Written Representations

for the

Royal Society for the Protection of Birds

Submitted for Deadline 1

16 January 2019

Planning Act 2008 (as amended)

In the matter of:

Application by Norfolk Vanguard Limited for an
Order Granting Development Consent for the
Norfolk Vanguard Offshore Wind Farm

Planning Inspectorate Ref: EN010079

Registration Identification Ref: 20012785

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1 Introduction

1.1.1 These representations have been prepared with Dr McCluskie, whose qualifications and experience are provided in Annex 1.

1.2 The RSPB

1.2.1 The Royal Society for the Protection of Birds (the RSPB) was set up in 1889. It is a registered charity incorporated by Royal Charter and is Europe's largest wildlife conservation organisation, with a membership of 1.22 million (RSPB, 2018). The principal objective of the RSPB is the conservation of wild birds and their habitats. The RSPB therefore attaches great importance to all international, EU and national law, policy and guidance that assist in the attainment of this objective. It campaigns throughout the UK and in international fora for the development, strengthening and enforcement of such law and policy. In so doing, it also plays an active role in the domestic processes by which development plans and proposals are scrutinised and considered, offering ornithological and other wider environmental expertise. This includes making representations to, and appearing at, public inquiries and hearings during the examination of applications for development consents.

1.3 The RSPB's interest in offshore wind development

1.3.1 Faced with the threats of climate change to the natural world the RSPB considers that a low-carbon energy revolution is essential to safeguard biodiversity. However, inappropriately designed and/or sited developments can also cause serious and irreparable harm to biodiversity, and damage the public acceptability of the necessary low-carbon energy transition technologies.

1.3.2 The UK is of outstanding international importance for its breeding seabirds, including northern gannet for which the UK supports over 50% of the world population. As with all Annex I and regularly migratory species, the UK has particular responsibility under the Birds Directive[1] to secure the conservation of this important seabird's population.

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[1] Directive 2009/147/EC of the European Parliament and of the Council of 30 November 2009 on the conservation of wild birds (codified version) (the Birds Directive).

1.3.3 The available evidence suggests that the main risks of offshore wind farms for birds are collision, disturbance/displacement, barriers to movement (e.g. migrating birds, or disruption of access between the breeding areas and feeding areas), and habitat change particularly with associated changes in food availability and the cumulative and in-combination effects of these across multiple wind farms.

1.3.4 Such impacts are avoidable, and the RSPB has spent considerable time working with stakeholders in the UK offshore wind industry to ensure that decisions about deployment of renewable energy infrastructure take account of environmental constraints and seek to avoid or minimise impacts wherever possible. The RSPB therefore strongly advocates the use of rigorous, participative environmental assessments to inform the development of projects.

1.4 Summary of the RSPB's Position

1.4.1 The RSPB's primary concerns about the Norfolk Vanguard proposal result from a number of methodological concerns about the assessment of various impacts and the implications those concerns have for the overall conclusions about the impacts of the Norfolk Vanguard proposal. Our concerns focus on the following aspects:

- The impact of collision mortality on the kittiwake population of the Flamborough and Filey Coast SPA alone and in-combination with other plans and projects;
- The impact of collision mortality on the gannet population of the Flamborough and Filey Coast SPA alone and in-combination with other plans and projects; and
- The impact of collision mortality on the lesser black-backed gull population of the Alde-Ore Estuary SPA alone and in-combination with other projects.
- Cumulative collision mortality to North Sea populations of kittiwake and great black-backed gull; and
- Cumulative operational displacement to North Sea populations of red-throated diver, guillemot and razorbill.

1.4.2 Our key methodological concerns are listed below:

- Use of Potential Biological Removal in assessment of impacts on SPA populations;
- Use of an unverified stochastic Collision Risk Model (CRM) which underestimates collision mortality;

- Use of median bird densities within the deterministic CRM;
- Use of revised Nocturnal Activity Rates;
- Use of migration-free breeding season;
- Approach to apportioning of mortality to SPAs for kittiwake and lesser black-backed gull;
- Breeding season gannet avoidance rate of 98.9%;
- Inclusion of unjustified criticisms of kittiwake tracking data; and
- Proposal for mitigation of impacts on the Alde-Ore Estuary SPA.

1.4.3 We therefore do not agree that there is sufficient robust evidence available to support a conclusion of no adverse effect on the integrity of the Flamborough and Filey Coast SPA or the Alde-Ore Estuary SPA, or to rule out significant effects on North Sea populations of kittiwake, great black-backed gull, red-throated diver, guillemot and razorbill.

2 Protected Sites and Species

2.1 The Flamborough and Filey Coast SPA

2.1.1 The Flamborough Head and Bempton Cliffs SPA was designated under Article 4(2) of the Birds Directive as an SPA in 1993 due to the presence of 83,370 pairs of black-legged kittiwake (*Rissa tridactyla*), representing 4% of the Eastern Atlantic breeding population. In 2001 the UK SPA Review found that it also qualified under Article 4(2) as a site regularly supporting at least 20,000 seabirds, due to at the time of designation, the site regularly supported 305,784 individual seabirds including: puffin (*Fratercula arctica*), razorbill (*Alca torda*), guillemot (*Uria aalge*), herring Gull (*Larus argentatus*), Gannet (*Morus bassanus*), and Kittiwake. Kittiwake and the seabird assemblage are therefore the qualifying features of this SPA.

2.1.2 In January 2014, Natural England opened a formal consultation on proposals to extend the SPA and rename it as the Flamborough and Filey Coast SPA. The proposals comprised changes to the designated site boundary and changes to the numbers of qualifying species. The site was formally designated in November 2018 and the Flamborough Head and Bempton Cliffs SPA has been incorporated into it.

2.1.3 Natural England has also conducted a review of the seabird populations using contemporary data. A summary of Natural England's review (Natural England, 2014) of the ornithological interest of the SPA is as follows with the key species set out in more detail in Table 2.1 below:

> The application of SPA selection guidelines (JNCC 1999) to current data for this site confirm that it qualifies by regularly supporting internationally important numbers of breeding black-legged kittiwakes, northern gannet, common guillemot and razorbill and an assemblage of European importance of over 20,000 breeding seabirds. Black-legged kittiwake, northern gannet, common guillemot and razorbill are all main components of the assemblage and present in internationally important numbers. However, northern fulmar is also present in sufficient numbers to warrant being listed as main component species of the assemblage, since numbers exceed 2,000 individuals (10% of the minimum qualifying assemblage of 20,000 individuals). In addition, Atlantic puffin, herring gull, European shag (*Phalacrocorax aristotelis)* and great cormorant (*Phalacrocorax carbo)* are also part of the breeding seabird assemblage.

Table 2.1: Summary of Ornithological Interest of the SPAs

Species	**Count (period)**	**% of subspecies or population (pairs)**	**Interest Type**
Flamborough Head and Bempton Cliffs SPA			
Black-legged kittiwake *Rissa tridactyla*	83,700 pairs (1987)	4% Western Europe	Migratory
Flamborough and Filey Coast SPA			
Black legged kittiwake *Rissa tridactyla*	44,520 pairs 89,041 breeding adults (2008-2011)	2% North Atlantic	Migratory
Northern gannet *Morus bassanus*	8,469 pairs 16,938 breeding adults (2008-2012)	2.6% North Atlantic	Migratory
Common guillemot *Uria aalge*	41,607 pairs 83,214 breeding adults (2008-2011)	15.6% (*Uria aalge albionis*)	Migratory
Razorbill *Alca torda*	10,570 pairs 21,140 breeding adults (2008-2011)	2.3% (*Alca torda islandica*)	Migratory
	Count period	**Average number of individuals**	
Seabird assemblage	2008-2012	215,750	

2.1.4 The Conservation Objectives for the Flamborough and Filey Coast SPA are therefore as follows: *Ensure that the integrity of the site is maintained or restored as appropriate, and ensure that the site contributes to achieving the aims of the Wild Birds Directive, by maintaining or restoring;*

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

2.1.5 Since this site was originally designated as an SPA, the national populations of both kittiwake and some assemblage species have suffered substantial declines. For example the UK breeding kittiwake population has reduced by 70% since 1986 (Hayhow *et al*., 2017). Within the SPA there has been a reduction from the 83,370 breeding pairs of kittiwakes (at time of designation, 1993) to an average of 44,520 breeding pairs between 2008 and 2011.

2.2 The Alde-Ore Estuary SPA

2.2.1 The main feature of the Alde-Ore Estuary SPA affected by the Application is the breeding lesser black-backed gull population, the majority of which breed at Havergate Island (RSPB reserve) and Lantern Marshes on Orfordness (National Trust reserve).

2.2.2 The Alde-Ore Estuary SPA was classified in 1996 on the basis of supporting an average of 14,070 lesser black-backed gull Apparently Occupied Nests (AONs) between 1994 and 1998, or 12% of the biogeographic population. Following classification, the lesser black-backed gull population experienced a rapid increase in the late 1990s, peaking in 2000. This is reflected in the population of 21,700 pairs described in the Alde-Ore Estuary SPA site account in the UK SPA Review (Stroud *et al*. 2001). Since this time, the population has experienced a severe decline, such that in 2018 there were only 1,424 breeding pairs recorded in the Alde-Ore Estuary SPA. Further details of population figures can be found in Annex 2.

2.2.3 Natural England has determined that the target population of the SPA is 14,074 pairs of lesser black-backed gulls if the SPA is to meet its conservation objectives.

2.2.4 The Conservation Objectives for the Alde-Ore Estuary SPA are as follows:

Ensure that the integrity of the site is maintained or restored as appropriate, and ensure that the site contributes to achieving the aims of the Wild Birds Directive, by maintaining or restoring;

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

2.2.5 The Alde-Ore Estuary is the only SPA for lesser black-backed gull on the east coast of England. As such it plays an important role with respect to the UK population of this species. Even at its now much reduced size the most recent population estimate (1,424 pairs) represents 1.27% of the UK population of 112,000 AON (JNCC, 2018b).

3 Legislation and Policy Background

3.1 Introduction

3.1.1 The suite of Energy NPSs set out the Government's approach to ensuring the security of energy supplies and the policy framework within which new energy infrastructure proposals are to be considered. The presumption in favour of granting consent, as identified in NPS EN-1, *Overarching National Policy Statement for Energy*, is subject to the tests set out below in section 104 of the Planning Act (see NPS EN-1 paragraphs 4.1.2 and 1.1.2).

3.1.2 Section 104 of the Planning Act 2008 provides that an application for development consent for energy infrastructure must be decided in accordance with the relevant National Policy Statement (NPS) except where in doing so it would lead to the UK:

- being in breach of its international obligations;
- being in breach of any statutory duty that applies to the Secretary of State;

or would:

- be unlawful;
- result in adverse impacts which would outweigh the benefits; or
- be contrary to regulations about how decisions are to be taken.

3.1.3 The statutory duties include the Conservation of Habitats and Species Regulations 2017 (the Habitats Regulations) (NPS EN-1 paragraph 4.3.1) and the wider objective of protecting the most important biodiversity conservation interests (see NPS EN-1 section 5.3 generally). It notes the Habitats Regulations' statutory protection for important sites including Ramsar sites, listed under the Ramsar Convention[2], SPAs designated under the Birds Directive and Special Areas of Conservation (SACs) designated under the Habitats Directive[3].

3.1.4 NPS EN-1 also confirms that for the purposes of considering development proposals affecting them, listed Ramsar sites should also, as a matter of policy, receive the same protection (NPS EN-1 paragraph 5.3.9).

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[2] The Convention on Wetlands of International Importance 1971.

[3] Council Directive 92/43/EEC on the conservation of natural habitats and of wild fauna and flora.

3.1.5 NPS EN-3, *National Policy Statement for Renewable Energy Infrastructure*, specifically identifies birds as a biodiversity concern to be taken into account (paragraph 2.6.59 and 2.6.68). Whilst it is stated that the designation of an area as a protected European site does not necessarily restrict the construction or operation of offshore wind farms (paragraph 2.6.69), the legislative requirements identified above are still to be met. The protection afforded by legislation, to which the 2008 Act and the NPSs refer, are addressed briefly below.

3.2 The Conservation of Habitats and Species Regulations 2017 and the Conservation of Offshore Marine Habitats and Species Regulations 2017

3.2.1 SACs and SPAs are protected as “European sites” in inshore waters (up to 12 nautical miles from the baselines) under provisions within the Conservation of Habitats and Species Regulations 2017 (Habitats Regulations); and in offshore waters (i.e. from 12-200 nautical miles) under provisions within the Conservation of Offshore Marine Habitats and Species Regulations 2017 (Offshore Regulations).

3.2.2 The Habitats & Offshore Regulations set out the sequence of steps to be taken by the competent authority (here the Secretary of State for Business, Energy and Industrial Strategy) when considering authorisation for a project that may have an impact on a European site and its species before deciding to authorise that project. These are as follows:

a. Step 1: consider whether the project is directly connected with or necessary to the management of the SPA and its species [regulation 63 (1)]. If not –
b. Step 2: consider, on a precautionary basis, whether the project is likely to have a significant effect on the SPA and its species, either alone or in combination with other plans or projects (the Likely Significance Test) [regulation 63 (1)].
c. Step 3: make an appropriate assessment of the implications for the SPA and its species in view of its conservation objectives. There is no requirement or ability at this stage to consider extraneous (non-conservation e.g. economics, renewable targets, public safety etc) matters in the appropriate assessment [regulation 63 (1)].
d. Step 4: consider whether it can be ascertained that the project will not, alone or in combination with other plans or projects, adversely affect the integrity of the SPA and its species, having

regard to the manner in which it is proposed to be carried out, and any conditions or restrictions subject to which that authorisation might be given (the Integrity Test) [regulation 63 (6)].

e. Step 5: In light of the conclusions of the assessment, the competent authority shall agree to the project only after having ascertained that it will not adversely affect the integrity of the SPA, alone or in combination with other plans or projects [regulation 63 (5)].
f. Step 6: only if the competent authority is satisfied that, there being no alternative solutions, the plan or project must be carried out for imperative reasons of overriding public interest (which, subject to [regulation 64(2)], may be of a social or economic nature), they may agree to the plan or project notwithstanding a negative assessment of the implications for the European site [regulation 64 (1)].
g. Step 7: in the event of the imperative reasons of overriding public interest and alternative solutions tests being satisfied, the Secretary of State must secure that any necessary compensatory measures are taken to ensure that the overall coherence of the Natura 2000 network is protected [regulation 68].

3.2.3 It is important to add that in addition to the requirements set out above, in relation to both inshore area and the offshore marine area, any competent authority must exercise its functions so as to secure compliance with the requirements of the Habitats Directive and the Birds Directive; and in particular to take such steps as it considers appropriate to secure the preservation, maintenance and re-establishment of a sufficient diversity and area of habitat for wild birds[4], having regard to the requirements of Article 2 of the Birds Directive.[5] And for offshore SPAs and SACs regulation 26, Offshore Regulations requires competent authorities to exercise their functions (as far as possible) to secure steps are taken to avoidance the disturbance of species and the deterioration of habitats or habitats of species within those sites.

3.3 Appropriate assessment

3.3.1 As part of the assessment requirements, regulation 63, Habitats Regulations (regulation 28, Offshore Regulations) require the application of the precautionary principle. Meaning that if it

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[4] As required by Article 3, Birds Directive

[5] See regulation 9(1) and 10(1)(2)(3) and (8) of the Habitats Regulations and regulation 6 of the Offshore Regulations. Article 2 Birds Directive imposes a requirement on Member States to maintain all wild bird populations at a level which corresponds in particular to ecological, scientific and cultural requirements, while taking account of economic and recreational requirements, or if necessary to restore the population of these species to that level (Article 2).

cannot be excluded, on the basis of objective information, that it is likely to have a significant effect on a SPA or SAC and its species an appropriate assessment will be required: see *Waddenzee* case.[6]

3.3.2 Following that appropriate assessment, a project may only be granted consent if the competent authority is convinced that it will not have an adverse effect on the integrity of the European site(s) and their species concerned, having applied the precautionary principle and taken account of the conservation objectives for those sites and their habitats and species. *Waddenzee* confirmed that where doubt remains as to the absence of adverse effects on the integrity of the site, approval should be refused[7] (subject to the considerations of alternative solutions, imperative reasons of overriding public interest and the provision of compensatory measures as set out in regulations 64 & 68).

3.3.3 An appropriate assessment requires all aspects of the project which could affect the site, its species and its conservation objectives to be identified in the light of the best scientific knowledge in the field.[8] The competent authority, "taking account of the conclusions of the appropriate assessment of the implications…for the site concerned, in the light of the conservation objectives, are to authorise such activity <u>only if they have made certain</u> that it will not adversely affect the integrity of the site. That is the case where <u>no reasonable scientific doubt remains as to the absence of such effects</u>"[9].

3.3.4 Office of the Deputy Prime Minister (ODPM) Circular 6/2005 states that the 'integrity of the site' should be defined as 'the coherence of the site's ecological structure and function, across its whole area, or the habitats, complex of habitats and/or populations of species for which the site is or will be classified'.[10] A site can be described as having a high degree of integrity where the inherent potential for meeting site conservation objectives is realised, the capacity for self-repair and self-renewal under dynamic conditions is maintained, and a minimum of external management support is required. When looking at the 'integrity of the site', it is therefore

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[6] CJEU Case-127/02; [2004] ECR-7405 at [45].

[7] [56]-[57].

[8] [61].

[9] [59].

[10] See ODPM Circular 6/2005 para. 20.

important to take into account a range of factors, including the possibility of effects manifesting themselves in the short, medium and long-term".[11]

3.3.5 As is clear from the requirements of the Habitats and Offshore Regulations, the assessment of integrity is to be considered by reference to the impact of the project alone and in combination with other plans and projects. As clearly set out in *Waddenzee*, para 61:

> **61** In view of the foregoing, the answer to the fourth question must be that, under Article 6(3) of the Habitats Directive, **an appropriate assessment of the implications for the site concerned of the plan or project implies that, prior to its approval, all the aspects of the plan or project which can, by themselves or in combination with other plans or projects, affect the site's conservation objectives must be identified in the light of the best scientific knowledge in the field**. The competent national authorities, taking account of the appropriate assessment of the implications of mechanical cockle fishing for the site concerned in the light of the site's conservation objectives, are to authorise such an activity only if they have made certain that it will not adversely affect the integrity of that site. That is the case where no reasonable scientific doubt remains as to the absence of such effects. (emphasis added)

3.4 Environmental Impact Assessment

3.4.1 The Infrastructure Planning (Environmental Impact Assessment) Regulations 2017 state that development consent cannot be granted for EIA development unless the decision-maker has taken into account environmental information including an environmental statement which describes the significant effects, including cumulative effects, of the development on the environment. This will include effects on all wild bird species whether SPA species or not.

3.4.2 NPS EN-3 acknowledges that offshore wind farms have the potential to impact on birds through collision with rotating blades, direct habitat loss, disturbance from construction activities, displacement during the operational phase (resulting in loss of foraging/roosting area) and impact on bird flight lines (i.e. barrier effect) and associated increased energy use by birds for commuting

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[11] See too the European Commission Guidance; Wind Energy Developments and Natura 2000, 2011, page 82-83, paragraph 5.5.3.

flights between roosting and foraging areas.[12] These potential impacts have been taken into account by the RSPB and its remaining concerns with the applications are set out below, in the context of the legislative provisions summarised above, in particular those relating to appropriate assessment.

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[12] Paragraph 2.6.101; see paragraphs 2.6.100-110 and 2.6.58-71 generally. Effects on foraging areas outside an SPA are to be taken into account when assessing the effects on bird populations of the SPA: see *Hargreaves v Secretary of State for Communities and Local Government* [2011] EWHC 1999 (Admin), which concerned effects on pink-footed geese which commuted inland from their roosting sites in the SPA to feed on grain and winter cereal crops on fields adjacent to the proposed development site.

4 Offshore Ornithology

4.1 Introduction

4.1.1 Our comments in this section relate primarily to the following documents:

- Environmental Statement, Ch. 13 Offshore Ornithology (doc. 6.1.13)
- Environmental Statement Appendix 13.1 Ornithology Technical Appendix Annex 3 (doc. 6.2.13.1)
- Information for Habitats Regulations Assessment (doc. 5.3)
- The Applicant's Response to S51 Advice (doc. AS-006)

Note that our comments refer to both the original application documents and the updated information provided in the Applicant's Response to S51 Advice.

4.1.2 We have significant concerns regarding the findings of some of the impact assessments. As a result of the methodological concerns (set out below) and our initial indicative[13] recalculation of collision risk using the Band (2012) model and the preferable Marine Scotland version of the stochastic model (McGregor *et al.*, 2018), the RSPB considers that the impacts have not been adequately assessed and, as such consider that an adverse effect on the integrity of the following SPAs and their species cannot be ruled out as follows:

- The impact of collision mortality on the kittiwake population of the Flamborough and Filey Coast SPA alone and in-combination with other plans and projects;
- The impact of collision mortality on the gannet population of the Flamborough and Filey Coast SPA alone and in-combination with other plans and projects; and
- The impact of collision mortality on the lesser black-backed gull population of the Alde-Ore Estuary SPA alone and in-combination with other projects.

4.1.3 In addition, we consider that insufficient evidence has been provided to rule out potential

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[13] Although we did run both deterministic and the stochastic versions of the model, as stated in our Relevant Representations, using the data presented in Appendix 13.1 and associated Annexes, as some of the limitations of the data were not immediately apparent and are poorly explained, particularly the use of means of medians rather than true means, we therefore did not appreciate that these data were not appropriate for the analysis. As detailed in the main text, the Applicant has not provided full details of the parameters for the collision risk models and therefore our initial calculations can only be indicative (and therefore are not presented). However once the Applicant has provided all the information required [noting the Examining Authority's written question 3.3 overall but specifically 3.3(e), (f) and (j)] we wish to re-examine this issue, including re-running our calculations, which we will provide along with our concerns to the Applicant. If it is not possible to resolve those concerns, we will present them in our next submission to the Examining Authority.

significant impacts on the following North Sea populations:

- Cumulative collision mortality to North Sea populations of kittiwake and great black-backed gull; and
- Cumulative operational displacement to North Sea populations of red-throated diver, guillemot and razorbill.

4.2 Overarching Concerns Regarding the Assessment of Collision Risk

4.2.1 In this section, we describe our overarching methodological concerns with the Applicant's assessment of collision risk. Later in the document, we explain how this has affected the outcomes for individual species.

Stochastic Collision Risk Model

4.2.2 In order to predict the collision risk mortality of an offshore wind farm in the UK, the Band (2012) collision risk model (CRM) has previously been used in assessment. This model uses a number of input parameters, such as bird size, flight speed and turbine blade dimensions, to calculate the probability of a bird that passes through the swept area of a turbine blade colliding with that blade. For this deterministic model these input parameters were defined as single values with no indication of variability around them. In reality, most of the parameters will exhibit a considerable degree of variability and stochastic collision risk modelling has been developed to allow this to be incorporated into the model and thus generate a potential range of output predicted collision mortalities.

4.2.3 Masden (2015) created a stochastic version of the model as proof of concept, in order to demonstrate the feasibility of doing so, although the model remained incomplete. McGregor *et al*., (2018), under commission of Marine Scotland Science and overseen by an expert steering panel, produced a revised and fully tested stochastic model which has received widespread stakeholder acceptance (see, for example, NE's answer to the Hornsea Project Three Examiners' Question Q1.2.56[14]). By contrast, the Applicant has presented an entirely new and untested version that does not follow a recognised methodology, with insufficient detail provided as to

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[14] https://infrastructure.planninginspectorate.gov.uk/wp-content/ipc/uploads/projects/EN010080/EN010080-001239-EN10080%20261379%20Annex%20A%20Hornsea%20Three%20-%20NE%20response%20to%20ExA%20Written%20Questions.pdf

how it incorporates variability and uncertainty in the input parameters or how it overcomes the statistical difficulties of non-independence (the degree of interrelation) of some of these parameters. The RSPB therefore does not agree that the model presented by the Applicant is fit for purpose and recommend that the Marine Scotland (McGregor *et al*., 2018) model version is used in preference.

4.2.4 We have carried out an initial recalculation of potential collision risk using both the original Band deterministic model and the preferred Marine Scotland stochastic model. However these results are only indicative as the Applicant has not presented the full information needed to carry out a more complete re-modelling process. However these initial results indicate that the Applicant's collision risk modelling is likely to significantly underestimate mortality for key species.

4.2.5 The Band (2012) model guidance is very clear on how novel methods of collision risk assessment should be carried out:

> "Developers and their advisors are encouraged where appropriate to go beyond the core requirements set out in this guidance; but where they do so, the standard approach of this guidance should also be pursued so as to make clear how the results of any improved methods differ from that of the standard approach."

In not presenting the results for the standard approach to CRM or providing sufficient data to allow such an assessment to be done, the Applicant has failed to follow the guidance and so in terms of collision risk modelling this is an inadequate assessment from which it is impossible to draw meaningful conclusions (and as set out above appears to result in an underestimation of potential impacts).

4.2.6 The Applicant's presentation of the CRM is inadequate because it does not present the full list of parameters required to run the model including:

- Width of windfarms
- Latitude of Windfarms
- Density of birds in flight including mean density, not median (see below)

- Details of whether the proportion of birds within the rotor swept area (PCH) for Option 1 was derived from values for birds in the windfarm footprint or for birds in the windfarm plus buffer.

Notwithstanding these caveats (above and in footnote 13) our preliminary indicative results suggest that the Applicant's approach seriously underestimates collision risk.

Use of median bird densities within the CRM

4.2.7 The Applicant has presented both deterministic and stochastic versions of the CRM (see above). For the deterministic version (Band 2012) of the CRM the correct value to use for bird density is the mean monthly value. As detailed in Environmental Statement Appendix 13.1 Ornithology Technical Appendix Annex 3 (doc. 6.2.13.1), the values used by the applicant appear to be median values, which will result in the model predicting considerably lower collision mortalities. The Applicant presents mean values for density in Annex 3 of Appendix 13.1, but these are not true means, they are means of medians. These have been calculated by taking the median from monthly surveys and then calculating the mean for that month across the two, or sometimes three, years. This is a different mean from that normally used in CRM, which would give a higher predicted mortality.

Nocturnal activity rate

4.2.8 We do not agree with the changes in nocturnal activity rates proposed. For example for gannet, we welcome the latest published evidence review (Furness *et al*., 2018), however we are concerned that the Applicant has not used the values presented in this paper, 8% and 3% for the breeding and non-breeding seasons respectively, rather they have used 4.3% and 2.3%, which will result in lower predicted mortalities. We are also concerned that by using revised nocturnal activity rates for gannet (and this is also applicable to kittiwake) mortalities are potentially underestimated because in doing so there is no account for the potential interaction between survey timing and diurnal behavioural patterns. Peaks in foraging activity at first and last light (see for example Fig. 3 in Furness *et al*. 2018) will not be accounted for in the assessment if these did not coincide with surveys (the timings of which are currently unknown, but likely to be midday if aerial), and the survey may have been carried out at a time of much lower activity. Thereby the application of the revised nocturnal activity rates either recommended by Furness *et al*., (2018) or the rates suggested by the Applicant could result in underestimates of collision risk. We therefore request that details of the timings of survey are presented.

4.2.9 For kittiwake the Applicant cites a paper in preparation that has neither been peer reviewed or made available to the Examining Authority, and therefore cannot be accepted, particularly when this unseen evidence for a change in nocturnal activity rates will result in an unjustified reduction in predicted mortalities. No evidence at all is presented for the use of lower nocturnal activity rates for other species.

4.2.10 It is also not clear how these revised rates account for the distinction between the definition of daylight as used in the Band model and with the official concept of 'twilight' and 'night'. This is an issue as the Band (2012) model considers the nocturnal period as between sunset to sunrise and so treats flight activity that occurs at twilight as being within the nocturnal flight period. Evidence from tagging shows that an important number of seabirds actively forage at twilight.

4.2.11 The Applicant's Response to S51 Advice (doc. AS-006) incorporates a proposal to apply the reductions in nocturnal activity rates to other windfarms as part of the cumulative/in-combination assessment. We do not agree with this proposal for the reasons explained above, noting in particular that it is unlikely that the timings of surveys undertaken for other windfarms are known. Furthermore any change in nocturnal activity rate cannot be applied *post hoc* to collision mortality, the model itself needs to be rerun as the modelling calculates the reduction in activity at night through the interaction of nocturnal activity and the latitude of specific wind farm, which therefore is a calculation specific to that wind farm, necessitating a rerun of the model.

