelf waters. During the incubation stage, birds fed on average for 2–3 days at sea and ranged up to 142 km (max 198 km) from the colony, whereas when tending chicks, feeding trips tended to last for a single day and birds remained within 95 km (max 140 km) of the colony.

Six chick-rearing birds tracked from Illauntannig and High Island, Ireland, had a mean trip duration of 38 hours (maximum 67 hours), mean total distance travelled of 518 km (maximum 1,113 km) and mean range of 170 km (maximum 336) (Wilkinson, 2021). The maximum foraging range provided in the review by Woodward et al. (2019) is taken from these data alone, as presented by Critchley et al. (2018), so 336 km is the current standard foraging range used for offshore wind casework., although confidence in this value is poor. Most of the foraging locations for these birds were near or beyond the shelf edge at the Porcupine Bank and Porcupine Seabight and one bird apparently foraged close to the coast overnight (Wilkinson, 2021). There was also evidence of dual foraging, with some birds making shorter trips and remaining much closer to the colony (Wilkinson, 2021), and the possibility of birds using this strategy should be considered when using foraging range data to assess the risks posed by wind farm developments. All tracking to date in Britain and Ireland has been conducted between mid-July and mid-August, representing a fairly narrow time window relative the full extent of the breeding season (May to October).

The trips of European Storm-petrels tracked in the Atlantic are shorter than those of GPS-tracked Mediterranean storm-petrels, which lasted up to five days, covered total distances of up to 1,727 km, and ranged up to 469 km from the colony (De Pascalis et al., 2021, Rotger et al., 2021). Birds breeding at Benidorm Island in the Mediterranean had foraging areas that encompassed submarine canyons (Rotger et al., 2021). Birds breeding at Sardinia, Italy, foraged in shallow water with strong currents during their longer incubation trips but closer to the colony during chick-rearing, in shallow, productive waters (De Pascalis et al., 2021).

4.3.4.1 Diet

On Skokholm, Pembrokeshire, in the 1960s, regurgitates contained young herring *Clupea harengus* and sprat *Clupea sprattus* as well as amphipods and decapods (Scott, 1970). Two main prey groups were identified in birds breeding in the Bay of Biscay by D'Elbee and Hemery (1997). The first consisted of oceanic and neritic organisms such as ichthyoplankton (Gadidae, Ammodytidae, Myctophidae) and microzooplankton (Copepoda, Euphausiacea, Chaetognatha, Anthomedusae and meroplanktonic larvae), with an average body length of 4 cm (range 0.5–9 cm) and included bioluminescent species that migrate vertically in the water column at night. The second group consisted of littoral (Gobiidae) and suprabenthic intertidal organisms such as isopods (Cirolanidae), which were presumably collected during nocturnal inshore foraging (D'Elbee and Hemery, 1997). DNA analysis of faeces and regurgitates collected from European Storm-petrels breeding on Mousa, Shetland (Wood, 2017) identified fish in 97% of faecal samples and 92% of regurgitates, and invertebrates in 71% of faecal samples and 3% of regurgitates. No faecal samples contained squid DNA. The most common fish species were Norway pout *Trisopterus esmarkii* and Whiting *Merlangius merlangus*. European Storm-petrels will also scavenge on carcasses and waste from fishing boats and can be found foraging in high concentrations around fish farms (Hudson and Furness, 1989, Medeiros Mirra, 2010, Borg, 2012, Josa et al., 2021).

4.4 Leach's Storm-petrel

4.4.1 Spatial and temporal distribution in Scotland

4.4.1.1 Breeding distribution

Most Leach's Storm-petrels breeding in Britain and Ireland do so at colonies in Scotland's Western Isles, with smaller numbers on islands off the west coast of Ireland, and two small colonies (± 20 pairs) in Shetland (Figure 4; Mitchell et al., 2004). However, for two of the seven Scottish colonies for which breeding was confirmed in the Seabird 2000 census, there is no recent evidence of breeding (Foula, Shetland and Old Hill, Lewis). Breeding was confirmed at Gloup Holm, Shetland in 2020 (Miles et al., 2021). Birds are present at the breeding colonies between April and October.

4.4.1.2 Marine distribution

4.4.1.2.1 Derived from at-sea surveys

Based on data from at-sea surveys, during May to August Leach's Storm-petrels were highly concentrated over the shelf edge, Rosemary Bank seamount and the deeper waters to the north-west of Scotland (Skov et al., 1994, Stone et al., 1995, Kober et al., 2009). Numbers were highest in the waters around St Kilda, with occasional sightings in the northern North Sea (Hall et al., 1987, Stone et al., 1995). The species was more widely dispersed between September and November, with low numbers around the Outer Hebrides, Shetland, and the east of Scotland (Stone et al., 1995). There were no sightings of Leach's Storm-petrels in Scottish waters between December and April.

Poisson kriging of ESAS data collected between 1979 and 2006 showed distributions of Leach's Storm-petrels during the breeding season (June – October) similar to those described above, with the highest densities on the northern slope of the Rosemary Bank seamount (Kober et al. 2009).

4.4.1.2.2 Derived from tracking data

Tracking of breeding Leach's Storm-petrels from St Kilda in 2021 confirmed their use of deep waters (>1,000 m) around the Rosemary Bank seamount that were identified as hotspots by at-sea surveys (RSPB unpublished data). There is no information from either tracking or ringing regarding the marine distribution of Leach's Storm-petrels (of any age-class) from Scottish colonies outside the breeding season. Twenty Leach's Storm-petrels breeding on St Kilda were equipped with GLS tags in 2021, for retrieval in 2022 (RSPB unpublished data).

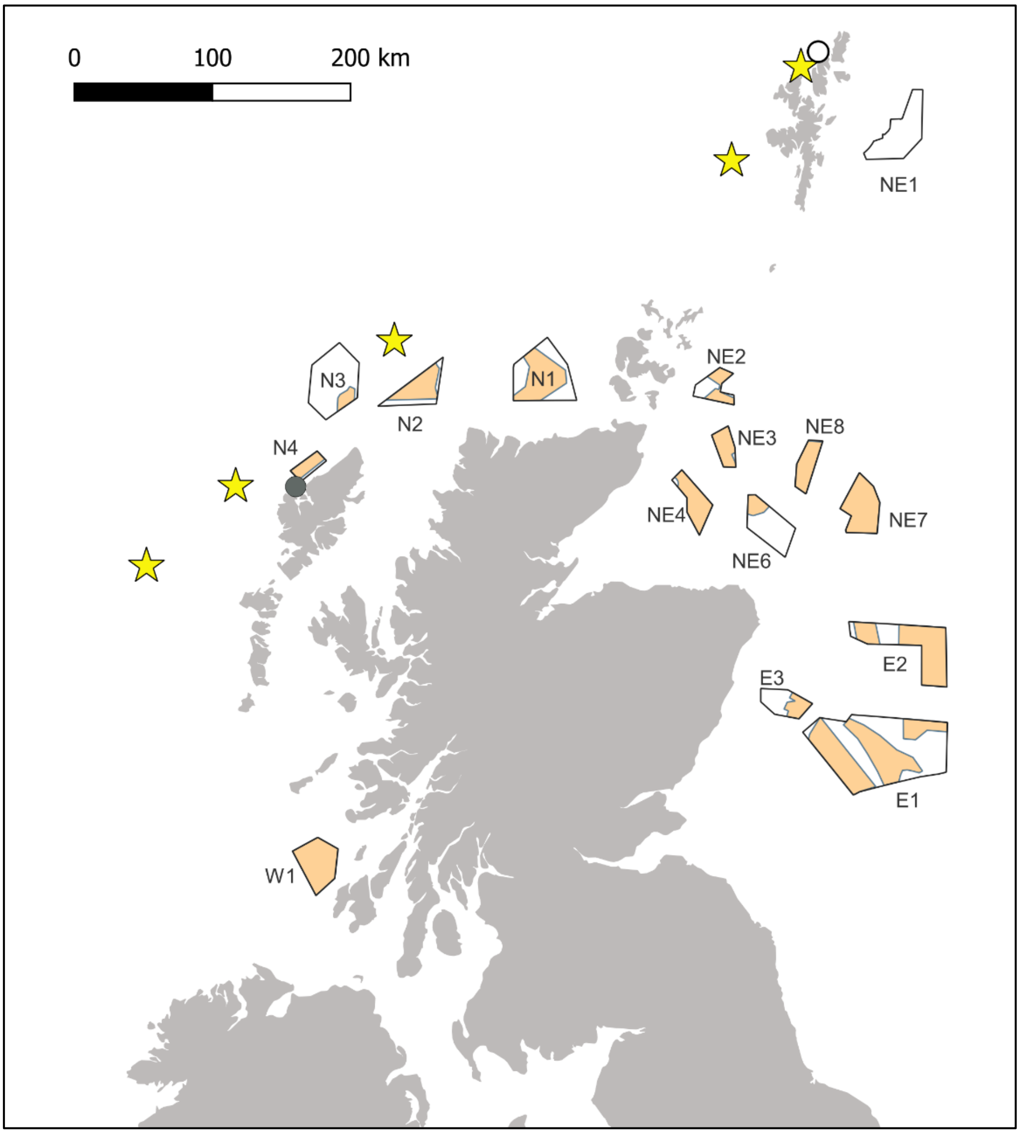

Figure 4. Leach's Storm-petrel colonies in relation to Plan Option areas. Yellow stars represent SPAs with Leach's Storm-petrel as a designated feature. Grey circles are non-SPA colonies surveyed in the Seabird 2000 census (Mitchell et al., 2004). White circles are colonies identified since Seabird 2000. Orange polygons within Plan Option areas indicate sites awarded lease option agreements in the ScotWind leasing round. Note that breeding colonies in Ireland (not shown) may also be at risk of impacts from developments in Scottish waters. (Contains information from the Scottish Government (Marine Scotland) licensed under the Open Government Licence v3.0 and from Crown Estate Scotland under Crown Copyright.)

4.4.2 Population status and abundance

4.4.2.1 Breeding

The global population of Leach's Storm-petrels is estimated at 6.7–8.3 million breeding pairs, with 40–48% of these in the Atlantic (>90% of which breed in the western Atlantic) and 52–60% in the Pacific (BirdLife International, 2022c). There is genetic evidence for long-distance dispersal between colonies in the Atlantic, suggesting that Leach's Storm-petrels in the North Atlantic should be considered as a metapopulation for conservation and management purposes (Bicknell et al., 2012). At the time of the Seabird 2000 census Scotland held more than 99% of Britain and Ireland's breeding Leach's Storm-petrels, with 94% of these on the St Kilda archipelago (Table 3; Mitchell et al., 2004). Additional surveys of Dùn, the island of the St Kilda archipelago with the largest Leach's Storm-petrel population, indicated a 54% decline between 1999 and 2006 (Newson et al., 2008). Data from a further survey of the four main St Kilda islands in 2019 suggest a 68% decline across the archipelago in the 20 years since Seabird 2000, with the population currently estimated at 8,869 (95% CLs 7,787–10,102) AOS (Deakin et al., 2021). This decline led to the species being up-listed from Amber to Red in the UK's Birds of Conservation Concern assessment (Stanbury et al., 2021). The population on North Rona was estimated at 713 AOS in 2009, a 34% decline since 2001, and a 2009 survey of Sula Sgeir found no Leach's Storm-petrels (Murray et al., 2010).

Table 3 Numbers of breeding Leach's Storm-petrel Apparently Occupied Sites (AOS) in Scotland 1999–2002 (Mitchell et al, 2004). All colonies except Loch Roag are SPAs with Leach's Storm-petrel as a designated feature.

Administrative area	Colony	AOS	95% lower confidence limit	95% upper confidence limit
Shetland Islands	Foula	15	3	30
	Gruney	20		
Orkney Islands	Sule Skerry	0		
Western Isles	St Kilda	45,433	34,310	61,398
	Flannan Isles	1,425	1,232	1,708
	Loch Roag	17	15	20
	North Rona	1,132	849	1,700
	Sula Sgeir	5	3	7
Total		**48,047**	**36,432**	**64,883**

4.4.2.2 Non-breeding

An estimated 100,000 individuals are present in Scottish waters on passage (Furness and Wade, 2012).

4.4.3 Productivity and survival

4.4.3.1 Age at first breeding

While the age of first breeding is not known for populations breeding in Scotland, in Canada Leach's Storm-petrels typically breed for the first time at 4–5 years of age (Huntington and Burtt, 1970).

4.4.3.2 Productivity

Breeding success was estimated at a minimum of 0.59 young per egg laid in burrows inspected with an endoscope on St Kilda in 2007 (Money et al., 2008) and 0.65 in 2008, with most failures occurring at the egg stage (Bicknell et al., 2009). The number of Leach's Storm-petrels breeding in nest boxes on St Kilda has increased from a single pair in 2006 to nine pairs in 2021 and breeding success has risen each year from 0.00 in 2006 to 0.89 in 2021, presumably due to increased breeding experience of nest box occupants (Nisbet, 2021). On Kent Island, New Brunswick, breeding success varied with breeding age, increasing by 0.39 ± 0.04 (SE) per year for the first two years, and declining by 0.20 ± 0.04 per year for the final two years of breeding (Mauck et al., 2012).

4.4.3.3 Survival

Our literature search found one estimate of adult survival rate (0.880) for Leach's Storm-petrel in Scotland (1984), cited in MacDonald et al. (2015, Table 1). This value was estimated in 1984 based on ringing conducted during the 1970s and early 1980s. Since that time the number of Great Skuas *Stercorarius skua* breeding on St Kilda has increased substantially, and they have been found to exert high levels of predation on Leach's Storm-petrels (Votier et al., 2006, Miles, 2010), with inevitable, but currently unquantified, impacts on adult survival rates. There is a need to assess current survival rates of Leach's Storm-petrels on St Kilda.

Mean annual survival was 0.78 ± 0.04 at Bon Portage Island, Nova Scotia in 2009–14 (Fife et al., 2015), and 0.79 for colonies at both Baccalieu Island and Gull Island, Newfoundland (Pollet et al., 2019), which is considered low for Procellariiformes and appears to be driving the decline of the species in the north-west Atlantic. Estimates on Kent Island, New Brunswick, Canada suggest that survival increased with age, with annual survival of 0.749 ± 0.046 from breeding year 1 to 2, 0.802 ± 0.040 from 2 to 3, and 0.870 ± 0.030 thereafter (Mauck et al., 2012). Estimates of adult survival for two breeding locations in the East Pacific (Rock Island and Cleland Island, British Columbia) were considerably higher at 0.975 ± 0.011 and 0.975 ± 0.001, respectively (Rennie et al., 2020).

4.4.4 Foraging ecology

Leach's Storm-petrels are highly pelagic, foraging over deep waters, although the species has been observed foraging on the shoreline in eastern Newfoundland, apparently as a result of extreme food stress caused by a marine heatwave (D'Entremont et al., 2021). Incubating adults tracked from Country Island and Bon Portage Island in Nova Scotia travelled to or beyond the continental shelf with foraging ranges of 1,015 ± 238 km and 612 ± 166 km, respectively (Pollet et al., 2014). Foraging range varied between years (Pollet et al., 2014). Tracking of incubating Leach's Storm-petrels at seven colonies in eastern Canada (including Country Island and Bon Portage) revealed little overlap between foraging areas among colonies, although there was no evidence that this was due to intra-specific competition rather than the availability of pelagic foraging habitat (Hedd et al., 2018). Foraging range was not affected by colony size but tended to be shorter for birds breeding at the southern end of the range (Hedd et al., 2018). Trips averaged 4.0 ± 1.4 days, with birds travelling to or beyond the continental slope to highly pelagic waters 400–830 km from the colonies, on average (Hedd et al., 2018). Birds from all but one colony showed a preference for deeper waters, with tracked birds from five of the seven colonies foraging over waters with median depths of more than 1,950 m and average chlorophyll *a* concentrations less than 0.6 mg/m^3 (Hedd et al., 2018). In contrast, birds from the most southerly colony, Kent Island, mainly foraged in shallower neritic waters with a median depth of 181 m (Hedd et al., 2018). Woodward et al. (2019), the current standard reference for foraging ranges for UK offshore wind casework, present a mean foraging range of 657 km, which is based on 11 individuals from Gull Island, Newfoundland, which were the only birds tracked by Hedd et al. (2018) using GPS devices, rather than geolocators, which are less precise. Further GPS tracking has been carried out at Gull Island and the maximum foraging range for 182 complete trips during 2016–2021 was 953 km (Collins et al., 2022).

Fourteen Leach's Storm-petrels tracked from St Kilda in 2021 (RSPB unpublished data) foraged over deep (>1000 m) oceanic water to the north and north-west of St Kilda. During the incubation stage, birds fed on average for 2–3 days at sea and ranged up to 301 km (max 412 km) from the colony, whereas when tending chicks, feeding trips tended to last for two days and birds ranged up to 260 km (max 294 km) from the colony.

In their wintering areas off southern Africa, Leach's Storm-petrel abundance peaked in waters more than 2,000 m deep, particularly in areas with large salinity and sea surface temperature gradients (Camphuysen, 2007).

4.4.4.1 Diet

As far as we are aware there have been no studies of the diet of Leach's Storm-petrels in the UK, but analysis of 18 regurgitate samples from the Faroes found them to be comprised of fish (primarily Gadiformes), crustaceans and other invertebrates (Hey, 2019). The fish species consumed included the demersal species Norway Pout *Trisopterus esmarkii* and Whiting *Merlangus* and, in contrast to studies in the western Atlantic (see below), Hey (2019) found no pelagic fish species, suggesting that foraging habits may differ for birds breeding in the Faroes.

In Newfoundland, fish and crustaceans appear to make up the bulk of the diet of chick-rearing birds, with fish increasing through the breeding season, and mature myctophids and sandeels being particularly abundant in regurgitate samples (Hedd and Montevecchi, 2006, Hedd et al., 2009). Prey items collected range in size from 0.3 to 18 cm (Hedd et al., 2009). While myctophids are deep-water species, migrating to the surface at night, the amphipod *Hyperia galba*, abundant in diet samples, is a parasite of jellyfish commonly found inshore, suggesting both offshore and nearshore foraging by Leach's Storm-petrels. Other prey items include euphausiids, decapods, copepods, isopods and cephalopods (Montevecchi et al., 1992, Hedd and Montevecchi, 2006, Hedd et al., 2009). A more recent study of the diet of Leach's Storm-petrels at colonies in Nova Scotia and New Brunswick found that fish occurrence was lower and cephalopod occurrence was higher than in the Newfoundland studies, although fish still comprised the bulk of the diet (Frith et al., 2020). Diet samples from Daikoku Island, Hokkaido, Japan, consisted of similar taxa to those collected in the north-west Atlantic, although the species differed and cephalopods were more prominent (Watanuki, 1985). Leach's Storm-petrels will also scavenge on fisheries discards (Frith et al., 2020).

Leach's Storm-petrels are surface-feeders and the prominence of diel vertically migrating prey such as myctophids in their diet suggests that they feed mainly at night (Hedd and Montevecchi, 2006). However, the species has been observed feeding during daylight on the midwater fish *Vinciguerria lucetia* in the tropical Pacific when large numbers of fish apparently came to the surface to feed on an abundance of copepods, concentrated by an oceanographic anomaly, so care should be taken when assuming foraging habits based on diet (Pitman and Ballance, 1990).

4.5 Northern Fulmar

4.5.1 Spatial and temporal distribution in Scotland

4.5.1.1 Breeding distribution

Before the mid eighteenth century, the only Northern Fulmar breeding sites within Britain and Ireland were at St Kilda but following a rapid population and range expansion between the late 19th and late 20th centuries the species now breeds all around the coasts of Britain and Ireland, both on mainland cliffs and offshore islands. The largest colonies and most breeding birds are in the north and west of Scotland (Mitchell et al., 2004).