Non-adult birds in breeding season mortality assessments

4.2.12 The Applicant notes in several species assessments that numbers of collisions are significantly reduced when breeding adults only are considered. The RSPB acknowledge that some birds present in the Norfolk Vanguard area during the breeding season will be juveniles and non-breeding adults from SPA colonies. However such individuals will all at some point in their life cycle be associated to a breeding colony and the significant proportion that go on to breed will do so at a colony. Further uncertainty exists as to which colony the birds will breed at and this will not necessarily be at the natal colony. As such, some appointing to specific colonies should be carried out. Consequently effects on these birds, even when not breeding, will impact on future breeding at the colony. Such effects should be incorporated into PVAs to enable assessment of their significance. We therefore do not agree that calculations of collision risk for breeding birds from SPAs can be reduced to the adults only figure.

Potential Biological Removal

4.2.13 Potential Biological Removal (PBR) is used in justification of conclusions of no adverse effect on the integrity of the SPAs for both gannet and kittiwake. The RSPB disagree with the use of PBR in this context following the publication of the RSPB Practitioner's Perspective (Green *et al*., 2016) and the reviews by Cook and Robinson (2015) and O'Brien *et al.* (2017), and therefore support NE's position (stated in their Relevant Representations, doc. RR-106) that PBR does not provide an appropriate threshold for this purpose.

4.2.14 PBR was designed to manage whaling quotas by detecting unsustainable mortality in a population leading to risk of its extinction. Whereas SPAs are required to be designated to protect key areas for rare, threatened and regulatory migratory species in order to maintain or where necessary restore, these populations of conservation importance. Levels of acceptable mortality derived from PBR will therefore be higher than those acceptable for a population to continue to meet the conservation objectives of a SPA.

4.2.15 Where population modelling is required to inform an assessment, this should be based on Population Viability Analysis (PVA). PVA enables comparison of the change in population size with and without the project after several years, thereby presenting an indication of the magnitude of change attributable to the proposal and is therefore more suitable for assessing the effects of a project on an SPA. We would also prefer to see these outputs presented in the form of counterfactuals of population size. These are a robust and informative metric (Green *et al*., 2016, Cook and Robinson, 2015 and O'Brien *et al.,* 2017, Jitlal *et al*., 2017) which indicate the percentage difference between the population with or without additional mortality at the end of the lifetime of the wind farm.

Density dependent outputs of PVA

4.2.16 We do not accept the arguments for the use of PVA outputs incorporating compensatory density dependence. The reasons for this are outlined in Green *et al*. (2016) and the reviews by Cook and Robinson, 2015 and O'Brien *et al*., 2017, and are not that density dependence does not exist, but rather that we do not have the means to accurately quantify the strength and form of it in a biologically meaningful way in order to incorporate it into PVA. Whilst we accept that density dependence is likely to exist in seabird populations, precise species and colony specific knowledge of its size and shape are needed to correctly parameterise the population models. This is important to acknowledge because density dependence is not always compensatory, but can also be depensatory, slowing the rate of population growth at lower population densities. In other words, a population decline arising from an offshore wind farm could have larger consequences on the population than are predicted by the compensatory density dependent or even density independent models. Horswill and Robinson (2015) identified depensation occurring in three gull species (black-legged kittiwake, black-headed gull and herring gull). As such it would be very wrong to simply assume that density independent outputs are highly precautionary, rather that density independent outputs are the most sensible to use for assessment.

Reductions in windfarm capacity post-consent

4.2.17 It is stated that many of the collision estimates for other windfarms are based on higher numbers of turbines than were actually installed. Based on a method of updating collision estimates presented by EATL (2016) this is stated to overestimate in-combination mortality by 400 for gannets of Flamborough and Filey Coast SPA and 550 for kittiwakes of Flamborough and Filey Coast SPA and Flamborough Head and Bempton Cliffs SPA, and 20 for lesser black-backed gulls of the Alde-Ore Estuary SPA. This is an acceptable point for windfarms where the DCO has been amended and therefore there is legal certainty regarding the reduction, but where windfarms still have their original DCOs, it is not appropriate to do anything less than assess the full extent of those DCOs when considering in-combination/cumulative effects.

4.3 Collision Risk to Kittiwakes of Flamborough and Filey Coast SPA

Underestimation of collision mortality

4.3.1 Due to the concerns explained above regarding the stochastic model used for the assessment of collision risk and the reductions to collision mortality applied as a result of the nocturnal activity rate reviews the RSPB have carried out an initial re-calculation of the collision risk modelling using the Band (2012) deterministic model and the Marine Scotland stochastic model (McGregor *et al.*, 2018), without the nocturnal activity rate reductions (indicative only and see caveats to the input data, above). This shows that the Applicant's model underestimates collision mortality for kittiwakes of Flamborough and Filey Coast SPA, and it will produce significantly increased predictions when it is re-run using mean (rather than median) monthly bird densities.

Nocturnal activity rate

4.3.2 We do not agree with the changes in nocturnal activity rate proposed for the reasons discussed above. We are also concerned that the value presented for kittiwake is based on unpublished evidence (which we understand has not yet been submitted for publication and has therefore not been peer-reviewed) which does not appear to form part of the examination documentation. We are therefore unable to assess the robustness of this study. We therefore recommend that the range of rates recommended by Natural England and based on Garthe and Hüppop (2004) are retained for this species.

Breeding season definitions

4.3.3 We have concerns about the manner in which the Applicant has defined biological seasons for kittiwake. The use of the 'migration-free breeding season' means that months where breeding and migration can overlap are excluded from the analysis of breeding season impacts, which artificially reduces the duration of the breeding season and hence risks underestimating collision mortality of breeding birds.

4.3.4 For kittiwake, the migration-free breeding season excludes March-April and August, which again reduces the number of collisions. The first kittiwakes arrive at the Flamborough and Filey Coast colony in February, with most birds back by March and remaining until August, hence there is a strong argument for considering March, April and August to be part of the breeding season.

4.3.5 Given that recent tracking data (Wischnewski *et al.*, 2018). shows connectivity of breeding kittiwakes from that colony with the project site, the definition of 'breeding season' as presented in Furness (2015), should be used, except where colony specific evidence clearly suggests otherwise. If figures for the migration-free breeding season are to be presented, we consider that it would be necessary to attribute birds in the crossover months to breeding and dispersal in order to ensure collision risk to breeding birds is not underestimated.

Apportioning of mortality to Flamborough and Filey Coast SPA

4.3.6 We have concerns about some of the figures used for apportioning of collision mortality to SPAs and the evidence used to support this. The estimated proportion of kittiwakes from Flamborough and Filey Coast SPA used in the HRA is 16.5% and is not based on site-specific historical estimates. Consequently we have seen no evidence presented to support the figure selected.

4.3.7 Notwithstanding the Applicant's criticisms of the FAME and STAR tracking of kittiwakes, which are dealt with below, the assessment does not take into account more recent tracking data. This means we cannot agree with the assertions in para. 229 of the Information for HRA (doc. 5.3) that only a very small percentage of breeding adults from the SPA will be at risk of collision at the Norfolk Vanguard site and we consider that the value used for apportioning kittiwake collision mortalities to the Flamborough and Filey Coast SPA will considerably underestimate the actual impact.

4.3.8 The RSPB advocate the use of the theoretical approach as laid out in SNH guidance (SNH 2018) amended, as per the guidance, to take into account recent tracking data from Flamborough and Filey Coast SPA. This theoretical approach is based on foraging range and three colony-specific weighting factors: colony size, distance of colony from site and the areal extent of the open sea within the foraging range of the relevant species.

4.3.9 Tracking of kittiwake from the Flamborough and Filey Coast SPA has been carried out from 2010 to 2015 and 2017-2018. The tags used between 2010 and 2015 were GPS tags that required recapturing of the birds and typically were only able to collect data for a period of a few days, around the time of late incubation and early hatching when the birds are likely to remain closest to the nest. The tags used in 2017-2018 were very lightweight tags that allowed for remote downloading of data so there was no need to recapture the birds. A different attachment method was also used which meant that the tags remained on for longer, between 20 and 29 days. This means that kittiwakes were tracked for a longer part of the breeding season including when adults were provisioning large chicks (that can be left for longer than small chicks). The tracking data for 2017 are presented in Wischnewski *et al*. (2018) and has been made available to the Applicant. The foraging ranges recorded during 2017 were greater than those previous recorded, with a maximum foraging range of 324km , and this is most likely to be a function of the longer tracking period. The tracking in 2017 also showed a high degree of overlap with Norfolk Vanguard. Data from 2018 is currently being analysed.

4.3.10 As such we recommend that the applicant, in discussion with NE and RSPB, revise and recalculate the apportioning value for kittiwake using the amended SNH method which takes into account these recent tracking data. This value is likely to be higher than the current arbitrary suggested value.

4.3.11 Marine Scotland have been developing a tool that uses the information from Wakefield *et al*., (2017) to apportion birds to colonies. This is currently under internal review at Marine Scotland and is likely to be available soon. Once available it is likely to provide the best method for apportioning, for some species, including kittiwake.

Kittiwake productivity at Flamborough and Filey Coast SPA

4.3.12 Para. 96 of the Applicant's Response to S51 Advice (doc. AS-006) refers to the "continued relatively high breeding success" of the Flamborough and Filey Coast SPA colony. However, recent census data (Aitken *et al*., 2017) has shown that kittiwake productivity has declined rapidly at the SPA and this will have severe long-term impacts on the population growth. This means that it may no longer be appropriate to apply the outputs from PVAs produced for historic projects without revision of the parameters employed.

Population modelling

4.3.13 The Applicant's Response to S51 Advice (doc. AS-006) in paras. 92-93 and the Information for HRA (doc. 5.3) paras. 248-249 discuss the outputs of population modelling for the Flamborough and Filey Coast SPA kittiwake population carried out for the Hornsea Project One offshore windfarm. Whilst we welcome the inclusion of the outputs of the density independent version of the model, the concerns above regarding more recent changes to demographic rates may apply. We would also prefer to see these outputs presented in the form of counterfactuals of population size. These are a robust and informative metric which indicate the percentage difference between the population with or without additional mortality at the end of the lifetime of the wind farm.

4.3.14 The Applicant also reports on the PBR calculation for Hornsea Project One, despite the recommendation from Natural England that this metric no longer be used in this context (see our comments in section 4.2). We do not agree, therefore, with the statement in para. 97 of the Applicant's Response to S51 Advice (doc. AS-006), that the in-combination kittiwake mortality remains below the "sustainable levels estimated using PBR". In light of the publication of the RSPB Practitioner's Perspective (Green *et al*., 2016) and the reviews by Cook and Robinson (2015) and O'Brien *et al*. (2017) as well as NE's position and advice (as stated in their Relevant Representations, doc. RR-106), PBR outcomes should not be included when considering potential impacts and whether it is possible ascertain that there will not be adverse effects on the integrity of SPAs designated to protect rare, threatened and regulatory migratory species in order to maintain or where necessary restore, these populations of conservation importance. Especially since determinations on levels of acceptable mortality derived from PBR will be higher than those acceptable for a population to continue to meet the conservation objectives of a SPA.

Criticisms of kittiwake tracking data

4.3.15 The Applicant raises a number of issues with regard to the suitability of tracking data obtained as part of the FAME and STAR projects for use in the assessment. However the Applicant's Information for the Habitats Regulations Assessment (doc. 5.3) contains a number of misinterpretations and erroneous assertions.

4.3.16 In para. 224 it is claimed that the longest foraging trips from FAME/STAR kittiwake data were largely from colonies where the breeding success was zero or close to zero. This is stated without reference and is incorrect. The longest trips were recorded from Flamborough and Filey, where breeding success was comparatively high over the time of tracking.

4.3.17 It is true, as stated in para. 224, that study birds tend to be reachable. This could be from the top of a cliff, or the bottom. This, however, does not necessarily mean that the birds are at the periphery of the colony. In some colonies all birds are reachable, especially with the long pole used at Flamborough and Filey. The periphery problem is true for Bempton due to accessibility issues at the high cliff sections and there have been studies showing lower breeding success at the edge of colonies, which is why we are currently trying to tag birds at the centre of the colony. However, an examination of breeding success in 2017 found that it was generally low and breeding success at the tagging site in Flamborough is similar to the average for the whole SPA (Wischnewski *et al.,* 2018).

4.3.18 The claim in para. 224 that tagged birds were more likely to have failed is also incorrect. For the FAME and STAR data, where remote download tags were used, birds were re-caught on the nest so it is impossible to re-catch tagged birds if they have failed breeding as they would not return to their nest, or sit tight on the nest, if they were not protecting chicks. Furthermore, the fact that recaptured birds must still be breeding means the tagging study could actually be selecting more successful birds. In addition, there are no studies that we are aware of which demonstrate the effect of colony position on the foraging behaviour of seabirds, since the uncatchable birds cannot be tested. We agree that it is plausible that there is an effect, but whether the effect is larger than other factors determining where these birds are feeding (food availability, competition from conspecifics, seabird type etc.) is doubtful.

4.3.19 It should be noted that the FAME and STAR data are viewable on seabirdtracking.org. It can also be requested from the RSPB. Data have also been freely provided to a number of developers and their consultants.

4.3.20 With reference to para. 225, it is true that logger effects deserve the utmost attention. This is why the RSPB conducted trials in the first year of tagging kittiwakes using this technique and found no effect on foraging trip duration or breeding success. The reference to adverse effects from devices weighing more than 3% of a bird's body weight (Phillips *et al*, 2003) was for procellarids (petrels, prions and shearwaters) using long term deployments. In the study by Chivers *et al*. (2016), birds cited as having a 30% reduction in flight activity were actually equipped with two devices at once - a GPS tag of the same type used in FAME/STAR, plus an additional accelerometer. The paper does not give the weights of the devices separately but the tags are significantly larger than those used in FAME and STAR so a comparison is not necessarily valid. Furthermore, while it is true that Chivers *et al*. (2016) found that there was a reduction in flight behaviour in tagged kittiwakes carrying very heavy tags of more than five grams compared to birds carrying tags of only a gram, they also found that there was no difference in trip duration and the number of trips in 24 hours, and suggested that birds with heavier tags actually travelled shorter distances rather than longer ones (which also seems a bit more intuitive). Thus, tag effects do not really explain the longer ranging trips in tagged kittiwakes.

4.3.21 Despite the need for the scientific community to better understand and minimise device effects, tagging represents the best way to determine foraging locations of birds from a specific colony. The tagging conducted in 2017 from Flamborough and Filey Coast SPA using tags that were less than 2.5 % of the birds' body weight observed even longer foraging ranges with multiple actively breeding birds visiting the Norfolk Vanguard site (Wischnewski *et al*., 2018).

4.3.22 The study by Heggøy *et al*. (2015) (also referred to in para. 225) showing increased stress hormone in kittiwakes carrying loggers is potentially not comparable with RSPB tracking as it used tail attachments that have potential to increase flight costs by shifting the centre of gravity.

4.3.23 With reference to para. 226, Kidawa *et al*. (2012) found a reduction in body mass of chicks from birds that had been tagged and also recorded longer lasting trips but not longer distance ones. Also, they tagged little auks, which are a diving seabird species similar to penguins and for which some studies indicated that dorsal tag attachment increased drag and reduced their diving efficiency, thus increasing potential tag effects (ie. Ballard *et al*. 2001, Hamel *et al*. 2004).

4.3.24 It is also important to note that foraging trip duration is not the same as trip range. Birds going on longer lasting trips are not necessarily travelling to more distant sites; it is only known that they are away from the colony for longer. Therefore trip duration does not give any insight into the birds' distribution.

4.3.25 The quoted study by Passos *et al*. (2010) looked at the effect of additional weight on Cory's shearwater trip characteristics using geolocators. However shearwaters are, from a flight energetics perspective, very different from kittiwakes. They use dynamic soaring a lot which helps them to cover large distances without expending much energy, similar to albatrosses (Arnould *et al*. 1996), causing them to have regular foraging ranges that are more than four times larger (in this case) than foraging ranges of kittiwakes. Furthermore, geolocators can have errors of around

200 km therefore the conclusion drawn from this that attaching loggers increases the duration of foraging trips may not be applicable to kittiwakes in this case.

4.3.26 Ponchon *et al*. (2015) did show prospecting movements in birds that fail early during the breeding season (during incubation). However, para. 227 again incorrectly implies that FAME/STAR birds were unsuccessful breeders. Furthermore none of the tracks collected from birds that failed in the 2017 chick rearing period within the SPA include visits to other colonies. They exclusively show offshore trips to similar foraging areas to the ones visited by actively breeding birds.

4.3.27 The statement that "in winter" kittiwake distribution is pelagic in para. 231 is not specific enough. The time that some, not all, birds discussed in Bogdanova *et al*. (2017) are in the mid-Atlantic is not the whole non-breeding period. Furthermore, Bogdanova *et al*. (2017) found that successful breeders tended to stay closer to the colony, therefore the windfarm could be in contact with the most successful breeders, meaning that any collision mortality could have a greater impact on the population since it is affecting 'core' breeders.

4.3.28 The Applicant cites Carroll *et al.,* (2017) as evidence of limited connectivity between Flamborough and Filey Coast SPA and Norfolk Vanguard. Carroll *et al.,* (2017) used data from the tracking of kittiwake from the Flamborough and Filey Coast SPA from 2010 to 2015. Subsequent tracking was carried out in 2017 and 2018. The tags used between 2010 and 2015 were GPS tags that required recapturing of the birds and typically were only able to collect data for a period of a few days, around the time of late incubation and early hatching when the birds are likely to remain closest to the nest. The tags used in 2017-2018 were very lightweight tags that allowed for remote downloading of data so there was no need to recapture the birds. A different attachment method was also used which meant that the tags remained on for longer, between 20 and 29 days. This means that kittiwakes were tracked for a longer part of the breeding season including when adults were provisioning large chicks (that can be left for longer than small chicks). The tracking data for 2017 are presented in Wischnewski *et al.* (2018) and have been made available to the Applicant. The foraging ranges recorded during 2017 were greater than those previous recorded, with a maximum foraging range of 324km, and this is most likely to be a function of the longer tracking period. Furthermore the tracking in 2017 showed a high degree of overlap with Norfolk Vanguard. These more recent data should be used in assessment of connectivity. Data from 2018 are currently being analysed.

4.3.29 In summary, we do not consider that the Applicant has presented information which justifies the exclusion of the FAME/STAR (or subsequent) tracking data from that used to inform consideration of kittiwake foraging range and connectivity with the Norfolk Vanguard site. Therefore, our recommendation that apportioning is revisited using these data still applies.

Conclusions regarding kittiwake collision mortality and adverse effects on the integrity of the Flamborough and Filey Coast SPA

4.3.30 The Applicant concludes that there will no adverse effect on the integrity of the Flamborough and Filey Coast SPA as a result of collision mortality to kittiwakes alone (para. 253 of the Information for HRA (doc. 5.3)) or in-combination (para. 97 of their Response to S51 Advice (doc. AS-006) and para. 254 of the Information for HRA (doc. 5.3). We do not agree there can be sufficient confidence in these conclusions, as we consider the Applicant's calculations and presentation are flawed and that their figures underestimate mortality. As such, we do not agree that there can be confidence in conclusions of no adverse effect on the integrity of the site.

4.4 Collision Risk to Gannets of Flamborough and Filey Coast SPA

Underestimation of collision mortality

4.4.1 Due to the concerns explained above regarding the stochastic model used for the assessment of collision risk and the reductions to collision mortality applied as a result of the nocturnal activity rate reviews the RSPB have carried out an initial re-calculation of the collision risk modelling using the Band (2012) deterministic model and the Marine Scotland stochastic model (McGregor *et al*., 2018), without the nocturnal activity rate reductions (indicative only and see caveats to the input data, above). This shows that the Applicant's model underestimates collision mortality for gannets of Flamborough and Filey Coast SPA, and it will produce significantly increased predictions when it is re-run using mean (rather than median) monthly bird densities.

Nocturnal activity rate

4.4.2 We do not agree with the changes in nocturnal activity rates proposed for the reasons discussed above. We are also concerned that the nocturnal activity rate presented for gannet in the application documents is also not in accordance with the latest review (Furness *et al*., 2018) which recommends 8% in the breeding season and 3% in the non-breeding season. The values used in the assessment, 4.3% and 2.3% respectively, are not supported by the peer-reviewed paper and will result in predictions of fewer collisions. This approach therefore represents an unjustified lack of precaution in the assessment.

Breeding season definitions

4.4.3 We have concerns about the manner in which the Applicant has defined biological seasons for gannet. The use of the 'migration-free breeding season' means that months where breeding and migration can overlap are excluded from the analysis of breeding season impacts, artificially reducing the duration of the breeding season and hence risks underestimating collision mortality of breeding birds.

4.4.4 For gannet, the migration-free breeding season excludes March and September, which reduces the number of predicted collisions. However, gannets start arriving at the Flamborough and Filey Coast colony in January and establishing their nest sites in March. Whilst peak fledging is in August, some birds are still fledging in September, hence there is a strong argument for considering March and September to be part of the breeding season.

4.4.5 Given that Norfolk Vanguard is within the mean-maximum foraging range of gannets from Flamborough and Filey Coast SPA, the definition of 'breeding season' as presented in Furness (2015), should be used, except where colony specific evidence clearly suggests otherwise. If figures for the migration-free breeding season are to be presented, we consider that it would be necessary to attribute birds in the crossover months to breeding and dispersal in order to ensure collision risk to breeding birds is not underestimated.

Gannet avoidance rate

4.4.6 We maintain our position that, whilst we agree with the use of a 98.9% avoidance rate for non-breeding gannets, in the breeding season, a 98% avoidance rate is appropriate. Cleasby *et al*., (2015), while not discussing avoidance rates, demonstrated that foraging birds are at more risk of collision than commuting birds. In order to provision chicks, gannets will need to forage more during the breeding season and will also be constrained by central place foraging. Such behavioural differences are likely to result in changes in avoidance behaviour (Cook *et al*., 2018), and since the figures used for the calculation of avoidance rates advocated by the SNCBs are largely derived from the non-breeding season for gannet (Cook *et al.*, 2014 and Cook *et al*., 2018) we recommend a more precautionary avoidance rate of 98% should be presented for the breeding season. The current SNCB advice also highlights that due consideration should be given to uncertainty in collision risk estimates, including the use of confidence intervals around the avoidance rates and flight height estimates.

Effects of harvesting on gannet populations

4.4.7 Para. 86 of the Applicant's Response to S51 Advice (doc. AS-006) and para. 218 of the Information for HRA (doc. 5.3) use harvesting at Sula Sgeir to argue that gannet populations are robust to human impacts. The effect of harvesting by humans would be dependent on demographic rates of the individual colony and we therefore do not agree that such generalisations are robust.

Population modelling

4.4.8 The Applicant's Response to S51 Advice (doc. AS-006) in paras. 81-84 and the Information for HRA (doc. 5.3) paras. 213-216 discuss the outputs of population modelling for the Flamborough and Filey Coast SPA gannet population carried out for the Hornsea Project Two offshore windfarm. Whilst we welcome the presentation of the outputs of the density independent version of the model, we would prefer to see these outputs presented in the form of counterfactuals of population size. These are a robust and informative metric which indicate the percentage difference between the population with or without additional mortality at the end of the lifetime of the wind farm.

4.4.9 The Applicant also reports on the PBR calculation for the East Anglia ONE offshore windfarm, despite the recommendation from Natural England that this metric no longer be used in this context (see our comments in section 4.2). We do not agree, therefore, with the statement in para. 88 of the Applicant's Response to S51 Advice (doc. AS-006) and para. 221 of the Information for HRA (doc. 5.3), that the in-combination gannet mortality remains below "sustainable levels". In light of the publication of the RSPB Practitioner's Perspective (Green *et al*., 2016) and the reviews by Cook and Robinson (2015) and O'Brien *et al.* (2017) as well as NE's position and advice (as stated in their Relevant Representations, doc. RR-106), PBR outcomes should not be included when considering potential impacts and whether it is possible ascertain that there will not be adverse effects on the integrity of SPAs designated to protect rare, threatened and regulatory migratory species in order to maintain or where necessary restore, these populations of conservation importance. Especially since determinations on levels of acceptable mortality derived from PBR will be higher than those acceptable for a population to continue to meet the conservation objectives of a SPA.

Conclusions regarding gannet collision mortality and adverse effects on the integrity of Flamborough and Filey Coast SPA

4.4.10 The Applicant concludes that there will be no adverse effect on the integrity of the Flamborough and Filey Coast SPA as a result of collision mortality to gannets from the Norfolk Vanguard project alone (para. 220 of the Information for HRA (doc. 5.3)) or in-combination with other projects (para. 88 of the Response to S51 Advice (doc. AS-006) and para. 221 of the Information for HRA (doc. 5.3)). We do not agree there can be sufficient confidence in these conclusions, we consider the Applicant's calculations and presentation are flawed and that their figures underestimate mortality and cannot be relied on. As such, we do not agree that there can be confidence in conclusions of no adverse effect on the integrity of the Flamborough and Filey Coast SPA.

4.5 Collision Risk to Lesser Black-backed Gulls of the Alde-Ore Estuary SPA

Underestimation of collision mortality

4.5.1 Due to the concerns explained above regarding the stochastic model used for the assessment of collision risk and the reductions to collision mortality applied as a result of the nocturnal activity rate reviews the RSPB have carried out an initial re-calculation of the collision risk modelling using the Band (2012) deterministic model and the Marine Scotland stochastic model (McGregor *et al.*, 2018), without the nocturnal activity rate reductions (indicative only and see caveats to the input data, above). This shows that the Applicant's model underestimates collision mortality for lesser black-backed gulls of the Alde-Ore Estuary SPA, and it will produce significantly increased predictions when it is re-run using mean (rather than median) monthly bird densities.

Nocturnal activity rate

4.5.2 We do not agree with the changes in nocturnal activity rate proposed for the reasons discussed in our overarching concerns, and because the cited evidence (EATL, 2015) has not been subject to peer-review. Nocturnal activity is one of a number of variables included in the Band model process, and recent work has indicated how important consideration of these variables is.

4.5.3 For large gulls, there is no peer-reviewed evidence for a change in the factor that is being used. The current factor is derived from the expert opinion collected by Garthe and Hüppop (2004) and this use is endorsed by Band (2012). A review of seabird vulnerability to offshore wind farms (Furness *et al*., 2013) recommended that no changes be made to the nocturnal activity scores for these species. Partial analysis of data from thermal imaging cameras was carried out in the Skov *et al*. (2018) ORJIP Bird Collision Avoidance report, but was incomplete and did not fully account for the distinction between the definition of daylight as used in the Band model and with the official concept of 'twilight' and 'night'. This is an issue as the Band (2012) model considers the nocturnal period as between sunset to sunrise and so treats flight activity that occurs at twilight as being within the nocturnal flight period. Evidence from tagging shows that a number of seabirds actively forage at twilight. We therefore do not consider that any change should be made to the recommended nocturnal activity rates.

Apportioning of mortality to SPAs

4.5.4 The methods used for apportioning collision mortality of lesser black-backed gulls to the Alde-Ore Estuary SPA are inadequately explained, with insufficient reference to current knowledge and a lack of precaution. Furthermore in calculating the number of non-SPA birds the Applicant gives a rounded up figure of 5400 birds, then simply doubles it (and rounds up further) to 11000, with scant justification other than saying 5400 was a likely underestimate, but presenting no supporting evidence. By overstating the non-SPA population in this way, the potential impact on the Alde-Ore Estuary SPA is again significantly understated.

4.5.5 Whilst we acknowledge the difficulties arising from the lack of recent census data for urban gull colonies, the approach taken by the Applicant to estimate the urban gull population in Norfolk and Suffolk is speculative and lacking in precaution. A key source of information, the Seabird 2000 census (Mitchell *et al*., 2004), is missing from the cited colony counts and no evidence is provided for the rate chosen to account for colony growth since the last counts. The Seabird 2000 census carried out in 1999 – 2002 recorded 1149 apparently occupied nests (AON) in Suffolk roof-nesting colonies, 1605 AON in Norfolk coastal colonies, and 1456 in Suffolk Coastal colonies (excluding the SPA colony). This gives a total of 4210 AON outside the SPA, or 8420 adult birds. We acknowledge that these data do not include roof-nesting birds in Norfolk, and that the counts of roof-nesting birds are thought to be underestimated. More recent work by Coulson and Coulson (2015) suggests that results from the vantage point surveys of roof-nesting birds carried out for Seabird 2000 should be multiplied by 1.33 to correct for under-detection of nests. This would raise the number of adult birds in Norfolk and Suffolk to 9178 when the roof-nesting numbers for Suffolk are corrected in this way. Given that Norfolk is likely to be similar to Suffolk in terms or urban habitats available, it may be appropriate to double the numbers of urban birds in Suffolk to account for the missing Norfolk data. This would give a total non-SPA population of 12,234 adult birds, or 21,093 birds of all ages (assuming adults comprise 58% of the population, Furness, 2015), of which 10,539 are from urban colonies in Norfolk and Suffolk.

4.5.6 JNCC (2018b) discuss the growth rate of lesser black-backed gull colonies since the Seabird 2000 census, and conclude that there is insufficient evidence to allow a trend to be identified. Colonies display differing trends, due to differing in factors affecting their growth rate. Many large coastal colonies have undergone significant declines, including that of Orfordness, whilst some urban colonies, particularly in the south-east and north-west are known to have increased significantly. Given that JNCC (2018b) cannot specify trend figures, and that the non-SPA population for Norfolk and Suffolk includes both urban colonies (likely to have increased) and rural coastal colonies (may have decreased), we therefore do not consider it safe to propose an overall level of population change for the non-SPA population since the Seabird 2000 census.

4.5.7 There is also no discussion of the differences in foraging behaviour between urban and inland colonies and rural, coastal colonies. Whilst the evidence available is limited, some studies of lesser black-backed gull diet are available. Coulson and Coulson (2008) found no offshore marine component (i.e. fish or fish offal) in the diet of the lesser black-backed gull colony in Dumfries, in an analysis of regurgitated pellets. Food sources were predominantly agricultural (55% of pellets), from landfill sites (23%) or intertidal habitats (12%). Similarly, at an inland colony in the Netherlands (c.30km from the North Sea), Gyimesi *et al*. (2016) found no marine remains in an analysis of pellets and boluses, and found only 2 of 710 trips recorded by GPS tags visited the North Sea. Conversely, at two rural island colonies in the south-eastern North Sea, Kubetzki and

Garthe (2003) found that 80% of lesser black-backed gull pellets contained prey from coastal waters. Given this difference, we do not consider it safe to assume that birds from urban colonies will forage at sea to the same extent as those birds from rural coastal colonies, including the Alde-Ore Estuary SPA. There is an argument therefore, to exclude urban populations when considering apportioning to the SPA.

4.5.8 Using the Applicant's calculation of 6,700 birds of all ages associated with the SPA, the apportioning to the Alde-Ore SPA would therefore be between 24.1% if urban birds are included (6700/21093 + 6700) and 38.8% when urban birds are excluded (6700/10555 + 6700). Given the discussion above, the lower figure (which is close to the Applicant's proposed 25%) is clearly unrealistic, and a figure likely to be at least 35% would be more appropriate.

4.5.9 However, the RSPB further advocate the use of the theoretical approach as laid out in SNH guidance (SNH, 2018). This theoretical approach is based on foraging range and three colony-specific weighting factors: colony size, distance of colony from site and the areal extent of the open sea within the foraging range of the relevant species.

Presentation of cumulative and in-combination collision risk figures

4.5.10 Table 2.4 in the Applicant's Response to S51 Advice (doc. PB4476-008-001) appears to present the cumulative, rather than in-combination (ie. apportioned to the SPA), figures for lesser black-backed gulls of the Alde-Ore Estuary SPA. This should be checked and revised.

4.5.11 Table 13.15 in the Applicant's Response to S51 Advice (doc. PB4476-008-001) Appendix 1 (and Table 6.5 in the Information for HRA, doc. 5.3) also appear to show only cumulative mortality totals, not mortality apportioned to the AOE SPA. Although the apportioning is explained in the text, the lack of tabulated in-combination figures reduces the clarity and transparency of the assessment. The table should therefore be revised (in line with the tables produced for other species) to show the actual figures on which the in-combination assessment is based.

4.5.12 Table 13.15 also appears to use the migration-free breeding season figures for Norfolk Vanguard, despite noting that the extended (standard) breeding season is used. The Applicant has acknowledged that the extended breeding season figures are the most appropriate to use for this species and site and committed to using these in the assessment (para. 49, ES Ch.13., doc. 6.1.13). Table 13.15 therefore requires complete revision to show in-combination figures as explained above, and the correct breeding season figures for Norfolk Vanguard.

4.5.13 Para. 70 of the Applicant's Response to S51 Advice (doc. PB4476-008-001) explains that only windfarms within mean-maximum foraging range of the AOE SPA (141km) are included in the in-combination assessment of the breeding season, but it also states that it is likely that the breeding season total should be based only on windfarms within 72km (mean foraging range) of the SPA. We would not agree that the use of mean foraging range is appropriate to determine inclusion of projects in the in-combination assessment and note that no justification is provided for this assertion.

Galloper PVA demographic rates

4.5.14 Para. 75 of Appendix 1 to the Applicant's Response to S51 Advice discusses the outputs of the PVA produced for the Galloper offshore windfarm and refers to the set of demographic rates known as the 'medium scenario'. It is stated that this set of demographic rates was based on those expected as a result of management measures being carried out at the SPA. We disagreed with the use of the medium scenario during the Galloper examination, as there was insufficient evidence that colony management would enable the achievement of these rates. Colony numbers have since fluctuated, from 1907 apparently occupied nests in 2012 (when the Galloper examination took place), reaching a peak of around 2460 nests in 2015, with a subsequent decline to 1424 nests/pairs in 2018 (see Annex 4 for details). We therefore consider that the use of the medium scenario does not form a reliable basis for the assessment.

Potential for mitigation of impacts on the Alde-Ore Estuary SPA

4.5.15 The RSPB are concerned at the Applicant's interest in mitigating or 'offsetting' impacts on lesser black-backed gull through predator management at the Alde-Ore Estuary SPA, as outlined in Information for HRA (doc. 5.3), para. 201 and para. 77 of the Applicant's Response to S51 Advice (doc. PB4476-008-001) Appendix 1.

4.5.16 The consenting of the Galloper offshore windfarm in 2013 included provision for a fund to provide mitigation measures for mortality of lesser black-backed gulls on the Alde-Ore Estuary SPA. The

RSPB raised concerns about the principle of and likely success of this approach to mitigation during the examination. Since 2013 Natural England have been unable to identify and deliver measures that could provide successful mitigation by raising productivity, mainly because there is uncertainty as to the relative importance of factors affecting this population. As yet therefore, it has not been possible to implement the required mitigation.

4.5.17 Due to the uncertainty around the relative importance of the various factors affecting this population, we consider the Applicant's assertion that the main driver affecting gull numbers at the colony is management of predation to be an oversimplification. Whilst this does undoubtedly affect productivity, given the current levels of predator control across the SPA (carried out as part of normal site management), it is unlikely that further predator control alone would lead to sufficient further increases in productivity. Therefore, given that we query whether the proposed measures could lawfully be accepted as mitigation, the likely effectiveness of the proposed management measures on productivity is uncertain, and the lack of evidence that they would be over and above the management required for the site, we do not agree that measures of this sort should be considered as mitigation for SPA impacts[15].

Conclusions regarding lesser black-backed gull collision mortality and adverse effects on the integrity of the Alde-Ore Estuary SPA

4.5.18 The Applicant concludes no adverse effects on the integrity of the Alde-Ore Estuary SPA as a result of collision mortality to lesser black-backed gulls from Norfolk Vanguard alone is predicted (para. 199 of the Information for HRA (doc. 5.3)), and that the likelihood of adverse effects on integrity arising from collision mortality in-combination with other projects is "sufficiently small that it can

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[15] It is worth noting that this issue has more recently than the Galloper decision and consent, been considered in Case C-521/12 Briels and Others v Minister van Infrastructuur en Milieu, 15 May 2014 where the Court concluded (page 5)

> "The assessment carried out under Article 6(3) of the Habitats Directive cannot have lacunae and must contain **complete, precise and definitive findings and conclusions capable of removing all reasonable scientific doubt** as to the effects of the works proposed on the protected site concerned (see, to that effect, *Sweetman and Others* EU:C:2013:220, paragraph 44 and the case-law cited).

Consequently, the application of the precautionary principle in the context of the implementation of Article 6(3) of the Habitats Directive requires the competent national authority to assess the implications of the project for the Natura 2000 site concerned in view of the site's conservation objectives and taking into account the protective measures forming part of that project aimed at avoiding or reducing any direct adverse effects for the site, in order to ensure that it does not adversely affect the integrity of the site.…

be ruled out" (in para. 88 of their Response to S51 Advice (doc. AS-006) and para. 200 of the Information for HRA (doc. 5.3)). This does not demonstrate the required level of confidence that adverse effects on the integrity can be excluded, as it does not meet the standard from the *Waddenzee* Judgement that "no reasonable scientific doubt remains as to the absence of an adverse effect". On the basis of the Applicant's figures (which we consider underestimate mortality for the reasons explained above) the in-combination mortality will result in an increase on background mortality of 3.5%, which clearly indicates that adverse effects on the integrity cannot be excluded. We recommend that a revised PVA is developed which incorporates evidence-based demographic parameters, and that the density independent outputs in the form of counterfactuals of population size are presented in order to properly assess whether it is possible to exclude adverse effects on the integrity of this site.

4.6 Cumulative Collision Risk to Kittiwakes

Underestimation of the Norfolk Vanguard contribution to cumulative collision mortality

4.6.1 Due to the concerns explained above regarding the stochastic model used for the assessment of collision risk and the reductions to collision mortality applied as a result of the nocturnal activity rate reviews the RSPB have carried out an initial re-calculation of the collision risk modelling using the Band (2012) deterministic model and the Marine Scotland stochastic model (McGregor *et al*., 2018), without the nocturnal activity rate reductions (indicative only and see caveats to the input data, above). This shows that the Applicant's model underestimates the Norfolk Vanguard contribution to cumulative collision mortality for kittiwakes, and it will produce significantly increased predictions when it is re-run using mean (rather than median) monthly bird densities.

Nocturnal activity rate

4.6.2 We do not agree with the changes in nocturnal activity rate proposed or their potential application to other windfarms in the cumulative assessment for the reasons discussed above. We are particularly concerned that the value presented for kittiwake is based on unpublished evidence (which we understand has not yet been submitted for publication and has therefore not been peer-reviewed) which does not appear to form part of the examination documentation. We are therefore unable to assess the robustness of this study.

Density dependent outputs of PVA

4.6.3 The Applicant presents the density dependent outputs of a kittiwake PVA produced for East Anglia THREE in para. 42 of their Response to S51 Advice (doc. AS-006) and para. 461 of ES, Ch. 13 (doc. 6.1.13). As explained in section 2, we do not accept the arguments for including compensatory density dependence put forward by the Applicant, as we do not have the means to accurately quantify the strength and form of density dependence in a biologically meaningful way in order to incorporate it into PVA. Furthermore, density dependence is not always compensatory, as implied by the Applicant, but can also be depensatory, slowing the rate of population growth at lower population densities. In other words, a population decline arising from an offshore wind farm could have larger consequences on the population than are predicted by the compensatory density dependent or even density independent models. Horswill and Robinson (2015) identified depensation occurring in three gull species including black-legged kittiwake. As such we agree with Natural England's position, that the density independent outputs are those that should be considered in the assessment of impact significance.

Kittiwake population changes

4.6.4 Para. 42 of the Applicant's Response to S51 Advice (doc. AS-006) and para. 461 of ES, Ch. 13 (doc. 6.1.13) go on to discuss the changes in the UK kittiwake population over three 15 year periods and use this as evidence that a decline of up to nearly 11% due to windfarm mortality over 25 years would be undetectable against this level of natural change.

4.6.5 JNCC (2018a) discusses the rapid decline in the UK kittiwake population observed since the early 1990s and link this to declining productivity and adult survival, with declines in sandeel prey and the effects of climate change on sea surface temperatures noted as likely contributory factors. Frederiksen *et al*. (2004) also demonstrated the vulnerability of kittiwake populations to human activities through a study based on the Isle of May. Their population modelling showed that this population was unlikely to increase should the local sandeel fishery remain active and would be likely to decline further if sea surface temperature also increased, due to effects on both productivity and adult survival.

4.6.6 Given this context of continued declines in the UK population since the early 1990s and the effect of anthropogenic impacts on adult survival and productivity, we strongly disagree with the

Applicant's assertion that declines of the level predicted by the PVA due to offshore windfarm mortality alone would be undetectable against these background changes. Rather, we consider that this could add significantly to the multiple stressors affecting this population and reduce the likelihood of population recovery.

Conclusions regarding cumulative kittiwake collision mortality

4.6.7 The density independent outputs of the East Anglia THREE PVA raise significant concerns regarding the impact of offshore windfarm mortality on the kittiwake population, with a decline of 10.3-10.9% predicted over 25 years. Given the sensitivity of the kittiwake population to human impacts, we cannot agree that this magnitude of effect is low (as stated in para. 44 of the Applicant's Response to S51 Advice, doc. AS-006 and para. 463 of ES, Ch. 13 doc. 6.1.13) nor that this would equate to impacts of minor adverse significance.

4.6.8 We are also concerned that the Applicant's CRM calculations underestimate the contribution of Norfolk Vanguard to cumulative mortality.

4.7 Cumulative Collision Risk to Great Black-backed Gulls

Underestimation of the Norfolk Vanguard contribution to cumulative collision mortality

4.7.1 Due to the concerns explained above regarding the stochastic model used for the assessment of collision risk and the reductions to collision mortality applied as a result of the nocturnal activity rate reviews the RSPB have carried out an initial re-calculation of the collision risk modelling using the Band (2012) deterministic model and the Marine Scotland stochastic model (McGregor *et al*., 2018), without the nocturnal activity rate reductions (indicative only and see caveats to the input data, above). This shows that the Applicant's model underestimates the contribution of Norfolk Vanguard to cumulative collision mortality for great black-backed gulls, and it will produce significantly increased predictions when it is re-run using mean (rather than median) monthly bird densities.

Nocturnal activity rate

4.7.2 We do not agree with the changes in nocturnal activity rate proposed (or their proposed application to other windfarms in the cumulative assessment) for the reasons discussed in our overarching concerns, and because the cited evidence (EATL, 2015) has not been subject to peer-review. Nocturnal activity is one of a number of variables included in the Band model process, and recent work has indicated how important consideration of these variables is.

4.7.3 For large gulls, there is no peer-reviewed evidence for a change in the factor that is being used. The current factor is derived from the expert opinion collected by Garthe and Hüppop (2004) and this use is endorsed by Band (2012). A review of seabird vulnerability to offshore wind farms (Furness *et al*., 2013) recommended that no changes be made to the nocturnal activity scores for these species, and an update, including the same authors (Wade *et al*., 2016) maintained this recommendation. Partial analysis of data from thermal imaging cameras was carried out in the Skov *et al*., 2018 ORJIP Bird Collision Avoidance report, but was incomplete and did not fully account for the distinction between the definition of daylight as used in the Band model and with the official concept of 'twilight' and 'night'. This is an issue as the Band (2012) model considers the nocturnal period as between sunset to sunrise and so treats flight activity that occurs at twilight as being within the nocturnal flight period. Evidence from tagging shows that a number of seabirds actively forage at twilight. We therefore do not consider that any change should be made to the recommended nocturnal activity rates.

Density dependent outputs of PVA

4.7.4 The Applicant presents the density dependent outputs of a great black-backed gull PVA produced for East Anglia THREE in para. 62-63 of their Response to S51 Advice (doc. AS-006) and para. 479-480 of ES, Ch. 13 (doc. 6.1.13). As explained in section 2, we do not accept the arguments for including compensatory density dependence put forward by the Applicant, as we do not have the means to accurately quantify the strength and form of density dependence in a biologically meaningful way in order to incorporate it into PVA. Furthermore, density dependence is not always compensatory, as implied by the Applicant, but can also be depensatory, slowing the rate of population growth at lower population densities. In other words, a population decline arising from an offshore wind farm could have larger consequences on the population than are predicted by the compensatory density dependent or even density independent models. Horswill and Robinson (2015) identified depensation occurring in three gull species. As such we agree with Natural England's position, that the density independent outputs are those that should be considered in the assessment of impact significance.

Conclusions regarding great black-backed gull cumulative collision mortality

4.7.5 The density independent outputs of the East Anglia THREE PVA (presented in para. 63 of Appendix 1 to the Applicant's Response to S51 Advice, doc. AS-006) raise significant concerns regarding the impact of offshore windfarm mortality on the great black-backed gull population, with a decline of 21.3-21.5% predicted over 25 years based on an additional mortality of 900 per year. We cannot agree that this magnitude of effect is low nor that this would equate to impacts of minor adverse significance.

4.7.6 We are also concerned that the Applicant's CRM calculations underestimate the contribution of Norfolk Vanguard to cumulative mortality.

4.8 Cumulative Operational Displacement of Red-throated Divers

Displacement and mortality rates

4.8.1 For red-throated diver, displacement rates of 80% and mortality of 1-5% have been used in the assessment. As there are few robust studies of displacement, results differ, and we do not know the consequences for mortality or population trajectories, it is appropriate to consider a range of putative displacement and mortality rates. We therefore agree with Natural England that displacement of up to 100% and mortality of up to 10% represents an appropriate level of precaution and should be used in the assessment.

Use of 4km buffer

4.8.2 Para. 9 of the Applicant's Response to S51 Advice (doc. AS-006) and para. 426 of ES, Ch. 13 (doc. 6.1.13) state that the inclusion of the 4km buffer in the assessment is a source of precaution as evidence suggests that displacement decreases with distance, in some cases reaching zero by 2km. However, we highlight that there is increasing evidence to show that divers can be displaced from a greater distance, not only from operational wind farms but also from the associated boat traffic (e.g. Mendel *et al*., 2019). As such we consider that a 4km buffer is an absolute minimum rather than representing a precautionary approach and that impacts are possible over an even greater scale.

Conclusions regarding cumulative operational displacement of red-throated divers

4.8.3 The assessment concludes that there is a "high likelihood that cumulative displacement would be lower than the worst case totals" due to the precaution in the assessment. This negates the purpose of the precautionary approach to assessment, and overlooks the use of lower than recommended mortality rates. Even with these lower rates, the increase on baseline mortality is still 1.1-2.2%, therefore we also disagree with the subsequent statements that mortality will be likely to be less than 1% and therefore of negligible magnitude.

4.9 Cumulative Operational Displacement of Guillemots

Displacement and mortality rates

4.9.1 For guillemots, displacement rates of 70% and mortality of 1% have been used in the assessment. ES Ch. 13 (doc. 6.1.13) para. 432 states that the use of a 70% displacement rate is supported by a 68% avoidance rate shown in studies of windfarm avoidance by guillemots. The paragraph does note that avoidance rate applies to birds flying through the area, while displacement affects swimming birds and therefore the value may not be appropriate. We view this as a non-standard approach as we have not seen evidence presented to justify the argument that it is appropriate to apply avoidance rates to displacement.

4.9.2 There are few robust studies of displacement, results differ, and we do not know the consequences for mortality or population trajectories, hence it is appropriate to consider a range of putative displacement and mortality rates. This is illustrated by the fact that a rise on the mortality rate considered to 2% is sufficient to lift the increase on baseline mortality to over 1%. Whilst the use of a 1% increase on baseline mortality as a threshold of significance is arbitrary, this illustrates the sensitivity of the assessment of significance to the mortality rates used. We therefore agree with Natural England that displacement of up to 100% and mortality of up to 10% represents an appropriate level of precaution and should be used in the assessment.

Conclusions regarding cumulative operational displacement of guillemots

4.9.3 The assessment concludes that the magnitude of effect is negligible and that the impact significance is minor adverse. Given the concerns above around the lack of precaution in the choice of displacement and mortality rates, we cannot agree with these conclusions.

4.10 Cumulative Operational Displacement of Razorbills

Displacement and mortality rates

4.10.1 For razorbills, displacement rates of 70% and mortality of 1% have been used in the assessment. The Applicant refers to the discussion of displacement rates presented for guillemot to justify the use of a 70% displacement rate for razorbill. We disagree with the arguments presented as explained above in section 4.9. Again, we consider it appropriate to consider a range of putative displacement and mortality rates. This is illustrated by the fact that increasing the mortality rate considered to 2% is sufficient to lift the increase on baseline mortality to over 1%. Whilst the use of a 1% increase on baseline mortality as a threshold of significance is arbitrary, this illustrates the sensitivity of the assessment of significance to the rates used. We therefore agree with Natural England that displacement of up to 100% and mortality of up to 10% represents an appropriate level of precaution and should be used in the assessment.

Conclusions regarding cumulative operational displacement of razorbills

4.10.2 The assessment concludes that the magnitude of effect is negligible and that the impact significance is minor adverse. Given the concerns above around the lack of precaution in the choice of displacement and mortality rates, we cannot agree with these conclusions.

5 Comments on the Draft DCO – Provisions for Post-construction Monitoring

5.1.1 The In Principle Monitoring Plan - Offshore (doc. 8.12) explains that project level monitoring is not proposed for offshore ornithology. Whilst the RSPB welcomes the Applicant's inclusion of strategic offshore monitoring within its proposals, we are concerned that provision for project level monitoring has not been included.

5.1.2 The current lack of empirical evidence of the scale of impact on bird populations from offshore wind farms means the high levels of uncertainty in the conclusions of predicted population level impacts used for the decision making process remain. Post consent monitoring would help address and reduce these uncertainties for future deployment of offshore renewables, and is needed to validate the conclusions reached by the various assessments that have been undertaken.

5.1.3 To provide this required empirical evidence, monitoring must include both strategic monitoring at a large spatial scale (e.g. biogeographic, regional or country-level) and project level monitoring, although it is likely that there will be significant overlap between activities needed to deliver these elements.

5.1.4 The resources required must be made available for this monitoring and should be directed to two main tasks, surveillance (to observe and react to population scale impacts) and targeted monitoring (aimed at investigating focused questions, understanding impacts and their mitigation (and whether that migration is effective) and informing future planning).

5.1.5 It is crucial that the questions to be answered are clearly defined from the start. This will allow debate as to the practicality of different means of answering the questions and in particular:

- **Focus effort to make efficient use of limited resources.** A tailored approach is required to single out specific species and/or impacts. This is in preference to generic monitoring across all receptors;
- **Ensure change can be detected.** Power analysis should be undertaken to gauge level of effort against likelihood of detecting an effect; and

- **Align methodologies to gain consistency and comparability.** Consistency of approach will build the empirical dataset and enable analysis at regional and bio-geographic scales to detect population level effects. Seeking early dialogue between developers, government, agencies and stakeholders (including the RSPB) is recommended to define approaches.

Strategic Monitoring

5.1.6 In those instances where the expected impact is collision (and therefore direct mortality), monitoring, notably of breeding adults, should be possible through annual colony counts. More detailed information about individual mortality events may be provided for example by regular abundance estimates through the breeding season (which would be akin to observing nest desertions in productivity monitoring).

5.1.7 In those instances where displacement or barrier effects from the windfarm footprint and buffers zones are expected, the population level impact will derive from poorer foraging success and reduced breeding productivity, through for example reduced clutch sizes or fledging success. Productivity monitoring will therefore be required for these populations.

Project Level Monitoring

5.1.8 Beyond strategic monitoring, project level monitoring is needed to understand the impact pathways, test hypotheses that have been used in planning decisions, such as avoidance and collision rates, to seek approaches to mitigate impacts and to improve marine planning for future applications.

5.1.9 Novel approaches may be required to address these questions. As a first step there must be discussion, justification and decisions made on the study objectives and the most appropriate methods of data collection. These approaches themselves will likely require testing and validation. Operators with suitable expertise will be required to deliver the most effective studies.

5.1.10 The focal issues and species have been identified during baseline data collection. The main topics for post-construction monitoring and research are collision risk and displacement/barrier effects. Studies benefit from before/after comparison, whilst data collection during construction is also helpful to identify whether construction per se is the cause of observed changes and whether effects persist during the operational phase. Reference site(s) help to interpret any changes observed in the wind farm. Gradient studies enable assessment of the effects of increasing distance away from wind turbines.

5.1.11 Post-construction studies need to be of sufficient duration to permit the distinction between short-term and longer-term effects attributable to the presence of the wind farm. Reviews at pre-determined time intervals will enable decisions to be taken with respect to any necessary refinements of the study methods (bearing in mind the problems associated with changing methodology), as well as reviewing the results and whether there are indications of adjustments in behaviour.

5.1.12 The RSPB will discuss these requirements (particularly the need to include project level monitoring) with the Applicant and we request that a Scientific Steering Group is established to determine the details of the monitoring methods. However, we wish to highlight at this stage that monitoring cannot be regarded as a mitigation measure since it has no ability to reduce or offset possible adverse effect on the SPAs nor their species.

6 Overall Conclusion and Recommendations

6.1.1 Given the concerns we have discussed in the preceding sections, we do not agree that there is sufficient robust evidence available to support conclusions of no adverse effect on the integrity of the Flamborough and Filey Coast SPA or the Alde-Ore Estuary SPA, or to rule out significant effects on North Sea populations of kittiwake, great black-backed gull, red-throated diver, guillemot and razorbill.

6.1.2 In order to present robust evidence on which a sound assessment can be based, we consider that the Applicant should provide the following updates:

- Stochastic collision risk modelling using the accepted Marine Scotland (McGregor *et al*. 2018) version of the model
- Deterministic collision risk modelling (Band, 2012) using mean monthly bird densities in the calculations (rather than median monthly densities as currently presented)
- Presentation of the full set of parameters required to replicate the collision risk modelling
- Nocturnal activity rates:
    - For all species: presentation of the survey timings to enable understanding of whether likely peaks in activity at first and last light are accounted for.
    - For gannet: if survey timings are known and peaks in activity are accounted for, use of the rates recommended in Furness *et al*. (2018), rather than the Applicant's subsequent revision to these. If survey timings are not known or peaks are not accounted for, use of the rates based on Garthe and Hüppop (2004)
    - For all other species (including kittiwake): use of the rates based on Garthe and Hüppop (2004)
- Where population modelling is used to assess impacts on an SPA, density independent outputs of PVA should be presented, in the form of counterfactuals of population size
- Use of the standard breeding season in assessment of collision risk for kittiwake, gannet and lesser black-backed gull
- Apportioning of impacts to lesser black-backed gull of the Alde-Ore Estuary SPA to be recalculated using SNH (2018) guidance
- Apportioning of impacts to kittiwake of the Flamborough and Filey Coast SPA to be

recalculated using SNH (2018) guidance and informed by recent tracking data

- Use of a 98% avoidance rate for gannets in the breeding season
- Consideration of displacement rates of up to 100% and mortality rates of up to 10% in assessments of displacement for auks and red-throated diver

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Annex I - Qualifications and Experience of the RSPB's Expert

Dr. Aly McCluskie

Dr. Aly McCluskie is a Senior Conservation Scientist in the RSPB He has worked for the RSPB for 10 years, focussing on predator and human conflict and understanding the potential environmental consequences of the development of renewable energy. He now works within the small team that provides scientific support for site conservation. This involves both active research and review in order to provide the science required to underpin policy and casework specifically in relation to the effective conservation of protected sites. In particular this examines the interactions between wind farms, both terrestrial and offshore, and birds.

This work involves working with a range of statutory conservation bodies, government agencies and developers as well as the RSPB's own casework team in attempt to best facilitate potential wind farm developments. Previously he has assessed and trained environmental consultancy field workers, acted as a scientific advisor for BBC wildlife films, and as an ecologist for CEH, SNH, Natural Research and the University of Oxford WildCRU

He has sat on a variety of scientific steering and advisory groups and the expert panels including:

- Vattenfall European Offshore Wind Development Centre Scientific research programme, including expert sub-group for Bird Collision Avoidance study
- Joint Natural Conservation Council, *Bird Collision Avoidance: Empirical evidence and impact assessments*
- Marine Scotland Science *Testing and validating metrics of change produced by Population Viability Analysis (PVA)*
- Offshore Renewables Joint Industry Programme (ORJIP) *Bird avoidance behaviour and collision impact monitoring at offshore wind farms*
- Natural England and the Crown Estate *Seabird Flight Height Comparability Project*
- University of Highlands and Islands, *Incorporating Variability and Uncertainty into Collision Risk Modelling*
- Marine Scotland Science, *The avoidance rates of collision between birds and offshore turbines*
- German Federal Ministry for Economic Affairs and Energy*: PROGRESS, (Prognosis and Assessment of Bird Collision Risks at Wind Turbines)*
- Moray Offshore Windfarm Limited, *Large Gulls Foraging Behaviour Study*
- Marine Scotland Science, *Strategic assessment of collision risk of offshore wind farms to migrating birds*
- Marine Scotland Science, *Statistical modelling of bird and cetacean distributions in offshore renewables development areas*

Recent relevant publications:

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Annex II - The Alde-Ore Estuary SPA Species of Concern

Table 1: Lesser black-backed gull breeding population at the Alde-Ore Estuary SPA

Year	**SPA Total** Pairs/AON
1986	5043
1987	
1988	7500
1989	
1990	8223
1991	
1992	
1993	9050
1994	10008
1995	11256
1996	14817
1997	20218
1998	21704
1999	22514
2000	23400
2001	5790
2002	6838
2003	6249
2004	6264
2005	4708
2006	5325
2007	2446
2008	2769
2009	1974
2010	1603
2011	1580
2012	1907
2013	
2014	
2015	2459
2016	
2017	1914
2018	1424

Note: Varying data sources are available – figures in blue are based on reserve records (from RSPB, National Trust and Landguard Bird Observatory) and figures in black are from the JNCC SMP database. It should also be noted that methods used between years and between sources may not be strictly comparable.