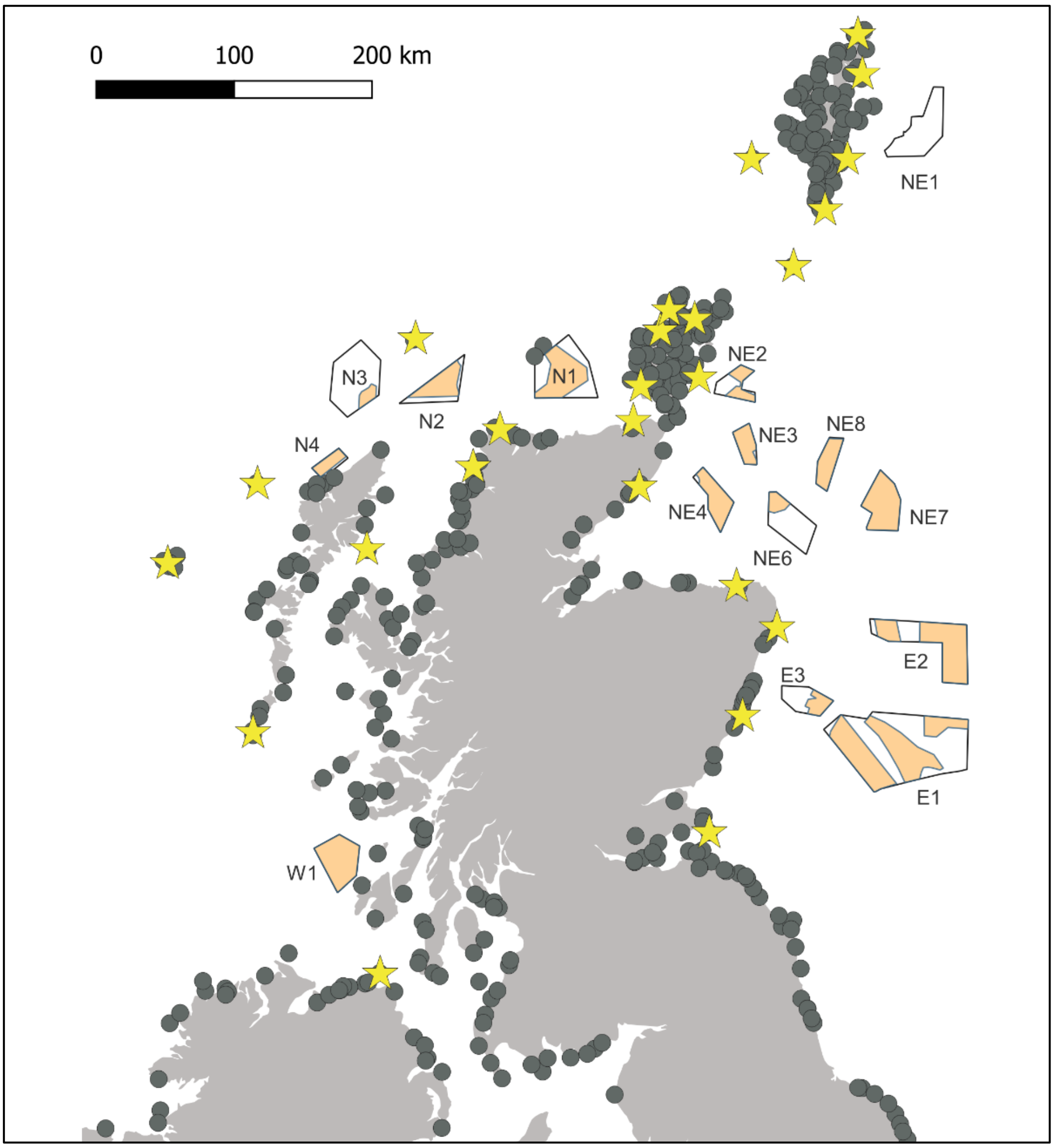

Figure 5. Northern Fulmar colonies in relation to Plan Option areas. Yellow stars represent SPAs with Northern Fulmar as a designated feature. Grey circles are non-SPA colonies surveyed in the Seabird 2000 census (Mitchell et al., 2004). Orange polygons within Plan Option areas indicate sites awarded lease option agreements in the ScotWind leasing round. Note that breeding colonies in Ireland, England and Wales (not shown) may also be at risk of impacts from developments in Scottish waters. (Contains information from the Scottish Government (Marine Scotland) licensed under the Open Government Licence v3.0 and from Crown Estate Scotland under Crown Copyright.)

4.5.1.2 Marine distribution

4.5.1.2.1 Derived from at-sea surveys

Fulmars breed in Scotland between April and mid-September but are present in Scottish waters throughout the year and may visit nest sites from November (Fisher, 1952). During March and April, the highest densities of Fulmars observed on at-sea surveys in Scotland were around the shelf edge to the north and west, including Shetland (Stone et al., 1995). During May to July the highest densities remained at these shelf edges, but the densities increased around colonies in Shetland, Orkney and the Hebrides. Fulmars leave the colonies during September, and from August to November high densities were found throughout the northern North Sea, as well as to the north and west of Scotland, and around Shetland and Orkney (Stone et al., 1995). The species remains in Scottish waters over winter. The results from Stone et al. (1995) are supported by those of (Kober et al., 2009) and Waggitt et al. (2020) who modelled at-sea survey data, and Darby et al. (2021), who modelled habitat preference based on tracking of 102 breeding adults from 11 colonies between 2009 and 2019.

4.5.1.2.2 Derived from tracking data

Tracking from Eynhallow revealed high variation in foraging trips during incubation, with Fulmars foraging in the North Sea, Norwegian Sea, Barents Sea and the central North Atlantic (Edwards et al., 2013, Edwards, 2015, Edwards et al., 2016), but trips during chick-rearing remained largely over the Scottish continental shelf and northern North Sea (Edwards, 2015). A male Fulmar tracked from Eynhallow to the Mid-Atlantic Ridge during incubation foraged over areas of persistent thermal fronts along the Charlie-Gibbs Fracture Zone (Edwards et al., 2013).

4.5.2 Population status and abundance

4.5.2.1 Breeding

The global population of Northern Fulmar is estimated at approximately 7 million pairs, or 20 million individuals (Carboneras et al., 2016), with 3.38–3.50 million pairs in Europe (BirdLife International, 2022b). At the time of the Seabird 2000 census, there were an estimated 537,991 AOS in Britain and Ireland, of which 90% (485,852 AOS) were in Scotland (Mitchell et al., 2004). The population growth in Britain appeared to have stopped between the 1980s and 2000, with overall numbers remaining stable but a mixture of increases and decreases at individual colonies (Mitchell et al., 2004). Scottish populations have been declining since the 1990s and in 2019 (the latest year for which data are available) were at their lowest since nationwide monitoring began in 1986 (JNCC, 2021b).

4.5.2.2 Non-breeding

An estimated 1 million Fulmar are present in Scotland outside of the breeding season (Furness and Wade, 2012).

4.5.3 Productivity and survival

4.5.3.1 Age at first breeding

The mean age at first breeding in the Northern Fulmar is 9 years (range 6–12 years; Dunnet and Ollason, 1978b), but this may be an overestimate (Dunnet et al., 1979). More recent estimates would be useful considering the change in the trajectory of Scotland’s Fulmar population since the 1970s.

4.5.3.2 Productivity

In Scotland, Fulmar productivity has been declining since the mid-1990s, and the number of young fledged per pair decreased from c. 0.55 to 0.39 between 1986 and 2019 (JNCC, 2021b). Productivity is monitored annually at the Isle of May (Newell et al., 2016), Fair Isle (Shaw et al., 2002), Canna (Swann, 2000) and Eynhallow, Orkney (Lewis et al., 2009). Mavor et al. (2008) present multi-year data for a large number of colonies in Scotland.

4.5.3.3 Survival

Annual adult survival has been estimated over several decades at Eynhallow by Grosbois and Thompson (2005). Survival estimates for the first time interval of the study (1962–1963) were 0.951 (95% confidence interval = 0.911–0.973) for females and 0.975 (0.9592–0.9888) for males, but declined over time and for the last time interval (1994–1995) were 0.898 (0.843–0.936) for females and 0.8674 (0.8105–0.9091) for males. Previous studies provide estimates of adult survival at Eynhallow going back to the 1950s (Dunnet et al., 1963, Dunnet and Ollason, 1978b, Dunnet and Ollason, 1978a).

Data on the survival of juvenile and immature Fulmars are lacking. Dunnet and Ollason (1978b) used indirect methods to estimate that mean annual survival of pre-breeders in the 1950s–1970s was between 0.88 and 0.93, but this may be an overestimate.

4.5.4 Foraging ecology

Fulmars are predominantly surface-seizers but will also make shallow dives (Garthe and Furness, 2001). The species is a common scavenger at fishing vessels (Garthe and Hüppop, 1994), where it can congregate in large numbers (Camphuysen et al., 1995), but the evidence for fisheries driving the at-sea distributions of Fulmars is mixed, with some studies suggesting the spatial distribution of fisheries and Fulmar are not correlated (Camphuysen and Garthe, 1997), while others suggest a strong correlation (Darby et al., 2021).

Woodward et al. (2019), the current standard reference for foraging ranges for UK offshore wind casework, present a mean foraging range of 134.6 ± 90.1 km, a maximum range of 2,736 km, and a mean maximum range of 542.3 ± 657.9 km.

At some colonies, including St Kilda, the timing of colony attendance and/or the prevalence of nocturnally vertically migrating species such as lantern fish in the diet suggest that Fulmars are largely foraging at night (Furness and Todd, 1984, Danielsen et al., 2010, Danielsen, 2011), while elsewhere there appears to be little or no nocturnal foraging (Furness and Todd, 1984, Ojowski et al., 2001). Tracking data also indicates nocturnal foraging around fishing vessels (Dupuis et al., 2021), and Fulmars have been observed to feed on discards at night (Garthe and Hüppop, 1993).

4.5.4.1 Diet

The diet of Northern Fulmars in Scotland has been relatively well studied during the breeding season, and far more information is available than for other Procellariiformes in the region. The studies published to date reveal that Fulmars have a very broad diet, which can vary substantially between colonies and years (Furness and Todd, 1984, Camphuysen and van Franeker, 1996, Phillips et al., 1999b), as well as between the sexes and breeding stages (Ojowski et al., 2001, Owen et al., 2013).

Diets of Fulmars at Scottish colonies include fish, crustaceans, jellyfish, squid, pelagic zooplankton and offal (Furness and Todd, 1984, Fowler and Dye, 1987, Camphuysen and van Franeker, 1996, Bourne, 1997, Hamer et al., 1997, Phillips et al., 1999b). Several studies have suggested that sandeels (Ammodytidae) are particularly important in the diets of Fulmars breeding in Shetland (Furness and Todd, 1984, Hamer et al., 1997, Phillips et al., 1999b), and Gray et al. (2003) found that chick survival was reduced in a year of low sandeel abundance. In contrast, Ojowski et al. (2001) found that sandeels comprised only 1% of diet on Foula and Unst, where Gadidae, and Norway pout in particular, were the most common prey.

4.6 Sooty Shearwater

4.6.1 Spatial and temporal distribution

4.6.1.1 Breeding distribution

Sooty Shearwaters mainly breed on offshore islands around New Zealand and Chile, with smaller colonies recorded in southern Australia and the Falkland Islands (Brooke, 2004, Reyes-Arriagada et al., 2007, Catry et al., 2019, Clark et al., 2019, BirdLife International, 2022a). They are present at the breeding colonies between late September/early October and April (Richdale, 1963, Warham et al., 1982, Brooke, 2004).

4.6.1.2 Marine distribution in Scotland

The species undertakes one of the longest known avian migrations and tracking of non-breeding Sooty Shearwaters has uncovered a clockwise circular route in the Atlantic (Hedd et al., 2012, Bonnet-Lebrun et al., 2021). Those found around Scotland are likely to be birds that migrate from colonies around Cape Horn and the Falkland Islands (Phillips, 1963, Hedd et al., 2012, Catry et al., 2019). The species has been observed in Scottish waters between May and December, but mostly at low densities (Phillips, 1963, Camphuysen, 1995, Stone et al., 1995). Long-term data from vessel-based surveys show relatively low densities during July and August, with the highest concentrations in the Minch, and moderate densities around the Rockall Bank (Stone et al., 1995). The species is widespread but at mostly low densities over the continental shelf to the north and west of Scotland, around Shetland, the Moray Firth and the North Sea. During September to November, the highest densities are around Orkney and Caithness, with widespread low or moderate densities elsewhere and more widespread, low densities off the north-east coast and in the Firth of Forth than earlier in the year (Stone et al., 1995).

4.6.2 Population status and abundance

4.6.2.1 Breeding

Sooty Shearwaters are one of the most abundant shearwaters in the southern hemisphere, but the global population is thought to be in moderately rapid decline due to the impact of fisheries bycatch, climate change and the harvesting of chicks by indigenous communities, and the species is listed as Near Threatened (Newman et al., 2008, Newman et al., 2009b, BirdLife International, 2022a). The global population is thought to number around 4.4 million breeding pairs, with 19–23.6 million individuals in total (Newman et al., 2009b, Waugh et al., 2013, BirdLife International, 2022a).

4.6.2.2 Non-breeding

An estimated 7,500 Sooty Shearwaters are present in Scottish waters outside of their austral breeding season (Furness and Wade, 2012). Declines have been observed in other wintering locations (Veit et al., 1997, Oedekoven et al., 2001), but data for Scotland are lacking.

4.6.3 Productivity and survival

4.6.3.1 Age at first breeding

The mean age at first breeding of Sooty Shearwaters is 4.8 years (range 2–10 years; Fletcher et al., 2013).

4.6.3.2 Productivity

All the productivity data found in our literature search are from New Zealand colonies, where productivity is highly variable and non-synchronous. Mean breeding success across studied colonies ranges from 3% to 76% (Jones et al., 2003, Newman et al., 2009a). Hamilton (1998) found that at colonies with high levels of predation, chick survival was 0–41%, but where there was lower natural predation or predator management had been implemented, 64–100% of chicks survived to fledging age.

4.6.3.3 Survival

Adult survival is between 86–98% (Clucas et al., 2008). Fletcher et al. (2013) estimate the survival rate of juveniles in their first two years as 41–54% per year.

4.6.4 Foraging ecology

Sooty Shearwaters forage by surface-seizing and pursuit diving (Weimerskirch and Sagar, 1996). They are capable of diving deeper than other petrels (Dunphy et al., 2015), using a zigzag pattern to reduce buoyancy (Oka, 1994). Weimerskirch and Sagar (1996) give an average dive depth of 38.7 ± 20.1 m (range 2–67 m), but Shaffer et al. (2009) found that 90% of birds dived no deeper than 30 m.

4.6.4.1 Diet

Diet data for Sooty Shearwaters in Scotland are lacking, but elsewhere they are known to feed on fish, squid and crustaceans, mostly euphausiid krill and hyperiid amphipods (Brown et al., 1981, Jackson, 1988, Shiomi and Ogi, 1992, Kitson et al., 2000, Cruz et al., 2001, Petry et al., 2008). They will also follow fishing boats to feed on discards (Wahl and Heinemann, 1979, Otley et al., 2007).

4.7 Existing documented pressures

Pressures arising from human activities in Scottish seas are comprehensively assessed in the Feature Activity Sensitivity Tool (FeAST; Rogerson et al., 2021). Here, we summarise the most important pressures for the species considered in this review.

4.7.1 Climate change

Although the effects of climate change on seabird food availability, productivity and population trends are well-documented from some seabird species and locations (Arnott and Ruxton, 2002, Frederiksen et al., 2004, Frederiksen et al., 2006, Daunt and Mitchell, 2013) there is little evidence of such direct effects on the focal species of this review. Their large potential foraging areas and generalist diets argue for resilience to climate change impacts for these species. However, Scotland's breeding populations of Leach's and European Storm-petrels are predicted to be considerably reduced or extinct by the end of the 21st century, as a result of climate change (Russell et al., 2015), likely as a result of bottom-up effects on their food resources (Daunt and Mitchell, 2013, Mitchell et al., 2020).

Climate change may result in an increase in the incidence of heavy rainfall during incubation, which is negatively correlated with Manx Shearwater hatching success on Rum and Canna as a result of burrows being flooded (Thompson and Furness, 1991). Cold temperatures early in the breeding season may also delay egg laying, which could further affect productivity (Thompson, 1987). Although Manx Shearwaters are generally believed to be robust to changes in food availability due to their ability to travel long distances and consume varied prey (Mitchell et al., 2004), milder winters have been associated with reduced prey quality, later breeding, reduced adult attendance and lower peak and fledging weights of chicks at the Pembrokeshire colonies (Riou et al., 2011). Climate change may also result in an increase in soil erosion and the consequent loss of nesting habitat, or an increase the prevalence of diseases (see below).

Milder winters may also increase the survival of invasive predators such as rats, resulting in increased predator populations and a higher rate of predation on seabirds (Swann, 2000).

Climate change may result in an increase in the incidence and severity of extreme weather events, such as storms that cause large-scale wrecks of Leach's Storm-petrels in the north-east Atlantic (Wynne-Edwards, 1953, Boyd, 1954, Teixeira, 1987).

Ocean acidification has implications for calcium-based marine organisms (Orr et al., 2005). Storm-petrels produce one of the largest eggs, in relation for female body size, of any bird (Davis, 1957a), which exerts high demands on internal calcium stores for eggshell formation. Ocean acidification could affect the ability of these species to produce well-calcified eggshells.

4.7.2 Bycatch

Bradbury et al. (2017) assessed the risk to seabirds of surface, pelagic and benthic fisheries bycatch in UK waters based on (i) species-specific sensitivity (related to conservation status, life history, behavioural traits, bycatch literature and expert opinion) and (ii) the overlap in species distribution ("vulnerability") and relevant fishery activity in summer and winter ("exposure"). They concluded that Fulmars had extremely high sensitivity to bycatch at the sea surface (ranked second of 61 species considered, with a score of 90), Manx Shearwater ranked 8th (score = 66), Sooty Shearwater ranked 16th (score = 53), Leach's Storm-petrel ranked 39th (score = 31) and European Storm-petrel ranked 46th (score = 26). All species scored highly for their response to fishing activity, indicating their tendency to follow vessels. The storm-petrels scored lower than the shearwaters and Fulmar for surface entrapment risk (2 versus 4 and 5 for storm-petrels, shearwaters and Fulmar respectively), although instances of entrapment of both European and Leach's Storm-petrels in fishing nets have been recorded (Bradbury et al., 2017, Costa et al., 2020). All five species were assessed as having lower, and similar, sensitivities to bycatch in pelagic fisheries, due to lower entrapment risk. To date there has been a limited programme of seabird bycatch monitoring in UK waters, although instances of Fulmar bycatch in longline fisheries in Scotland have been recorded, suggesting several thousand individuals may be bycaught each year (Northridge et al., 2020). Currently there are no systematic data on rates of bycatch for other focal species in Scotland, or elsewhere in UK. Manx and Sooty Shearwater are also at risk of bycatch in fisheries operating in the southern hemisphere during their non-breeding and breeding periods, respectively (Uhlmann, 2003, Bugoni et al., 2008), but the level of risk will vary depending on the areas used by the birds (Bonnet-Lebrun et al., 2020).