Flamborough and Filey Coast pSPA Seabird Monitoring Programme

2017 Report

Aitken, D., Babcock, M., Barratt, A., Clarkson, C., Prettyman, S.

RSPB Bempton Cliffs, 9 - 11 Cliff Lane, Bempton, East Riding of Yorkshire, YO15 1JD

Front cover image: Black-browed Albatross amongst Northern Gannets at Staple Newk, RSPB Bempton Cliffs, East Yorkshire, Wednesday 28 June 2017. © David Aitken

CONTENTS

SUMMARY

The Flamborough and Filey Coast seabird monitoring programme is a partnership between the RSPB and Natural England, set up to monitor and report on the condition of this internationally important seabird colony. Established in 2008, the project aims are to establish repeatable baseline census monitoring of the colony, and to pursue a number of key areas of research and surveillance required to inform the population trends. For 2017, the results of seabird monitoring at Flamborough/Bempton and Filey have again been consolidated into a single report covering the Flamborough and Filey Coast proposed Special Protection Area (pSPA).

The results inform the pSPA and underlying Site of Special Scientific Interest (SSSI) condition assessments and provide critical data to inform casework and the establishment of a Marine Protected Area (MPA) network.

In 2017 the seabird monitoring programme was successfully completed by a dedicated team of staff, volunteers, a Seabird Research Assistant and a residential seabird research volunteer.

On balance, 2017 was an average year for most species. A run of poor weather in late June during Black-legged Kittiwake chick provisioning may have affected adult birds' ability to forage and as a result, many nests were observed unattended leaving chicks susceptible to exposure and vulnerable to predation, factors which may account for the poor productivity output. In addition, Common Guillemot recorded the lowest productivity result since monitoring began in 2009 and Razorbill experienced below average productivity. Plot-specific factors such as displacement by prospecting Northern Gannet and Carrion Crow predation may still be affecting auk breeding success. Northern Fulmar productivity remained average while European Herring Gull continue to produce poor breeding results. Northern Gannet productivity dropped below that of the previous two years, though output remains robust.

The productivity results were as follows:

- Northern Fulmar productivity – the addition of two plots at Cunstone Nab in 2017 increased monitoring to seven sites. 128 pairs were monitored from which 64 chicks successfully fledged producing a mean productivity of 0.58 (SE ± 0.0679) chicks per apparently occupied site (AOS) and an aggregated productivity of 0.50 chicks per AOS.

- Northern Gannet productivity – 273 nests were monitored across five plots, from which 220 chicks fledged producing a mean productivity of 0.81 (SE ± 0.0347) chicks per apparently occupied nest (AON) and an aggregated productivity of 0.81 chicks per AON.

- Black-legged Kittiwake productivity at Flamborough/Bempton – a remarkable 900 nests were monitored across eighteen plots, from which 527 chicks fledged producing a mean productivity of 0.58 (SE ± 0.0353) chicks per AON and an aggregated productivity of 0.59 chicks per AON.

- Black-legged Kittiwake productivity at Filey – due to a full colony count in 2017, only 150 nests were monitored across three plots, from which 59 chicks fledged producing a mean productivity of 0.39 (SE ± 0.0742) chicks per AON and an aggregated productivity of 0.39 chicks per AON.

- European Herring Gull productivity – 88 nests were monitored across five plots, from which 65 chicks fledged producing a mean productivity of 0.73 (SE ± 0.0475) chicks per AON and an aggregated productivity of 0.74 chicks per AON.
- Common Guillemot productivity – 304 pairs were monitored across six plots, from which 185 chicks fledged producing a mean productivity of 0.59 (SE ± 0.0896) chicks per AOS and an aggregated productivity of 0.61 chicks per AOS.
- Razorbill productivity – 377 pairs were monitored across eight plots, from which 244 chicks fledged producing a mean productivity of 0.56 (SE ± 0.0884) chicks per AOS and an aggregated productivity of 0.65 chicks per AOS.

Alongside the seabird monitoring programme, a boat and land-based whole colony count was completed in 2017. This mammoth task – including 214 person/hours of boat based counting – was coordinated by Keith Clarkson, recently retired RSPB Bempton Cliffs Senior Site Manager, and a team of five experienced seabird counters.

The whole colony count results (which are presented here as SPA followed by pSPA) were as follows:

- Northern Fulmar – 846 AOS recorded for the SPA. An additional 411 AOS were recorded at Filey, bringing the pSPA total to 1,257 AOS.
- Northern Gannet – 13,392 AOS recorded for the SPA. In addition, a further 1,169 non-breeding birds were also present in 'clubs' which are likely to form future extensions of the colony. There are no breeding Northern Gannets outside the SPA boundary.
- European Shag – 25 AON recorded, typically, at the base of the cliffs within the SPA.
- Great Cormorant – 27 AON recorded. The breeding range of the Great Cormorant is restricted to the high sandstone cliffs north of Filey, which sits outside the SPA boundary but within the pSPA extension, where breeding numbers have remained stable over the past 7 years.
- Black-legged Kittiwake – 45,504 AON recorded for the SPA. An additional 6,031 pairs were recorded at Filey bringing the pSPA total to 51,535 AON.
- European Herring Gull – 351 AON recorded for the SPA. An additional 115 pairs were recorded at Filey bringing the pSPA total to 466 AON.
- Common Guillemot – 84,647 individuals recorded on the cliffs across the SPA. An additional 6,214 individuals recorded at Filey bringing the pSPA total to 90,861 individuals. Using a conversion factor of 0.67 (Birkhead, 1978; Harris 1989) translates to 60,877 pairs or 121,754 breeding individuals within the pSPA.
- Razorbill – 27,967 individuals recorded on the cliffs across the SPA. An additional 2,261 individuals recorded at Filey bringing the pSPA total to 30,228 individuals. Using a conversion

factor of 0.67 (Birkhead, 1978; Harris 1989) translates to 20,253 pairs or 40,506 breeding individuals within the pSPA.

An early season count of Atlantic Puffin was completed in March this year, which included Filey Cliffs to the north, allowing total pSPA coverage for the first time. Approximately 2,879 Atlantic Puffin were counted staging on the sea during the pre-breeding period. This count allows an index of colony size with year-to-year comparisons, but should not be treated as a definitive population count.

The Black-legged Kittiwake study-plot counts produced a mean of 1,943 AON; this is an improvement on 2016 but still on the low side of the median, although Black-legged Kittiwake study-plot means have been more variable than those for Common Guillemot or Razorbill.

The Common Guillemot study-plot counts produced a mean of 1,348 individuals, a slight reduction on the previous two years but in line with an overall upward trend since 2009 (the highest being 1,454 in 2014).

The Razorbill study-plot counts produced a mean count of 676 individuals, a new record high mean and in line with the general upward trend since the first counts in 2009.

A dedicated volunteer was available for three weeks to assist with the Common Guillemot diet composition study. A total of 284 prey items were recorded, of which 69% were clupeids, while 20% were sandeel spp. and 11% were other/unidentified. Although absolute numbers were relatively small, more cephalopods—eight prey items—were recorded than in previous years.

This year, the reserve supported a Black-legged Kittiwake tracking project, led by RSPB’s Conservation Science team and funded by Ørsted (formerly DONG Energy). The project informs The Flamborough and Filey Seabird Monitoring Group (RSPB, Flamborough Head European Marine Site, and Natural England), and builds on tracking work previously undertaken at this colony. For the first time, automatic-download tracking devices were fitted to adult birds; employing this technology meant that a bird only needs to be caught once in order to attach the device, after which data is downloaded to a base station whenever the device is in range. The device is designed to fall off after a few weeks. In total, twenty auto-download devices were deployed, eighteen of which successfully collected data.

Recreational disturbance continues to be a threat to the breeding success of the colony. A Personal Watercraft Code of Conduct is now in effect from 1 March to 30 September; users are asked to maintain a no-wake speed within 300m of the cliffs and near rafts of birds. Despite this, the Flamborough Head EMS study of disturbance incidents across the pSPA in 2017 identified numerous disturbance events involving Jet Skis, motorised boats and kayakers. The Flamborough Head EMS Project Officer has also facilitated an agreement with the Chief Pilot of the Humberside Search and Rescue helicopter whereby from 15 March to 15 August crews will not carry out training exercises between North Landing and High Stacks, Flamborough Head. The voluntary Angling Code of Conduct developed with local angling clubs for Bempton and Speeton Cliffs, incorporating a closed season for cliff-top angling from 1 March to 30 September, is largely considered to be a success. Review meetings will continue to ensure the code of conduct remains relevant and effective.

INTRODUCTION

Background

Seabird population data has been collected at Flamborough and Bempton since at least 1969. In 1969, all species but European Shag and Atlantic Puffin were counted as part of the 'Operation Seafarer' national seabird census. In 1987, all species were counted during the 'Seabird Colony Register' census. All species were counted again in 2000 for 'Seabird 2000' and again in 2008. Whole-colony counts of Northern Gannet were completed in 1970-77, 1985-94, 1996-99, 2002, 2004-05, 2008-09, 2012 and again in 2015. In addition, whole colony counts for European Herring Gull were completed in 2010 and 2014 and for European Shag in 2014.

Before the commencement of the Flamborough Head and Bempton Cliffs seabird monitoring programme in 2009, breeding success data for Flamborough/Bempton was collected for Northern Gannet during 1973-79, 1986-94, 1996-98, and 2006. Black-legged Kittiwake breeding success has been monitored continuously since 1986. Common Guillemot productivity was monitored during 1991-98 and 2005-06 and Razorbill productivity was monitored in 2005-06. Northern Fulmar and European Herring Gull breeding success were monitored for the first time in 2009, and is ongoing. Unfortunately, it is not possible to monitor breeding success for Atlantic Puffin at this cliff-nesting colony and only limited monitoring of European Shag is possible depending on nest site use.

At Filey, a whole colony count was carried out in 1986 (Williams 1996). In 2002 the 'Seabird 2000' census team identified a significant seabird colony nesting on the cliffs to the north of Filey Bay (Mitchell et al. 2004). The significance of this colony came to light in 2008 in response to large numbers of Common Guillemot and Razorbill being caught and killed in gillnets set by fishermen in Filey Bay. It was recognised that birds caught in the nets could have originated from either the Flamborough/Bempton or Filey colonies. Unfortunately, at that time there was little current data about the state of the colony at Filey.

The Flamborough and Filey Coast pSPA Seabird Monitoring Programme

Flamborough and Filey Coast pSPA supports the largest mainland seabird colony in England, the only mainland gannetry in England and one of the largest mainland Black-legged Kittiwake colonies in the UK. The landward boundary of the pSPA generally follows the coast at Flamborough Head from South Landing in the south to Speeton in the North with an additional section from the forefront of Filey Brigg headland to Cunstone Nab. The seaward boundary extends approximately 2km parallel to the coast from the landward boundaries before moving seawards and extends approximately 2km into the marine environment (see maps at Appendix 1).

Flamborough Head is a highly protected site both for its wildlife and unique chalk habitats. The site is designated as a European Marine Site, a Special Area of Conservation, a Special Protection Area, a Site of Special Scientific Interest and a Heritage Coast site which includes three Local Nature Reserves, as well as RSPB Bempton Cliffs nature reserve and the Yorkshire Wildlife Trust Flamborough Cliffs nature reserve.

At the north end of the pSPA the Filey Brigg SSSI falls within the pSPA and the Gristhorpe Bay and Red Cliff SSSI is just to the north of the pSPA.

The Flamborough and Filey Coast pSPA qualifies under Article 4.2 of the Birds Directive for the following reasons:

- It supports over 1% of the biogeographical population of four regularly occurring migratory species: Black-legged Kittiwake (*Rissa tridactyla*); Northern Gannet (*Morus bassanus*); Common Guillemot (*Uria aalge*); and Razorbill (*Alca torda*).
- It is used regularly by over 20,000 seabirds in any season: during the breeding season the area regularly supports over 200,000 seabirds.

Due to the importance of the seabird colony and level of site protection, Natural England and the RSPB proposed in 2008 a project to enable a baseline count, population monitoring and further research to collect data on the health of the colony and the Flamborough Head and Bempton Cliffs SPA and underpinning SSSIs. This proposal led to the establishment of the Flamborough Head and Bempton Cliffs seabird monitoring programme, which began with the 2009 seabird breeding season.

There was evidence to suggest that the cliffs north of Bempton supported a sizeable colony that might also meet the EU Birds Directive criteria. So, in 2009, a boat-based whole colony count of the breeding seabird assemblage nesting on the cliffs between Filey and Cayton was carried out by the RSPB. The results suggested that the total number of breeding seabirds in this colony exceeded 20,000 birds, and therefore this colony also met SPA qualifying criteria. In response to this evidence the RSPB, with funding support from Natural England, completed five consecutive years of colony count data to verify these findings. This data supported the proposed extension of the existing SPA to include Filey Cliffs to create the pSPA.

The data collected by this now enlarged Flamborough and Filey Coast seabird monitoring programme will inform the condition and management of the Flamborough and Filey Coast pSPA and underpinning SSSIs. In addition, the results will also inform current and new planning enquiries and environmental assessments e.g. the proposed Hornsea and Dogger Bank offshore wind arrays that may have a detrimental impact on the features of the designated sites. It is also hoped that seabird tracking data collected from the colony will inform potential new offshore MPAs.

Data collected will also be used to inform the Seabird Monitoring Programme (SMP) coordinated by Joint Nature Conservation Committee (JNCC), the RSPB's Annual Reserve Monitoring (ARM) programme, the RSPB Bempton Cliffs reserve management plan and the Yorkshire Wildlife Trust's reserve management.

The key aims of the seabird monitoring programme, and how they are currently implemented, are as follows:

- **Understanding variation and trends in seabird productivity**

  Northern Fulmar, Northern Gannet, Black-legged Kittiwake, European Herring Gull, Common Guillemot and Razorbill plots have been monitored for breeding productivity annually since 2009.

- **Understanding population numbers and trends**

  Black-legged Kittiwake, Common Guillemot and Razorbill study-plot counts have been carried out annually since 2009. A whole colony census was carried out in 2008 and repeated in 2017. It is intended that a whole colony count be completed every five years within the reserve's management plan cycle.

- **Understanding the relationship between the colony and the larger marine environment**

  As the relevant technologies improve we hope to better understand foraging behaviours of birds breeding in the colony and to identify preferred foraging areas and trends in provisioning such as determining key feeding areas for key species, and the factors that influence their location. This includes ongoing seabird tracking, currently focused on Black-legged Kittiwake, and monitoring of Common Guillemot diet composition. In the future this could extend to range finders, remote tracking, and increased use of fish population modeling data and benthic mapping.

- **Understanding how RSPB Bempton Cliffs relates to wider pSPA and potential impacts on disturbance by developing research proposals to address the following management issues**

  What are the types of human activities that could disturb the colony and what are their effects? Currently recreational disturbance is monitored and recorded by Bempton Cliffs and others on an ad-hoc basis. For those activities that are of particular concern, we hope to develop specific research proposals which assess level of impact.

The annual programme of monitoring is coordinated by the RSPB Bempton Cliffs seabird team lead by the reserve Warden, the Seabird Research Assistant, and a team of dedicated volunteer seabird researchers including members of Flamborough Bird Observatory (FBO) and Filey Bird Observatory & Group (FBOG).

The results of the 2017 Flamborough and Filey Coast seabird monitoring and research programme are detailed in this report.

Raw Data

Access to the productivity and population monitoring data collected during the seabird monitoring programme is available to researchers and conservation organisations by agreement with RSPB

METHODS

The Flamborough and Filey Coast seabird monitoring programme follows the methods and guidelines set out in the '*Seabird monitoring handbook for Britain and Ireland'* (Walsh et al., 1995) – ("the Handbook" hereafter). The Handbook summarises census and productivity monitoring techniques for seabirds relevant to colonies in Britain and Ireland. The appropriate methods are followed for each species at this colony taking into account the resources available and the physical geography of the colony. Please refer to the Handbook for more details on methodologies for each species and survey undertaken.

N.B. This year, within each section of the report, species are presented in International Ornithological Congress (IOC) taxonomic order which the British Ornithologists' Union (BOU) adopted as of 1 January 2018. In addition, all species are referred to by their IOC international name or vernacular name with the exception Common Murre, which is referred to as Common Guillemot.

Productivity monitoring

Productivity monitoring was completed for the ninth consecutive year for six of the eight breeding seabird species found in the colony: Northern Fulmar, Northern Gannet, Black-legged Kittiwake, European Herring Gull, Common Guillemot and Razorbill. In 2017, a small number of European Shag nests at Flamborough Head were monitored for the third year. Unfortunately, it is not possible to monitor Atlantic Puffin productivity at this cliff-nesting colony.

All productivity monitoring was based on marking apparently occupied sites (AOS) or apparently occupied nests (AON) on a laminated photograph of the relevant plot. For a detailed description of the relevant methodology, please refer to the section of the Handbook for the relevant species.

The productivity monitoring plots were identified when the Flamborough Head and Bempton Cliffs seabird monitoring programme was established in 2009. Plots were selected with a view to providing, where possible, a sample size in the region of 50 AOS or AON per plot and a total sample in excess of 250 AOS/AON for each species, while providing safe vantage points for the observer with little or no disturbance to breeding seabirds. In 2011, five additional monitoring plots for Black-legged Kittiwake were established at Filey Cliffs; in 2014 one of these was dropped and an additional monitoring plot added on Filey Brigg. In 2017, two additional Northern Fulmar plots were added at Cunstone Nab at the North end of Filey Cliffs. Indicative maps of the productivity plot locations are included in Appendix 2.

This year, we present productivity calculated as the mean of the individual plot results for each species as well as presenting species productivity data by aggregating the results of each plot (total chicks fledged / total nests (or sites) monitored).

Northern Fulmar *Fulmarus glacialis*

Seven productivity plots were monitored, including two new plots at Cunstone Nab at the North end of Filey Cliffs. Plots are photographed in early May and AOSs are marked on the laminated photographs over three visits over the late May/early June period. A final visit is made in mid-August and large chicks present at that time are assumed to fledge.

Northern Gannet *Morus bassanus*

Five productivity plots were monitored between late April and October. Plots are photographed in mid to late April and up to 50 or 60 AONs are marked on the laminated photographs. The plots are then visited every 7 – 10 days. Average visit time early in the season is 2 – 3 hours per plot, but reduces once chicks get larger and are more visible. Presence of an egg or chick is recorded (if seen) each visit.

European Shag *Phalacrocorax aristotelis*

In 2017, informal productivity monitoring was carried out on six European Shag nests at Breil Nook, Flamborough alongside the nearby auk productivity monitoring plots. These nests were not selected at random – being the nests it was possible to see from existing seabird monitoring points – and three of them were in recesses in the cliff and not always possible to see clearly. The nest sites were marked on laminated photographs of the cliffs and checked approximately weekly from 30 April until the end of July. No more than 30 minutes per visit was spent. Presence and number of eggs or chicks (to the extent visible) were recorded each visit.

Black-legged Kittiwake *Rissa tridactyla*

Twenty-one productivity plots were monitored across the pSPA between May and August, eighteen plots – 900 AONs – were monitored between Flamborough and Bempton and three plots – 150 AONs – at Filey. Plots are photographed in early to mid-May and up to 50 or 60 AONs are marked on the laminated photographs. Plots are then visited every week, ideally on the same day so visits are 7 days apart. Presence and number of eggs or chicks at each AON is recorded (if seen) each visit. Volunteers are also asked to record chick size using standard codes, but not all do. Average visit time varies according to the volunteer, but 1 to 1.5 hours per visit is probably typical.

European Herring Gull *Larus argentatus*

Five productivity plots were monitored between May and August. Two of the plots are linear and include all safely observable nests found on a defined stretch of cliff. One linear plot is at Bempton Cliffs and one is at Flamborough Head. Plots are photographed around early/mid-May and AONs are marked on the laminated photographs over two visits. Additional AONs may be added over the course of the season. Plots are then visited every week, ideally on the same day so visits are 7 days apart. Presence and number of eggs or chicks at each AON is recorded (if seen) each visit. Volunteers are also asked to use size codes for chicks. Average visit time varies according to the volunteer, but 1 to 2 hours per visit is probably typical.

Common Guillemot *Uria aalge*

Six productivity plots were monitored between late April and end of July. Plots are photographed in late April/early May and up to 50 to 60 AOSs are marked on the laminated photographs over two visits. Plots are then visited every third day. Additional sites may be added over the course of the season, especially if it has been hard to get 50 AOSs. Presence of an egg or chick is recorded (if seen) each visit. Average visit time early in the season is 2 – 2.5 hours, but reduces once chicks get larger and are more visible.

Razorbill *Alca torda*

Eight productivity plots were monitored between late April and the end of July. Plots are photographed in late April/early May and up to 50 to 70 AOSs are marked on the laminated photographs over two visits. Plots are then visited every third day. Additional sites may be added over the course of the season, especially if it has been hard to get 50 AOSs. Presence of an egg or chick is recorded (if seen) each visit. Average visit time early in the season is 2 – 2.5 hours, but reduces once chicks get larger and are more visible.

Whole colony counts

Colony Count

A whole colony count was completed between 18 May and 14 June 2017 for all breeding seabirds found within the pSPA, except for Atlantic Puffin. Counts were carried out by a team of six experienced observers using a combination of boat and land–based counts. The count took 253 hours or 34 person days to complete and included a total of 214 person hours of boat-based counts and 40 person hours of land based counts. The boat-based surveys were assisted by the Emmerson family of Flamborough and the hire of their fishing coble. Survey methods followed the methodologies and guidelines set out in the Handbook. For the purpose of boat (and land)-based colony counts, the Flamborough to Bempton section of the Flamborough and Filey Coast pSPA is divided into 178 sub-sections, comprising 15km of coastline. These subsections have been marked on a set of laminated photographs which cover the length of the cliffs.

As part of the whole colony count, a count at Filey was completed on Friday 9 June and took approximately 6 hours to complete. It was a boat-based survey and was undertaken by two RSPB staff with assistance from Filey Sailing Club and the use of their RIB. Survey methods followed the methodologies and guidelines set out in the Handbook. The Filey colony is divided into five recording areas, taken from the JNCC Seabird Monitoring Programme (SMP) website; within these recording areas, 24 sub-sections have been established and marked on photographs of the cliffs to assist the counts.

Atlantic Puffin *Fratercula arctica* **at Flamborough, Bempton and Filey**

Given the impossibility of monitoring the population or productivity of cliff-nesting Atlantic Puffin the species was not included in the Flamborough/Bempton (now Flamborough and Filey Coast) seabird monitoring programme. However, after the species was added to the Red List in 2015 it was decided to attempt to monitor the breeding population to the extent possible. In 2016, advice was sought from Puffin expert Professor Mike Harris, who recommended that we count adults on the sea when large numbers of birds return en masse pre-breeding season (M Harris pers. comm., 2016). Although this cannot be considered to provide an accurate census, it does provide an index with changes from year to year can be compared. Accordingly, for the second consecutive year, in March 2017, RSPB Bempton Cliffs viewpoint volunteers, who are out watching the sea every day, were asked to inform staff as soon as large numbers of Atlantic Puffin appeared staging on the sea. The first reports were in the morning of Friday 24 March. Based on the advice received, a team of two RSPB staff and a volunteer from FBOG, all with extensive bird monitoring experience, walked the distance from High Stacks at Flamborough Head (just south of the lighthouse) to the end of Speeton Cliffs before Hunmanby Gap – approximately 15 kilometers – and Filey Brigg to Cunstone Nab – approximately 4 kilometers. Using visual markers the sea was divided into sections and the number of Atlantic Puffin on the sea in each section was counted using binoculars and/or telescopes as appropriate. Due to the length of the cliffs to be covered and staff resources available meant that only one count was possible over one day.

Study-plot counts

The size and nature of the Flamborough and Filey Coast colony mean that is not practicable to conduct annual whole colony population monitoring. Accordingly, study-plots for population monitoring of Black-legged Kittiwake, Common Guillemot and Razorbill were established at Flamborough and Bempton in 2009. Plots were selected to be dispersed through the colony as randomly as possible given the need to provide a safe vantage point and minimise disturbance to breeding birds. Counts have been conducted each year since 2009. In 2011, counts of Common Guillemot and Razorbill were abandoned due to an early breeding season.

For each species the same plots are used each year as required by the Handbook; plot boundaries, based on clear cliff features, are marked on laminated photographs of the relevant area of cliff. Indicative maps of the study-plot locations at Flamborough and Bempton are included in Appendix 3.

The Handbook suggests that study-plot counts are not recommended for general use when counting Black-legged Kittiwake, as population changes may not be detected due to movements within the colony or colony extensions, or losses rather than through changes of density across the colony. However, as Flamborough and Bempton holds one of the largest mainland populations in the UK, it is important that trends are monitored.

Black-legged Kittiwake study-plot counts

Seven study-plots were counted between 0800 and 1600 on at least two occasions during the period from 1 June to 22 June.

Common Guillemot study-plot counts

Seven study-plots were counted between 0800 and 1600 on five occasions during the period from 1 June to 18 June.

Razorbill study-plot counts

Seven study-plots were counted between 0800 and 1600 on five occasions during the period from 1 June to 20 June.

Common Guillemot diet study

In 2017 a dedicated residential volunteer undertook a Common Guillemot diet study at Flamborough/Bempton. The study was carried out from 30 May to 18 June 2017 to cover the peak chick rearing season using methodology adapted from Jeavons (2015). As the observer did not have previous experience it was elected to only monitor Common Guillemot and their productivity plots at Grandstand South and Carter Lane were used for these observations. Observations were made between 0600 and 0800 and 1730 and 2000 on most days during the survey period. The observations were rotated between plots to get a similar number of surveys under morning and evening conditions at both sites. During each session the observer watched the cliff face without binoculars or a telescope, for birds flying in and landing, focusing predominantly on areas where the nests were located. When birds landed, the observer used binoculars (Viking 10x40 HD) or a telescope (RSPB HD 82mm with 25-50x zoom eyepiece) to identify the species of each prey item, to assign it to a size category (small, medium, large). Identification and sizing was done using an identification guide created for the RSPB STAR project and modified for this study. Size categories were based on the size of the prey in relation to the bill.

European Shag roost counts and colour ring re-sighting

Winter roost counts of European Shag, as identified in 2014 at Breil Nook, Flamborough Head, were discontinued this year after several years of counts did not show the numbers anticipated. It remains possible that this is due to the impracticality of seeing some of the areas used to roost from land and with little to no access to boat-based observing in winter.

Throughout the seabird monitoring season, however, colour ring codes are recorded, where practicable; over the winter specific visits are made in appropriate conditions. In each case records are submitted to the Centre for Ecology & Hydrology's (CEH) winter Shag distribution project.

Seabird tracking

Building on the previous GPS tracking work on Black-legged Kittiwakes within the Flamborough and Filey Coast pSPA between 2010 and 2015 (Aitken et al. 2014; Babcock et al. 2015), a further 20 chick rearing birds were tracked in the 2017 breeding season. The aim of the study was to trial a new methodology that will eventually enable the collection of fine-scale distribution data over longer deployment periods, and measurement of behavioral information by determining flight heights and three-dimensional acceleration. Both are crucial factors in understanding the risks of birds colliding with wind turbines and present a major gap in current environmental assessment methods for offshore wind developments, particularly for Kittiwake. Funding for the 2017 tracking work was secured from Ørsted (formerly DONG Energy) as part of a package of strategic monitoring related to the Hornsea Project One offshore windfarm development. Keith Clarkson (former RSPB Bempton Cliffs Senior Site Manager, now 'retired'), Lucy Wright (RSPB Principal Conservation Scientist) and Aly McCluskie (RSPB Senior Conservation Scientist) worked together to secure the funding for this work, with input to the proposals from members of the Flamborough and Filey Coast Seabird Monitoring Group.

Fieldwork was carried out under licenses from the British Trust of Ornithology (BTO) and Natural England (NE) by Saskia Wischnewski, RSPB Conservation Scientist, and Derren Fox, RSPB Senior Research Assistant, with help from Michael Babcock, RSPB Bempton Cliffs Seabird Research Officer, and David Aitken, RSPB Bempton Cliffs Warden. Solar powered University of Amsterdam (UvA) tags with GPS, accelerometer and remote download capabilities, were glue mounted to the mantle of adult chick-rearing birds caught at the nest. Eighteen birds were tracked from the usual sites at either end of the pSPA, Flamborough Head and Filey Brigg. A further two tags were deployed at a new study site at Speeton Cliffs, closer to the centre of the pSPA. Because data from tags were remotely downloaded to a base station via a relay network, birds did not need to be recaptured, meaning that data collection could continue later in the breeding season for as long as the tags stayed glued to the bird and it returned to the colony to download data to the relay network.

The capture and handling of birds followed strict protocols to minimise stress for the birds and disturbance to the colony. Glue mounting compared to the usual taping of tags to the bird increased the duration of attachment from a maximum of four days to 29 days. Tags weighed between 7.4 and 7.7g. Weight increased with attachment material to ~8g, representing ~2% of the body weight of a Kittiwake (~400g) and staying well below the recommended maximum tag burden of 3%. However, since this was the first time this tag attachment and tag was used on Kittiwakes, tagged and control nests were monitored until chicks were fledging age (>30 days) to assess tagging effects. Failure rates were high across control (58% for Filey and 33% for Flamborough) and tagged nests (75%) but tagged nests were also significantly lower on the cliff than control nests by 1.17 ± 0.42 m, and nest height also affected failure rate. Models accounting for nest height did not detect a significant effect of tagging on nest failure at either site.

Recreational disturbance

The Flamborough Head EMS study into recreational activity has identified recreational disturbance as an ongoing issue across the pSPA. Recreational disturbance can include: anglers descending the cliffs to access the foreshore; cliff-top angling during the breeding season; boat, jet ski and kayak use; paragliders, powered hang gliders and low flying aircraft. Wherever possible, disturbance events are recorded and photographed. The information is passed to the EMS project officer, Heather Davison, who is gathering data on recreational disturbance across the pSPA in an attempt to build a better picture of the activities taking place, the frequency that they occur and the possible impact they may have on the pSPA. A future challenge is to determine how to assess the impact of these activities on the breeding seabird assemblage and then manage this impact.

RESULTS

Productivity monitoring

Northern Fulmar *Fulmaris glacialis*

Mean productivity for Northern Fulmar was 0.58 (SE ± 0.0679) chicks per AOS. A total of 128 AOS were monitored across seven plots, from which 64 chicks successfully fledged (Table 1, Figure 1). The mean productivity for Northern Fulmar recorded between 1986-2005 from between thirteen and forty-one colonies annually was 0.41 (SE ± 0.01) chicks per AOS, (Mavor et al., 2008).

Table 1: Northern Fulmar productivity 2017: two new productivity plots at Cunstone Nab at the North end of Filey Cliffs were added to the monitoring programme this year. The new plots are marked *.

Plot	AOS	Fledged chicks	Productivity ch/pr
*Cunstone Nab A	34	17	0.50
*Cunstone Nab B	24	14	0.58
New Roll-up	4	3	0.75
Old Dor	30	7	0.23
Newcombe	11	8	0.73
Breil Nook	10	7	0.70
Swineshaw Hole	15	8	0.53
Aggregate productivity	128	64	0.50
Mean of plot results		**0.58**	**± 0.0679 SE**
Original plots total	70	33	0.47
Mean of original plots		0.59	± 0.0967 SE

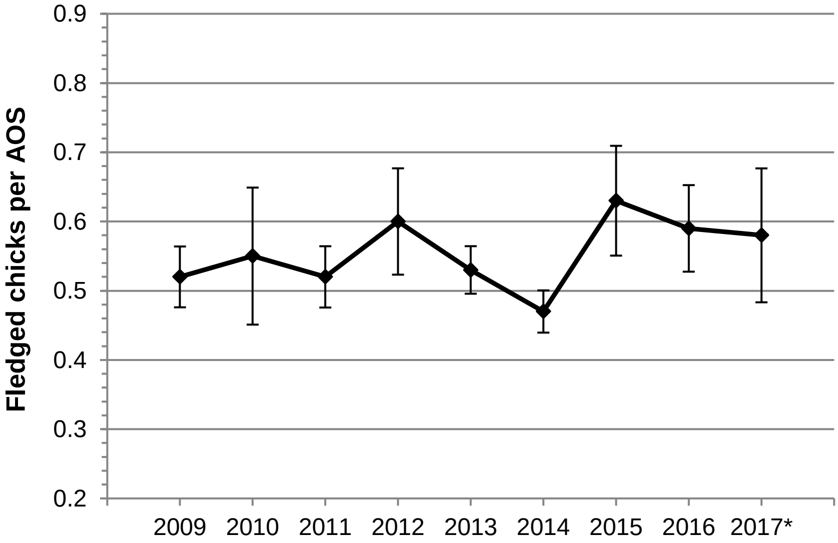

Fig. 1: Northern Fulmar productivity 2009 – 2017, mean of plot productivity results plus/minus SE. *Note: two new plots added in 2017.

The productivity on one plot at Old Dor is notably poor and may be as a result of Northern Gannet activity. On the final monitoring visit, Northern Gannets were noted on several ledges that had previously held apparently incubating Northern Fulmar.

Northern Gannet *Morus bassanus*

Mean productivity for Northern Gannet was 0.81 (SE ± 0.0347) chicks per AON. A total of 273 AON were monitored across five plots, from which 220 chicks successfully fledged (Table 2, Figure 2). The mean productivity for Northern Gannet recorded between 1986-2005 from between three and six colonies annually was 0.69 (SE ± 0.01) chicks per AON, (Mavor et al., 2008).

Table 2: Northern Gannet productivity 2017

Plot	AON	Fledged chicks	Productivity ch/pr
Jubilee Corner	53	44	0.83
Nettletrip	57	48	0.84
Staple Newk 1	52	44	0.85
Staple Newk 2	53	45	0.85
Staple Newk 3	58	39	0.81
Aggregate productivity	273	220	0.81
Mean of plot productivity ch/pair		**0.81**	**± 0.0347 SE**

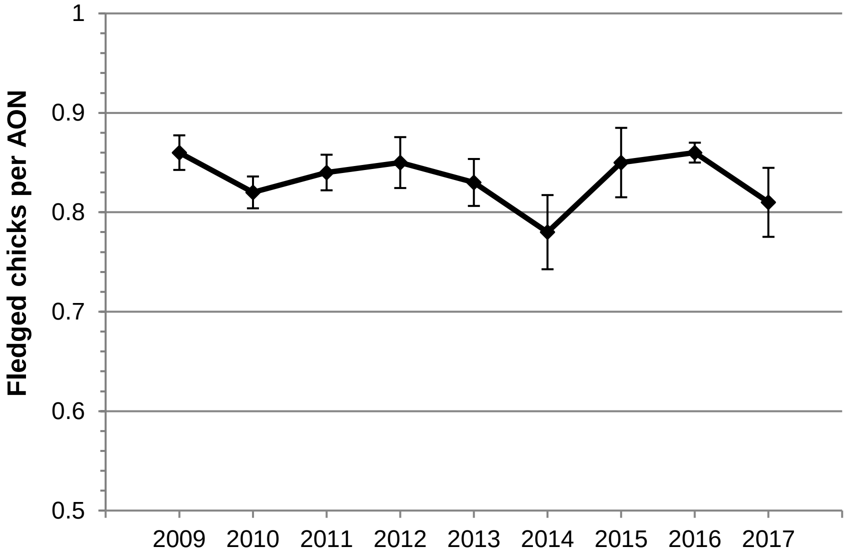

Fig. 2: Northern Gannet productivity 2009 – 2017, mean of plot productivity results plus/minus SE.

European Shag *Phalacrocorax aristotelis*

Five European Shag nests in the area of Breil Nook, Flamborough Head were monitored. Due to line of sight it was not always possible to see the chicks clearly, but it is estimated that at least 12 chicks fledged. Accordingly, productivity for this group of nests was a minimum of 2.4 chicks per pair.

Black-legged Kittiwake *Rissa tridactyla* **– Flamborough and Bempton**

Mean productivity for Black-legged Kittiwake at Flamborough and Bempton was 0.58 (SE ± 0.0353) chicks per AON. A total of 900 AON were monitored across 18 plots, from which 527 chicks successfully fledged (Table 3, Figure 3). The mean productivity for Black-legged Kittiwake recorded between 1986-2005 from between thirty and sixty-one colonies annually was 0.68 (SE ± 0.03) chicks per AON (Mavor et al., 2008).

Table 3: Flamborough/Bempton Black-legged Kittiwake productivity 2017

Plot	AON	Fledged chicks	Productivity ch/pr
Jubilee Far	50	26	0.52
Bartlett Nab Near	50	34	0.68
Bartlett Nab Far	50	30	0.60
Grandstand North Near	50	29	0.58
Grandstand North Near Edge	50	32	0.64
Grandstand North Mid	50	28	0.56
Grandstand North Far Edge	-	-	-
Grandstand North Low	50	26	0.52
Old Dor	50	31	0.62
Newcombe	47	9	0.19
Back of Newcombe	50	33	0.66
Carter Lane 1	50	25	0.50
Carter Lane 2	-	-	-
Saddle Nook 1	50	35	0.70
Saddle Nook 2	50	33	0.66
Saddle from Breil	50	16	0.32
Breil Nook North	50	42	0.84
Breil Nook South	50	32	0.64
Back of Breil Nook	53	28	0.53
Swineshaw Hole	50	38	0.76
Lighthouse	-	-	-
Aggregate productivity	900	527	0.59
Mean of plot productivity ch/pair		**0.58**	**± 0.0353 SE**

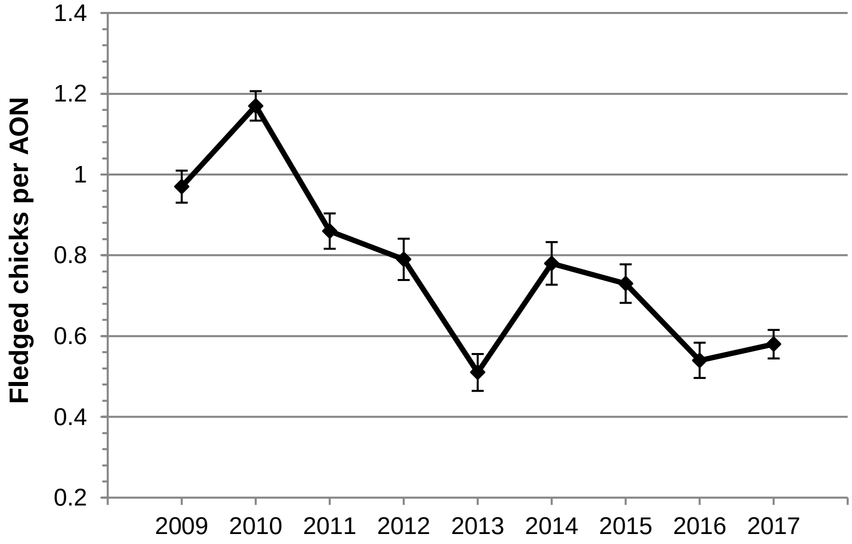

Fig. 3: Flamborough/Bempton Black-legged Kittiwake productivity 2009 – 2017, mean of plot results plus/minus SE.

Black-legged Kittiwake *Rissa tridactyla* – Filey Cliffs

Mean productivity for Black-legged Kittiwake at Filey was 0.39 (SE ± 0.0742) chicks per AON. A total of 150 AON were monitored across 3 plots, from which 59 chicks successfully fledged (Table 4, Figure 4). The mean productivity for Black-legged Kittiwake recorded between 1986-2005 from between thirty and sixty-one colonies annually was 0.68 (SE ± 0.03) chicks per AON (Mavor et al., 2008).

Table 4: Filey Black-legged Kittiwake productivity 2017

Plot	AON	Fledged chicks	Productivity ch/pr
Plot 1	50	27	0.54
Plot 7	50	15	0.30
Plot 8	50	17	0.34
Plot 9(a)	-	-	-
Plot 10(a)	-	-	-
Aggregate productivity	150	59	0.39
Mean of plot productivity ch/pair		**0.39**	**± 0.0742 SE**

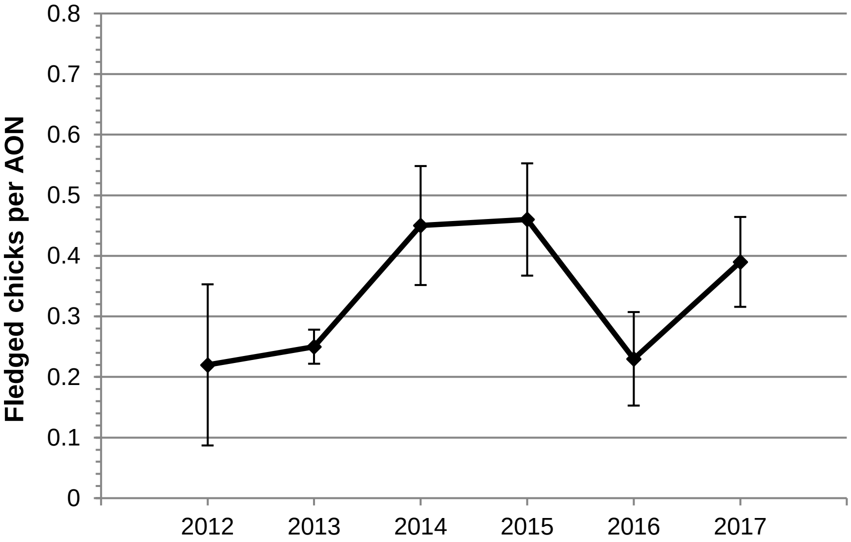

Fig. 4: Filey Black-legged Kittiwake productivity 2012– 2017, mean of plot results plus/minus SE. In 2017 only three plots were monitored with a total sample size of 150 AON. Note that in 2012 productivity on 3 plots was 0.0 due to landslip.

European Herring Gull *Larus argentatus*

Mean productivity for European Herring Gull was 0.73 (SE ± 0.0475) chicks per AON. A total of 88 AON were monitored across five plots, from which 65 chicks successfully fledged (Table 5, Figure 5).

Table 5: European Herring Gull productivity 2017

Plot	AON	Fledged chicks	Productivity ch/pr
Jubilee to Old Dor	25	20	0.80
Newcombe North	6	4	0.67
The Saddle Rock	23	15	0.65

Breil Nook Stack	17	11	0.65
Newcombe to Breil	17	15	0.88
Aggregate productivity	88	65	0.74
Mean of plot productivity ch/pair		**0.73**	**± 0.0475 SE**

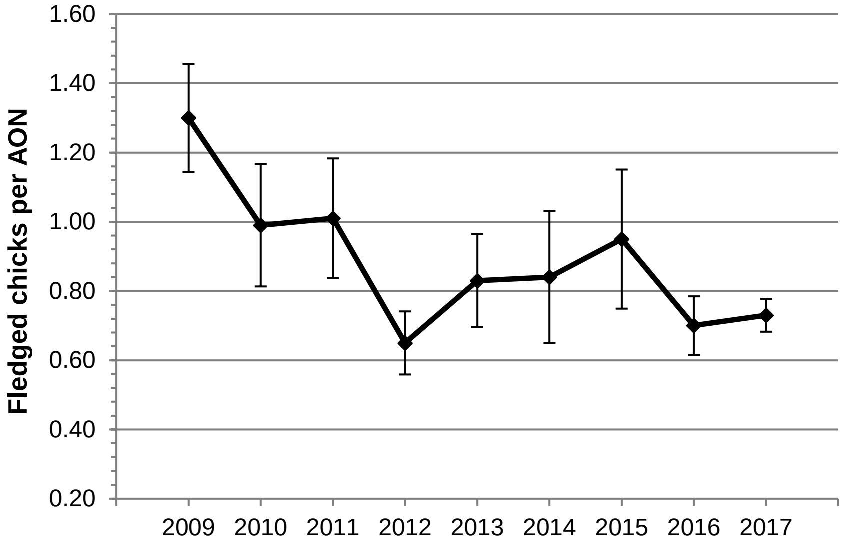

Fig. 5: European Herring Gull productivity 2009 – 2017, mean of plot results plus/minus SE.

Common Guillemot *Uria aalge*

Mean productivity for Common Guillemot was 0.59 (SE ± 0.0896) chicks per AOS. A total of 304 AOS were monitored across six plots, from which 185 chicks successfully fledged (Table 6, Figure 6). The mean productivity for Common Guillemot recorded between 1986-2005 from between three and fifteen colonies annually was 0.69 (SE ± 0.02) chicks per AOS (Mavor et al., 2008).

Table 6: Common Guillemot productivity 2017

Plot	AOS	Fledged chicks	Productivity ch/pr
Nettletrip	47	29	0.62
Grandstand North	44	7	0.16
Grandstand South	55	42	0.76
Carter Lane 1	47	30	0.64
Carter Lane 2	56	40	0.71
Breil Nook	55	37	0.67
Aggregate productivity	304	185	0.61
Mean of plot productivity ch/pair		**0.59**	**± 0.0896 SE**
Mean productivity excluding Grandstand North		0.68	± 0.0264 SE

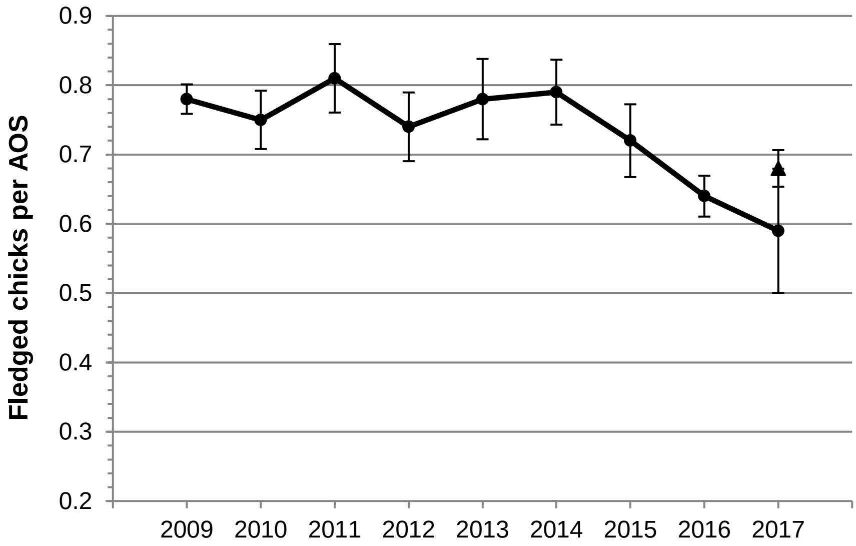

Fig. 6: Common Guillemot productivity 2009 – 2017, mean of plot results plus/minus SE. ▲ = mean of plot results excluding Grandstand North plus/minus SE.

This year, productivity on Grandstand North was notably low. Carrion Crow predation on eggs and displacement by prospecting Northern Gannet seemed to be responsible. These factors mean that this plot is likely not representative of the colony as a whole and this hypothesis is supported by the results of the full colony count. Exclusion of this plot improves the mean productivity from 0.59 to 0.68. However, the Nettletrip plot, which is also affected by Northern Gannet activity (new nests and prospecting birds noted again this year) showed an upturn in productivity in 2017.

Razorbill *Alca torda*

Mean productivity for Razorbill was 0.56 (SE ± 0.0884) chicks per AOS. A total of 377 AOS were monitored across eight plots, from which 244 chicks successfully fledged (Table 7, Figure 7). The mean productivity for Razorbill recorded between 1986-2005, from between one and seven colonies annually, was 0.65 (SE ± 0.02) chicks per AOS (Mavor et al., 2008).

Table 7: Razorbill productivity 2017

Plot	AOS	Fledged chicks	Productivity ch/pr
Grandstand Gully	13	2	0.15
Grandstand North	46	17	0.37
Grandstand South	22	7	0.32
Newcombe	80	67	0.84
Back of Newcombe	49	29	0.59
Saddle Nook	53	37	0.70
Breil Nook	56	46	0.82
Swineshaw Hole	58	39	0.67
Aggregate productivity	377	244	0.65
Mean of plot productivity ch/pair		**0.56**	**± 0.0884 SE**
Mean of plot productivity on RSPB Reserve		0.28	

Mean of plot productivity at Flamborough Head	0.72

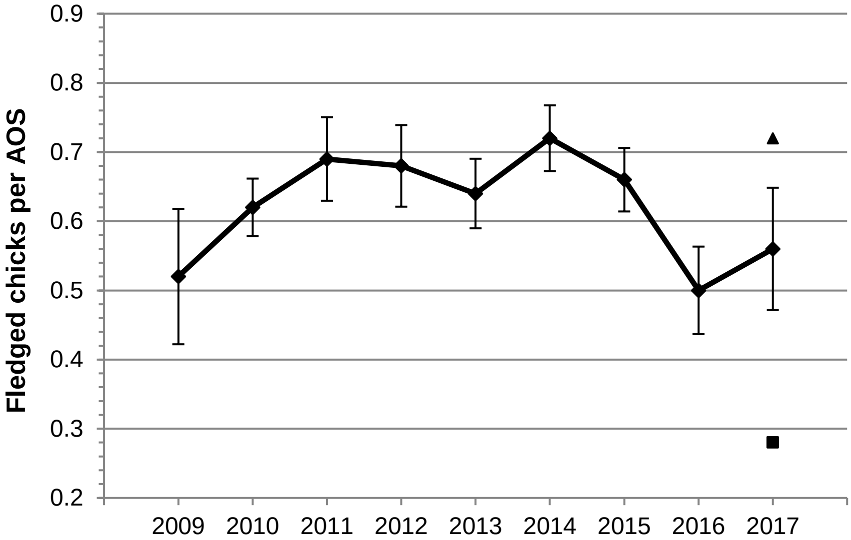

Fig. 7: Razorbill productivity 2009 – 2017, mean of plot results plus/minus SE. ■ = mean of Grandstand plot results; ▲ = mean of Flamborough plot results.

Once again, Carrion Crow activity may have had an impact on these results, particularly for the three Grandstand plots at the RSPB Bempton Cliffs reserve. There were near daily observations of predated eggs or actual Carrion Crow predation of Razorbill eggs in the vicinity of the Grandstand viewpoint. It is therefore likely that productivity on these plots is not typical of the larger colony. For this reason Figure 7 shows separate means for the three Grandstand plots on the RSPB Reserve and for the plots at Flamborough Head as well as the overall mean of plot results.

Whole colony counts

N.B. Complete whole colony count data for the Flamborough and Filey Coast pSPA exists for 2017 only. For this reason, the following population graphs presented here are for the original Flamborough Head and Bempton Cliffs SPA.

Northern Fulmar *Fulmaris glacialis*

846 AOS were recorded across the SPA (Figure 8). An additional 411 AOS were recorded at Filey, bringing the pSPA total to 1,257 AOS or 2,514 breeding individuals.

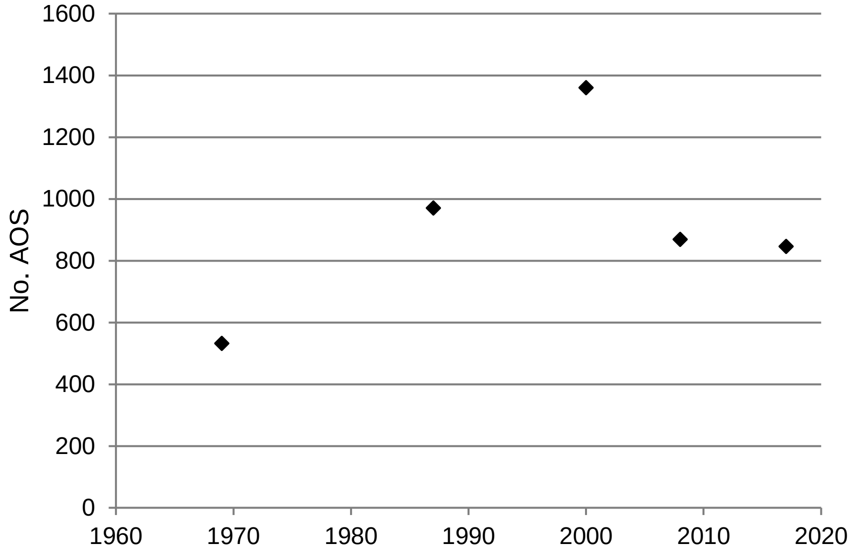

Fig. 8: Trend in Northern Fulmar *Fulmaris glacialis* breeding population within the Flamborough Head and Bempton Cliffs SPA (1969-2017).

Northern Gannet *Morus bassanus*

13,392 AOS were counted (Figure 9). In addition to the apparently occupied sites, a further 1,169 non-breeding birds were also present in ‘clubs’ which are likely to form future extensions of the colony.

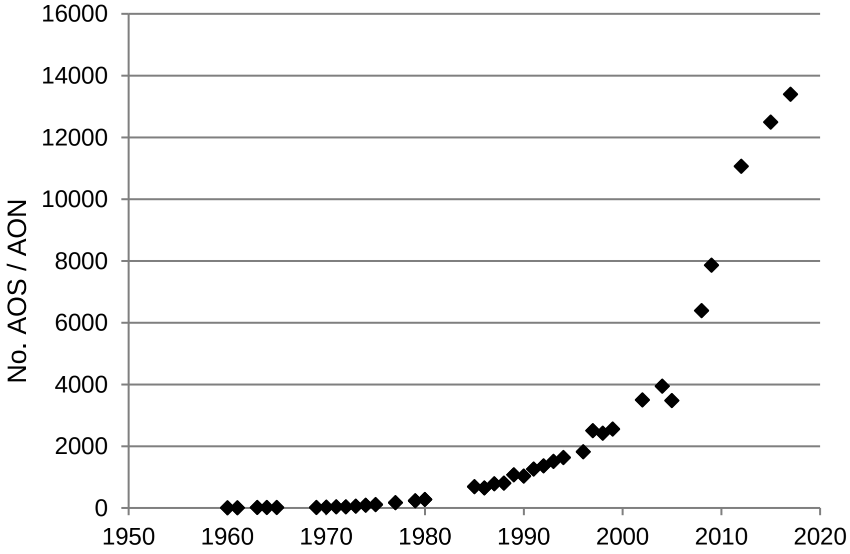

Fig. 9: Trend in Northern Gannet *Morus bassanus* breeding population within the Flamborough Head and Bempton Cliffs SPA (1960-2017).

European Shag *Phalacocorax aristotelis*

25 pairs with apparently occupied nests were counted, typically, at the base of the cliffs within the SPA (Figure 10).

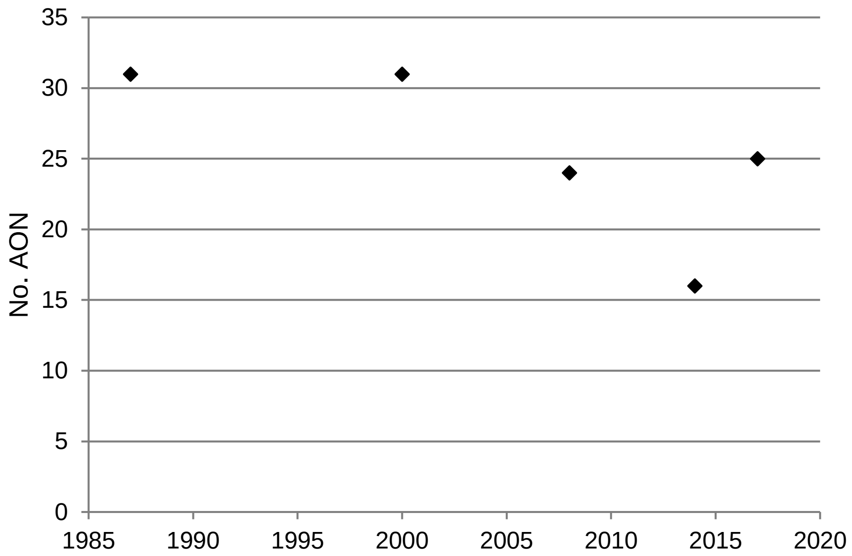

Fig. 10: Trend in European Shag *Phalacrocorax aristotelis* breeding population within the Flamborough Head and Bempton Cliffs SPA (1987-2017).

Great Cormorant *Phalacocorax carbo*

27 pairs with apparently occupied nests were recorded. The breeding range of the Great Cormorant is restricted to the high sandstone cliffs north of Filey in the pSPA extension, where breeding numbers have remained stable over the last 7 years.

Black-legged Kittiwake *Rissa tridactyla*

45,504 AON were counted across the SPA (Figure 11). An additional 6,031 AON were recorded at Filey, bringing the pSPA total to 51,535 AON or 103,070 breeding individuals.