4.7.3 Pollution

Levels of oil contamination of seabirds have fallen in recent decades both in UK waters (Schmitt, 2019) and elsewhere in the North Sea (e.g. Stienen et al., 2017) and chronic oil pollution is generally considered not to be an important driver of seabird population declines in Scotland (NatureScot, 2021). Procellariiformes are particularly vulnerable to plastic ingestion (Moser and Lee, 1992, O'Hanlon et al., 2017) and a programme of monitoring plastic ingestion in Fulmars in the North Sea has operated since 2002 (Van Franeker et al., 2021). Whether plastic ingestion leads to mortality and exerts population level effects on the focal species in the UK is currently unknown. In Canada, Leach's Storm-petrels have been found to contain some of the highest known levels of mercury contamination of any seabirds, but no deleterious effects have been identified (Pollet et al., 2017).

4.7.4 Invasive Non-Native predators

Invasive Non-Native (INN) species such as Brown Rats *Rattus norvegicus*, Black Rats *Rattus rattus*, various species of mustelid and feral cats *Felis catus* have been implicated in population declines and extirpation of Manx Shearwaters, Leach's and European Storm-petrels in Scotland and elsewhere in the UK. The breeding population of Manx Shearwaters on Canna declined by 99% between 1995 and 2004, with predation by Brown Rats and cats implicated as the primary cause (Swann, 2000, Patterson, 2006). Productivity on Canna averaged 0.6 in the 1980s, declining to <0.2 in the mid-1990s due to predation by Brown Rats *Rattus norvegicus* (Luxmoore et al., 2019), but increased following rat eradication in 2006, averaging 0.74 fledglings/pair between 2009 and 2017 (Luxmoore et al., 2019) and 0.90 between 2015 and 2019 (JNCC, 2021b). While breeding productivity has greatly increased, the Manx Shearwater population has not recovered and remains very small (Luxmoore et al., 2019), in contrast to the recoveries of Manx Shearwater populations following rat eradication on Ramsey (Bell et al., 2019) and Lundy (Booker and Price, 2014). Brown Rats were implicated in the decline of a Manx Shearwater colony on Eigg, where predation by native Eurasian Otters *Lutra lutra* may have also been a factor (Evans and Flower, 1967). Brown Rats are also present on Rum and, although in the 1980s it appeared that the location of the Manx Shearwater colony on a mountain 'island' surrounded by unproductive moorland offered the species some protection from rat predation (Thompson, 1987), more recently there have been concerns that the rats may be causing a problem (Mitchell et al., 2004). There is mixed evidence regarding the effect of rats on the productivity of breeding Manx Shearwaters on Rum (Thompson, 1987, Mitchell et al., 2004, Ratcliffe et al., 2009, Lambert et al., 2015) and some evidence that negative effects of Wood Mice *Apodemus sylvaticus* can substitute those of rats when rat numbers are reduced (Lambert et al., 2021).

Invasive mammalian predators are a key threat to both Leach's and European Storm-petrel populations, with the species breeding almost exclusively on rat-free islands, and rarely coexisting with other introduced mammalian predators (De León et al., 2006). The colonies of both species on Foula, where cats are present, were formerly more numerous but by the time of the Seabird 2000 census had apparently been restricted to a small number of inaccessible ledges (Mitchell et al., 2004). Despite an abundance of suitable habitat, European Storm-petrels had not been recorded on the Shiant Isles prior to the eradication of Black Rats in 2016 (Main et al., 2019), but calling birds were heard in 2017 and successful breeding was detected in 2018 (First Storm-petrel chick for Shiant Isles [redacted]).

During the Seabird 2000 census an American Mink *Neogale vison* was found on Old Hill, Loch Roag, which threatened the survival of the small colony of Leach's Storm-petrels that existed there at the time (Mitchell et al., 2004). Preventing the colonisation of Leach's Storm-petrel breeding islands by mammalian predators is essential for their survival. The majority of Leach's Storm-petrels in the UK breed at St Kilda, where there is a high risk of invasion by mammals as the main island of Hirta is regularly visited by supply vessels and leisure boats.

4.7.5 Native predators

Native avian predators such as gulls and skuas may become problematic if populations increase or if changes in other food sources result in increased predation of seabirds (Votier et al., 2004b, Bicknell et al., 2013, Church et al., 2019).

The large decline of Leach's Storm-petrels on St Kilda has been attributed primarily to increased predation by the Great Skua population (Votier et al., 2004a, Votier et al., 2006, Miles, 2010, Deakin et al., 2021), which increased from 10 to 271 pairs between 1971 and 1997 (Phillips et al., 1999a). Great Skuas were estimated to consume approximately 14,850 Leach's Storm-petrels at St Kilda in 1996 (Phillips et al., 1999c) and 21,000 a year in 2007–2009 (Miles, 2010).

Great Skuas on St Kilda were estimated to consume 455 Manx Shearwaters in 1996 (Phillips et al., 1999c), which is roughly equivalent to 30% of the total estimated adult mortality (Mitchell et al., 2004), and 7,450 European Storm-petrels, which was more than three times the archipelago's estimated number of breeding adults in 1999/2000 (Mitchell et al., 2004).

Leach's Storm-petrel populations on North Rona and the Flannan Isles likely also face high levels of predation by Great Skuas and Great Black-backed Gulls *Larus marinus* (Mitchell et al., 2004). However, populations in the western Atlantic that are not subject to intense predation have also declined (Wilhelm et al., 2020) and the species may face additional important pressures away from the breeding colonies.

Storm-petrels are vulnerable to predation by a variety of owl species, including Little Owl *Athene noctua* (Lockley, 1947), Short-eared Owl *Asio flammeus* and Long-eared Owl *Asio Otis*

(Bried, 2003). Predation by breeding owls may contribute to storm-petrel population decline or hinder attempts to establish new colonies following INN eradication, as on Ramsey, Pembrokeshire (M. Bolton pers. obs.). Whilst no owl species breed at any Scottish storm-petrel colonies, Short-eared Owls do occur as regular migrants at colonies such as St Kilda and Mousa, but are unlikely to cause population-level impacts.

Otters will predate seabirds and have been found to impact the breeding success of storm-petrels at some colonies outside of Scotland (e.g. Quinlan, 1983), but more often their impacts at the population level are not large (Bolton et al., 2017, D'Entremont et al., 2020).

The endemic subspecies of Field Mouse on St Kilda *Apodemus sylvaticus hirtensis* is present at some Leach's Storm-petrel sub-colonies and seabirds are known to make up part of its diet, but it is unclear whether this is through predation or scavenging (Bicknell et al., 2009, Bicknell et al., 2020).

4.7.6 Breeding habitat degradation and loss

Decline of the European Storm-petrel population on Auskerry, Orkney, has been attributed to an increase in the number of sheep on the island, which led to the trampling and destruction of 65% of the rabbit burrows (Mitchell et al., 2004) that were previously the main nesting habitat for storm-petrels (Wood, 1997). Similarly, trampling and manuring by ground-nesting seabirds at a European Storm-petrel colony in Brittany, France, resulted in changes to vegetation cover, increased erosion and eventually the collapse of the old rabbit burrows that the storm-petrels nested in, leading to a population decline (Cadiou et al., 2010).

Competition for habitat with other ground-nesters can also be a problem. An increase in the Atlantic Puffin *Fratercula arctica* population on Sule Skerry apparently reduced the available habitat for European Storm-petrels by displacing them from burrows (Mitchell et al., 2004). Similarly, Manx Shearwaters breeding on Mingulay, Outer Hebrides, were apparently extirpated when their burrows were taken over by an increasing Atlantic Puffin population (Elwes and Guards, 1869).

In the western Atlantic, habitat changes at Leach's Storm-petrel breeding colonies have been associated with population declines (D'Entremont et al., 2020) and the colony on North Rona appears to be confined to the walls of man-made structures due to extensive soil erosion limiting the habitat available for burrows (Mitchell et al., 2004). High densities of Soay sheep on Hirta, St Kilda render the sward over most of the island unsuitable for breeding Leach's Storm-petrels, and most birds nest in boulder crevices. In contrast, the absence of sheep on Dùn, St Kilda allows the development of a deep tussock sward and, despite its small area, Dùn supports the majority of the UK's breeding Leach's Storm-petrels.

4.7.7 Parasites, disease and natural toxins

An outbreak of H5N1 Avian Influenza in Great Skuas in 2021, affected colonies at which Manx Shearwaters, Leach's and European Storm-petrel breed (e.g. St Kilda; Banyard et al., 2022). It is not known if Procellariiformes were also infected as none have been found dead, but they may benefit from lowered predation pressure from Great Skuas in the short term. The cause of recent wrecks of seabirds (mainly auks) in the North Sea from September 2021 to January 2022 is currently unknown, though one possibility is that they may be related to toxins associated with algal blooms, which have been detected in the blood of asymptomatic shearwaters in the Mediterranean Sea (Soliño et al., 2019). It is currently unclear to what extent petrels and shearwaters may be exposed to harmful algal toxins in Scottish waters. However, if toxic algal blooms increase in range and frequency due to climate change (Gobler, 2020), they could exert complex effects on seabirds and their predator-prey dynamics.

The disease puffinosis kills approximately 4% of Manx Shearwater fledglings on Skomer and Skokholm each year but is not considered to have population-level impacts (Brooke, 1990). While puffinosis has been linked to a coronavirus (Nuttall and Harrap, 1982), the cause is still not fully understood. A fatal case of avian malaria was recently detected for the first time in a Manx Shearwater in its wintering grounds in Brazil (Vanstreels et al., 2020).

The prevalence of parasites and disease in storm-petrels has not been well-studied at Atlantic colonies, but research on the Mediterranean subspecies of the European Storm-petrel indicates slower mass gain and reduced body condition and survival in chicks with higher parasite loads (Merino et al., 1999, Sanz-Aguilar et al., 2020).

4.7.8 Disturbance from recreation

The localised impact of disturbance by human recreational activities can be significant, with European Storm-petrels breeding within 10 m of a tourist trail on Mousa having significantly lower breeding success than those nesting in less disturbed areas (Watson et al., 2014), but due to the geographic remoteness of most colonies, and inaccessibility of the nesting areas at those colonies, most Scottish breeding colonies are not exposed to recreational disturbance. The impacts of disturbance at sea have not been studied, but Manx Shearwaters may be particularly vulnerable to disturbance by leisure craft when rafting in dense groups prior to visiting colonies.

5 Risks from collision, displacement and lighting attraction

Assessment of the risk of bird collisions at wind farms principally focuses on risks associated with a bird being struck by a rotating blade when passing through the rotor-swept area. The probability of collision, for a bird on a collision course with a turbine, depends on (i) the flight height of the bird, (ii) the likelihood of the bird altering its flight path to avoid the rotor swept area (i.e. avoidance), and (iii) if the bird passes through the rotor-swept area, whether it is struck by a rotating blade. Before considering these components in turn it should be noted that other collision risks may be associated with wind farms and their operations, such as collision with masts and aerials on the support vessels, or with moorings associated with floating wind platforms.

Whilst some components of the overall assessment of the collision risk posed by wind farms, and their population-level consequences, can be computed with estimable precision and accuracy, other components, such as the avoidance rate, or in the case of nocturnal procellariform seabirds, the attraction rate, are subject to considerably greater uncertainty, which render estimates of collision rate and population consequences highly speculative.

In this section we review the available published information to parameterise the collision risk models, and information which may assist the estimation of avoidance rates. Critical to the latter is the extent to which nocturnally active seabirds such as shearwaters and storm-petrels may be attracted to the illuminations required for turbines, support vessels and the construction or expansion of ports. We firstly consider factors other than illumination which may contribute to attraction of nocturnal Procellariiformes for offshore windfarms. In the final section, we explicitly consider the evidence for light attraction.

Flight height estimates presented below are obtained from aerial and vessel-based surveys, necessarily conducted under adequate weather and lighting conditions and usually including ship-following birds. These values may change under different weather and lighting conditions. Many sources providing assessments of the time a species spends at collision risk height do not specify the assumed turbine dimensions, and since turbine technology is rapidly evolving, collision risk levels may also change. Data on flight speeds have been obtained from tracking studies and refer to ground speeds, taking no account of non-linear flight paths and measured at the interval of the tracking device. They will therefore underestimate instantaneous flight speed to an unknown degree. Further, most tracking studies have been conducted on breeding adult birds, and parameter values may differ for immatures or juveniles or for different times of year.

5.1 Attraction of shearwaters and storm-petrels to offshore structures

A number of studies in Canada have found clear evidence that shearwaters and storm-petrels may be attracted to offshore structures such as drilling platforms, likely due to local prey enhancement as the structure acts as an artificial reef (Baird, 1990, Montevecchi, 2006, Burke et al., 2012). The foundations associated with offshore turbines may similarly act as artificial reefs, and cause changes in patterns of sediment transport and accumulation that could provide spawning grounds for benthic species. Whilst there is limited evidence for attraction of shearwaters and storm-petrels to oil and gas platform in the UK (Bourne, 1979, Sage, 1979), likely due to low densities of these species in the northern North Sea where seabird interactions with oil platforms have been studied, other authors report attraction of a variety of diurnal seabird species to oil platforms, likely as a result of local prey enhancement (Tasker et al., 1986). If fishery activity is reduced within windfarms, then local increases in fish density may result in these areas attracting seabirds, such as Manx Shearwaters, storm-petrels, and their avian predators such as large gulls and skuas. Aguado-Giménez et al. (2016) found that European Storm-petrels were attracted to fish farm cages 5 km from the coast during daylight, likely due to local prey enhancement. Procellariiform species are highly pelagic and are extremely unlikely to be attracted to offshore structures for the purposes of roosting, as is seen in species such as cormorants and shags (Dierschke et al., 2016).

5.2 Collision risk

5.2.1 Manx Shearwater

5.2.1.1 Flight style

Manx Shearwaters are classed as glide-flappers (Spear and Ainley, 1997b), using both flapping and gliding flight and engaging in slope-soaring behaviour (Thompson, 1987, Spivey et al., 2014). Gliding and soaring flight may increase with increasing wind speed (Gibb et al., 2017). Flight speed (see below), wing shape, relatively high wing loading, and tail shape (rounded, not forked) suggest that Manx Shearwaters have only moderate flight manoeuvrability (Warham, 1977, Furness and Wade, 2012).

5.2.1.2 Flight height

The species is generally considered to have low collision risk as it apparently spends limited time flying at rotor blade height (i.e. usually flies less than 20 m above sea level; Garthe and Hüppop, 2004, King et al., 2009, Cook et al., 2012, Furness and Wade, 2012, Furness et al., 2013, Bradbury et al., 2014, Certain et al., 2015). However, current flight height data for this species is based on aerial or vessel-based at-sea surveys, which can only take place during daylight and in relatively calm weather and may not be representative of the behaviour of Manx Shearwaters under all conditions. The species rarely uses level, flapping flight, but usually engages in slope-soaring, which leads to constant variation in flight height, although generally birds will remain low to the sea surface where the shear is strongest (Spivey et al., 2014). Flight heights may increase in stronger winds (Spear and Ainley, 1997b, Ainley et al., 2015) and modelling by Johnston and Cook (2016) indicated an increase in mean flight height between April and September.

Of 6,957 Manx Shearwater recorded during vessel-based surveys at 10 offshore wind farm sites, 0.04% (95% confidence interval <0.01–10.1%) were flying at heights that would put them within the rotor-swept zone (assumed to be 20–150 m above sea level), and models suggested their flight height distribution was unlikely to vary with distance to the coast (Cook et al., 2012). Models by Johnston and Cook (2016) estimated the proportion of flight time within the rotor-swept zone was 0.0 (95% confidence interval 0.0–0.0), based on boat survey data, and 0.0 (95% credible interval 0.0–0.02) based on digital aerial survey data.

5.2.1.3 Flight speed

Breeding Manx Shearwater GPS-tracked from Skomer, Wales, by Guilford et al. (2008) had a mean ground speed of 11.13 ± 9.55 m/s during flight. Behavioural models of GPS data for birds breeding on Skomer and Lighthouse Island, Northern Ireland, indicate median ground speeds of 8.9 m/s during direct or transiting flight and 2.01 m/s during foraging, when flight is more tortuous (Dean et al., 2013). Breeding Manx Shearwaters tracked from Great Blasket and High Island, Ireland in 2014 and 2015 had a mean ground speed across whole trips of 1.58 m/s (SD = ± 0.79 m/s, range 0.36–5.88 m/s), although ground speeds within trips would have shown greater variation (Wischnewski et al., 2019). Tracking from Lundy Island indicated mean ground speeds of 10.89 ± 3.31 m/s during flight, with clusters around 11 and 15 m/s in low wind speeds and greater variation in higher wind speeds, when birds were more likely to engage in soaring flight (Gibb et al., 2017).

Mean ground speeds differed between adults and immatures GPS-tracked from Skomer, with mean (± SE) speeds of 7.0 m/s ± 0.32 m/s for adults and 4.97 ± 0.25 m/s for immatures on short trips and 5.83 ± 0.17 m/s for adults and 5.14 ± 0.22 m/s for immatures on long trips (Fayet et al., 2015).

5.2.1.4 Temporal activity patterns

For breeding Manx Shearwaters tracked from Skomer and Lighthouse Island (Copeland) in July and August of 2009–2011, the percentage of time spent in different behaviours varied between breeding stages and colonies, with birds spending an average of 10% of their time in direct flight (i.e. transiting/commuting) and 63% foraging during incubation, and 15% in direct flight and 57% foraging during chick-rearing (Dean et al., 2013). Direct flight and foraging increased in the hour before sunrise, peaked just after sunrise and were lowest around midday when birds spent more time resting on the water (Dean et al., 2013). There was then a second peak in flight before sunset and a rapid decline at the onset of darkness. Foraging occurred almost entirely within daylight and twilight and birds roosted on the water in the evening and at night. Other GPS tracking studies from Skomer show similar activity patterns during incubation and chick-rearing (Guilford et al., 2008, Fayet et al., 2015). However, dietary analysis of Manx Shearwaters on Rum indicates that birds may have been foraging at night during the pre-laying period (Thompson, 1987).

5.2.1.5 Avoidance behaviour

Limited data are available on wind turbine avoidance behaviour of Manx Shearwaters given that there is little overlap between the species' distribution and currently operational wind farms, but Dierschke et al. (2016) preliminarily classified the species as weakly avoiding wind farms. Surveys of the Robin Rigg offshore wind farm in the Solway Firth detected a decline in the number of Manx Shearwaters in the area during construction and operation, compared with pre-construction (Canning et al., 2013b, Canning et al., 2013a), suggesting some macro-avoidance, but birds were observed close to turbines (Dierschke et al., 2016). An obvious gap in Manx Shearwater distribution was observed at North Hoyle wind farm in Liverpool Bay (Dierschke et al., 2016).

Flight speed, wing and tail morphology suggest that Manx Shearwaters may have limited manoeuvrability for micro-avoidance of turbine blades and associated structures. Flight agility is likely to be influenced by wind speed. Warham (1977) noted that in low winds shearwaters often come in fast and crash land at the colony but on windy evenings can stall and land lightly. In the context of collisions with turbine, shearwaters are likely to have lowered manoeuvrability under conditions when blades are turning more slowly. Adults, sub-adults and fledgling shearwaters of various species are known to collide with human-made structures on land, and this can sometimes result in high numbers of fatalities (Podolsky et al., 1998, e.g. Albores-Barajas et al., 2016), further indicating low levels of micro-avoidance.

5.2.2 European Storm-petrel

5.2.2.1 Flight style

European Storm-petrels fly with a combination of flapping and short glides, often moving in zig-zags and sometimes shearing in strong winds (Flood and Thomas, 2007). When feeding they hover or patter on the surface of the water, dipping to seize food items (Flood and Thomas, 2007). Smaller-bodied Procellariiformes have greater manoeuvrability in flight due to lower wing loading (Warham, 1977) and storm-petrels are highly manoeuvrable in snatching prey for the sea surface.

5.2.2.2 Flight height

Vessel-based observations suggest European Storm-petrels generally fly within 2 m of the sea surface, but occasionally up to 5 m (Flood and Thomas, 2007). They may fly lower in strong winds to shelter in wave troughs, as observed in the oceanitid and *Oceanodroma* storm-petrels (Ainley et al., 2015). Largely as a result of its low flight height, the European Storm-petrel is generally considered to be at low risk of collision (King et al., 2009, Cook et al., 2012, Furness and Wade, 2012, Furness et al., 2013, Bradbury et al., 2014, Certain et al., 2015), but data on flight heights for this species are limited. Observations of 52 European Storm-petrels on surveys of two offshore wind farm sites included a mean of 2% (range 0–2.5%) flying at heights that would put them at risk of collision with wind turbine blades (Cook et al., 2012).

5.2.2.3 Flight speed

European Storm-petrels tracked from Ireland had a mean trip speed of 4.05 (range 2.62–4.93) m/s and the maximum ground speed of any bird between two consecutive GPS locations was 11.18 m/s (Wilkinson, 2021). Mediterranean Storm-petrels tracked from Sardinia during incubation in 2020 had a mean speed of 4.0 ± 0.9 (range 2.1–5.2) m/s and a maximum speed of 9.8 ± 2.0 (6.7–12.5) m/s, while those tracked during chick-rearing in 2019 had a mean speed of 2.63 ± 0.9 (1.1–4.1) m/s and maximum speed of 7.38 ± 1.7 (4.5–9.8) m/s (De Pascalis et al., 2021). For Mediterranean storm-petrels tracked from Benidorm Island, the mean (± SD) speed was 4.18 ± 0.68 m/s (range 3.46–4.82 m/s) and the maximum travel speed was 10.17 ± 3.33 m/s (range 6.41–22.46 m/s) (Rotger et al., 2021). The mean speed for birds engaging in area-restricted search behaviour (i.e. foraging) was 2.03 ± 0.86 m/s (range 0.63–3.95 m/s) (Rotger et al., 2021).

5.2.2.4 Temporal activity patterns

European Storm-petrels depart from and return to the colony at night and while on foraging trips will forage both diurnally and nocturnally (D'Elbee and Hemery, 1997, Bolton, 2021). A two-state hidden Markov model for European Storm-petrels tracked from west Ireland assigned 60.6% of locations from High Island birds as foraging behaviour and 39.4% as transiting, while for Illauntannig foraging and transiting were assigned to 59.2% and 40.8% of locations, respectively (Wilkinson, 2021). Note that resting behaviour was not considered by Wilkinson (2021), but Mediterranean storm-petrels tracked from Benidorm Island spent a mean (± SD) of 35.23% ± 9.77 (range 19.00–54.00%) of the time resting on the water (Rotger et al., 2021).

5.2.2.5 Avoidance behaviour

We found no information in the literature regarding the extent of macro-, meso- or micro-scale avoidance by European Storm-petrels.

5.2.3 Leach’s Storm-petrel

5.2.3.1 Flight style

Leach’s Storm-petrel is classed as a glide-flapper, using a combination of flapping and long, shearing glides and hovering or pattering on the surface of the water to seize food items (Spear and Ainley, 1997b, Flood and Thomas, 2007). It has a very low wing loading (Warham, 1977) and its flight path can be irregular and unpredictable, with rapid changes of speed and direction, and becoming highly erratic in strong winds (Spear and Ainley, 1997b, Flood and Thomas, 2007).

5.2.3.2 Flight height

Vessel-based observations suggest that Leach's Storm-petrels generally stay within 5 m of the sea surface (Flood and Thomas, 2007) and they may fly lower during strong winds to shelter in wave troughs (Ainley et al., 2015). The species is usually assumed to have a low risk of collision, but data are limited and information for the European Storm-petrel is often used as a proxy (King et al., 2009, Langston, 2010, Furness and Wade, 2012, Furness et al., 2013, Bradbury et al., 2014).

5.2.3.3 Flight speed

Our literature search did not identify any estimates of flight speed for Leach's Storm-petrel, but Pollet et al. (2019) suggest it is relatively slow, similar to the 4 m/s given by Withers (1979) for Wilson's Storm-petrel.

5.2.3.4 Temporal activity patterns

Leach's Storm-petrels depart from and return to the colony at night (Ainslie and Atkinson, 1937) and are believed to forage both diurnally and nocturnally (Pitman and Ballance, 1990, Hedd and Montevecchi, 2006). More detailed information on their at-sea activity is lacking.

5.2.3.5 Avoidance behaviour

We found no information in the literature regarding the extent of macro-, meso- or micro-scale avoidance by Leach's Storm-petrels

5.2.4 Northern Fulmar

5.2.4.1 Flight style

The Norther Fulmar is a flap-glider, uses gliding flight extensively during foraging (Pennycuick, 1987) and increases gliding behaviour with higher wind speeds (Ainley et al., 2015). It has intermediate flight manoeuvrability (Warham, 1977, Garthe and Hüppop, 2004).

5.2.4.2 Flight height

Fulmars are generally considered to be at low risk of collision as they apparently spend limited time at collision risk height (Garthe and Hüppop, 2004, Cook et al., 2012, Fijn et al., 2012, Krijgsveld, 2014, Leopold and al., 2014, Harwood et al., 2018). Modelling based on 29,168 vessel-based observations estimates that the proportion of Fulmars flying at collision risk height (where the lower limit of the rotor-swept area is 20 m above sea level) is 0.002 (95% CI 0.000–0.061; Johnston et al., 2014). However, the species may fly higher in stronger winds (Spear and Ainley, 1997b, Ainley et al., 2015) and this behaviour is unlikely to be captured in vessel-based surveys, which are conducted only in relatively calm conditions.

5.2.4.3 Flight speed

The mean air speed of Fulmars measured off Foula, Shetland, using an ornithodolite was 13.0 m/s (Pennycuick, 1987). A male Fulmar GPS-tracked from Eynhallow, Orkney, during incubation, had an overall ground speed of 7.9 m/s and a maximum hourly ground speed of 17.6 m/s during its outward journey to the Mid-Atlantic Ridge, and an overall ground speed of 7.7 m/s and maximum hourly ground speed of 13.5 m/s on its return journey, although the return leg was largely against a headwind (Edwards et al., 2013). Hourly transit ground speeds were faster during the day (median = 9.4, range = 0.9–17.6 m/s) than at night (median = 4.6, range = 0.2–9.5 m/s), but hourly ground speeds during area restricted search (median = 1.2, range = 0.1–6.9 m/s) did not differ between day and night (Edwards et al., 2013). However, ground speeds from tracking data tend to be underestimates, and Weimerskirch et al. (2001) suggest the species regularly attains ground speeds of 19.4 m/s, aided by wind. Elliott and Gaston (2005) found that ground speeds of Fulmars in Nunavut, Canada, were lower during incubation (9.2 m/s) than chick-rearing (10.8 m/s) and, in contrast to Edwards et al. (2013), found that ground speeds were significantly lower for outgoing birds (8.8 m/s) than incoming birds (10.2 m/s).

5.2.4.4 Temporal activity patterns

The diurnal pattern of colony attendance by breeding Fulmars is very variable, suggesting different levels of nocturnal foraging at different breeding sites (Dott, 1975, Furness and Todd, 1984, Ojowski et al., 2001, Danielsen, 2011). Analysis of tracking data also suggests a combination of diurnal and nocturnal foraging (Edwards et al., 2013). Observations at sea near Shetland in the breeding seasons of 1992-94 found that Fulmars spent 81% of time resting or swimming and only 19% of time flying (Ojowski et al., 2001), but tracking by Edwards et al. (2013) suggested that foraging bouts involve short searching flights and only brief periods on the water, when prey is captured and consumed. Given the wide range of prey and varied foraging ecology of the Fulmar, it is likely that the time it spends in different behaviours is also highly variable.

5.2.4.5 Avoidance behaviour

Dierschke et al. (2016) classified Fulmars as weakly avoiding offshore wind farms, based on post-construction studies at 20 sites, but the authors note that data for this species are limited and it may actually display strong avoidance behaviour. It is possible that the lack of fishing vessels within wind farm areas makes them unattractive to Fulmars (Neumann et al., 2013, Braasch et al., 2015), but there is conflicting evidence regarding the influence of fishing vessels on Fulmar distributions (see section 4.5.4).

5.2.5 Sooty Shearwater

5.2.5.1 Flight style

Like Manx Shearwaters, Sooty Shearwaters are glide-flappers (Spear and Ainley, 1997b) with intermediate flight manoeuvrability (Warham, 1977, Garthe and Hüppop, 2004).

5.2.5.2 Flight height

Sooty Shearwaters are considered to have low collision risk as they generally fly very close to the sea surface and therefore below blade height (usually assumed to be 20–150 m above sea level), but this is based on very small sample sizes (Paton et al., 2010, Cook et al., 2012) and an assumption that Sooty and Manx Shearwaters fly at similar heights (Furness and Wade, 2012). Like Manx Shearwaters, Sooty Shearwaters may fly higher in stronger winds (Spear and Ainley, 1997b, Ainley et al., 2015).

5.2.5.3 Flight speed

Our literature search did not identify any estimates of flight speed specifically for Sooty Shearwater, but Spear and Ainley (1997b) estimated average ground speeds for diving shearwaters, a group which includes Sooty Shearwater, as 10.7 ± 2.3 m/s with a headwind, 14.0 ± 3.5 m/s with a tailwind, and 13.2 ± 4.6 m/s with a crosswind. Flying with a cross wind is by far the most common method used by Procellariiformes, including Sooty Shearwater (Spear and Ainley, 1997a).

5.2.5.4 Temporal activity patterns

While in the northern hemisphere, Sooty Shearwaters spend a large proportion of their time on the water and just 23.9 ± 15.2% of their time in flight, although this increases to 67 ± 24.1% once they begin their return migration to their breeding ground (Hedd et al., 2012, Bonnet-Lebrun et al., 2021). When on the water, they are resting, feeding, digesting (Bonnet-Lebrun et al., 2021) or moulting (Keijl, 2011). In July 2007, Keijl (2011) photographed 76 individuals in a flock gathered off Rockall, to the west of the Scottish mainland, 46% of which were in active primary moult. On their wintering grounds Sooty Shearwaters are particularly stationary at night, when they are on the water for 89% of the time (Hedd et al., 2012), although they are more active on nights with increased moonlight (Bonnet-Lebrun et al., 2021).

5.2.5.5 Avoidance behaviour

We found no information in the literature regarding the extent of macro-, meso- or micro-scale avoidance by Sooty Shearwaters.

5.3 Displacement and barrier effects

There is a lack of empirical evidence relating to displacement, disturbance and barrier effects for these procellariiform seabirds, and therefore high levels of uncertainty regarding their vulnerability (Wade et al., 2016, Kelsey et al., 2018). These species are all generally considered to have a low vulnerability to displacement and disturbance from offshore wind farms and associated activities such as ship and helicopter traffic, and often rank lower than all other Scottish seabird species in terms of population impacts (Furness and Wade, 2012, Furness et al., 2013, Bradbury et al., 2014, MMO, 2018, Rogerson et al., 2021). They will associate with vessels at sea and display limited escape behaviour and short flight distances when approached by boats (Furness et al., 2013). However, there is some evidence of Manx Shearwaters and Fulmars avoiding offshore wind farm developments during the construction and operational phases (see sections 5.2.1.5 & 5.2.4.5; Dierschke et al., 2016), and the deficiency of data for the other species does not indicate a lack of impact. A higher level of disturbance may occur during the construction phase, when activity, noise and light levels may be greatest. The impacts of artificial light on nocturnally active species may also result in increased levels of displacement (see section 5.4).

Habitat specialisation is a key consideration when assessing vulnerability to displacement, with the negative impacts likely to be greater for specialists than generalists. Manx Shearwaters, European Storm-petrels and Leach’s Storm-petrels cover large distances when foraging during the breeding season and appear to forage on a broad range of taxa (see section 4), which could suggest a lack of specialisation. However, all three species apparently travel long distances to target specific oceanographic features (see section 4; Scott et al., 2013, Dean et al., 2015, Hedd et al., 2018, Wilkinson, 2021), and displacement from these important foraging areas would likely have negative consequences. Displacement of Manx Shearwaters from key rafting sites may also result in population-level impacts, if displacement requires them to spend energy on flight, and thereby consume resources that would otherwise have been devoted to their chick on arrival at the colony.

Older chicks can be left unattended and unfed for several days at a time due to their accumulation of large lipid reserves (Ricklefs and Schew, 1994, Bolton, 1995b, Hamer et al., 1998), which could help to buffer them against a reduction in provisioning frequency due to increases in parental foraging trip durations caused by barrier effects. However, during the first week after hatching, chicks are unable to thermoregulate adequately and need to be brooded by adults. In this period they are particularly vulnerable to starvation and inclement weather as adults must divide their time between nest attendance to brood the chick and foraging at sea. Most breeding failures occur at this stage, as the energetic demands on adults, in relation to time available for foraging, are greatest (Bolton, 1995a). Increased energy demands on adults, for example due to displacement from profitable feeding areas, or increased flight paths due to barrier effects, would likely lower chick survival rates.

Several studies have shown that the flight paths of petrels and shearwaters are orientated to maximise the energetic benefits of crosswinds (Spear and Ainley, 1997b), often resulting in circular (rather than direct “out and back”) patterns to foraging trips (Ventura et al., 2020). Displacement and barrier effects may prevent the optimisation of foraging tracks to maximise the energetic benefits of cross winds. During the pre-laying exodus, female Manx Shearwaters undertake long foraging trips to oceanic waters (Dean, 2012) to acquire the nutrients required for egg formation. The single large egg represents a considerable resource investment, and the inward flight to the colony for egg laying is likely to be energetically expensive, at a critical time in the breeding cycle. Increased flight costs imposed by barrier or displacement effects during this period may have particularly high costs on breeding success. The foraging behaviour of female storm-petrels during the pre-laying period is unknown, but since they lay one of the largest eggs in relation to body size of any bird (approximately 30% of female body weight; Davis, 1957a), increased foraging costs imposed by displacement or barrier effects are likely to be particularly severe.

5.4 Lighting attraction and disorientation

The nocturnal attraction of birds to light, often with fatal consequences, has been known for several centuries. Early settlers of the Azores archipelago lit bonfires on the cliffs at night to attract seabirds, which they beat from the air with sticks, to be used as feed for their pigs (Fructuoso, 1561). Studies in the USA estimated that in the mid-1960s more than one million nocturnal migrant birds died annually by collision with illuminated communications towers (Gauthreaux and Belser, 2006) and observers in both the UK and Canada have reported that hundreds, or even thousands, of seabirds, predominantly species of storm-petrel, are killed by attraction to the gas flares of hydrocarbon platforms (Sage, 1979, Wiese et al., 2001, Baillie et al., 2005, Montevecchi, 2006, Burke et al., 2012), although Bourne (1979) disputed the identification of birds killed at platforms in the North Sea. Tasker et al. (1986) did not report shearwaters and storm-petrels attracted to platforms in the central North Sea, but observations were from a region and season associated with low densities of these species (Waggitt et al., 2020). Attraction distances and the possible influence of light position relative to flight paths are considered further in following sections.

There have been several reviews of the attraction of seabirds to artificial light (Montevecchi, 2006, Laguna et al., 2014, Rodríguez et al., 2017) and we do not repeat that information here.

Rather, we review the literature in the specific context of the issues surrounding the assessment of the impacts on seabirds of wind farm development and operation. Wind farms are required to be illuminated in accordance with marine navigation regulations (DECC, 2011, IALA, 2013, MCA, 2021) and the Air Navigation Order (CAA, 2016). In addition, a large programme of port expansion is underway in Scotland to support the construction and maintenance of new offshore wind farms, and this will result in increased illumination in coastal areas. The central issue is the extent to which illumination of wind farm structures, associated infrastructure (such as wet storage), construction activities and the vessels and ports associated with wind farm operations will: (i) attract seabirds, and (ii) modify seabird behaviour in their proximity.

The literature on light attraction in birds does not always make a clear distinction between: (i) attraction *per se* (i.e. "phototaxis"), which could potentially operate over ranges of tens of km, and (ii) the alteration of flight paths of birds when in close proximity (i.e. within tens of metres) of illuminated structures (i.e. "disorientation"). Long-range light attraction may result in birds being displaced from foraging areas and activities. Light-induced disorientation may cause birds to circle light sources for many hours (Gauthreaux and Belser, 2006) with obvious implications for collision risk. Existing models of collision risk assessment (Band et al., 2007, Band, 2012, Masden, 2015, McGregor et al., 2018) do not explicitly model the scenario of birds circling a turbine, but rather consider a straight flight path only. In the context of the assessment of impacts of wind turbines on seabirds, it is helpful to make a clear distinction between these two effects (attraction and disorientation), and the spatial scales at which they operate. The first will affect the number of birds brought into the vicinity of wind turbines and associated structures, vessels and shore facilities ("macro" and "meso" scales sensu Cook et al., 2018), and the second will affect the length of time birds remain within the proximity of potential collision surfaces and the number of occasions an individual bird may pass through the rotor-swept area ("micro" scale sensu Cook et al., 2018). These two effects of artificial light may have different drivers, and impact juveniles and adults differently, as discussed below. We do not consider light attraction to be a separate impact pathway, but it may exacerbate one or more of the recognised impact pathways (e.g. collision, displacement).

5.4.1 Evidence for light-induced disorientation

There is abundant evidence of light-induced disorientation for a wide range of avian groups, including shearwaters and petrels. Such evidence includes: the grounding of fledgling Manx Shearwaters, Leach's and European Storm-petrels in lit areas of the village on Hirta, St Kilda (Miles et al., 2010); collision of Manx Shearwaters with lighthouses and other illuminated structures (Archer et al., 2015, Guilford et al., 2019); grounding of European Storm-petrels onto rocks lit by researchers' head torches (Albores-Barajas et al., 2011); grounding of European and Leach's Storm-petrels on hydrocarbon platforms (Sage, 1979, Wiese et al., 2001, Baillie et al., 2005, Montevecchi, 2006, Burke et al., 2012, Gjerdrum et al., 2021), and the grounding of Leach's Storm-petrels on vessels (Wynn, 2005, Wakefield, 2018, Wilhelm et al., 2021) and industrial developments (Wilhelm et al., 2021).

While the distance from which birds have been attracted to such light sources is usually unknown, observers report that, once attracted to the vicinity (i.e. within several tens of metres) of a powerful light source, birds seem unable to escape and become vulnerable to collision. Rodríguez et al. (2022) showed formally that flight tortuosity of fledgling Cory's Shearwaters *Calonectris borealis* heading from inland breeding sites to the sea increases with the level of light radiance over which they are flying. Tracks of tagged individuals reveal that they remain in flight within the lit areas for several hours before grounding.

Many studies describe procellariform seabirds being drawn downwards towards bright light shining from below (e.g. Rodríguez et al., 2015a, Rodríguez et al., 2017, Rodríguez et al., 2022). In such cases the birds' natural flight height is lowered by light attraction/disorientation. It is not clear to what extent light attraction/disorientation may result in birds that are flying close to the sea (below rotor swept height), being drawn upwards to heights within the rotor swept area, although this is likely to be the case for storm-petrels stranded on oil platforms. The impact of light attraction on flight height must be considered.