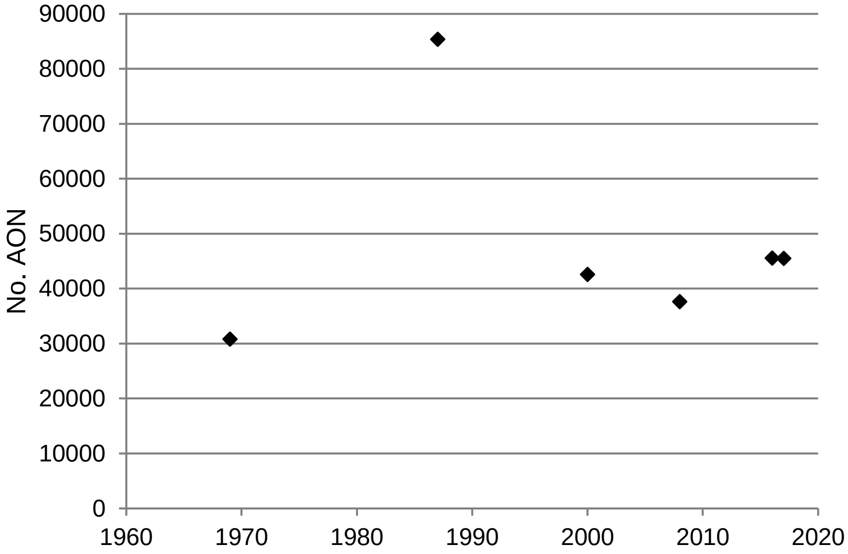

Fig. 11: Trend in Black-legged Kittiwake *Rissa tridactyla* breeding population within the Flamborough Head and Bempton Cliffs SPA (1969-2017).

European Herring Gull *Larus argentatus*

351 AON were counted across the SPA (Figure 12). An additional 115 pairs were recorded at Filey, bringing the pSPA total to 466 AON or 932 breeding individuals.

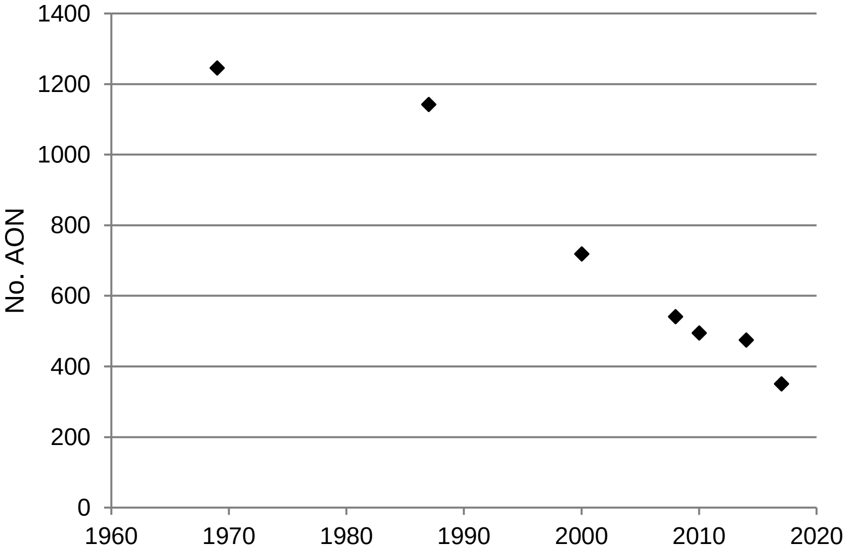

Fig. 12: Trend in European Herring Gull *Larus argentatus* breeding population within the Flamborough Head and Bempton Cliffs SPA (1969-2017).

Common Guillemot *Uria aalge*

84,647 individuals were counted on the cliffs across the SPA (Figure 13). An additional 6,214 individuals were recorded at Filey, bringing the pSPA total to 90,861 individuals. Using a conversion factor of 0.67 (Birkhead, 1978; Harris, 1989) translates to 60,877 pairs or 121,754 breeding individuals within the pSPA.

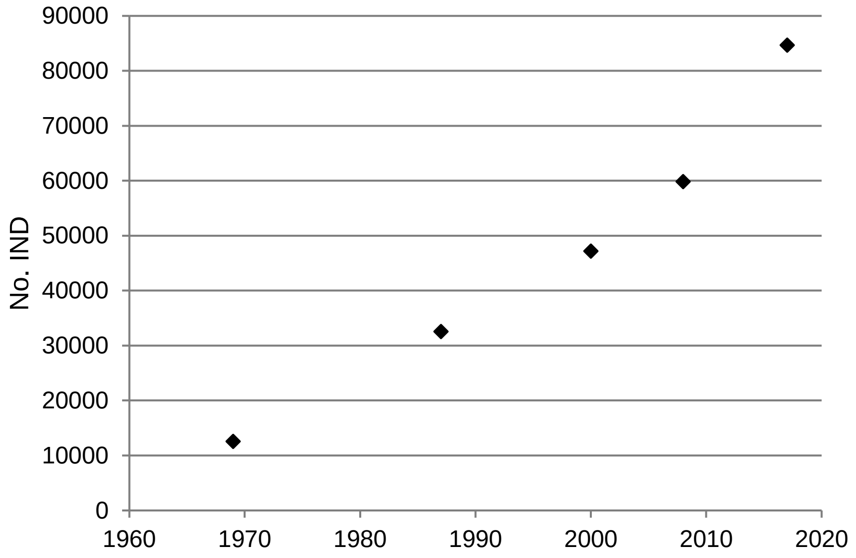

Fig. 13: Trend in Common Guillemot *Uria aalge* breeding population within the Flamborough Head and Bempton Cliffs SPA (1969-2017).

Razorbill *Alca torda*

27,967 individuals were counted on the cliffs across the SPA (Figure 14). An additional 2,261 individuals were recorded at Filey, bringing the pSPA total to 30,228 individuals. Using a conversion factor of 0.67 (Birkhead, 1978; Harris, 1989) translates to 20,253 pairs or 40,506 breeding individuals within the pSPA.

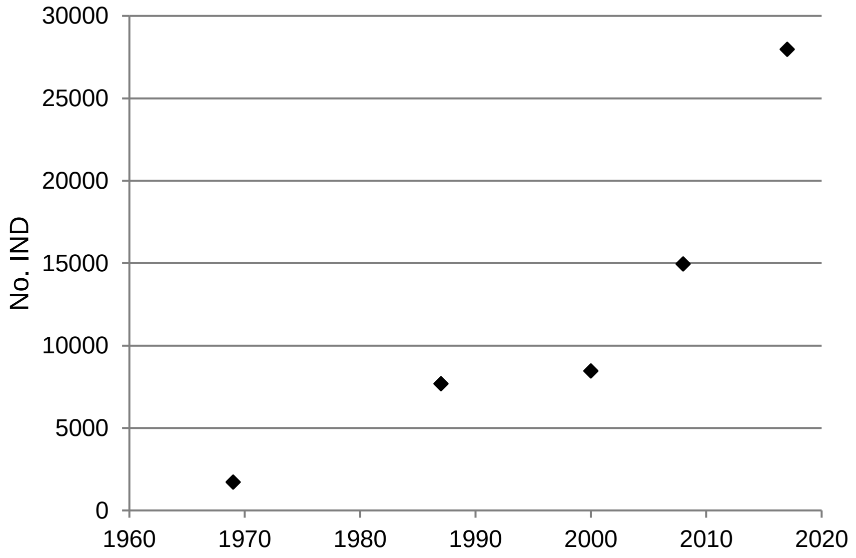

Fig. 14: Trend in Razorbill *Alca torda* breeding population within the Flamborough Head and Bempton Cliffs SPA (1969-2017).

Atlantic Puffin *Fratercula arctica* pSPA whole-colony count

A total of 2,879 Atlantic Puffin were recorded staging on the sea on Friday 24 March 2017. The count was broken down into sections as follows (from South to North):-

Flamborough Head to Thornwick Bay:	712
Thornwick Bay to Speeton Cliffs:	1,924
Filey Brigg to Cunstone Nab:	243

It should be stressed that these numbers are not to be treated as a count of the breeding population; they are intended to serve as an index and enable detection of relatively large scale year-to-year variation.

Study-plot counts

Black-legged Kittiwake study-plot counts

Seven study-plots were each counted on two separate occasions in the first three weeks of June. The mean of the two counts was 1,943 AONs, an increase after the poor year in 2016 but still relatively low (Table 8).

Table 8: Black-legged Kittiwake study-plot count results - last 6 years

Visit	2012 AON total	2013 AON total	2014 AON total	2015 AON total	2016 AON total	2017 AON total
1	1967	1554	1917	1966	1858	1945
2	1952	1508	1996	1977	1816	1940
Mean	**1960**	**1531**	**1957**	**1972**	**1837**	**1943**

Common Guillemot study-plot counts

Seven study-plots were each counted on five separate occasions in the first three weeks of June. The mean of the study-plot counts for Common Guillemot was 1,348 IND (Table 9). Both the high count and mean were similar to the last two years.

Table 9: Common Guillemot study-plot count results - last 5 years

Count	2013 IND total	2014 IND total	2015 IND total	2016 IND total	2017 IND total
1	1193	1411	1396	1491	1335
2	1226	1486	1410	1342	1428
3	1333	1327	1494	1361	1424
4	1323	1475	1420	1351	1323
5	1318	1573	1226	n/a	1231
Mean	**1279**	**1454**	**1389**	**1386**	**1348**

Razorbill study-plot counts

Seven study-plots were each counted on four separate occasions in the first three weeks of June. The mean of the study-plot counts for Razorbill was 676 IND; this is the highest mean count recorded and is in line with the general upward trend since the first counts in 2009 (Table 10).

Table 10: Razorbill study-plot count results - last 5 years

Count	2013 IND total	2014 IND total	2015 IND total	2016 IND total	2017 IND total
1	552	584	592	570	731
2	584	694	535	654	700
3	556	565	662	686	657
4	624	591	607	660	689
5	613	754	482	n/a	658
Mean	**586**	**638**	**576**	**643**	**676**

Continuation of study-plot counts

The completion of the whole colony count allowed us to compare the trend in colony counts of Black-legged Kittiwake, Common Guillemot and Razorbill in 2000, 2008 and 2017 with the trend shown by study-plot counts since 2009. Overall, the study-plot counts reflect the increase in Common Guillemot and Razorbill numbers shown by the whole colony counts starting in 2009 and the more or less steady Black-legged Kittiwake numbers. Accordingly, the study plot counts will be continued in the belief that they do at reflect changes in the larger colony even if they do not capture the potential magnitude of these changes.

Common Guillemot diet study

A total of 284 prey items were recorded during the study, reflecting the lack of dedicated diet observation sessions. Of these 69% were clupeids, 20% were sandeel spp., 3% were cephalopods and 8% were other/unidentified (Figure 15). Although the absolute number is quite small, this is the first time in recent years that more than one or two cephalopods have been recorded.

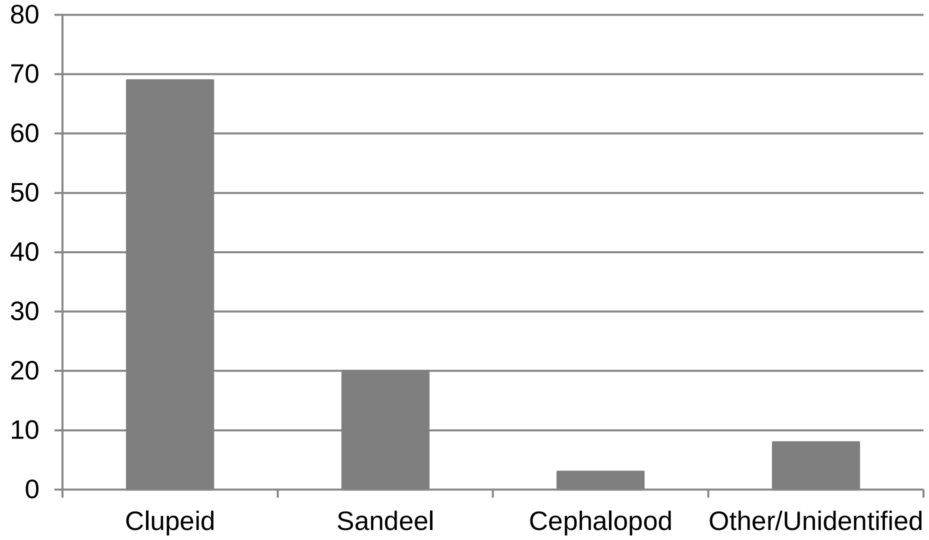

Fig. 15: 2016 Common Guillemot diet composition – percentage of observed prey items at Bempton Cliffs/Flamborough Head. n = 284 prey items.

European Shag roost counts and colour ring re-sighting

Winter roost counts were discontinued in 2016-17 as they continued to fall well short of anticipated numbers. Colour ring re-sightings continue to provide valuable insight in to the origins and movements of European Shag using the colony and effort was concentrated on collecting this data. To date, thirty individuals have been re-sighted, including nine new individuals in 2017 (Table 11).

Table 11: European Shag colour ring re-sightings at Flamborough Head 2014 – 2017

Code	**BTO ring number**	**Date**	**Age**	**Colony**	**First and last date recorded at Flamborough Head**
EUH		2014	Pullus	Fidra	10/11/14 - 13/05/17
CLR		2014	Pullus	Farnes	10/11/14 - 21/10/15
END	1478565	2014	Pullus	Inchmickery	10/11/14 - 22/05/17
CHC	G8898	2006	Pullus	Isle of May	25/11/14 - 04/08/17
CNE		2014	Pullus	Farnes	04/12/14
ACE	1472974	2014	Adult	Craigleith	19/01/15 - 18/02/16
ESB	1478625	2014	Pullus	Inchmickery	19/01/15
ARI		2014	Pullus	Craigleith	24/07/15
NEJ		2015	Pullus	Farnes (Inner)	21/10/15
DAN	1485389	2016	Pullus	North Sutor, Inverness-shire	06/02/17
UWE		2016	Pullus	Farnes (Inner)	20/05/17
FTA		2016	Pullus	Isle of May	21/05/17
IAX		2016	Pullus	Isle of May	21/05/17
HUD		2016	Pullus	Isle of May	22/05/17 - 07/08/17
DAN		2016	Pullus	Isle of May	04/08/17
TPC	1396622	20/06/09	Adult	Craigleith	10/11/14 - 28/02/17
RZF		2013	Adult	Farnes	21/10/15
PCA		2010	Pullus	Farnes (Staple)	21/10/15 - 26/08/16
AUL	1483281	2015	Adult	Isle of May	18/02/16 - 06/02/17
AFP		25/06/14	Pullus	Isle of May	10/11/14
AUH	1483074	2014	Pullus	Isle of May	18/02/16 - 28/02/17
ADA	1473962	2014	Pullus	Isle of May	18/02/16
IPJ		30/05/16	Pullus	Isle of May	13/09/16 - 19/09/16
DAP	1472058	2015	Pullus	Isle of May	11/03/17
IDT		2016	Pullus	Isle of May	04/08/17
CUX	1472024	2015	Pullus	Isle of May	13/02/17
HZA		2015	Pullus	Isle of May	19/06/17
NDC		16/06/14	Pullus	Isle of May	10/11/14
FTX		2012	Pullus	Isle of May	16/12/14
AFN	1453306	2011	Pullus	Isle of May	28/02/17

Seabird tracking

Overall, 168 foraging trips (longer than 1km and 1h) were collected from 18 of the 20 deployed tags (Figure 16), which showed large variability in trip duration, distance and range (Table 12). The utilization distribution for all trips collected in 2017 highlighted a substantial overlap with many of the offshore windfarm developments already in place or planned in front of the Yorkshire coast (Figure 17). However, there was a strong North/South divide between trips from Filey and Flamborough in the 2017 data set resulting in a decreased overlap with the Hornsea zones compared to tracking data from previous years (Figure 18). One successfully tracked bird from the new Speeton study site appeared to go to foraging sites further North, following a similar pattern to the Filey birds. However, as the centre of the colony is home to the biggest proportion of the population, it needs to be assessed whether the divide between northern and southern foraging areas is maintained, or if birds from the centre of the colony fill the gap in foraging distributions.

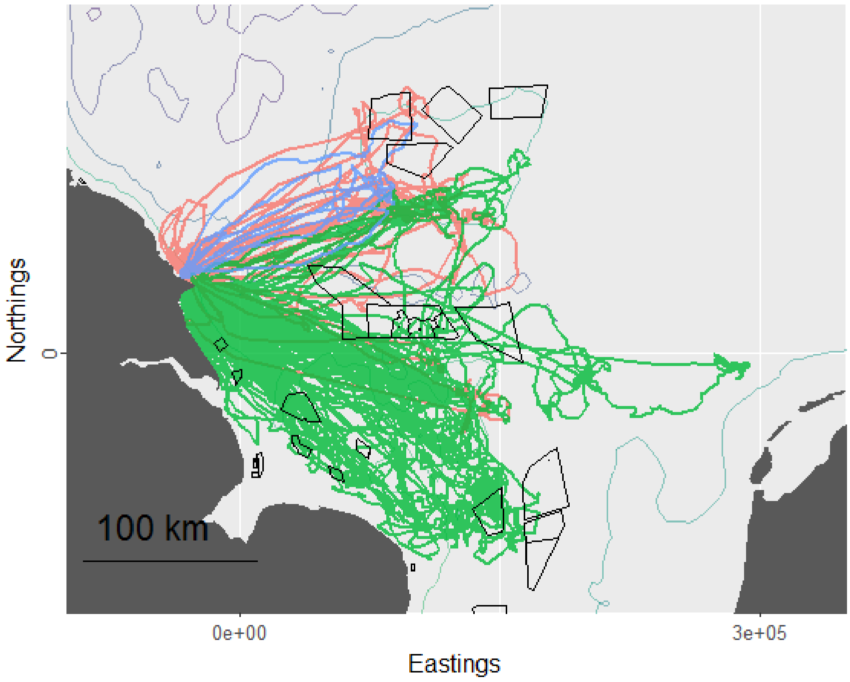

Fig. 16: Kittiwake GPS trips collected during the 2017 breeding season at the Flamborough and Filey Coast pSPA. All trips are shown from Flamborough (Green, N=133 trips from 13 birds), Filey (Red, N=29 trips from 4 birds) and Speeton (Blue, N=6 trips from 1 bird). Bathymetric contours and scale bars are shown, with land in dark grey (UK, left; the Netherlands, right) and the footprints of constructed, consented and planned offshore wind farms outlined in black (the Hornsea Zone includes the large group of windfarms in the centre of the map). The map is projected to the Azimuthal Equal Area centred on the mid-point of all the tracking data.

Table 12: Summary of the three trip metrics calculated for all 168 Kittiwake trips. Trips ranged over a maximum period of 29 days across the chick rearing period and also included trips from failed individuals. Note that the reported large standard deviations are due left skewed distribution of all three trip metrics. N=168 from 18 birds.

Trip metric	**Mean (±SD)**	**Range**
Trip duration (h)	22.12 (±28.69)	1.00 - 168.67
Foraging range (km)	88.65 (±74.22)	3.20 - 323.85
Travelled distance (km)	256.62 (±261.88)	7.91 - 1249.70

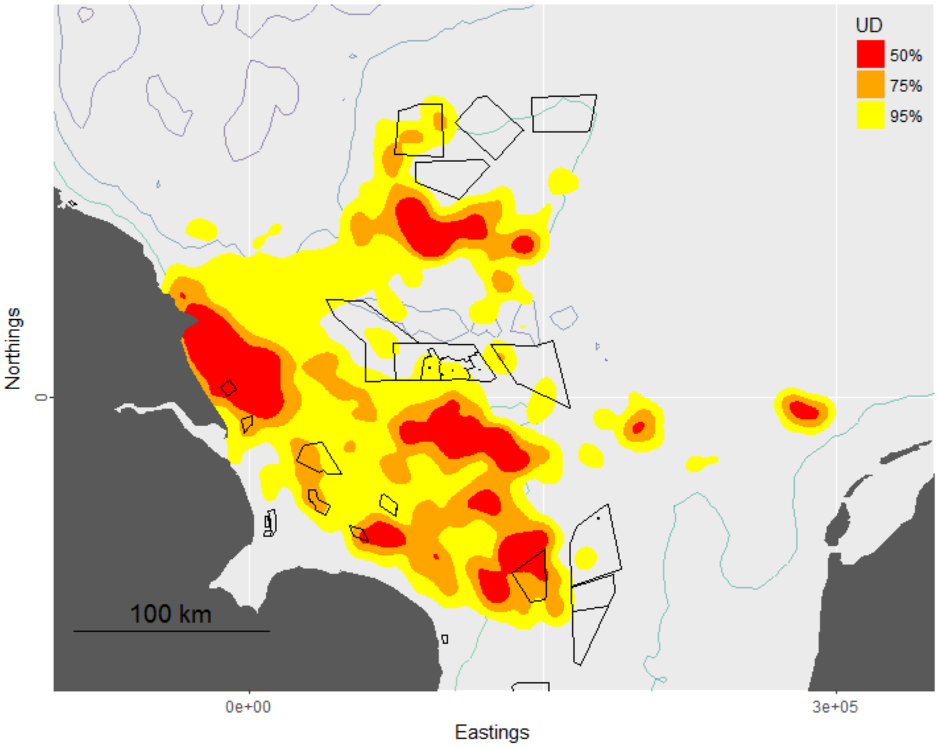

Fig. 17: Utilization distributions of all Kittiwakes tracked at Flamborough and Filey Coast pSPA during the 2017 breeding season. N=168 trips from 18 birds. 50, 75 and 95% contours are shown. Bathymetric contours, scale bar and outlines of all proposed, planned or active windfarm zones are shown, with land in dark grey (UK, left; the Netherlands, right). The map is projected to the Azimuthal Equal Area centred on the mid-point of all the tracking data.

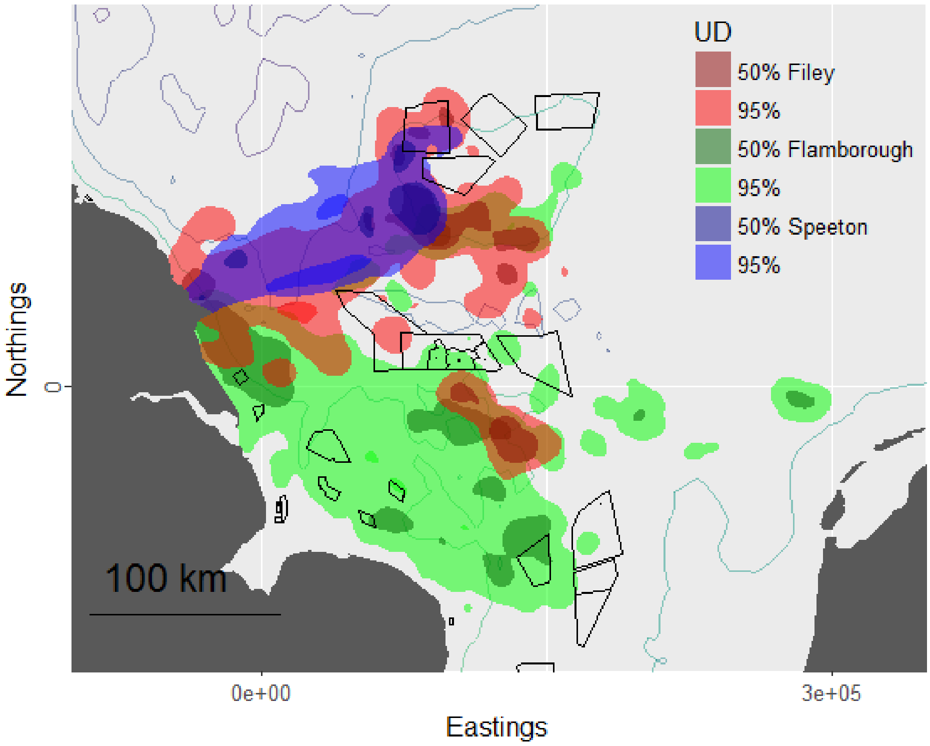

Fig. 18: Utilization distributions of all Kittiwakes tracked at Flamborough and Filey Coast during the 2017 breeding season by study site. 50 and 95% contours are shown. Green tones refer to Flamborough (N=133

trips from 13 birds), red tones to Filey (N=29 trips from 4 birds) and blue tones to Speeton (N= 6 trips from 1 bird). Bathymetric contours, scale bar and outlines of all proposed, planned or active windfarm zones are shown, with land in dark grey (UK, left; the Netherlands, right). The map is projected to the Azimuthal Equal Area centred on the mid-point of all the tracking data.

Note that construction of the Hornsea wind farms did not start, yet, and no turbines are in place in this area (though some of the smaller wind farms nearer the coast are already operational). In 2017 accelerometer samples were only collected for 3 birds due to a combination of small battery size and tag settings that were not able to compensate for shading effects in cliff nesting seabirds and therefore had a slow solar re-charge rate. These issues were discussed with the tag manufacturer and should be successfully addressed for future years by modifications to the tag design and operating system.

Because of the longer attachment method (glue mounting) and the remote download of data that does not require re-trapping of birds, this study not only provides a unique and scarce insight into the distribution of birds later in the breeding season when they are on larger chicks, it also includes some trips collected from failed breeders. Therefore, the next step (currently work in progress led by Saskia Wischnewski) is to assess how distributions and foraging trip characteristics change across the breeding season and after breeding failure.

Recreational disturbance

Recreational disturbance continues to be a threat to the breeding success of the colony. The Flamborough Head EMS partnership study of disturbance incidents across the SPA identified incidents involving Jet Skis, motorised boats and kayakers in 2017.

The voluntary code of conduct developed with local angling clubs for Bempton and Speeton Cliffs, incorporating a closed season for cliff-top angling from 1 March to 30 September, was largely considered to be a success. Review meetings will continue, however, to ensure the code of conduct remains relevant and effective.

A personal watercraft (Jet Ski) Code of Conduct applies from 1 March to 30 September; users are asked to maintain a no-wake speed within 300m of the cliffs and near rafts of birds. The Flamborough Head EMS Project Officer continues to work with personal watercraft users, the Personal Watercraft Partnership, local authorities, Natural England and the RSPB in order to ensure that the voluntary agreement is effective

Last year, the EMS Project Officer also facilitated an agreement with the Chief Pilot of the Humberside Search and Rescue helicopter whereby from 15 March to 15 August crews will not carry out training exercises between North Landing and High Stacks (just south of the lighthouse). This agreement compliments the existing Ministry of Defense ‘Environmental Avoidance’ area around Bempton Cliffs and does not cover emergency responses, which will continue as normal.

A PhD student from Leeds University has been conducting research around the EMS, in partnership with the Yorkshire Wildlife Trust and the Flamborough Head EMS Management Scheme, to look at recreational activity issues in comparison with another marine protected area in Bulgaria. It is hoped that this work will further the Management Scheme’s understanding of recreational disturbance and potentially provide some ideas for new management measures.

DISCUSSION

The Flamborough and Filey Coast pSPA (formally Flamborough Head and Bempton Cliffs SPA) supports the largest mainland seabird colony and largest Black-legged Kittiwake colony in the UK, as well as the only Northern Gannet colony in England. It is also the most southerly large cliff-nesting seabird colony on the North Sea coast. The Flamborough and Filey Coast pSPA Seabird Monitoring Programme has been operating since 2009, providing a real insight into trends in breeding seabird productivity and populations.

Productivity was average at best for most species monitored: Common Guillemot breeding success was the lowest recorded (0.59 chick/pair) since detailed monitoring commenced in 2009 and Razorbill productivity was low at 0.56 chick/pair. Plot specific factors such as displacement by prospecting Northern Gannet and corvid predation may account for this. Both of these factors will continue to be monitored closely. Black-legged Kittiwake productivity was also low (0.58 chick/pair) when compared to the national mean of 0.69 chick/pair (Mavor et al., 2008). A period of bad weather during chick provisioning in June may have affected adult birds' ability to forage, thus resulting in unattended nests susceptible to exposure and vulnerable to predation. This is the six consecutive year that productivity has been below the 0.80 chicks/pair believed to be necessary to sustain a population (Coulson, 2011). Northern Fulmar and Northern Gannet produced average results and European Herring Gull was again below average.

A total assemblage colony count was successfully completed in 2017 when favorable weather conditions and sea state allowed 10 days of boat-based surveys to be completed during the survey window, conditions which had not been replicated since the 2008 census. The data from 2017 will contribute to the next national census, 'Seabirds Count', scheduled to take place during the 2018-19 breeding seasons as confirmed by JNCC (Daisy Burnell, pers. comm., 2017). The full colony results are encouraging for most species with the exception of Northern Fulmar and European Herring Gull; both of which have experienced a steady decline since the last national census, 'Seabird 2000'. Conversely, the Northern Gannet population has grown exponentially with a 425% increase in that same time; similarly, Razorbill has undergone a 230% increase and Common Guillemot 79% since 'Seabird 2000'. The Black-legged Kittiwake population has seen a small 7% increase which is positive when compared with colonies elsewhere in the UK. A paper on the 2017 count is being prepared (Clarkson, 2017) and will provide more detail about the results, trends and methodology employed.