Seabird species that rear their young underground seem particularly, if not exclusively, sensitive to light-induced attraction/disorientation. In the case of fledglings this is perhaps because young fledge with somewhat under-developed visual acuity due to a lack of visual stimulation in the darkness of the nest chamber (Atchoi et al., 2020). It is notable that measurements from eyes of two Manx Shearwaters captured on the point of fledging indicated that their optical structure was slightly myopic (i.e. would not produce a focussed image on the retina; Martin and Brooke, 1991). Hence, the young of burrow-nesting shearwaters, storm-petrels, and puffins appear particularly vulnerable to grounding in well-lit areas on their fledging flights from the colony (Atchoi et al., 2020), whereas the young of closely related surface-nesting species, such as Fulmars, are not vulnerable to light-induced grounding. While numerous studies have shown that light-induced grounding is much more prevalent among recently-fledged juveniles, the timing of some grounding events of Leach's Storm-petrels on offshore oil platforms (in April–August before any young of the year have fledged; Gjerdrum et al., 2021), and the stranding of likely breeding Leach's Storm-petrels on a docked seismic vessel (Wilhelm et al., 2021), show that adults may be light-attracted on occasion too. Collins et al. (2022) found no impact of oil platforms on the behaviour of breeding Leach's Storm-petrels GPS-tracked in Newfoundland, but only 1.1% of trips involved exposure to oil platforms at night and around 30% of tracked birds were not recaptured, so their fate is unknown. While juveniles are clearly more susceptible than adults to light-induced grounding, it is not clear for how long post-fledging such susceptibility persists, and whether birds grounded weeks or months after fledging were forced to land by severe weather (e.g. Teixeira, 1987) rather than light attraction.

Petrels and shearwaters are more likely to be disorientated by artificial light under conditions of low ambient light (i.e. a new moon), and during conditions of fog, mist or light rain. Guilford et al. (2019) showed experimentally that, during foggy conditions but not clear nights, light emanating from windows resulted in disorientation of adult Manx Shearwaters, causing them to collide with the building. They suggested that when the birds were suddenly close to a relatively bright light, the light-scatter caused by fog compromised the birds' dark-adapted visual guidance. Alternatively, they suggested that Manx Shearwaters may use a light-dependent magneto-receptor, located in the eyes, for navigation (Hore and Mouritsen, 2016), which could become temporarily disrupted by saturation in the presence of bright light. However, several experimental studies have failed to find evidence for the existence of such a magnetic compass in either adult (Padget, 2017) or fledgling (Syposz, 2020) shearwaters, and the sensory basis of navigation in Procellariiformes remains unclear.

Experimental reduction in artificial lighting (outside lights turned off and the majority of windows shielded with blinds) in the village of Hirta, St Kilda resulted in fewer grounded fledgling Leach's Storm-petrels, but the number of grounded Manx Shearwaters remained high (Miles et al., 2010). The authors concluded that Manx Shearwaters may be more vulnerable than storm-petrels to disorientation, or that they also navigate towards low-frequency sounds, since many grounded individuals were located close to generators or extractor fans, one being found impaled in the outlet duct of an extractor. Potential attraction of shearwaters to low-frequency noise, and implications for attraction to wind turbines and associated structures and vessels, requires further consideration.

5.4.2 Evidence for light attraction

While there is clear evidence for the disorientation of burrow-nesting Procellariiformes by artificial light sources, the extent of long-range attraction is more difficult to quantify. There are reports of European Storm-petrels being attracted to garden fireworks and moth traps (Miles et al., 2010), which they are unlikely to have been overflying, and suggests they were attracted by the artificial illumination. The number of individuals recovered in campaigns to rescue grounded fledglings are typically very low in relation to the local population size (e.g. Miles et al., 2010, but see Le Corre et al., 2002, Rodríguez et al., 2015b, Rodríguez et al., 2022), suggesting that birds are not attracted over large distances, or if so, only a small proportion of individuals are affected, or recovered. For example, the number of fledgling Manx Shearwaters recovered in the town of Mallaig, Scotland (Syposz et al., 2018), broadly corresponds, given the size and distance of the colony that is the likely source of the majority of individuals (Rum, 27 km away), with the number predicted if birds disperse randomly in all directions and the small proportion that orientate towards Mallaig are then attracted from very short range.

Two cases where large numbers of fledglings, representing large proportions (up to 40%) of the local population, are encountered grounded in brightly illuminated urban areas are Barau's Petrels *Pterodroma baraui* on Reunion Island, Indian Ocean (Le Corre et al., 2002) and Cory's Shearwaters on Tenerife (Rodríguez et al., 2015b, Rodríguez et al., 2022). The grounding of large proportions of the cohort of fledglings may imply that birds are attracted from large distances. In both cases, nesting sites are mainly located in high altitude areas in the island interior, and fledglings fly over brightly lit coastal areas (some more than 10km distant from the nearest colonies) to reach the sea. When flying over these areas birds become vulnerable to disorientation from powerful light sources below them. The sensitivity of birds to disorientation when overflying powerful light sources projected upwards is evidenced from the disorientation of very large numbers of nocturnal migrants by ceilometers (bright lights shone vertically to measure the height of the cloud base; Rich and Longcore, 2006), and the effectiveness of spotlights directed upwards to ground and capture storm-petrels returning to the colony at night (Ishmar et al., 2015). Whilst the minimum distance between colony locations and some grounding sites is more than 10km in these studies, the distance from which birds are attracted by light may be considerably less. Since birds may overfly these coastal areas on route to the sea, the high disorientation sensitivity of procellariiform seabirds to light sources from below could potentially account for the high rate of grounding in the case of Reunion Island and Tenerife, without birds being attracted from large range.

Several recent, and highly innovative, studies have started to assess the behaviour of fledgling Procellariifomes in response to artificial light. The first (Troy et al., 2013) modelled the numbers of Newell's Shearwaters *Puffinus newelli* recovered in different sectors of Kauai Island, Hawaii, in relation to location and size of colonies, light radiance levels across the island, and models of fledgling movement. They concluded that the observed spatial pattern of groundings indicated that fledglings were attracted back to the island by coastal illumination after they had reached the sea, and from distances of up to 10 km from the coastline. These modelled estimates of attraction range receive empirical support from two studies (Rodríguez et al., 2015b, Rodríguez et al., 2022) that tracked fledgling Cory's Shearwaters as they overflew brightly lit coastal areas in Tenerife on their flights to the sea. Both studies were conducted over multiple years, and each found that c. 14% of fledglings were later recovered grounded. Although neither study attempted to estimate the distance from which fledglings may become attracted towards artificial light, inspection of the tracks suggests that abrupt course deviations towards lit areas could occur from a range of several kilometres. All birds recovered by Rodríguez et al. (2015b) were grounded within 16 km of their breeding colonies, and 50% were found within 3 km of their nest site. Once above brightly lit areas many birds showed highly tortuous flight paths, circling to remain within the lit areas, before descending to ground level, as illustrated here. On multiple occasions birds that had reached the sea, and were up to 2.5 km from land, returned to brightly lit areas on the coast.

On St Kilda, considerable numbers of Leach's and European Storm-petrels breed within 2 km and in direct line of sight of the village illuminations, but the number of grounded fledglings is very small in relation to the size of the breeding populations, representing <<1% of the number of young likely to fledge annually (Miles et al., 2010). If the number of fledglings encountered grounded is an accurate reflection of the numbers attracted and disorientated, these findings suggest that fledglings are not susceptible to attraction to these light sources from long range, although the level of illumination in the village was relatively low (32 outside lights and 11 buildings with indoor lighting; Miles et al., 2010). In contrast, the vast majority of Manx Shearwaters breeding on St Kilda do not fledge in sight of the village and would not pass within sight on a direct route to the sea, raising the likelihood that they are attracted to illumination after having reached the sea, and may be attracted from a considerable range (>2 km) to illuminated areas. Similar differences in the numbers of storm-petrels and shearwaters encountered grounded in Hawaii and the Canary Islands have led other authors to suggest that the larger species of Procellariiformes may be more vulnerable to light attraction (Telfer et al., 1987, Rodríguez and Rodríguez, 2009). Any such conclusions may be premature however, since the smaller size and largely dark plumage of storm-petrels may result in lower detection rates during searches for grounded birds and storm-petrels may be able to take flight after grounding in enclosed situations more readily than shearwaters, which require an open space in which to take a “run up” to become airborne. Due to their smaller size, storm-petrels are also more likely to be depredated (e.g. by cats and dogs) and removed (Wilhelm et al., 2021). The susceptibility of storm-petrels to light-attraction and disorientation may be higher than implied by the numbers of individuals encountered grounded.

5.4.3 Attraction to vessels

In addition to attraction to or disorientation by lights at ports and on turbines, the potential for interaction of Procellariiformes with wind farm service vessels should also be considered. There are many anecdotal accounts of nocturnal seabirds, especially storm-petrel species, alighting on ships at night. For example, Wakefield (2018) reports that on several occasions during a research cruise by RRS Discovery to the mid-Atlantic, Leach’s Storm-petrels were found on the ship’s decks at night and caught by hand. These groundings usually occurred in misty conditions and were likely caused by birds being attracted to or disorientated by the deck’s flood lights. Of 1,823 seabirds (all burrow-nesting Procellariiformes) recorded on board rock lobster fishing vessels around the Tristan da Cunha archipelago and Gough Island between 2013 and 2021, 4% died after being attracted to/disorientated by artificial lights (Ryan et al., 2021). As discussed above, it is not clear to what extent the grounding of storm-petrels on vessels results from macro- or meso-scale light attraction, or whether they are attracted to vessels by other cues (such as olfaction, low frequency sounds, or visual cues associated with a food source). Storm-petrels are known to follow a wide range of vessels, probably in search of food brought to the surface by the wake or vessel lighting or, in the case of fishing vessels, for offal. They can also be attracted to stationary vessels if any oily waste is released. In calm conditions European Storm-petrels may be attracted from distances of >1 km (M. Bolton pers. obs.) and may aggregate in large numbers. In the context of use of vessels for service operations for wind turbines, nocturnally active Procellariiformes (especially storm-petrels) are sensitive to attraction (by phototaxis, olfaction, or visual cues associated with food sources), and may subsequently become disorientated, either by lighting associated with the vessel, or navigation lights on nearby turbines.

5.4.4 Implications of the capabilities and sensitivities of the visual system of petrels and shearwaters for light disorientation/attraction

Petrels and shearwaters have been a particular focus for studies of avian vision for many decades (Lockie, 1952, Hayes and Brooke, 1990, Martin and Brooke, 1991) due to the species’ need for visual capabilities to fly and forage under a wide range of light intensities, and in air and water, where the refractive properties of light differ. As a result, a considerable amount of detailed information exists on the microscopic and optical structure, and the visual fields, of the eyes of Manx Shearwaters, Fulmars and storm-petrels (Mitkus et al., 2016), which can inform our understanding of their behaviour in the vicinity of lit structures at sea (Atchoi et al., 2020). In brief, the retinas of Manx Shearwater, Fulmar and Leach’s Storm-petrel all possess a central region (variously termed “*Area centralis*” (Lockie, 1952), “horizontal strip” (Hayes and Brooke, 1990), “visual streak” (Mitkus et al., 2016)), which receives light input from the horizon when the bird’s head is normally orientated. The central part of this region is equipped entirely with cones—photoreceptors that operate under high light intensities (i.e. daylight) that are capable of colour vision and are responsible for high spatial acuity. This horizontal central structure is found in a range of seabirds and other species inhabiting open landscapes and provides high acuity to detect objects at, or close to, the horizon in well-lit environments. The outer margin of the central horizontal strip is equipped with rods—photoreceptors that operate under low light conditions—which have low spatial acuity. The density of rods increases from the central strip to the periphery of the retina (Lockie, 1952). The density of rods in the peripheral retina (which receives light from above and below the horizon when the head is normally orientated) is two-fold greater in the Manx Shearwater than Fulmar, and four times greater than the House Sparrow *Passer domesticus*, which is not active at night.

Martin and Brooke (1991) measured the visual field of the eyes of the Manx Shearwater and found that the eyes are directed slightly forwards and downwards when the head is normally orientated, with a blind spot above and behind the crown. In normal flight the eyes will therefore receive greater light input from in front and below the bird than from above and behind. During daylight, when the pupil is contracted to restrict the amount of light entering the eye, light falls on the centre of the retina, and objects on or close to the horizon are rendered with high spatial acuity, while objects further from the horizon are rendered with lower acuity. In low light levels at night the pupil opens to allow more light to enter and this is detected by the high density of rods located towards the periphery of the retina. Thus, the optic system of shearwaters and petrels provides high acuity for objects close to the horizon during daylight, and high sensitivity (though low acuity) to low light levels at night. Disorientation of shearwater fledglings when overflying brightly lit areas may result from saturation of the visual pigments of the rods (Verheijen, 1985), which cannot be adequately rectified by contraction of the pupil to limit entry of light to the eye. Birds are in effect blinded and can no longer see visual details that they could detect when dark-adapted. Alternatively, bright light may cause contraction of the pupil, so little light falls on the peripheral rods, and the birds are unable to discern poorly lit objects beyond the brightly lit areas, and so circle to remain within the illuminated field.

5.4.5 Influence of light wavelength on visual perception of shearwaters and storm-petrels

Manx Shearwaters are known to forage at depths of up 55 m (Shoji et al., 2016). Since light of shorter wavelengths (blue) penetrates water to greater depths than that of longer wavelengths (red), to maximise acuity when foraging at depth it is likely that the cones of Manx Shearwaters have greater sensitivity to blue than red light. Since storm-petrels dive to a very limited degree (max 5 m; Albores-Barajas et al., 2011), they have less need for enhanced sensitivity to blue light.

Experiments to examine the response of adult Manx Shearwaters in flight over the colony to different intensities and wavelengths of light showed that birds were more responsive to (avoided) bright white than dim white light and showed greater avoidance of blue and green light than red light (Syposz et al., 2021a). There was no difference in the birds' behaviour when exposed to red light compared to no light. These results indicate that Manx Shearwaters have greater sensitivity to light of shorter wavelengths (blue and green) than long (red).

These findings appear to contrast with a number of largely observational (not experimental) studies that have examined the effect of light wavelength and pattern of illumination (constant vs flashing) on the collision rate of nocturnal migrants (principally passerines) with communication masts and onshore wind turbines in North America. These studies have compared the flight paths and/or number of birds found dead under structures with different types of illumination and may suffer from uncontrolled bias. However, they broadly indicate that flashing red lights causes less attraction and collisions than steady constant red light (Gehring et al., 2009, Kerlinger et al., 2010), and whilst constant red light caused greater attraction than flashing white light (Gauthreaux and Belser, 2006), Gehring et al. (2009) found no difference in the number of collisions at masts with flashing red or flashing white light. It has been suggested that red light may interfere with magnetoreception in migrating passerine birds: three passerine species showed normal orientation under dim monochromatic light from the blue-green range of the spectrum, while they were disoriented under yellow and red light (Wiltschko and Wiltschko, 2002). Gauthreaux and Belser (2006) recommend the use of flashing white lights in place of steady red lights to reduce the risk of collision of nocturnal (mainly passerine) migrants with communication masts in USA.

Several studies have failed to find evidence of magneto-reception in shearwaters (Padget, 2017, Syposz et al., 2021b) and it is possible that differences in the sensory systems used for navigation in nocturnal Procellariiformes and passerines may result in important differences in their sensitivities to attraction/disorientation by light of particular wavelengths. Several hundred million migrant birds cross the North Sea annually, at risk of collision with wind turbines (Hüppop et al., 2006), and the benefits of a particular lighting regime to reduce collisions of nocturnal Procellariiformes, such as the use of red navigation lights, must be weighed against likely impacts on other species.

5.4.6 Non-collision consequences of light attraction of seabirds that may affect their survival and productivity

If light-induced disorientation leads to individual birds circling the navigation lights on the nacelle or tower of turbines for protracted periods (as has been reported for birds disorientated by lighthouses or gas flares) the probability of collision with turbine blades or other surfaces is vastly increased, and may approach unity. However, individuals that are attracted to and disorientated by light associated with wind farms may become vulnerable to other lethal and sub-lethal impacts. If wind farms provide roosting opportunities for large gulls, or other predatory species (skuas, falcons), storm-petrels and Manx Shearwaters are likely to be vulnerable to predation (Hey at al. 2020), particularly if wind farm illuminations provide sufficient ambient light for effective hunting by these predators (Watanuki, 1986). Sub-lethal affects that may influence survival in the longer term, or the ability to rear young, could accrue from the wasteful expenditure of energy in circling flight for protracted periods. This may lead to loss of body condition resulting in birds becoming more vulnerable to starvation or predation. Flight costs of European Storm-petrels have been estimated at 3.9 times basal metabolic rate (Bolton, 1995a), close to the maximum sustainable work rate (Drent and Daan, 1980). Prolonged periods of flight, without opportunity to feed or rest, may lead to dehydration or exhaustion of birds that escape collision. Conversely, many fisheries use artificial light to attract prey and there is a possibility that birds could benefit from increased foraging opportunities if artificial lighting around wind farm developments increases prey availability by attracting it close to the sea surface. The evidence base around

Procellariiformes exploiting prey resources concentrated near the surface by artificial light is limited, but European Storm-petrels have been observed foraging around illuminated fish farms at night in the Faroe Islands (B. Porter, pers. comm.).

5.5 Options for mitigation

The second of the two expert workshops held as part of this project focussed on mitigation options to reduce the impacts on Procellariiformes of offshore wind farm developments and associated activities and infrastructure. Table 4 summarises the mitigation options discussed at the workshop and in the published literature. Full reports of both workshops are provided in Appendix 1. NatureScot (2020) have suggested several potential mitigation options for reducing the impacts on birds of lights placed on wind farms for the purposes of aviation safety. These mitigation options do not relate specifically to offshore wind farms or Procellariiformes but we include them in Table 4.

Table 4. Suggested mitigation options for reducing the impacts on Procellariiformes of offshore wind farm developments and associated activities and infrastructure, collated from the published literature and discussions during the two expert workshops held as part of this project.

Option	**Evidence base**	**Comments**	**Technical/legislative feasibility**
Alter pattern of illumination (flashing rather than steady lights)	Good evidence from numerous studies in USA that flashing lights cause less attraction/collision of migrant nocturnal passerines.	Not systematically tested for Procellariiformes. Bardsey lighthouse changed to a red flashing light in 2014 and this resulted in a huge reduction in collisions of Manx Shearwaters.	Need consistency in lighting across wind farms to avoid confusion to mariners and to comply with international standards, which precludes modification. Even apparently simple changes in lighting require intervention at early stage of turbine design/construction
Alter wavelength of lights	Studies conducted primarily on passerines provide little empirical evidence that white light causes less attraction/collision than red light (white light contains red). Green may be much better than white.	Experiments conducted on Manx Shearwater showed greater **avoidance** of white, blue and green than of red light. Not clear what the attraction properties of red vs white light are for Procellariiformes. Most vertebrate rods are maximally sensitive to green wavelengths and whether particular species are attracted to or repelled by green light would require specific behavioural	Need consistency in lighting across wind farms to avoid confusion to mariners and to comply with international standards, which precludes modification. Even apparently simple changes in lighting require intervention at early stage of turbine design/construction Search and rescue (SAR) lights need to be red to avoid reducing the night vision of crew.

		studies. Green light should only be used if it is highly directed.	
Directional intensity / shielding of lights	Some suggestion in the literature that birds are most sensitive to attraction of light from below. Fitting of shields to prevent upwards light radiation at a coastal resort in Hawaii reduced the number of grounded Newell's shearwaters by 40% over 2 seasons (Reed et al., 1985)	Birds may also be attracted upwards towards light, as is likely the case for storm-petrels stranded on offshore oil and gas platforms, which tend to be several tens of metres above the sea surface.	Already set out in ICAO requirements and EASA CS-ADR-DSN Chapter Q. This focusses the 2000 cd lighting in the horizontal plane and reduces the intensity of the light from above and below. Both regulations stipulate minimum requirements as well as additional recommended vertical angles, which cannot be ignored without justification. Most lights will incorporate this as standard. Marine lighting is also focused on the horizontal plane but needs to remain visible to all sizes of vessels both close to turbines and at the extreme range of the light.
Reduce intensity of lights	The effectiveness for reducing bird collisions is unknown, but likely to reduce the range from which any "attraction" might occur.	Not enough evidence on the impact this would have on different seabird species. Intensity more important than colour in bird night vision. Impact of different intensities	Already set out in CAA guidance CAP 764. Lights can be dimmed to 200 cd in good visibility (greater than 5km). 200 cd lights can still be visible to the human eye > 20 km in good visibility conditions.

		depends on atmospheric conditions. Any conditions creating large, diffuse pools of light likely to be a problem.	
Reduce number of turbines illuminated	Dependent on the range at which any “attraction” of birds to light might occur, the reduction in the number of turbines illuminated is likely to reduce the number of individual birds brought into the proximity of turbines		If the number of turbines lit is reduced, the intensity of lighting may have to increase to compensate.
Reduce or cover lighting associated with maintenance vessels and associated activities and infrastructure (e.g. ports, wet storage)	Reduction of vessel lighting and the use of blinds has successfully reduced the number of collisions of burrow-nesting Procellariiformes with fishing boats (Ryan et al., 2021).		Blinds for vessels should be easy to implement, but changes to safety lighting are likely to be more difficult.
No lighting, or turning off lighting at key times (e.g. fledging period)	There is good evidence for light-induced disorientation (i.e. circling) of Procellariiformes (especially storm-petrels), so elimination of lighting is likely to reduce the number of occasions an individual passes through the rotor-swept area, on a flight past a turbine.	Lack of lighting may result in collisions by birds that cannot see the turbines on nights with particularly low ambient light.	Not possible for offshore wind farms due to safety concerns. Should not be considered as a mitigation option.

Radar-activated lighting		Reduction in collisions will depend on the proportion of time turbines are left unilluminated, during periods when light-induced collisions would otherwise occur.	CAA support this in principle and are considering the parameters in detail. In the meantime, CAA are happy to discuss the approach on a case-by-case basis. In use in other countries, to differing extents, but it is acknowledged that the costs are high. Detection systems are not currently possible for all marine vessels, especially ill-equipped recreational vessels, and lighting provision must cater for all users.
Additional lighting to guide birds away from wind farms	Currently unclear whether this would be effective. May result in further attraction / disorientation / displacement of target birds.	Would need to consider wider impacts on species other than Procellariiformes.	Additional lighting may be more feasible than reduced lighting.
Shut down turbines during meteorological conditions likely to result in high collision rate	Collision risk is reduced if turbines are not rotating.	Since conditions that generate high collision rate are usually associated with lower wind speed, little economic impact on electricity generation?	Unlikely to be acceptable given the importance of offshore wind for future UK energy production.
Increase minimum blade height	May help to reduce collisions at times/in conditions when birds are flying higher (e.g. Manx Shearwaters fly higher in stronger winds).	Has benefits outside of mitigation for birds.	Requires feasibility assessment on a case-by-case basis.

Increase detectability by marking blades / towers	Maximising visibility of blades is likely to reduce the number of collisions as birds would be better able to avoid them.		There needs to be consistency across wind farms to avoid confusion to mariners and to comply with international standards.
Deter birds (seabirds and / or avian predators) using sound	Currently unclear whether deterrence using sound would be effective.	If birds could be deterred by sounds outside of human hearing range this would avoid interference with regulation sounds used for maritime safety.	There needs to be consistency across wind farms in their use of fog horns.
Train crew in safe handling / release of stranded birds	Would not prevent collisions but may reduce mortality of grounded / stranded birds.	Posters at harbours in Pembrokeshire, Wales, provide guidance for mariners in case of Manx Shearwaters stranding on their vessels. Similar schemes have been implemented in other countries for other seabird species.	Has been done elsewhere and could be relatively cheap to implement. Could be built into relevant consenting conditions.

6 Remaining evidence needs to inform assessment

The information below was compiled based on the literature review and workshop discussions. While some data are available for many of the parameters relevant to assessing the impacts of offshore wind development, including from studies in Scotland (see 'Catalogue of data sources'), there are some key evidence gaps remaining. While data are incomplete for all three species considered here, information for Leach's Storm-petrel in Scotland is generally more limited than for Manx Shearwater and European Storm-petrel. The order of knowledge gaps presented here and in section 8 is based on Table 6, which follows the trajectory for assessments of offshore wind farm impacts. Note that the Offshore Wind Strategic Monitoring and Research Forum (OWSMRF) is currently conducting a detailed review of the knowledge gaps and research recommendations relating to parameters required for PVA for Manx Shearwater and European Storm-petrel.

6.1 Detectability and diel variation in marine distributions

The use of Digital Aerial Surveys (DAS) is currently the most commonly recommended method for providing the baseline characterisation for wind farm assessments. There are several advantages to DAS, such as providing an audit trail and potentially causing less behavioural change in seabirds than vessel-based surveys. However, there remain a number of potential biases and these could be exacerbated by the behaviour and morphology of some procellariform species. These issues arise from detectability and identification and have not formed part of the main body of the current review as there has been scant work published that examines them. The issues with detectability are twofold: whether the size and flight characteristics of the species make them harder to detect and whether the nocturnal and crepuscular nature of some of the at-sea behaviours means that they are not captured by the survey flights that are restricted to certain daylight hours. All the procellariiform species covered by this review can be active throughout the day and night and with different levels of activity at different times. For example, for Manx Shearwater tracked from Skomer, diving occurred during the day and peaked in the evening (Shoji et al., 2016), while nocturnal foraging was observed from tracking of birds from High Island, Ireland (Kane et al., 2020). These diel variations in activity may mean that key activity periods are not picked up by the constrained timings of DAS. Even where birds have been detected it remains unclear whether morphologically similar species such as European and Wilson's Storm-petrel can be successfully identified to species level. Full consideration should be given to both detectability and species identification in relevant impact assessments. These aspects of DAS for marine ornithology surveys are currently being considered as part of a review being carried out by NatureScot's Scientific Advisory Committee.

6.2 Basic morphometric data

The basic morphometric data used in collision risk modelling (i.e. body length, wingspan) may vary geographically and is lacking for birds of all three species in Scotland. However, while Scotland-specific data would be useful, differences from birds outside of Scotland are unlikely to be large.

6.3 Flight data

The flight speed data currently available come from relatively coarse-resolution tracking data and will tend to be underestimates of the true travel speeds of birds, and measured flight height data are limited or non-existent for these species. There is a need to understand the extent of flight activity and flight heights in different weather conditions, at different times of day and whether flight heights change in response to turbines. While there has been some work on diurnal activity patterns for Manx Shearwater and European Storm-petrel, information for Leach's Storm-petrel is lacking. Flight heights when arriving at or departing/fledging from high elevation nesting sites are also unknown. Fledglings may have particularly poor flight control in the first few days after fledging, making them more vulnerable, but our knowledge of fledgling behaviour is poor.

6.4 Avoidance/attraction behaviour

Since currently operational wind farms overlap very little with the distributions of Manx Shearwaters, European Storm-petrels or Leach's Storm-petrels, very little is known about their avoidance or attraction in relation to offshore turbines, support vessels and associated lighting, infrastructure and activities. This means that many assessments of the impacts of such developments are based largely on expert opinion rather than empirical data. The ability of these species to detect rotating turbine blades is unknown but could inform mitigation options. Whether or not sound influences avoidance or attraction behaviour is also unknown. There are some data available on the energetic requirements of adults of all species to inform assessment of impacts of displacement (resulting from avoidance), but the review found no data in the energy requirements of chicks of European Storm-petrels or Fulmars.

6.5 Light attraction/disorientation

Critical knowledge gaps relate to light attraction and disorientation. Specific aspects include: the range over which light attraction of nocturnal Procellariiformes may occur (and therefore the size of the light catch basin for wind farms and related activities or infrastructure); the extent to which light attraction is exacerbated by particular meteorological conditions (e.g. fog, rain); the influence of wavelength and pattern of illumination (flashing/steady); the extent to which light attraction differentially affects adults and juveniles, and for how long after fledging juveniles may remain particularly susceptible to light attraction.

6.6 Diet

The level of impact caused by displacement of seabirds from foraging areas is related to the degree of dietary specialisation and the distribution of food resources. Food availability within wind farm developments may also influence the degree to which birds are attracted to the area. There have been very few studies of the diet of these procellariiform species, especially within Scotland, which means that prey distributions, and how prey distributions may change around wind farm developments, are poorly understood. A small amount of metabarcoding of Manx Shearwater diet samples from colonies in Wales has been conducted by the University of Oxford (K. Davies, pers. comm.), and similar work has been carried out at Cardiff University for European Storm-petrels from Mousa (Z. Deakin, pers. comm.), but the results of these studies are not yet published.

6.7 Apportioning impacts to protected colonies

Understanding the connectivity between specific offshore developments and SPA colonies is essential for apportioning impacts to colonies. While long-term data on distributions has been collected by vessel-based surveys, these data do not provide information on the provenance or age of the birds observed, and therefore the connectivity between SPA populations and Plan Option Areas is generally not known. Manx Shearwaters have been tracked extensively from colonies in Wales and Northern Ireland, but limited tracking data are available for Scottish colonies of all three species, and the marine habitat associations of these species in Scotland are therefore poorly understood. It is important to note that birds from colonies outside of Scotland (i.e. in Wales and Ireland) also use Scottish waters and need to be considered in apportioning assessments.

6.8 Evaluation of remaining evidence needs

The evidence needs relating to the key factors involved in the trajectory of assessment of impacts of offshore windfarms on protected colonies of procellariform seabirds are summarised and evaluated in Table 6.

The assessment trajectory commences with quantifying the baseline marine densities of the species of interest and concludes with a Population Viability Assessment for protected colonies, considering both collision and displacement impact pathways. Firstly, the key factors for each stage of the assessment trajectory have been scored ("medium" or "high") in terms of their relative importance within their respective assessment process. In the absence of a formal sensitivity analysis of all factors, we have scored those that may be considered to have an approximately linear effect on the outcome of their respective assessment process as having "medium" importance (e.g. the effect of body length, wingspan or flight speed on collision risk), and those which act in a non-linear manner (i.e. through the existence of a threshold or power relationship, such as flight height) as "high" importance. No factors were considered to have "low" importance. For each species we scored the level of uncertainty surrounding the estimation of each factor as "low", "medium" or "high", based on the availability of evidence identified in the literature review. The "evidence need" for each factor was then scored on the basis of both the importance of the factor, and the level of current uncertainty as shown in Table 5 below.

Table 5. Ranking of evidence needs of key factors in assessment of offshore windfarms on seabirds on the basis of their respective importance and uncertainty.

Importance	Uncertainty	Evidence need
Low	Low	Low
Low	Medium	Low
Low	High	Medium
Medium	Low	Low
Medium	Medium	Medium
Medium	High	High
High	Low	Medium
High	Medium	High
High	High	High

Finally, the tractability of conducting new research to fill the current evidence gap was assessed using expert judgement as “low”, “medium” or “high”, taking into consideration factors such as: (i) whether methodologies currently exist; (ii) have been widely used on these (or similar) species elsewhere, or (iii) whether further technological development would be required. Note that costs were not considered in the tractability factor.

Table 6. Summary of evidence needs of key factors involved in assessment of impacts of offshore windfarms on petrels and shearwaters in Scotland.

Assessment Trajectory	Key factors for impact assessment	Importance	Manx Shearwater			European Storm-petrel			Leach's Storm-petrel		
			Uncertainty	Evidence need	Tractability	Uncertainty	Evidence need	Tractability	Uncertainty	Evidence need	Tractability
Baseline marine density	Diel activity	Medium	Medium	Medium	High	Medium	Medium	High	Medium	Medium	High
	Detectability	High	Low	Medium	Medium	High	High	Medium	High	High	Medium
Collision	Bird morphology	Medium	Low	Low	High	Low	Low	High	Low	Low	High
	Flight speed	Medium	Medium	Medium	High	High	High	High	High	High	High
	Flight height	High	High	High	High	High	High	Low	High	High	Medium
	Avoidance behaviour	Medium	High	High	High	High	High	High	High	High	High
	Nocturnal activity	Medium	Medium	Medium	High	Medium	Medium	High	Medium	Medium	High
	Light attraction	High	Medium	High	Medium	High	High	Medium	High	High	Medium
Displacement	Avoidance behaviour	Medium	High	High	High	High	High	High	High	High	High
	Light attraction	High	Medium	High	Medium	High	High	Medium	High	High	Medium
	Vessel response	Medium	Medium	Medium	High	Medium	Medium	High	Medium	Medium	High
	Energetics	Medium	Medium	Medium	High	Medium	Medium	High	Medium	Medium	High
Apportioning	Colony size	Medium	Low	Low	High	Low	Low	High	Medium	Medium	High
	Colony location	High	Low	Medium	High	Low	Medium	High	Low	Medium	High
	Foraging ranges	High	Medium	High	High	High	High	High	High	High	High
	Connectivity	High	Medium	High	High	High	High	High	High	High	High
PVA*	Colony Size	Medium	Low	Low	High	Low	Low	High	Medium	Medium	High
	Demographic rates	Medium	Medium	Medium	High	Medium	Medium	High	High	High	High

*Note that OWSMRF is currently conducting a detailed review of the knowledge gaps relating to parameters required for PVA for Manx Shearwater and European Storm-petrel.

7 Examination of challenges and recommendations for filling data gaps to assess the impacts of offshore windfarms

The suggestions below are based on the literature review and workshop discussions with respect to evidence gaps and approaches to filling these for Manx Shearwater, European Storm-petrel and Leach’s Storm-petrel. Note OWSMRF is currently conducting a detailed review of the knowledge gaps and research recommendations relating to parameters required for PVA for Manx Shearwater and European Storm-petrel.

7.1 Detectability and diel variation in marine distributions

There is an important need for experimental validation of potential biases in aerial survey methods, including detectability, identification and diel variation. Detectability could be tested by carrying out targeted digital aerial surveys (DAS) or vessel-based surveys with an experimental approach. For example, decoy models of birds could be used to assess detectability under different conditions, but only for birds drifting on the sea surface. Surveys could also be performed alongside large scale high-resolution tracking of birds, or decoys, but achieving large enough sample sizes of tagged birds is likely to be difficult. Radar or thermal imaging could be used to conduct nocturnal surveys of leased areas, but identification to species level may not be possible. Evidence needs with respect to detectability are greatest for the two storm-petrel species and for diel activity the highest priority is Leach’s Storm-petrel (Table 5).

7.2 Basic morphometric data

Body length and wingspan measurements would be relatively easily collected by ringers or fieldworkers working with Procellariiformes in Scotland. However, while these data could readily be collected, the evidence need is low for all three species (Table 5).

7.3 Flight data

The evidence needs for flight speeds and heights are high for all three species, with the exception of medium scoring for Manx Shearwater flight height (Table 5). Estimates of flight parameters such as speed and height can be gained from tracking data, but acquiring accurate estimates is difficult, even with high resolution data. Where possible, “instantaneous” flight speeds from GPS tags, based on Doppler-shift information derived from the movement of the tag relative to the movement of the satellites (Safi et al., 2013), will be more accurate than that derived from distance covered between successive fixes. Tags providing high resolution tracking data are available for Manx Shearwaters but the accuracy of flight height data from high resolution GPS tracking of this species is still low. Small (< 1.5 g) barometric pressure loggers can be used to estimate flight heights when deployed alongside GPS devices , but because of the need to calibrate to local environmental pressure, accuracy may sometimes be low. Since tags deployed on storm-petrels must be much smaller than those used on Manx Shearwaters, the limitations on battery life and data storage capacity mean that GPS data collected for storm-petrel foraging trips tends to be of lower resolution. However, it would be possible to collect higher resolution data for short periods of storm-petrel foraging trips. Tracking of fledglings is challenging due to the difficulty of retrieving tags for data download.

Flight height can be measured with radar, although these measurements have biases. Distinguishing between similar species (e.g. Manx vs Balearic Shearwater) is difficult, but mobile radar units could be deployed in areas where only a single species is expected (e.g. Rum for Manx Shearwaters). It may be possible to estimate flight heights from vessel-, or turbine-mounted cameras or from aerial or thermal imagery, but again, accuracy is likely to be low. Flight height can also be accurately measured using laser rangefinders (Largey et al., 2021).

It is important to note that flight within wind farm developments may differ from that elsewhere so work within wind farms is important, but currently limited for these species by the lack of overlap between their marine distributions and operational wind farms.

7.4 Avoidance/attraction behaviour

Assessment of macro-avoidance of windfarm development is best achieved by comparing marine distributions of seabird pre- and -post construction. In light of the limited tracking of the three focal species to date in Scotland, we recommend further tracking studies from key colonies to better understand the pre-construction movements and distribution of these species. Such tracking studies should continue as construction occurs and after it is completed, to inform understanding of meso- and micro-avoidance behaviour.

Currently there is little known overlap between operational offshore wind farms and shearwater and storm-petrel marine distributions, so there is limited scope for collecting data on the species' micro, meso and macro avoidance behaviour within and around wind farms, although the evidence need is high (Table 5). There is some overlap between Manx Shearwater distributions and wind farms in the Irish Sea/Solway Firth, and this could be an area in which to focus initial studies, although the number of birds moving close to/within wind farms may be small.

If a suitable site was available, GPS tracking birds could reveal macro- and meso-scale avoidance of wind farms. VHF receivers could be placed on turbines or other infrastructure, as has been done on oil and gas structures elsewhere, but obtaining sufficient sample sizes of VHF-tracked birds would be challenging. Radar can be used to quantify flight lines without the need for tagging birds, and changes to flight lines would provide evidence of avoidance or attraction. Portable radar devices are available, with a detection distance of 72 nautical miles.

Tracking or visual observations could be used to assess avoidance or attraction behaviour in relation to sound. Experiments with sound could be conducted using similar methods to those suggested for light attraction experiments, below.

Whilst no studies have been conducted to date on the energy requirements of chicks of European Storm-petrels or Fulmars, from which to inform assessment of the consequences on productivity of displacement of breeding adults from feeding areas, well—established methods are available and such studies would be feasible.

7.5 Light attraction/disorientation

The evidence need around light attraction/disorientation is high for all three species but there are challenges to addressing the knowledge gaps (Table 5). To address the current knowledge gaps regarding the spatial scale, age classes affected, environmental drivers, and influence of light characteristics on light attraction of nocturnal Procellariiformes, we recommend a series of experiments are conducted, for both Manx Shearwaters and storm-petrels, to examine the behaviour of both adults and fledglings at varying ranges from experimentally manipulated light sources. Such experiments will be logistically challenging to perform, but given magnitude of the current knowledge void, and the impact of potential light attraction on the estimates of collision rate, these studies could be considered a high priority.

Possible approaches could include use of thermal video equipment to record flight paths of adults attending the colony, and fledglings leaving the colony, in response to lights of differing wavelength, intensity and distance from the colony, under differing levels of ambient light, and visibility (i.e. foggy/clear). See Gauthreaux and Belser (2006) for an example of tracking flight paths of nocturnal migrants in relation to illuminated communications towers in USA. Monitoring for flight paths could be supplemented by targeted tracking of adults using GPS tags, and fledglings using coded VHF (MOTUS) tags. Tracking fledglings as they leave the burrow is challenging (see papers by Rodriguez et al. (2015b, 2022) for an account of the difficulties), but use of VHF tags, and a suitable array of detection stations, would overcome the difficulties of tag life and detection frequency.