The completion of the whole colony count allowed us for the first time to review the annual study plot counts and compare the trends with the counts in 2000, 2008 and 2017. Overall, the study-plot counts reflect the increase in Common Guillemot and Razorbill numbers shown by the whole colony counts starting in 2009 and the relatively stable Black-legged Kittiwake numbers. Accordingly, the study plot counts will be continued in the belief that they do at reflect changes in the larger colony even if they do not capture the potential magnitude of these changes.

This year, a dedicated residential volunteer was available to undertake the auk diet composition study. The observer had limited fieldwork experience and so only Common Guillemot prey items were monitored. It was valuable to test the feasibility of using a volunteer with limited experience to see if meaningful data could be collected and the results suggest that it is possible to collect useable data; although it is crucial that time is spent assessing the observer and ensuring quality control during the study period. The diet observations form an important indicator of prey availability in the North Sea and so it is essential that a sustainable model for this monitoring work is developed.

In 2017, the reserve supported a Black-legged Kittiwake tracking project, led by RSPB's Conservation Science team and part-funded by Ørsted (formerly DONG Energy). This tracking project informs the Flamborough and Filey Coast Seabird Monitoring Group (RSPB, Flamborough Head European Marine Site, and Natural England), and builds on tracking previously undertaken at this colony. For the first time, automatic-download solar tracking devices were fitted to adult birds; employing this technology meant that a bird only needs to be caught once in order to attach the device, after which data is downloaded wirelessly to a base station whenever the device is in range. In total, twenty devices were deployed, eighteen of which successfully collected data.

Construction of the first phase of Hornsea Project One is due in 2019 and so it is vital to collect as much data as possible before completion in order that bird behaviour before and after construction can be compared. In 2018, it is hoped to tag 20 additional Black-legged Kittiwake at Speeton and Flamborough, in addition to 20 Northern Gannet at Bempton. This year's tags will include altimeters as well as accelerometers, so that altitude as well as flight patterns can be collected. This data will be invaluable in understanding risk from collision with blades and turbines, as well as in future understanding changes in flight pattern associated with avoidance. In addition, the tags will collect higher resolution data in the core foraging areas, and less data when resting on the cliffs, where re-charging solar batteries is problematic. This higher rate of sampling within the windfarm footprint should provide greater insights into the behaviour of birds in this zone prior to construction.

A Black-legged Kittiwake colour ringing project is also proposed in 2018, and re-sighting data will help determine survivorship. These projects will be managed by RSPB and part-funded by Ørsted and Natural England.

The Flamborough and Filey Coast Seabird Monitoring Group will continue to meet regularly in 2018 to assess data collected and to understand what further data is required to understand this colony and its dependency on the Hornsea Zone. This collaborative working with Ørsted to gather data will allow us to focus our energies on data interpretation, rather than disputing validity of data or data collection methodologies.

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The National Lottery through the Heritage Lottery Fund (HLF) for funding a seasonal Seabird Research Assistant, helping to deliver monitoring and research projects across the SPA.

Natural England for granting the necessary permissions and continued support of research projects across the pSPA.

The Emmerson family of Flamborough who operate their fishing cobles out of North Landing for essential boat support to deliver the whole colony counts, with special thanks to skipper Mike Emmerson, crew member Bob and Joanne Emmerson who organised the logistics.

Filey Sailing Club for their boat support at Filey Cliffs who, with the help of skipper Dave Warburton and crew member Mick, allowed us to use their RIB to access the length of Filey Cliffs to carry out the colony count. I would also like to thank Filey Sailing Club Safety Officer, Chris Place for logistic support throughout.

The RSPB East Yorkshire Local Group and Flamborough Bird Observatory for their continued financial support and tireless fundraising efforts that help deliver seabird monitoring and research and members of Filey Bird Observatory & Group, for their continued commitment to monitoring Black-legged Kittiwake at Filey Cliffs.

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The Flamborough Head EMS Management Scheme Project Officer, Heather Davison, for her work on coordinating the recreational disturbance study and bringing together user groups to produce voluntary codes of conduct that will help protect the SPA.

LEADER for funding optics and other monitoring equipment that continue to provide essential tools for our volunteer team.

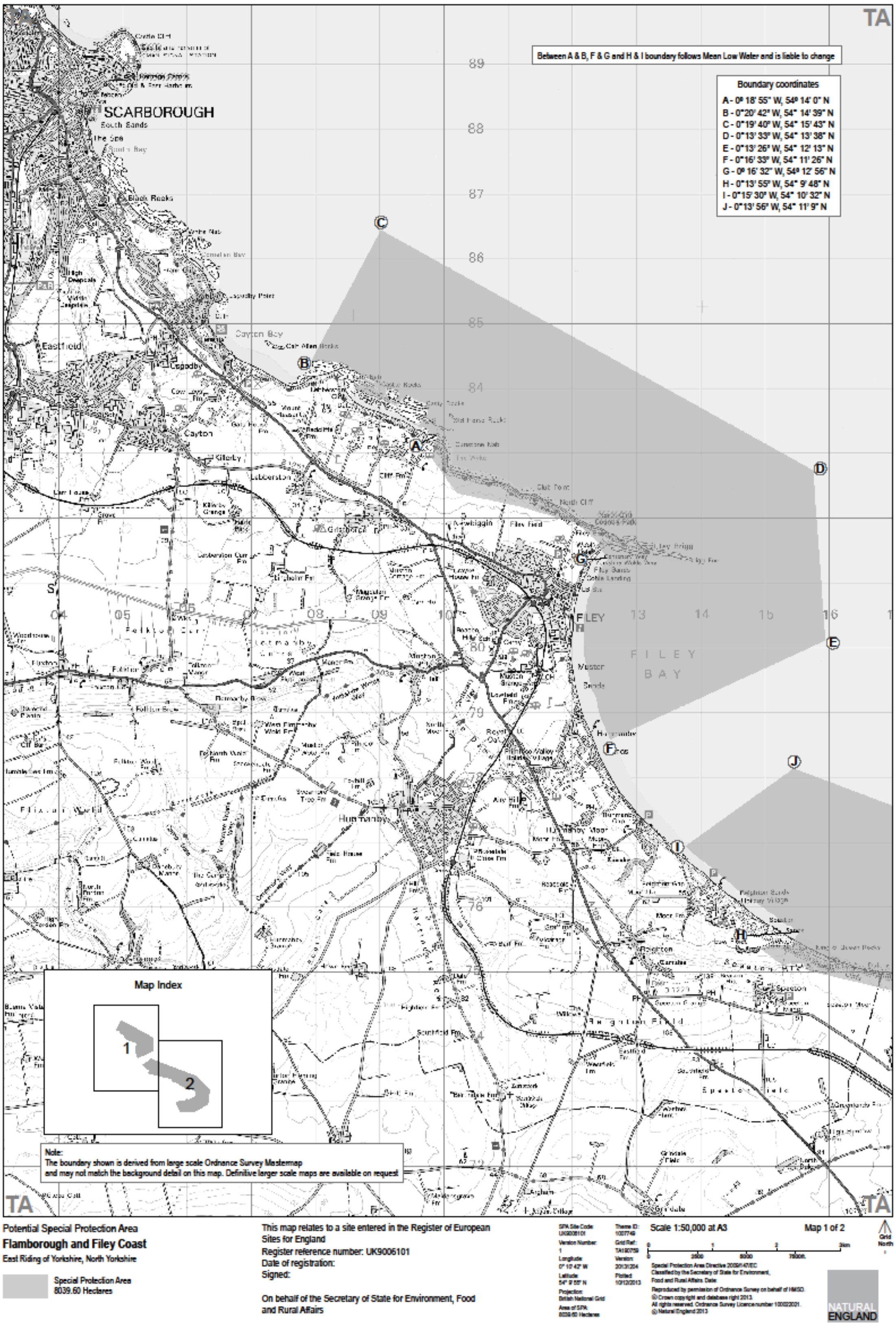
Blue Dolphin Holiday Park at Filey for allowing access to reach important sections of the colony for essential monitoring works.

And last but not least, the owners and management at Thornwick Bay Holiday Village at Flamborough for providing invaluable parking permits for North Landing car park.

Without all of the aforementioned, the Flamborough and Filey Coast seabird monitoring programme would not be the success that it is.

Appendix 1: Flamborough and Filey Coast pSPA Maps

North



South


Between P & Q boundary follows Mean Low Water and is liable to change
Boundary coordinates
K- 0°4' 56" W, 54° 9' 1" N
L - 0°2' 27" W, 54° 7' 23" N
M - 0°2' 53" W, 54° 5' 52" N
N - 0°5' 41" W, 54° 4' 57" N
O - 0°8' 21" W, 54° 5' 1" N
P - 0°9' 2" W, 54° 6' 3" N
Q - 0°7' 9" W, 54° 6' 10" N
K
L
M
N
O
P
Q
TA
FLAMBOROUGH HEAD
Flamborough
Bempton
BRIDLINGTON
B R I D L I N G T O N
Map Index
1
2
Note:
The boundary shown is derived from large scale Ordnance Survey Mastermap
and may not match the background detail on this map. Definitive larger scale maps are available on request
Potential Special Protection Area
Flamborough and Filey Coast
East Riding of Yorkshire, North Yorkshire
Special Protection Area
8039.60 Hectares
This map relates to a site entered in the Register of European Sites for England
Register reference number: UK9006101
Date of registration:
Signed:
On behalf of the Secretary of State for Environment, Food and Rural Affairs
Scale 1:50,000 at A3
Map 2 of 2
Grid North
NATURAL ENGLAND

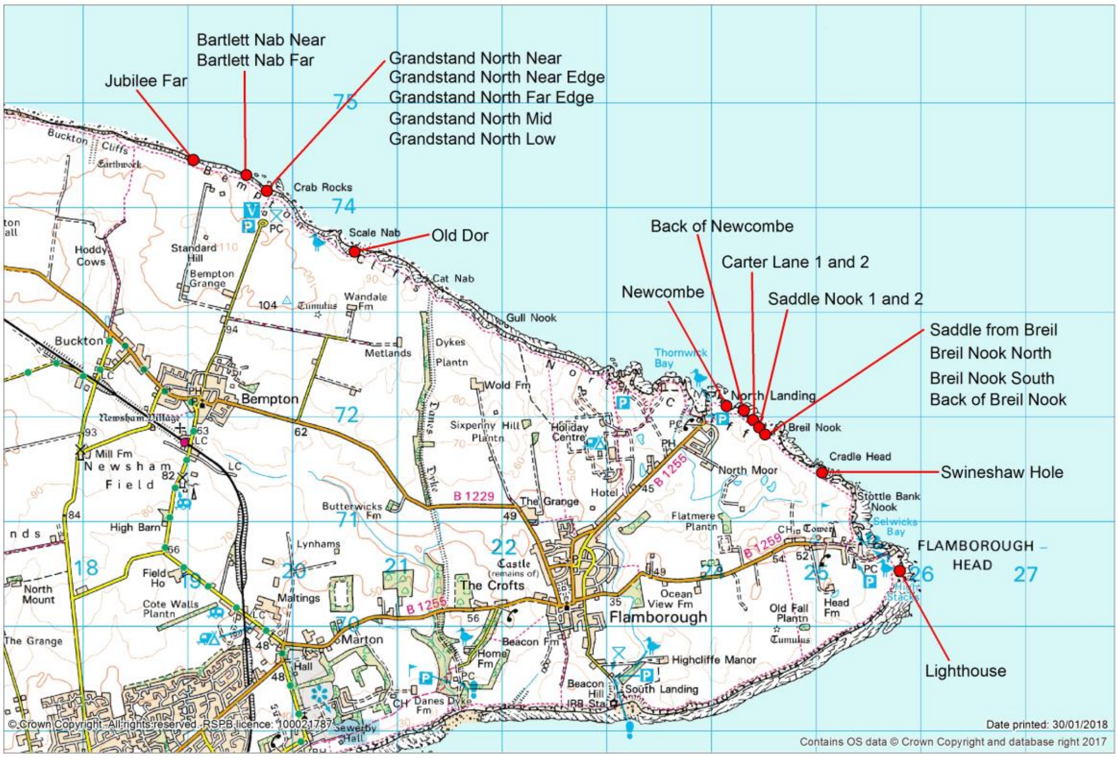
Appendix 2 - Productivity Plot Locations

Northern Fulmar productivity plots

Northern Gannet productivity plots

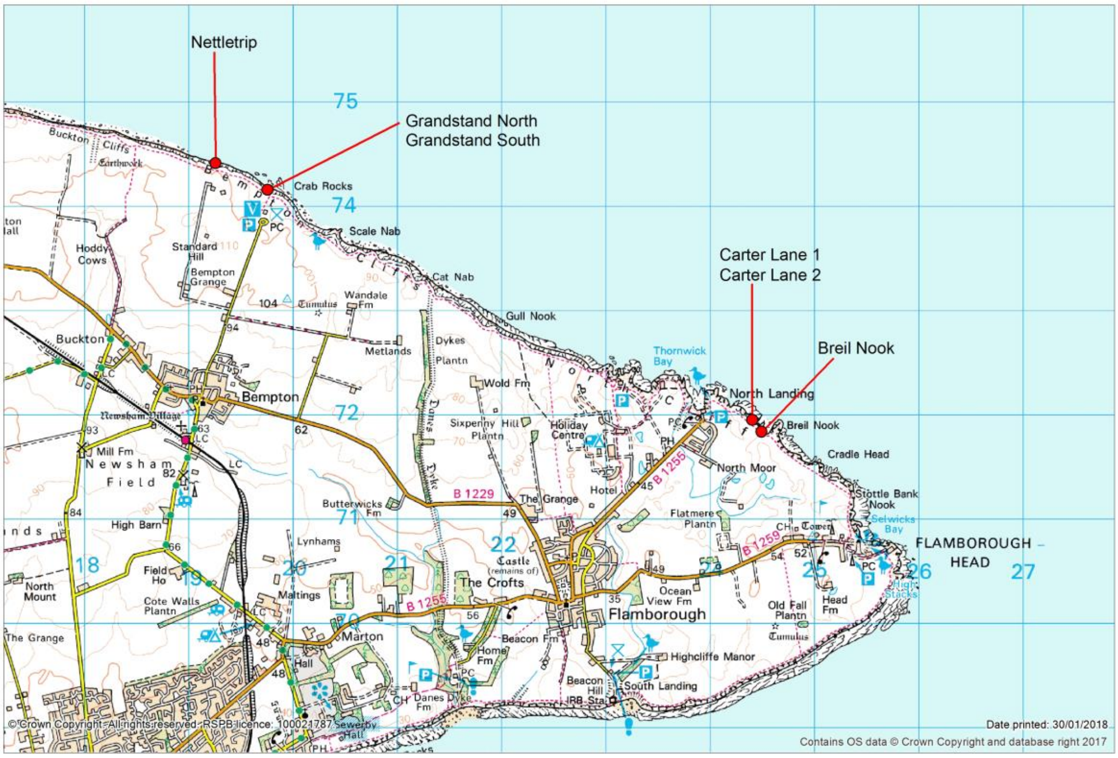
European Herring Gull productivity plots

Black-legged Kittiwake productivity plots – Flamborough and Bempton



Black-legged Kittiwake productivity plots – Filey

Common Guillemot productivity plots



Razorbill productivity plots

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Appendix 3 – Study-plot Locations

Black-legged Kittiwake study-plot locations

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Common Guillemot study-plot locations

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Razorbill study-plot locations

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The foraging behaviour and energetics of wandering albatrosses brooding chicks

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Abstract: The energy expenditure of ten (five male, five female) wandering albatrosses (*Diomedea exulans* Linnaeus 1758) brooding chicks on Bird Island, South Georgia, was measured using doubly-labelled water. At-sea foraging behaviour was measured in the same individuals using satellite telemetry and leg-mounted activity recorders. Mean mass-specific daily energy expenditure was 341 kJ kg^{-1} day^{-1} during a mean of 4.12 days at sea and did not differ between the sexes. This is significantly lower than previously reported for the species and the lowest recorded for any albatross. There were no significant relationships between energy expenditure and the proportion of time spent flying (59.7%), distance flown (1448 km) or average speed (16.5 km h^{-1}) suggesting that flying is not the most energetically expensive activity during foraging trips.

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Key words: energy expenditure, foraging behaviour, South Georgia, wandering albatross

Introduction

The behaviour of flighted pelagic seabirds at sea has traditionally been difficult to study because of their ability to cover great distances out of the range of both land- and ship-based observers. Recent advances in technology, however, have led to the miniaturization of a variety of devices for monitoring behaviour to the point where they can be deployed on, or in, medium to large seabirds without apparent adverse affects on their natural behaviour. This has enabled researchers to study various aspects of the physiology and behaviour of seabirds at sea, such as body temperature, heart rate, time spent flying, foraging times and locations, diving patterns and meal sizes in a range of species (e.g. Prince & Morgan 1987, Jouventin & Weimerskirch 1990, Weimerskirch & Wilson 1992, Afanasyev & Prince 1993, Bevan *et al.* 1994, 1995, Huin 1994, Prince *et al.* 1994, Falk & Moller 1995). Size constraints, however, still limit the number and type of devices that can be deployed and few studies have managed simultaneously to monitor more than one of these parameters; the energy costs of at-sea behaviour have been particularly neglected in recent studies.

The wandering albatross (*Diomedea exulans* Linnaeus 1758) is the largest and most pelagic of all seabirds. It breeds biennially on subantarctic islands throughout the Southern Ocean, a successful breeding season lasting nearly one year (Tickell 1968, Croxall *et al.* 1990). Several recent satellite tracking studies have revealed that wandering albatrosses have an extensive foraging range during the breeding season, travelling up to 15 000 km in a single foraging trip (Jouventin & Weimerskirch 1990, Prince *et al.* 1992, Weimerskirch *et al.* 1993). The monitoring of foraging times and meal sizes, using stomach temperature archival recorders (Weimerskirch & Wilson 1992, Wilson *et al.* 1995), has suggested that wandering albatrosses capture the majority of their prey during the day when, as indicated by data from leg-mounted activity recorders, they spend as little as 17% of the time on the water (Prince & Morgan 1987). Weimerskirch & Wilson (1992) estimate that over 50% of prey items weigh less than 200 g (but see Wilson *et al.* 1995) and, by combining satellite tracking with stomach temperature recording, they suggested that wandering albatrosses encounter prey at a rate of one item for a minimum of every 107 km travelled. It is not surprising, therefore, that these birds may spend up to 74% of the time away from the nest flying (Afanasyev & Prince 1993).

The wandering albatross uses a "dynamic soaring" mode of flight (Pennycuick 1982) which provides a substantial energy-saving advantage over flapping flight (Baudinette & Schmidt-Nielsen 1974). It is the energy efficiency of soaring flight that enables wandering albatrosses to cover great oceanic distances during foraging trips with the lowest energy expenditure (1.83 times basal metabolic rate) recorded for any free-ranging bird (Adams *et al.* 1986). Although recent studies of at-sea behaviour have provided new insights into the foraging ecology of this species, no information is available on how different behaviours relate to energy expenditure. In the present study, our aim was to investigate the relationship between foraging range, activity patterns and energy expenditure.

Methods

The study was conducted on Bird Island, South Georgia (54°00'S, 38°02'W), during the 1992 chick-rearing period. The energy expenditure, at-sea activity and foraging range of adult wandering albatrosses were measured during the early brood stage (March–April).

Energy expenditure

The energy expenditure of ten birds (five male, five female from separate nests) was determined using the doubly-labelled water method (Lifson & McClintock 1966, Nagy 1980, Speakman 1993). The energy expenditures of two individuals (one male, one female) were measured on two consecutive foraging trips.

After being weighed on a spring balance (± 0.1 kg), each individual was given an oral dose, by stomach tube, of 10 ml $H_2{}^{18}O$ 10% AP (Isotec Inc., Miamisburg, OH, USA) and a 1 ml weighed dose (± 0.01 g) of tritiated water (HTO; 200 μCi ml^{-1}) by intra-muscular injection. Each bird was kept in a portable enclosure next to its nest for an isotope equilibration period of four hours. A blood sample (5 ml) was then collected from a tarsal vein into a heparinized syringe before the bird was released. The times of the bird's departure from, and return to, the colony were recorded by direct observation.

Upon return to the colony each bird was recaptured, before it relieved its mate from brooding duties, weighed and a final blood sample was collected to determine the clearance rates of the administered isotopes. In two individuals another blood sample was also collected after they returned from their next foraging trip. To minimize the handling time and the effects of stress, background blood samples were not collected from the study individuals. Instead, blood samples were collected from six control animals to determine the background levels of isotopes in the albatross population. All blood samples were stored at 4°C for several hours before being centrifuged and the plasma fraction separated. For HTO analysis, subsamples (1–2 ml) of plasma were stored frozen (-20°C) in plastic vials until analysed in November 1993. For $H_2{}^{18}O$ analysis, aliquots (25–50 μl) of plasma were stored in flame-sealed capillary tubes until analysis in January 1994.

To measure the specific activity of tritium in plasma, samples were thawed and 0.2 ml subsamples were distilled into preweighed scintillation vials following the procedures of Ortiz *et al.* (1978). The vials were then reweighed to obtain the mass of the water sample, accurate to 0.1 mg. Scintillant (10 ml Ultima Gold; Canberra Packard, Brook House, Pangbourne, Berkshire, UK) was added to the vials which were then counted for 10 min in a Beckman LS1701 scintillation counter with correction for quenching by means of the sample channels ratio and an external standard to set the counting window for each vial. Samples were analysed in duplicate and each vial was counted twice. Subsamples (0.2 ml) of the injectant were counted in the same way, and at the same time, as the water from the plasma samples to determine the specific activity of the tritium injected.

Plasma samples were analysed for ^{18}O content at Centrum voor Isotopen Onderzoek, Gröningen, the Netherlands, following the methods of Masman (1986). Isotopic enrichments were determined in triplicate on a SIRA-9 mass spectrometer. Because the $H_2{}^{18}O$ was administered orally, we were not confident that the animals would receive an accurately measured dose and, hence, oxygen dilution space could not be determined. Total body water (TBW) was, therefore, calculated only from the initial dilution of HTO (Nagy 1983). Total body water at the end of the study period was calculated by multiplying the fractional water content at the beginning of the study by body mass upon recapture.

Carbon dioxide production was calculated from DLW measurements using the equations of Speakman *et al.* (1993) accounting for changes in TBW. A constant of 25.2 J ml^{-1} was used to convert CO_2 production to energy expenditure (Costa 1987) based on the average calorific value of chemical components of the diet (35% squid, 41.5% fish, 0.2% crustaceans and 18.8% carrion) of wandering albatross at Bird Island (Clarke & Prince 1979, Prince & Morgan 1987). Time ashore was calculated as the difference between the duration of the energy expenditure measurement and the time at sea (determined by direct observation). At-sea metabolic rate (MR, kJ kg^{-1} day^{-1}) was then calculated for each animal by solving the equation: Measured MR = [(Ashore MR) (Proportion of time ashore)] + [(At-sea MR) (Proportion of time at sea)], assuming the metabolic rate while ashore to be 248 kJ kg^{-1} day^{-1}, the rate reported for incubating wandering albatrosses (Brown & Adams 1984).

At-sea activity and foraging range

An activity recorder with a saltwater switch (24 g, 1.8 cm diameter tube of 8.2 cm length) was attached to a darvic ring on one of the legs of each bird (Afanasyev & Prince 1993). Whether the instrument was wet or dry was recorded at 3 s intervals and integrated hourly to give the proportion of every hour the bird spent flying or on the water. Night was defined as the dark period between the midpoints of civil twilight at South Georgia.

Information on foraging locations and range was obtained using satellite telemetry. Toyocom T2028C platform terminal transmitters (PTTs) weighing 180 g (1.3–2% of bird's body mass) were attached to the dorsal feathers of eight birds following the procedures detailed in Prince *et al.* (1992). Data on the position of instrumented birds were obtained via the ARGOS system and manipulated with purpose-built computer software. Only location estimates of ARGOS system Class = 0 (at least two transmitter messages received during the satellite pass) or better were used for mapping the general flight path of the bird's foraging trip whereas only Class 1–3 (accuracy of 100 m–1 km) were used for flight speed estimates (Prince *et al.* 1992). Both PTT and activity recorder were removed from the study animals at the end of the measurement intervals.

Statistical analyses followed methods outlined in Sokal & Rohlf (1981) and Zar (1984) using Unistat® Statistical Package (Version 4.5, Unistat Limited, London, UK). The Kolmogorov-Smirnov test was used to determine whether

Table I. Sex, measurement interval, body mass, total body water and CO_2 production rates of wandering albatrosses studied during the brood period.

Bird	Sex	Study period (days)	Body mass (kg) initial	Body mass (kg) final	TBW (kg) initial	TBW (kg) final	TBW (%)	CO_2 production (ml kg^{-1} min^{-1})
1	M	3.88	11.3	11.1	4.71	4.63	41.7	8.12
1	M	6.13	11.1	11.1	4.63	4.63	41.7	9.61
2	M	5.96	12.3	11.5	5.10	4.77	41.5	7.45
3	M	7.00	10.0	9.0	4.53	4.08	45.3	6.18
4	M	5.88	11.5	10.8	5.65	5.31	49.1	7.24
5	M	5.17	10.0	8.7	4.89	4.26	48.9	8.43
6	F	3.92	9.3	9.4	3.83	3.88	41.2	9.87
7	F	5.96	10.0	9.5	4.02	3.82	40.2	8.37
8	F	4.04	9.7	10.5	4.62	5.00	47.6	9.41
8	F	6.81	10.5	9.0	5.00	4.29	47.6	8.44
9	F	8.88	8.7	7.8	3.85	3.45	44.3	8.23
10	F	5.13	7.8	7.9	3.46	3.50	44.3	10.08
Mean		5.73	10.2	9.7	4.52	4.30	44.5	8.45
S.E.		0.40	0.4	0.4	0.17	0.16	0.9	0.32

data were normally distributed and an *F*-test was used to confirm homogeneity of variances. Unless otherwise stated, data are presented as means ± 1 s.e. and results were considered to be significant at the $P < 0.05$ level.

Results

The measurement interval, sex, body mass, TBW, and CO_2 production rates for the ten study birds are presented in Table I. Males were significantly heavier (10.7 ± 0.4 kg) and had larger TBW pools (4.9 ± 0.2 kg) than females (mass 9.2 ± 0.3 kg, TBW 4.1 ± 0.2 kg; $P < 0.02$ in both cases) but percent TBW (44.5 ± 0.9%) was not significantly different between the sexes ($P > 0.7$). Mass-specific CO_2 production rates during the whole of the measurement period were significantly greater in females (9.1 ± 0.3 ml min^{-1} kg^{-1}) than in males (7.8 ± 0.4 ml min^{-1} kg^{-1}; $t_{10,0.05} = 2.30$, $P < 0.05$). At-sea mass-specific CO_2 production rates did not differ significantly between the sexes (females 10.5 ± 1.0 ml min^{-1} kg^{-1}, males 8.3 ± 0.6 ml min^{-1} kg^{-1}; $t_{10,0.05} = 1.87$, $P > 0.09$, combined 9.4 ± 0.7 ml min^{-1} kg^{-1}). However, as no sex-specific on-land CO_2 production rates could be assigned to these birds, the lack of statistical difference in at-sea CO_2 production rates should be viewed with caution. Indeed, one could probably infer that even at the $P = 0.09$ level there may well have been a significant difference given a larger sample size. For the purposes of the remaining analyses in this study, however, we have assumed at-sea mass-specific CO_2 production rates are the same for each sex.

Due to instrument failure, data on at-sea activity and foraging range were each obtained for only eight foraging trips. These data and the duration of each foraging trip, daily

Table II. Time at sea, experimental protocol, mass change, energy expenditure, at-sea activity and distance flown by wandering albatrosses making foraging trips during the brood period.