Such fieldwork would require the erection of lights in view of a breeding colony, but ideally as close to the sea as possible. Lunga, Treshnish Isles might offer a suitable location where several small, low-lying skerries are situated between 700 m and 1500 m from nesting areas of European Storm-petrels and Manx Shearwaters. Such skerries would provide a suitable platform for installing lights, which could be varied in an experimental manner to systematically assess the attraction of light of differing wavelength, intensity, splay, pulse frequency etc. St Kilda may provide a suitable location for these studies on Leach’s Storm-petrel and Manx Shearwater, locating test lights on the coast opposite the breeding colony on Dùn, which currently hosts about 6,000 pairs of Leach’s Storm-petrel. Mousa, Shetland would provide a logistically favourable site for studies on European Storm-petrel.

Since behaviour in relation to lights near the colony may be different from behaviour at sea, experiments using lights on vessels or marine structures would also be beneficial. On-board observers and thermal imaging could be used to record the behaviour and number of birds in the vicinity.

Similar experiments, both on land and at sea, have been carried out in New Zealand by the Northern New Zealand Seabird Trust, University of Auckland and Saint Martin’s University, and Lukles et al. (2021) provide useful recommendations for future work. Studies on light attraction of Leach’s Storm-petrels are also being performed by researchers at Memorial University Newfoundland, using a portable radar system deployed at colonies. Coordination and discussion between research groups working on light attraction of Procellariiformes would be extremely beneficial.

7.6 Diet

While diet data do not explicitly feed into assessment methods and are therefore not considered a priority, an understanding of diet and the distribution of food resources is useful for predicting the level of impact caused by displacement or the likelihood of attraction to wind farms. Diet samples can be relatively easily collected by ringers or fieldworkers. Storm-petrels often produce regurgitates upon capture in mist nets, and faecal samples can be collected from nest sites (especially nest boxes). Obtaining diet samples from tracked birds would be particularly useful. Regurgitate and faecal samples from tracked (and untracked) European Storm-petrels have been collected on Mousa (Cardiff University/RSPB) and Treshnish Isles (RSPB), and for Leach's Storm-petrels on St Kilda (RSPB), but most have not been analysed. Molecular diet work (i.e. metabarcoding) is likely to provide more detailed information than traditional visual analysis, but visual analysis is also valuable, and much cheaper than molecular methods.

7.7 Apportioning impacts to protected colonies

Evidence needs with respect to foraging ranges and connectivity to Plan Options are high for all three species and medium with respect to colony locations (Table 5). Current knowledge of foraging ranges, locations and sizes of SPA colonies suggests that the following features of Scottish SPAs may be impacted by developments within Plan Options:

1) Manx Shearwater at St Kilda, Rum and Copeland
2) European Storm-petrel at Mousa, Auskerry, Sule Skerry, North Rona, Priest Island, Treshnish and St Kilda
3) Leach's Storm-petrel at the Flannan Isles and North Rona

Tracking of European Storm-petrels has been carried out on Mousa, Shetland over five years (2014-2018) between mid-July and mid-August, and at Lunga, Treshnish (19 individuals) for a single year in late July and August. Leach's Storm-petrels (14 individuals) have been tracked from St Kilda in a single year in July. Manx Shearwaters have been tracked from Rum, with GPS data for 20 trips from nine chick-rearing birds in 2010 and 58 trips from 15 chick-rearing birds in 2011 included in Dean et al. (2015). To establish ecological connectivity between Plan Options and these protected features, it would be beneficial to carry out tracking at the remaining breeding sites and to increase the sample of birds tracked, and the seasonal coverage of tracking, at Rum, St Kilda, Lunga and Mousa. The logistics of tracking on the Flannan Isles or North Rona would be extremely expensive and challenging, and success could not be guaranteed. Tracking at the remaining sites would be somewhat more straightforward, though not easy.

Storm-petrels breeding in the Northern Isles may be vulnerable to impacts of wind farms in the east and north-east of Scotland when departing south on migration, if they migrate southwards through the North Sea. Migration routes are currently poorly known, particularly for juveniles, which may be at particular risk of light attraction in the days and weeks after fledging. Storm-petrels can be tracked using geolocator (GLS) tags to identify migration routes and nocturnal illumination events. GLS tags require recapture of the bird, which is difficult (though possible) for adults but considerably more challenging for juveniles, which do not return to UK waters for two years. Visual inspection of light curves from eight birds tracked by RSPB (unpublished data) indicates nocturnal “light spikes” in wintering feeding areas, possibly as birds approach fishing vessels. While VHF/MOTUS tags are too large to be deployed on leg rings (for long-term studies) on storm-petrels, they could be used to collect multi-annual data on the movements of juvenile or non-breeding Manx Shearwaters. A network of receivers would need to be established to use VHF/MOTUS tags, but it may be possible to place these on turbines or other offshore structures, if incorporated at the planning stage.

It is important to note that, given the long-distance movements of these Procellariiformes, there may also be connectivity between ScotWind Plan Options and colonies outside of Scotland, and as far away as Canada in the case of Leach’s Storm-petrel (Bicknell et al., 2012, Bicknell et al., 2014). Tracking of the species from colonies elsewhere would be required to determine the extent of overlap.

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9 Glossary of terms and acronyms

AOS	Apparently occupied site. Unit used for counts of seabird breeding pairs or nest sites.
ARS	Area restricted search. A movement pattern in which an animal travels more slowly and with greater tortuosity while foraging or searching for prey, thereby remaining for longer in areas with higher food availability.
brooding	Breeding stage during which a small chick cannot thermoregulate and must be attended by an adult at all times.
dual foraging	Foraging strategy in which chick-rearing seabirds undertake a combination of short foraging trips for chick-provisioning and long foraging trips for self-provisioning.
ESAS	European Seabirds at Sea. A partnership project, managed by JNCC, that uses a standardised vessel-based survey technique to gather data on seabird abundance and distribution.
GLS	Global location sensor or light-level geolocator. A lightweight, archival tracking device that records ambient light levels and time, which can be used to determine latitude and longitude. Often used to track bird migration. Provides two locations per 24-hour period and is only accurate to within tens of kilometres.
GPS	Global positioning system. Tracking devices that use satellite technology, giving highly precise locations which are accurate to within a few metres.
incubation	Breeding stage between the laying and hatching of an egg, when the egg is generally attended by an adult at all times to maintain its temperature.
littoral	Relating to nearshore or coastal environments.
magnetoreceptor	A device or organ that detects the earth's magnetic field. Some form of magnetic sense is found in a wide range of animals, but the nature of the magnetoreceptor organs is often poorly understood.
morphology	The structure and form of organisms, especially their external form.
neritic	Relating to the shallow part of the sea near a coast and overlying the continental shelf, approximately 200 m deep.
NMP	National Marine Plan. Legislation adopted by the Scottish government in 2015 which provides a framework for managing all developments, activities and interests in or affecting Scotland’s marine area (territorial

	and offshore waters), setting out high-level objectives, general policies and sectoral policies.
Oceanitid	Any member of the family Oceanitidae of Southern or Austral Storm-petrels.
OWSMRF	Offshore Wind Strategic Monitoring and Research Forum. An industry-led collaborative forum that aims to better understand the impact of large-scale offshore wind development on marine birds.
pelagic	Relating to the open ocean.
phototaxis	Directional movement in response to a light source.
post-brooding	Breeding stage following brooding, when a chick can thermoregulate independently and can be left unattended while adults are foraging.
Procellariiformes	An order of seabirds, commonly known as tubenoses after their specialised nostrils that allow them to excrete salt. The order contains four families: the albatrosses, petrels and shearwaters, and two storm-petrel families.
RAS	Retrapping Adults for Survival. A programme run as part of the British Trust for Ornithology’s Ringing Scheme, in which ringers aim to ring and then catch or re-sight adult birds of a single species in a well-defined study area, enabling estimates of adult survival.
SMP	Sectoral Marine Plan for Offshore Wind. The strategy through which the Scottish government aims to identify the most sustainable Plan Options for the future development of commercial-scale offshore wind energy.
SMR	Scottish Marine Regions. 11 regions identified by Scottish Ministers under the Marine (Scotland) Act 2010 based on physical characteristics.
suprabenthic	Organisms that live on the sea floor but migrate above it seasonally or daily.
surface seizing	A method of foraging in which a seabird on the sea surface grasps food items just below the surface with its bill.
thermoregulation	Regulation of body temperature, whether physiological or behavioural.

Annex 1: Workshop Report

Workshop 1: 1300 – 1600 UTC, 10th March 2022

Workshop 1 brought together ecological experts on the three key procellariiform species (Manx Shearwater, European Storm-petrel and Leach's Storm-petrel), as well as experts on seabird vision and the impacts of artificial light on seabirds. A draft version of the literature was circulated to attendees in advance of the workshop and a summary of the draft review was presented at the start of the workshop, along with the knowledge gaps it had identified, the priorities amongst those, and brief suggestions for filling them.

Participants

43 participants attended the workshop and engaged well with the information presented and questions posed. The following 27 organisations were represented:

- Acadia University, Canada
- Azores University
- Bangor University
- BirdLife Greece
- Birmingham University
- BTO
- Cardiff University
- CORY'S (Spanish environmental consultant)
- Environment and Climate Change Canada
- Faroese Environment Agency
- JNCC
- LBHI (Agricultural University of Iceland)
- MacArthur Green
- Marine Scotland Science
- Memorial University of Newfoundland
- NatureScot
- RSPB
- Scottish Association for Marine Science
- South Iceland Nature Research Centre
- UKCEH
- University of the Azores
- Universidad Autónoma de Madrid
- University College Cork
- University of Gdansk
- University of Oxford
- Vilnius University

Discussion summary

Each participant was randomly allocated to one of three breakout rooms. Each breakout room included two members of the project team, one acting as a facilitator and one as a scribe. A Jamboard (online whiteboard) was set up for each breakout room in advance and a link provided for participants so that they could add ideas or comments during the session or at any time during the week following the workshop. The scribes aimed to ensure all comments were captured on the Jamboard, adding any points not written by the participants themselves.

Participants in each of the three breakout rooms discussed five key questions, during two sessions.

Session 1:

1. Have we missed anything important from the review?
2. Have we identified the knowledge gaps correctly?
3. What are the priority knowledge gaps in terms of wind farm risk assessments?
4. What risks/potential mitigation should we focus on (in the next workshop)?

Session 2:

5. Recommendations/challenges (technological, logistical, temporal, ethical, financial) for addressing knowledge gaps

Following each session, all participants returned to the main room and each of the breakout room facilitators provided a summary of the key points discussed within their group. It was generally agreed that the literature review was largely complete and that the knowledge gaps were correctly identified, but some suggestions for minor edits or additions were made. Each of the three breakout rooms came to similar conclusions regarding the priority knowledge gaps and there were lots of suggestions for addressing knowledge gaps, although it was acknowledged that many would be challenging to fill. A summary of the points raised, and the actions RSPB have taken as result, follows.

1. Have we missed anything important from the review?

Comment	RSPB Response	Notes
Did the review cover demographic consequences of displacement/collision etc? Possibly through individual based models?	No change	The review covers demographic parameters but implementing models is beyond the scope of this work.
Importance of considering dual foraging (where breeding birds alternate between long and short foraging trips) in these species.	Implemented	The review already mentions evidence for dual foraging in the Manx Shearwater and European Storm-petrel 'Foraging ecology' sections but we have now highlighted that it should be considered when using foraging range data for risk assessments.
Think about specific questions around light attraction to target future work, e.g. distance of attraction to lights - how large is the light "catch basin"?	Implemented	The review already mentions the range over which light attraction occurs as a knowledge gap, but we have added explicit mention of the 'light catch basin'.
Light attraction is important but note that there are two distinct processes: disorientation of adult petrels in foggy conditions, and response of fledglings which are orientated to move towards light to reach the sea.	No change	Already included
Important to consider lights at ports, harbours and other infrastructure developments associated with ScotWind as well as the actual wind farms and associated vessels.	No change	Already included
There is evidence in Canada of attraction to oil and gas platforms.	No change	Already included
Construction phase may be more important than operational phase as more lights/disruption.	Implemented	Added a sentence to point this out in relation to displacement and barrier effects and added 'construction activities' as part of light attraction considerations.
Many more collisions with buildings when lit (see Guilford *et al.* Bird Study paper).	No change	Already included

We need more information on the impact pathways; is light attraction/disorientation a separate pathway or are they part of impacts such as displacement, e.g. attraction displaces birds from where they would be otherwise. How do we incorporate that into impact assessments?	Implemented	Noted in lighting attraction introduction that light attraction may lead to displacement and added: "We do not consider light attraction to be a separate impact pathway, but it may exacerbate one or more of the recognised impact pathways (e.g. collision, displacement)."
Fledgling behaviour likely most important but has been the focus of other studies.	No change	Already included
Fledgling flight heights.	Implemented	Added to knowledge gaps: "There is a need to consider flight heights when arriving at or departing/fledging from high elevation nesting sites, as well as when birds are away from the coast."
Understanding of the first few days after fledging, when birds don't have good control of flight and are vulnerable to weather etc. Need to differentiate from light attraction.	Implemented	Added to knowledge gaps: "Fledglings may have particularly poor flight control in the first few days after fledging, making them more vulnerable, but our knowledge of fledgling behaviour is poor."
The review is bird-focused, what about structural elements of windfarm infrastructure.	No change	Beyond the scope of the review, although some discussion around mitigation options.
Maps in review don't show the Irish colonies. There will be connectivity with Scottish colonies, and non-Scottish colony birds might be using Scottish waters (e.g. Faroes).	Refer to MS	No change to maps requested, but note added to figure legends to highlight the need to consider colonies from outside of Scotland.
For apportioning, is it only the closest SPAs that are important, or all within foraging range? Is there colony segregation of foraging areas? e.g. Manx Shearwaters have a mixture of these effects depending on behaviour; on longer trips they aren't segregated to colony but they are on shorter trips.	No change	Foraging area overlap is discussed in the review. The details of the apportioning method (e.g. which colonies are included) are not part of the review.

Habitat preferences of prey/prey distributions (maybe limited data). How might prey distributions change in wind farm footprint and influence attraction?	Implemented	Added to knowledge gap around diet. Is already mentioned as a possible cause of attraction.

2. Have we identified the knowledge gaps correctly?

Comment	RSPB Response	Notes
Basic morphometric data		
Physiological/morphological differences between Welsh and Scottish birds are unlikely.	Implemented	Added to knowledge gap that differences are not expected to be large.
Flight data		
Need to understand how birds gain altitude when getting to burrows high up in colonies, e.g. Manx Shearwaters on Rum	Implemented	Added to knowledge gaps: "There is a need to consider flight heights when arriving at or departing/fledging from high elevation nesting sites, as well as when birds are away from the coast."
Current flight height assessment is unreliable, probably underestimates altitude. It is difficult/computationally complex to interpret altitude from biologging data.	No change	Limitations of current data are already discussed.
Flight heights in different weather conditions and day/night. If we can rule out that they fly at collision height under any circumstances, collision rate will be virtually zero.	Implemented	Added to flight heights knowledge gap.
Flight height/behaviour with wind speed/weather.	Implemented	Added to flight heights knowledge gap.
At-sea distributions/overlap with leased areas		
Likely to be site specific causes in variation	No change	
Diet		
Link between prey and habitat	No change	Already included.
Molecular techniques are an important method to carry out diet analysis	No change	Included in suggestions for filling knowledge gaps.

Some molecular analysis of diet is being carried out for Welsh colonies.	Implemented	Have noted in evidence needs section.
Behaviour (avoidance/attraction)		
Attraction/disturbance by boats	Implemented	Added to avoidance/attraction knowledge gap.
Light attraction/disorientation		
Distance over which light attraction may occur.	No change	Already included
Conceptual understanding whether it is attraction or disorientation that makes birds appear round lights.	No change	Already included
Multiple lights on multiple wind farms could appear more like a starscape and cause more problems than a single light.	No change	
Other		
Diurnal activity patterns.	Implemented	Added to knowledge gaps (especially for Leach's Storm-petrel)
Construction vs operational phase impacts.	No change	The same knowledge gaps apply to both phases. 'Associated activities' are included in current attraction/avoidance knowledge gap.
Break down broad light attraction category.	No change	This is already broken down in the knowledge gaps section.
Detection, particularly at the start of the assessment, plus biases in detection, both with DAS (not detecting ESP) and boat based (including potential attraction of birds to survey vessels).	No change	We have commented on the problems with aerial and vessel-based surveys.

3. What are the priority knowledge gaps in terms of wind farm risk assessment?

Comment	RSPB Response	Notes
Effects of light influencing collision, displacement and barrier effects.	No change	Already included
Need to understand whether flight height changes in response to turbines.	Implemented	Added to flight heights knowledge gap.
Potential attraction to vessel lighting.	No change	Light attraction already widely covered.
Is there more published on birds attracted to fishing vessels?	No change	Attraction to vessels (including fishing vessels) is already discussed, but a full review of attraction to fishing vessels is beyond the scope of this piece of work.
Fledgling risks on first migration (at night especially). How long are juveniles vulnerable to light pollution and is this linked to visual physiology?	No change	Already included
There may be other drivers of attraction to infrastructure, such as sound.	Implemented	Possibility of attraction to sound is mentioned in the review, but have added to knowledge gap around attraction/avoidance.
Indirect pathways associated with effects of light on prey.	No change	Possibility of lights increasing prey availability is mentioned. Knowledge gaps include changes in prey distributions around wind farms.
Any novel pathways?	No change	Everything additional identified during workshops has been added.
How do the limits of our knowledge affect our ability to carry out assessments?	No change	
Collision rate	No change	Lack of empirical data regarding impact assessments is already discussed.

The birds' ability to detect rotating blades. Could inform mitigation measures to increase detectability of the pylon and blades under different light levels and visibility.	Implemented	Added to avoidance/attraction knowledge gap.

4. What risks/potential mitigation should we focus on in the next workshop?

It was noted that any mitigation must apply to a broad suite of marine birds, and not just petrels and shearwaters.

Light attraction

- How restrictive are permissible lighting patterns/wavelengths?
- Experimental design to understand changing light features (with a dark control).
- Consider key periods of the year when lighting is an issue, e.g. fledging period.
- Absence of light is potentially worse if birds cannot see the turbines. We need to better understand birds' response to light at sea.
- There is some evidence of storm-petrels foraging at night at illuminated fish farms. It would be good to understand the mechanisms.
- Training of vessel crew in handling and releasing birds attracted and grounded on vessels, according to an established protocol (noting that there are existing examples for oil and gas platforms in Nova Scotia).

Other

- Changing the height of turbines.
- Methods to increase the detectability of pylons/blades under different visibility and light levels. Make them detectable at a sufficient distance for birds to change flight path and avoid them.
- Preventing predators from nesting/spending time around turbines.
- Is it possible that an underwater array might provide shelter for fish and so attract birds that way? Can this be mitigated for?
- A better understanding of attraction to noise, e.g. diesel generators on St Kilda.

5. Recommendations/challenges (technological, logistical, temporal, ethical, financial) for addressing knowledge gaps.

The importance of identifying the easiest gaps to fill was highlighted, as well as the need to consider which aspects should be addressed by academia and which by the offshore wind industry.

Basic morphometric data

Focus on easier work (e.g. morphometric data) first, then look at more complex studies.

Scottish morphometric data won't be much different from Wales and can be done in a couple of days, e.g. by an established ringer.

Flight data

Radar: Flight height can be measured with radar but has biases. Species ID (e.g. Manx vs Balearic Shearwater) is difficult with radar but perhaps mobile radar units could be deployed in places like Rum with only one species of shearwater. However, there is a trade-off between radar size and accuracy.. Could the equipment be placed on buoys to measure flight heights at sea?

Thermal imaging cameras could be used to detect storm-petrels at sea and record behaviour. Could flight heights be worked out from these images? It may not be possible to identify to species level (e.g. Manx vs. Balearic Shearwater), but that may not be important.

Hi-Def have been doing interesting work on flight heights from aerial images, but confidence intervals may be large. A report on this work is forthcoming: Humphries G, Fail T, Watson M, Bickley D, Peters-Grundy R, Scott M, Keogan K, and Webb A (in review). Aerial photogrammetry of seabirds from digital aerial video images using relative change in size to estimate flight height. *Marine Biology*.

Data from cameras attached to birds has been used to estimate the height of birds based on the tilt of the horizon, but this is challenging.

GPS and altimeters have been used to measure flight heights of Manx Shearwaters, but both have large errors.

Behaviour/flight height may be different to normal within a wind farm, as in gannets, so work within wind farms is important.

When does behaviour change from shearing to gaining altitude and vice versa?

Tracking:

Tags providing high resolution data are available for Manx Shearwaters.

Long term tag attachment is possible on storm-petrels with sutures, but that method would be challenging to license in the UK. Short-term tagging during the breeding season is adequate for adults/immatures prospecting but not for fledglings.

PathTrack tags record instantaneous speeds. These are not currently automatically available with data download but can be requested.

Accelerometers are now/soon to be available for storm-petrels, but still won't give flight heights.

Is there an alternative to altimeters? The accuracy of altitude data from high resolution GPS data has been tested but still had large errors. High resolution data for storm-petrels is not yet possible due to the small battery sizes needed to keep tags small enough.

MOTUS could have potential for measuring height as well as location, but it is difficult to establish an array of receivers offshore. Receivers have been installed on oil and gas platforms and supply vessels in Canada, and it may be possible to deploy them on buoys at development sites. Any such inclusion of receivers in infrastructure would need to be considered at the planning stage for offshore wind farms. . GPS might be better, but remote-download tags are not yet small enough for storm-petrels so they need to be recaptured.

At-sea distributions/overlap with leased areas

This information becomes more important to understand if it is used in deciding where wind farms go.

There are huge impacts of Leach's Storm-petrel predation by Great Skuas at St Kilda. There is some genetic evidence that some of the birds eaten may be from Canada. This is an example of the need to understand connectivity between Canada and Scotland, and not just focus on birds from Scottish colonies.

Safe access to many colonies is a logistical constraint for tracking and determining at-sea distributions.

Radar can be used to look at density at sea before/after construction, and in combination with other studies such as tracking could help us to understand juvenile/adult ratios.

MOTUS: Tags are currently too big for storm-petrels but could be used on Manx Shearwaters. Useful for non-breeders/fledglings as long-term deployment outside the breeding season is possible. Detection distance is limited to line of sight. There is currently no UK network, but there is one in Canada which would be useful to learn from. It is important to incorporate at the planning stage if putting receivers on offshore structures. Unclear whether MOTUS could provide data on flight height and avoidance as well as distributions.

Detection*:* Carry out targeted digital aerial surveys (DAS) with experimental approach to detectability, for example using decoy models of birds to assess detectability under different conditions.

Rescue programmes, such as that for grounded Manx Shearwaters in Mallaig, could be used in association with ringing to identify source colonies. Birds could be tracked from wind farm areas or if stranded on vessels or structures, but remote download of data would be required.

Dye marking of large numbers of birds at colonies could be used to look at whether adults or young birds interact with wind farms.

Birds could be caught at sea and stable isotope or genetic analysis used to identify their origin, although catching at sea is challenging and may not be possible.

Diet

Change in community of marine life in wind farms is likely. It is important to understand birds' diet to understand how these changes affect the birds.

University of Oxford is currently carrying out diet studies, including metabarcoding. Cardiff University has performed metabarcoding of diet samples for many species, including storm-petrels. There needs to be a larger study using DNA metabarcoding, but it would be easy to collect samples.

Visual diet analysis is valuable and much cheaper than molecular. Molecular diet work is likely to be better than traditional visual analysis as it is very hard to identify prey to species level when samples are tiny/degraded (especially faecal samples), and visual identification is biased towards less digestible items such as squid beaks. Any diet studies are much cheaper than tracking.

Could faecal/regurgitate samples be collected by ringers? Would need to consider the logistics of transporting samples to the lab, but this is not too challenging.

It would be useful to link diet data to tracking information from the same individuals.

Is diet or productivity/prey distribution more important? Some knowledge of diet is required before being able to infer predator distributions from productivity/prey distributions.

Behaviour (avoidance/attraction)

Fledglings could be tracked, but it would be necessary to recapture them if remote download is not possible. Track adults and juveniles from the same colony to see if interactions with wind farms are different for different age classes.

VHF: receivers have been placed on oil and gas structures, but it is difficult to get a good sample size.

Radar could be used to quantify flight lines. Changes in flight lines are evidence for attraction/avoidance. The detection distance of portable radar is 72 nautical miles. Memorial University (Newfoundland) has a mobile radar system that will deployed at Leach's Storm-petrel colonies to study their behaviour around artificial light.

Look at circumstances of behaviour in the vicinity of existing lights.

Combine multiple methods (tracking, radar, camera etc). how do we get control site/set of birds, does this need to be tracking?

Light attraction/disorientation

An experimental approach is important, despite the challenges. Look at light range, wavelength, pattern and adults vs immatures.

Behaviour at the colony might be different from at-sea behaviour. Could you conduct experiments at sea using vessels or structures? Experiments will need to use lights that are possible to use on turbines/vessels and these should be standardised across studies using different locations/species.

Is response to light related to colony proximity? This would need both at-sea and landfall groups.

Studies at SPAs would require Habitat Regulations Assessment. Experimental studies might injure the birds and would be harder to licence than studies that take advantage of existing/proposed differences between developments/locations. Perhaps some sort of cushioning could be used on boats/structures to protect birds from impact during experiments.

Focus research on current windfarms in the Irish Sea (e.g. Robin Rigg) as there are Manx Shearwaters in the area. Can lighting on existing turbines be changed to monitor changes in behaviour using GPS tracking?

Could expertise within the offshore wind farm sector be harnessed to construct bespoke experimental structures at appropriate locations?

Bardsey lighthouse changed to a red flashing light in 2014 and there have been virtually no collisions since. There were a lot of reviews in the 19th century of birds flying into lighthouses, this should be reviewed as a starting point.

A questionnaire could be sent to vessel operators about birds found on deck. Onboard fisheries observer programmes could be used to gather data but are limited in the UK and only take place during the day. There is currently a project in South Georgia and previously one in New Zealand using these methods. Cruises around the UK with naturalists onboard (e.g. National Geographic) could be used for better species ID. JNCC also runs seabirds at sea surveys. We might be able to ask wind farm maintenance vessels/crews to partake, or it could be made a requirement in certain areas/settings.

Look at existing studies for the proportion of adults killed when attracted. It would be difficult to monitor the number of collisions at offshore structures.

Bio-acoustic studies could be used to pick up calls around structures/vessels/colonies. Caution is needed in interpreting the data because of changes in vocalisations in response to stress.

Systematic, rather than opportunistic, surveys of oil and gas platforms would help to understand the drivers of light attraction and would allow a finer scale temporal analysis.

Radar/cameras (BACI studies) could answer some of these questions.

Understanding the impact of different weather conditions is very important.

Light attraction is the biggest unknown but may not be the biggest issue. We don't know enough about collision/displacement either.

Workshop 2: 1530 – 1700 UTC+1, 31st March 2022

Workshop 2 focused on mitigation, particularly in relation to the potential impacts on procellariiform seabirds of the artificial lighting associated with offshore wind developments. The workshop began with a presentation on the key impact pathways identified in the review and Workshop 1, and some suggestions of mitigation options. A second presentation from Anatec described the current lighting requirements of offshore wind farms and associated infrastructure and activities.

Participants

39 participants attended Workshop 2, from the following 26 organisations:

- Anatec
- APEM Ltd
- BirdLife Malta
- Cardiff University
- Civil Aviation Authority
- EDF Renewables
- Environment and Climate Change Canada
- HiDef Surveying
- Houston Audubon Society
- JNCC
- Marine Scotland Science
- Maritime and Coastguard Agency
- Natural England
- NatureScot
- Northern Lighthouse Board
- ORE Catapult
- ØRSTED
- RSPB
- Scottish Power
- South Iceland Nature Research Centre
- SSE
- UKCEH
- University of Birmingham
- University of Gdansk
- University of Oxford
- Vattenfall

Discussion summary

Discussion was structured around three broad topics: changing the nature of lighting, changing lighting infrastructure, and other mitigation options. It was generally agreed that changing the nature of lighting (e.g. reducing lighting or changing the wavelength, intensity or pattern of illumination) would not be possible as lighting of vessels and structures is highly standardised and aims to maximise safety of vessels and aircraft. Shutting down turbines at key times was also considered not to be feasible, but other options were discussed, as follows.

1. Changing the nature of lighting

No lighting (or turning off lighting at key times, e.g. fledging period)

Not an option from a safety perspective. Shouldn't be considered as mitigation because of serious concern about turbines not being lit for even a short period of time.

Aviation lights only come on at night.

Reduce intensity

There is provision within aviation lighting rules to reduce lights when visibility is above 5 km. Lights can be dimmed up to 90% during reasonable weather.

There is not enough evidence on the impact this would have on different seabird species.

Intensity is more important than colour in bird night vision. The impact of different intensities also depends on atmospheric conditions (fog, rain, etc.). Any conditions that create large diffuse pools of light is the problem.

Bird vision is usually fully functional after a few days. However, there are suggestions that for burrow-nesting seabirds this may not be the case.

Juvenile burrow-nesting seabirds develop their eyes fully after they fledge:

- Mitkus, M., Nevitt, G. A., & Kelber, A. (2018). Development of the visual system in a burrow-nesting seabird: Leach's storm petrel. *Brain, Behavior and Evolution*, 91(1), 4-16.
- Atchoi, E., Mitkus, M., & Rodríguez, A. (2020). Is seabird light-induced mortality explained by the visual system development*? Conservation Science and Practice*, 2(6), e195.

Reduce number of turbines illuminated

If the number of turbines lit is reduced, the intensity of lighting would have to increase to compensate so this isn't always viable.

Lighting of non-turbine infrastructure could also pull birds off course (especially fledglings). Reduce, turn off, or cover other lights associated with turbines (e.g. maintenance vessels). Decrease the activity of maintenance vessels, or other lighting that isn’t crucial, during high risk periods for birds (e.g. fledging).

Alter pattern of illumination

There needs to be consistency across wind farms to avoid confusion to mariners (especially recreational) and to comply with international standards. Marine navigation lights (UK) generally have a character of 1 flash every 5 seconds (Significant Peripheral Structures) or 2.5 seconds (Intermediate Peripheral Structures).

Alter colour of lights

Again, there needs to be consistency across wind farms and compliance with international standards so this is unlikely to be possible.

Search and rescue (SAR) lights are red to avoid affecting the night vision of crew. No other colour is possible. White lights are used in some circumstances for obstacle lighting for aviation but this has the potential to disrupt night vision of crews.

Is there a way of changing the wavelengths of the lights (e.g. reduce blue wavelengths) to alter the birds' perception of the lights without changing the colour as perceived by humans (so as still to comply with maritime standards)?

Additional lighting

Is more lighting possible, as less is not? Could additional high attraction lighting on buoys be used to navigate birds around high risk areas, perhaps temporarily during the fledging period?

2. Changing lighting infrastructure

Shielding/directionality of lights

The peak of an aviation light beam is between 3-4 degrees above the horizontal plane and less than 10% is 1-1.5 degrees below the horizontal plane, so only 10% is directed down.

Vertical divergence of marine lighting is only a couple of degrees off of the horizontal, but the light needs to remain visible to all sizes of vessels, either up close to the turbine, or at the extreme range of the light.

Could you light up the wind farm at night from a light placed on another structure, pointing at the turbine? To avoid attraction and collision with the light itself, place it below the sea surface. This is unlikely to be feasible as turbine lighting is standardised to maximise efficacy and safety.

Lighting on ships could be directed towards the deck to reduce scatter.

There is probably potential to have design discussions with turbine designers as to how access door lighting could be altered, as well as with vessel suppliers. As such, there might be some mitigation solutions which could be explored with relative ease, or are already being implemented as standard, to minimise attraction for Procellariiformes.

Radar-activated lighting

There are already systems in some parts of the world where lights only come on when an aircraft is detected, but this is not in UK EEZ regulations at the moment. There are different technologies that can enable this. Could this be possible for vessels on the sea too?

A detection system isn't possible for all marine vessels, especially ill-equipped recreational vessels. The ability to detect and track small yachts/vessels, particularly in poor weather is difficult and would not be reliable. As such, the provision of lighting must always be for the lowest level user.

It is crucial to keep the importance of lighting in perspective and cater for all situations and everything that might be flying, e.g. civil aviation and search and rescue could be present at any time.

Lighting is required to provide ships with sufficient warning to take avoiding action so needs to consider all types of vessels, including very slow turning vessels.

3. Other suggestions

Increase blade height

This has benefits outside of mitigation for birds. 22 m of clearance is the required minimum to reduce impact on marine vessels.

Increased hub height would require feasibility assessment on a case-by-case basis of manufacturing and installation of foundations tower sections.

The threshold for crossover from fixed bottom vs floating turbines is likely to increase over time (perhaps up to 100 m). There's a relationship between increasing turbine size and jack-up/crane lifting capabilities and the latter may be a limitation for increasing blade height

As turbines become larger, we will likely see a slight increase in the sea level to lower tip clearance. We will also see a larger rotor diameter but not necessarily a large increase in the overall rotor-swept area (i.e. typically fewer/larger WTGs meaning the overall swept area of the rotor in the ‘danger zone’ for birds could be proportionally reduced.

Shut down turbines at critical times

The UK will be reliant on offshore wind in the future so switching off turbines would be the worst-case scenario.

Increase detectability by marking blades

Review recently conducted for Natural England. It is likely that something can be done to make blades more visible to birds flying through, and therefore prevent collisions.

Collision with towers is also possible. Maximising contrast between towers and blades (e.g. black/white stripes on blades and towers) has been looked at. There is currently a project in Norway looking at this.

However, the need for consistency across wind farms must be considered, to avoid causing confusion.

Use of sound to deter seabirds or avian predators

Operational offshore wind turbines have relatively high noise emissions (~110+ decibels). This could be a consideration if these species actively avoid noise sources. There is some evidence that Manx Shearwaters are attracted to generators on St Kilda by sound.

There is an Irish project testing if sound could be used to deter birds. Sound signals (e.g. foghorns) on turbines at periphery of windfarms which sound during poor visibility (under 2 nautical miles). There are characteristics that must be met (1 long blast, 2 short blasts every 30 seconds).

It would be interesting to test whether birds respond to foghorns or if there are sounds beyond human hearing range that might be detected and deter birds during times when risk is high. Although obviously has implications for displacement.

Train vessel crew in safe handling/release of stranded birds

SOPs for stranded birds.

Posters have been created for vessels anchored in St Brides Bay, Pembrokeshire, explaining what to do if Manx Shearwaters land on boats at night. This method has been repurposed for different species and locations around the world.

Annex 2: Catalogue of data sources

Parameter/data group	Manx Shearwater			European Storm-petrel			Leach's Storm-petrel			Northern Fulmar			Sooty Shearwater		
	Scot	UK & I	World[a]	Scot	UK & I	World[a]	Scot	UK & I	World[a]	Scot	UK & I	World[a]	Scot	UK & I	World[a]
Marine distribution[R]	1, 2, 3, 4, 5, 6, 7, 8			2, 3, 5, 6, 9			2, 5, 6, 9			2, 3, 6, 9, 10			2, 9, 11, 12		
Tracking[R]	7, 8, 13, 14[b], 15[b], 16[b], 17[b]	18, 19, 20, 21, 22, 23, 24, 25, 26		27	28	29‡, 30‡			31, 32, 33, 34, 35, 36	10, 37, 38, 39, 40, 41		42, 43, 44, 45			46, 47, 48, 49, 50, 51, 52, 53, 54
Colony size & location[R]	55, 56			55, 57, 58, 59, 60, 61, 62			55, 62, 63, 64, 65			55, 66					67, 68, 69, 70, 71
Foraging range[R]	13	15, 16, 20, 25, 72, 73		27	28	29‡, 30‡			31, 32, 36		73				47, 48, 54
Habitat associations[R]	2, 13	14		2	28, 74	29‡, 30‡	2		32, 75	2, 10, 39	76		2		48, 53, 54, 77, 78
Age at first breeding[N]		79, 80		81					82	83, 84					85
Productivity[R]	86, 87, 88, 89	90, 91		92	93, 94, 95	96, 97‡, 98,	101, 102, 103		104	86, 105, 106, 107,					111, 112, 113

						99‡, 100‡				108, 109, 110					
Survival^{R}	114	79, 80, 115, 116, 117, 118		60, 114	94, 117, 119	120, 121‡	122		104, 123, 124, 125	84, 126, 127, 128, 129					85, 130
Body length^{G}			131			131			131			131			131
Wingspan^{G}			131			131			131			131			131
Flight type^{G}	89	132, 133	134, 135		136†	135		136†	134, 135	137		135, 138‡			134, 135
Flight height^{G}	139	140, 141	134‡, 138‡, 142†		136†, 139	142†		136†	142†	139, 143	140, 141	134, 138‡, 144		139†	134‡, 138‡, 142†, 145
Flight speed^{G}		15, 16, 23, 25, 132	134‡		28	29‡, 30‡, 134‡			134‡, 146‡	37, 137		42, 45, 134‡, 147			47, 134‡
Nocturnal activity^{R}	89†	14, 16, 23, 25		27†		148†	149†		150†	41, 151†, 152†, 153					53, 154
Adult body mass^{N}	89, 155	156, 157, 158	131, 159	160, 161	94, 162	131, 159, 163, 164, 165, 166	161, 167		131, 159, 168			131, 159			131, 159, 169, 170

Chick mass/growth rate^{R}	89	80, 157, 171, 172, 173, 174, 175	176	177, 178, 179	94, 180		102		181, 182, 183, 184	185, 186, 187					
Length of breeding season^{R}	89	24, 80, 157	131		93, 94	96, 131	102		131, 188	189		131			131, 169, 170
Energy requirement – adult^{N}	190	191	176	92, 192, 193	191			191	194, 195, 196, 197, 198, 199, 200, 201, 202, 203, 204, 205	190, 206, 207, 208	191	195, 209, 210, 211		191	195, 205, 212
Energy requirement – chick^{N}			176						183, 205						
Maximum brood size^{G}			131			131			131			131			131
Dive depth^{R}		19			213†	214‡				215					77, 216

Avoidance^{G}	217	218										218, 219, 220			
Non-light attraction^{G}	221†, 222		223‡, 224‡, 225‡	221†		223‡, 224‡, 225‡, 226‡	221†		223‡, 224, 225	221†		223‡, 224‡, 225‡, 227	221†		223‡, 224‡, 225‡
Light attraction/ disorientation^{G}	222, 228	229, 230, 231, 232	233†, 234, 235	222, 236, 237,		233†, 235‡, 238†	222		233†, 234, 239, 240, 241, 242, 243, 244†, 245			234			233†, 234, 246†

Notes:
Parameter superscripts refer to whether a parameter is likely to vary at a regional (R) or national (N) scale or is expected to be similar globally (G).
Superscripts elsewhere identify data source limitations:
† Qualitative data or expert opinion
‡ Data for closely related taxa (including Mediterranean Storm-petrel)
[a] ‘World’ includes general references or when unclear where data from geographically.
[b] These references relate to birds tagged at colonies outside of Scotland, but that used Scottish waters.

Whether or not an item was freely publicly available at the time of the review is stated at the end of each reference in the bibliography below as (Yes) or (No).

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