Bird	Time at sea (days)	Type of trip[1]	Instruments carried[2]	Mass change (kg day^{-1})	Energy expenditure at sea[3] (kJ day^{-1})	Energy expenditure at sea[3] (kJ kg^{-1} day^{-1})	Multiples of BMR[4]	% of time flying	Distance flown (km)	Average speed (km h^{-1})
1	2.17	shelf edge	AR, PTT	-0.05	3696	330	1.53	57.9	732	13.2
1	4.17	polar front	AR, PTT	0.00	4398	396	1.83	92.8	1941	28.1
2	5.00	*	AR, PTT	-0.13	3248	273	1.26	*	1687	14.5
3	6.08	polar front	AR, PTT	-0.14	2079	219	1.01	28.3	1869	10.9
4	3.17	*	AR, PTT	-0.12	2998	269	1.24	57.4	*	*
5	3.71	*	AR, PTT	-0.25	3007	322	1.49	*	*	*
6	3.71	shelf edge	AR, PTT	0.03	3407	364	1.69	*	722	10.0
7	4.50	*	AR	-0.08	3117	320	1.48	52.5	*	*
8	1.29	shelf edge	AR, PTT	0.20	5668	561	2.60	65.2	426	29.5
8	3.81	polar front	AR, PTT	-0.22	3287	337	1.56	69.4	2688	13.1
9	7.63	*	AR	-0.10	2514	305	1.41	*	*	*
10	4.19	polar front	AR, PTT	0.02	3080	392	1.82	54.2	1282	12.7
Mean	4.12			-0.07	3375	341	1.58	59.7	1418	16.5
S.E.	0.46			0.03	254	24	0.10	4.4	187	2.6

[1] Foraging trips fell into two broad categories: to the continental shelf edge (short) and the Antarctic Polar Frontal Zone (long); [2] AR = activity recorder and PTT = satellite transmitter. Instrument failure prevented data collection from some animals and this is indicated by an * in the table; [3] Calculated from CO_2 production rates using a conversion of 25.2 kJ per litre of CO_2; [4] BMR = 216 kJ -kg^{-1} -day^{-1} (Brown & Adams 1984).

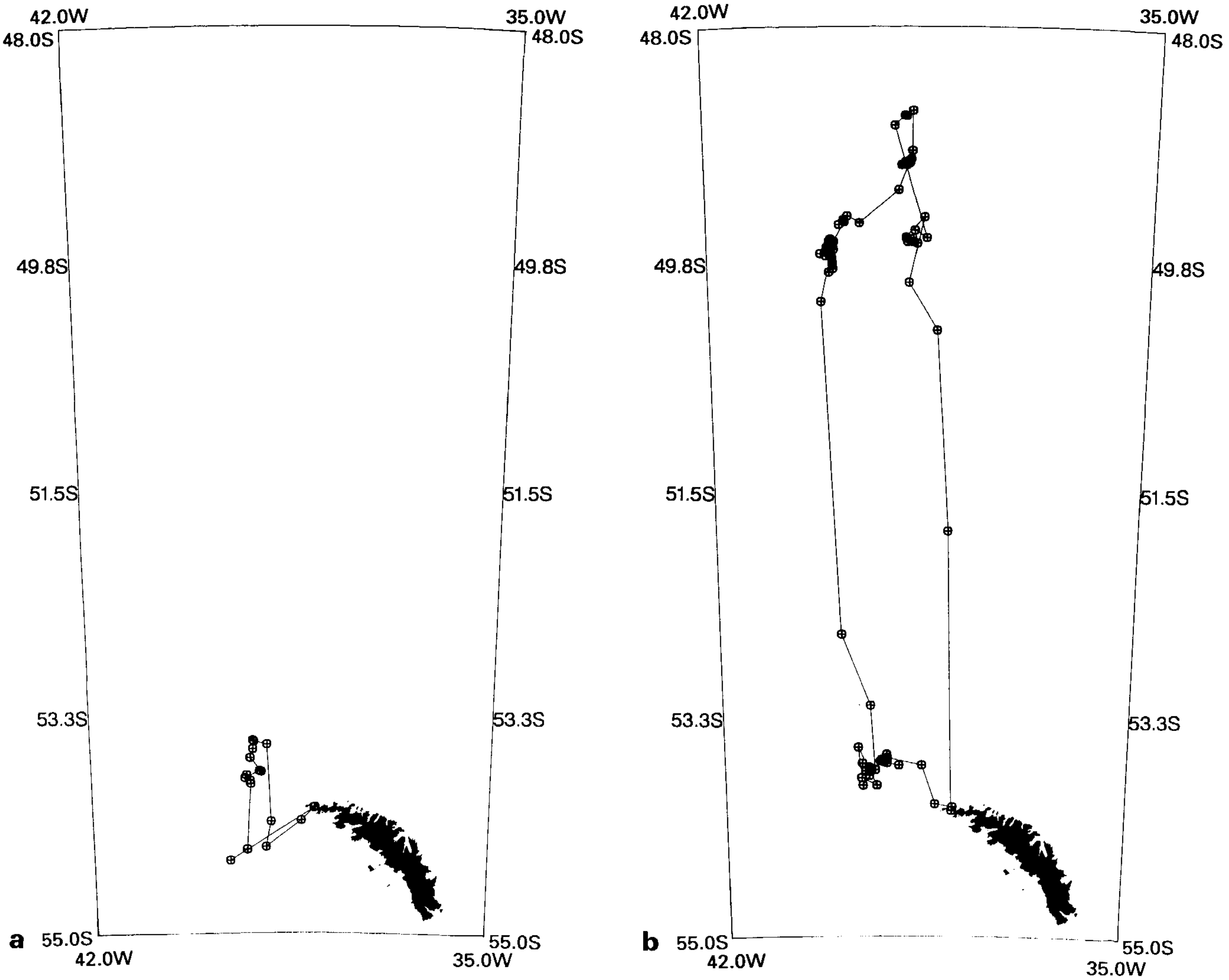

Fig. 1. Examples of the two distinct types of foraging trips observed in this study: **a.** short trips to the continental shelf edge; and **b.** longer trips to the Antarctic Polar Frontal Zone, shown here for Bird 8 (first trip) and Bird 3, respectively.

mass change, and energy expenditure are presented in Table II. There were no significant differences in the time spent at sea (4.12 ± 0.46 days), proportion of time spent flying (59.7 ± 4.9%), distance flown (1448 ± 211 km), daily mass change (-0.07 ± 0.03 kg $\cdot$day^{-1}) or at-sea metabolic rate (340.7 ± 23.8 kJ $\cdot$kg^{-1} $\cdot$day^{-1}) between the sexes ($P > 0.1$ in all cases) so the data were combined. Activity patterns differed temporally with birds spending 71.5 ± 4.1% of daylight time flying compared to only 44.0 ± 9.0% during the night.

There were generally two types of foraging trips undertaken by the study birds: short trips ranging no further than the continental shelf edge; and longer trips ranging much further to the Antarctic Polar Frontal Zone. Examples of the track typical of such foraging trips is given in Fig. 1; the appropriate categories for each bird are listed in Table II.

Both absolute and mass-specific daily energy expenditure were not related to the proportion of time spent flying, the distance flown, or average speed ($P > 0.2$ in all cases). Furthermore, there were no apparent relationships between these variables and the type of foraging trip undertaken (shelf edge or polar front). Mass-specific daily energy expenditure was, however, positively correlated with daily mass gain ($r^2 = 0.63$, $n = 12$, $P < 0.002$; Fig. 2a) and negatively correlated with time at sea ($r^2 = 0.37$, $n = 12$, $P < 0.04$, Fig. 2b). Daily mass gain was not related to either time at sea, the proportion of time spent flying, the distance flown or the average speed during a foraging trip ($P > 0.2$ in all cases). Absolute mass gain, however, was significantly negatively correlated with time at sea ($r^2 = 0.35$, $n = 12$, $P < 0.05$).

Discussion

Body composition and energy expenditure

The body masses of both the male and female wandering albatrosses recorded in the present study are similar to those

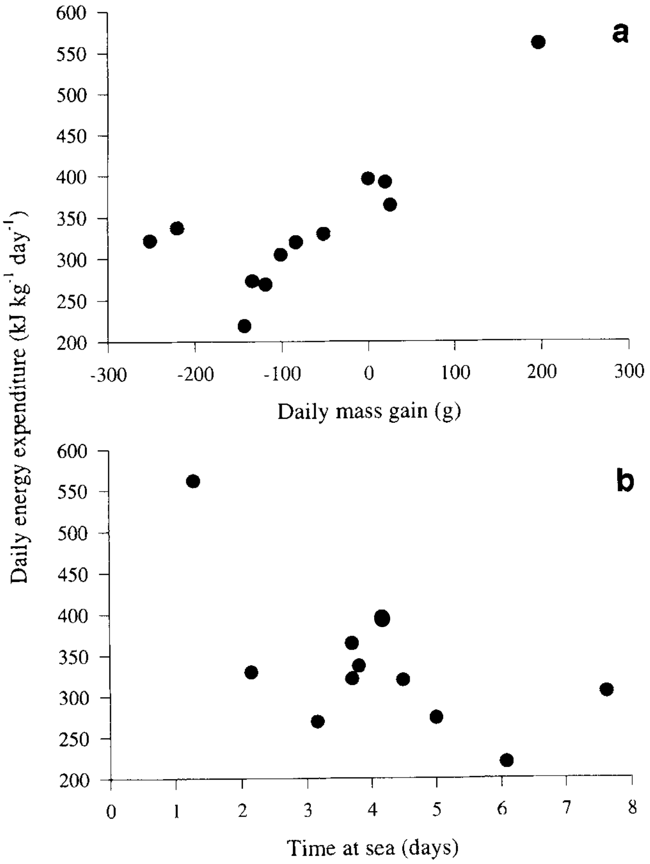

Fig. 2. a. The relationship between daily energy expenditure and daily mass gain during time at sea; and **b.** the relationship between daily energy expenditure and time at sea in wandering albatrosses during the brood period.

previously reported for this species at South Georgia (Croxall & Ricketts 1983) and, as in all albatross species, the males were heavier than the females (Warham 1990). The mean percent TBW of 44.5% found in this study is similar to the 47.5% reported by Adams *et al.* (1986) for wandering albatrosses breeding on Marion Island. Such TBW contents are lower than that reported for many marine and aquatic birds (Mahoney & Jehl 1984) but similar to that found in other procellariiforms (e.g. 45.9% in Wilson's storm petrel *Oceanites oceanicus* Kuhl 1820 and 51.9% in southern giant petrels *Macronectes giganteus* Gmelin 1789) (B.S. Obst, K.A. Nagy & R.E. Ricklefs unpublished data in Adams *et al.* 1986), including the Laysan albatross (*D. immutabilis* Rothschild 1893, 47%) (Pettit *et al.* 1988). It is thought this is due to the relatively long wings and flight feathers in procellariiforms contributing relatively more dry matter to the total body mass than in other species (Adams *et al.* 1986). In contrast, however, Costa & Prince (1987) recorded relatively high TBWs of 57.9% in the grey-headed albatross (*D. chrysostoma* Forster 1785).

The greater mass-specific CO_2 production rates, and hence metabolic rates, in females than in males found in the present study may simply reflect their smaller size (greater surface area to volume ratio). However, they may also reflect differences in activity levels. Whereas our methods were not able to document a significant difference in at-sea metabolic rates between the sexes, more detailed analyses of both on-land and at-sea activity patterns and their energetic costs may provide a better understanding of the observed sex-specific difference in CO_2 production rates.

The mean at-sea mass-specific energy expenditure recorded in the present study (341 kJ kg^{-1} day^{-1}) is significantly lower ($t_{19,0.05} = 6.05$, $P < 0.0001$) than that reported by Adams *et al.* (1986) for wandering albatrosses at Marion Island with 40–80 day old chicks (397 kJ kg^{-1} day^{-1}). This is due to differences in the equations used for calculating carbon dioxide production from isotope dilution and clearance rates. In the present study we used equations based on the two pool model (Speakman *et al.* 1993) whereas Adams *et al.* (1986) used equations for the single pool model (Lifson & McClintock 1966). The two pool model invariably produces an estimate lower than the single pool model and, indeed, the difference between the results of present study and that of Adams *et al.* (1986) disappears when the Lifson & McClintock (1966) equations are applied to our data (395 kJ kg^{-1} day^{-1}; $t_{19,0.05} = 0.149$, $P > 0.88$). Recent evidence in birds suggests the two pool model may be the most appropriate (Bevan *et al.* 1995) and, hence, for this reason we have presented our results using equations based on it. Nonetheless, the mass-specific energy expenditures reported in both studies are considerably lower than that observed in other species of albatross (Table III).

The grey-headed albatross, Laysan albatross and white-

Table III. Comparison of at-sea mass-specific energy expenditure in four species of albatross.

Species	Mass (kg)	At-sea energy expenditure (kJ kg^{-1} day^{-1})	Multiples of BMR	Stage of breeding season	Reference
Wandering albatross *D. exulans*	9.95	341	1.58*	brood period	This study
	8.42	397	1.83*	chick alone	Adams *et al.* (1986)
Grey-headed albatross *D. chrysostoma*	3.66	657	2.47#	incubation period	Costa & Prince (1987)
Laysan albatross *D. immutabilis*	3.06	676	2.60#	incubation period	Pettit *et al.* (1988)
White-capped albatross *D. cauta cauta*	3.92-4.2	670	2.57-2.62#	chick alone	Green & Brothers (1995)

*Measured basal metabolic rate (see Table II); #BMR predicted using BMR = 381.8 $M^{0.721}$ where BMR is in kJ day^{-1} and M is body mass in kg (Ellis 1984).

capped albatross (*D. cauta cauta* Gould 1841) all expend energy at 2.4–2.6 times basal metabolic rate while foraging whereas on average wandering albatrosses only expend energy at 1.58–1.83 times BMR (Adams *et al.* 1986, Costa & Prince 1987, Pettit *et al.* 1988, Green & Brothers 1995). All albatrosses are anatomically adapted for soaring flight (Pennycuick 1982) but there are no data available on energy expenditure during flight in any of these species to compare the cost of flying. Adams *et al.* (1986) suggested that the higher energetic cost of foraging in the grey-headed albatross compared to that in the wandering albatross may be partially due to differences in the amount of time spent flying between the two species. However, reports on the proportion of foraging trips spent flying vary considerably within species (Prince & Morgan 1987, Afanasyev & Prince 1993, this study) making it difficult to infer such time-energy budget differences between species. More recently, Prince *et al.* (1994) have observed that mollymawks submerge to much greater depths (2.5–4.7 m) than does the wandering albatross (0.3 m) and suggested that diving behaviour may be a typical part of foraging activity in mollymawks. Foot propelled diving is energetically expensive (Ellis 1984) and this activity may explain the higher foraging energy expenditure of mollymawks compared to wandering albatrosses.

At-sea behaviour

The mean time at sea observed in the present study (4.12 days) is longer than the previously reported means during the brood period for this species at South Georgia and Iles Crozet (2.7 and 2.8 days, respectively) but well within the ranges (1–10 days and 1.3–5 days, respectively; Tickell 1968, Weimerskirch *et al.* 1993). The range of trip durations in this study was also considerable (1.3–7.6 days). Due to small sample sizes, we were not able to assess the effect of instrumentation on foraging trip BMR. However, Weimerskirch *et al.* (1992) found no difference in the time spent at sea between birds instrumented with a PTT and controls.

Foraging trip durations of wandering albatrosses vary considerably throughout the breeding cycle (Tickell 1968, Weimerskirch *et al.* 1993). In a study of breeding birds at Iles Crozet, Weimerskirch *et al.* (1993) have shown that these changes are associated with changes in foraging ranges; birds at sea for short periods (2.4–2.8 days) forage within 260 km of the colony and travel distances of less than 1000 km whereas birds at sea for long periods (>10 days) forage up to 1500 km from the island and may travel as much as 6000 km. Similarly, South Georgia birds in the late chick-rearing period (August) travel up to 5000 km during mean foraging trip durations of seven days (Prince *et al.* 1992) whereas during the brood period (this study) birds travel less than 1500 km during short trips.

In the present study there was no relationship between the distance covered during a foraging trip and the time spent at sea. Similarly, whereas Weimerskirch *et al.* (1993) found a positive relationship between time at sea and distance covered during long trips of the incubation and chick-rearing periods, they found no such relationship during the brood period. The average flight speed (16.5 km h^{-1}) recorded in this study is approximately half that observed by Prince *et al.* (1992) for South Georgia birds during the chick rearing period (29 km h^{-1}) but similar to the 16 km h^{-1} reported for wandering albatrosses at the Auckland Islands during the incubation period (Walker *et al.* 1995). Whether these differences in flight speeds are due to differences in foraging effort or prevailing weather patterns is not known.

The proportion of time spent flying varied substantially between birds (range: 28–93%) but the mean (60%) fell within the range of means previously reported for the species (57–74%, Prince & Morgan 1987, Afanasyev & Prince 1993). The temporal distribution of the time spent flying concurs with Afanasyev & Prince's (1993) previous observation that most of the flying is done during the daylight hours. These results, in conjunction with information from stomach temperature loggers on the timing of foraging and satellite tracking, support Weimerskirch & Wilson's (1992) contention that wandering albatrosses generally rest during periods of darkness but actively search for food during the day.

Energetic costs of foraging

There were no relationships between at-sea energy expenditure and distance flown, the average speed or proportion of time spent flying. This suggests that, in general, flying is not the most energy expensive activity during foraging trips and is consistent with dynamic soaring being an energy efficient mode of flight (Baudinette & Schmidt-Nielsen 1974). Indeed, satellite tracking studies have shown that the flight paths of wandering albatrosses generally make use of prevailing winds, flying constantly with a crosswind in the hindquarter (Weimerskirch *et al.* 1992). At-sea daily energy expenditure was, however, negatively related to the time spent at sea. One possible explanation for this observation is that on longer foraging trips a smaller proportion of the overall time is spent in certain activities which may be energetically expensive. For example, birds that make long trips may spend most of the time in transit to the foraging area (Prince *et al.* 1992). During transit flight, birds may make use of the prevailing winds to reduce energy expenditure but manoeuvring within the foraging zone in search of food may require flying in many different directions over short distances, resulting in considerably greater energy expenditure. Investigations of energy expenditure with finer temporal resolution and on an activity-specific basis, as is now possible with the recent advances in heart-rate monitoring (Bevan *et al.* 1994, 1995), are needed to verify this hypothesis.

Daily mass gain was positively correlated with at-sea daily energy expenditure. This suggests that birds expending more energy in search of food are more successful foragers. Interestingly, the amount of mass gained during a foraging trip was negatively related to the time spent at sea. This is the opposite to what has been observed in wandering albatrosses during the incubation period at the Iles Crozet (Weimerskirch 1995).

In summary, the results of the present study further illustrate the energetic efficiency of flight in wandering albatrosses. The absence of any relationship between foraging behaviour and energy expenditure is consistent with the current interpretation of the foraging strategy in this species; using the prevailing weather conditions to cover great distances in search of food to scavenge (Prince *et al.* 1994). It would be of great interest, therefore, to conduct similar investigations on mollymawks, which have a similar mode of flight but a different feeding behaviour of greater potential energetic cost (Ellis 1984, Prince *et al.* 1994).

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EFFECT OF INSTRUMENT ATTACHMENT AND OTHER FACTORS ON FORAGING TRIP DURATION AND NESTING SUCCESS OF ADÉLIE PENGUINS

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Abstract. We compared foraging-trip duration of Adélie Penguins (*Pygoscelis adeliae*) carrying various combinations of radio-telemetry transmitters, implanted, passively interrogated transponder (PIT) tags, and time-depth recorders at two widely separated colonies of different size on Ross Island, Antarctica, during three austral summers. Trip duration was measured by electronic devices rather than human observation. Instrumentation had no significant effect on foraging trip duration. Most of the variation in foraging trip duration was attributed to individual and year. Males' trips were significantly shorter than females' in a subset of known-sex birds. No effect was evident in nesting success even for birds that wore instruments for >20 days. We recommend use of small, hydrodynamically designed and placed instruments to researchers who wish to collect data unaffected by instrument attachment.

Key words: *foraging, penguin, radio-transmitter, recorder, seabird, sea ice, weighbridge.*

Efectos de la Sujeción de Instrumentos y Otros Factores en la Duración de las Salidas de Forrajeo y el Éxito Reproductivo en *Pygoscelis adeliae*

Resumen. Se comparó la duración de las salidas de búsqueda de alimento de *Pygoscelis adeliae* a los que se les colocaron distintas combinaciones de radiotransmisores, emisores implantados de interrogación pasiva y medidores de tiempo y profundidad de buceo. El estudio fue realizado durante tres veranos australes en dos colonias ampliamente separadas de diferente tamaño en Ross Island, Antártica. La duración de las salidas fue medida por instrumentos electrónicos en lugar de observaciones humanas. Los instrumentos no tuvieron un efecto significativo en la duración de las salidas de búsqueda de alimento. La mayor parte de la variación en la duración de las salidas fue atribuida a individuos y años. Las salidas de los machos fueron significativamente más cortas que las de las hembras en una sub-muestra de aves previamente sexadas. No se detectaron efectos evidentes en éxito de nidificación incluso en individuos que llevaron instrumentos por más de 20 días. Recomendamos el uso de instrumentos pequeños diseñados y colocados hidrodinámicamente a fin de no influenciar los datos colectados.

INTRODUCTION

Adélie Penguins (*Pygoscelis adeliae*) are remarkably sturdy animals, supplying ample opportunities for increasing our knowledge of the Antarctic marine ecosystem (e.g., Croxall and Prince 1979, Wilson et al. 1994, Wilson 1995). Many previous investigators have attached radio-transmitters (Trivelpiece et al. 1986, Davis et al. 1988, Sadleir and Lay 1990) and time-depth recorders (Naito et al. 1990, Watanuki et al. 1993, Wilson et al. 1993) to penguins to learn more about various aspects of their at-sea behavior. Using relatively small samples, several researchers have reported effects of these different instrument packages, as well as effects of the methods and timing of attachment (for review see Wilson and Culik 1992). They have done this either by comparing foraging-trip duration or chick rearing of instrument-wearing birds to unencumbered controls (Croll et al. 1991, Watanuki et al. 1992, 1993, 1997), studying their energetic expenditures (Culik and Wilson 1991, Culik et al. 1994), or examining their hydrodynamic properties at various swim speeds (Bannasch et al. 1994). Results have indicated that many instruments slow penguin swimming speeds significantly (Culik and Wilson 1991, Wilson et al. 1991, Culik et al. 1994); cause ex-

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TABLE 1. Number of individuals (*n* trips) for weighbridge (WB), transmitter (RT), and time-depth recorder (TDR) groups by season and colony (individuals not repeated among treatments). Each treatment was applied to a separate subcolony at each colony.

Colony	Treatment	1997–1998	1998–1999	1999–2000	All years
Crozier	WB	52 (367)	38 (259)	57 (312)	96 (938)
	RT	21 (187)	17 (135)	17 (127)	55 (449)
	TDR			12 (23)	12 (23)
Royds	WB	65 (571)	52 (375)	64 (495)	120 (1441)
	RT	20 (207)	16 (135)	17 (118)	53 (460)
	TDR			10 (21)	10 (21)

cessive preening, pecking, or other ‘‘psychological’’ hindrance (Wilson et al. 1989, Wilson and Wilson 1989a); generally increase energetic expenditure (Culik and Wilson 1991, Bannasch et al. 1994); and decrease nesting success (Watanuki et al. 1992). It has been proposed that these effects can be minimized by attaching smaller, streamlined instruments on the lower back behind the area of greatest girth, where the thickest boundary layer (area of slowest flow velocity and resulting lowest drag) is found (Bannasch et al. 1994).

Higher energetic costs and associated slower swimming speeds have also been attributed to metal flipper bands on Adélie Penguins in a swim canal (Culik et al. 1993), though such an effect of bands in the wild has not been reported.

Clearly there are many potentially confounding variables involved with interpreting the effects of instrument attachment on trip duration, and these can be difficult to sort out without adequate sample size or other methodological considerations. For example, most works examining factors affecting foraging-trip duration have relied on a relatively small number of birds at a single colony during a single breeding season (but see Watanuki et al. 1992, 1993, 1997 for a previous multi-season study). Few investigators have reported the effect of sex or individual behavior irrespective of instrument attachment, although Clarke et al. (1998) found that male Adélie Penguins made consistently shorter (distance and duration) foraging trips than females. Others have noted differences between the sexes in energetics (Chappell et al. 1993) and timing of colony attendance (Ainley et al. 1983).

We investigated foraging-trip duration and breeding success for Adélie Penguins during the guard and crèche stages of chick rearing at two colonies for three seasons. As a part of this effort we compared banded birds carrying implanted, passively interrogated transponder (PIT) tags and unbanded birds fitted with a streamlined radio-transmitter (RT) or time-depth recorder (TDR) and RT. We tested whether device attachment affected nest success or trip duration for the different colony-season groupings, and whether trip duration was related to colony, season, day within season, individual, sex, or the cumulative number of trips in a season an individual had made at the time of the measured trip.

METHODS

STUDY AREAS

Our study covered most of the guard and crèche stages of chick rearing (21 December to 11 January) for three austral summers, 1997–1998 to 1999–2000. We will refer to the austral summers as seasons. Study colonies were Cape Crozier (120 000 pairs total) and, 77 km away, Cape Royds (4000 pairs) on Ross Island (77°30′S, 168°E) in the southern Ross Sea, Antarctica. At each of these colonies, we selected two subcolonies of approximately 200 pairs for foraging-trip comparisons using two methods of assessment. We will refer to these subcolonies as either RT (radio-transmitter) or WB (weighbridge). RT subcolonies were different each season at Cape Crozier, but were the same at Cape Royds (though individual RT birds were different). WB subcolonies were the same each season and included many of the same individuals among seasons. In the 1999–2000 breeding season we also attached TDRs and RTs to several individuals from an additional subcolony at each site (Table 1).

RADIO-TRANSMITTERS

Each season we attached RTs to approximately 15 adults at each colony, each having chicks in their nests on 20 December. Transmitters pulsed 0.92 times $\sec^{-1}$, were manufactured by Ad-

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vanced Telemetry Systems (ATS, model PN7, Isanti, Minnesota), weighed 15.7 g, measured 2.0 × 1.1 × 5.3 cm (2.2 cm^2 maximum cross section) and were tapered according to our instructions for better hydrodynamics (Bannasch et al. 1994, Culik et al. 1994). Using published measurements of Adélie Penguin maximum frontal cross-sectional body area of 314 cm^2 and 200 cm^2 (Oehme and Bannasch 1989, Wilson et al. 1989), our RTs measured $\leq$1% of the birds' cross-sectional area. Each RT had a 29 × 0.1 cm whip antenna that angled toward the tail at 42° relative to the bird's back. Birds were caught at their nests by hand and held for approximately 10 min while instruments were attached using black Tesa® tape (Wilson and Wilson 1989b). At Cape Royds in all seasons and at Cape Crozier in the first season, individuals were selected from different nests. At Cape Crozier in 1998–1999 and 1999–2000, we tried to capture both members of pairs as they changed between foraging and brooding duty (i.e., during nest exchanges). We attached most RTs to the lower rump, on the centerline just above the uropygial gland, but in 1998–1999 we attached some (21 of 33) to the area between the scapulas, at approximately the area of maximum girth. RTs were removed on or after 11 January each season.

We used a scanning receiver and an ATS DCCII data logger to record the presence or absence of RT birds in the colonies. The scanner was programmed to ''listen'' for transmitters for 30 sec on each frequency every 90 min. A reference transmitter was also logged, and the scanner was checked throughout the season to verify accuracy. We adjusted sensitivity as necessary. Transmitters that logged fewer than 10 pulses in a 30-sec scan were classified as absent from the subcolony, since background radio interference or other perturbations sometimes caused the scanner to record up to 10 pulses (GB, pers. obs.).

WEIGHBRIDGES

We used plastic fencing to surround subcolonies, leaving one access point, which we filled with a WB. The WB consisted of an electronic scale, direction indicator, and radio-frequency-identification reader connected to a data logger (see Kerry et al. 1993 for a complete description of a similar system). We implanted unique PIT tags in 216 Adélie Penguins between 1996 and 2000. Tags, manufactured by Avid Inc. (Norco, California), weighed 0.1–0.3 g and were 1.4–1.8 cm long and 0.2–0.3 cm wide. Tags were injected subcutaneously in the scruff of the neck, between and a little above the shoulders. Birds with tags were also banded on the left flipper with numbered stainless steel bands (National Museum, Wellington, New Zealand), of the type used on Magellanic Penguins (*Spheniscus magellanicus*) without apparent deleterious results (P. D. Boersma, pers. comm.). Handling time was generally <2 min per individual. We attempted to maintain at least 30 active nests at which both adults were tagged at each of the two study subcolonies. WBs were installed between 10 December and 15 December in all seasons. The identification, time, direction, and body mass for >95% of crossings were recorded (Table 1). In the WB subcolony at Cape Crozier, sex was determined for 80 individuals (83% of banded birds; 42 males and 38 females) using a combination of copulatory position, behavior, ''tread marks'' (evidence of female copulatory position), bill and head size differences, and body mass (Ainley et al. 1983, Kerry et al. 1992).

TIME-DEPTH RECORDERS

In a third subcolony at each colony, we selected one or two individuals every few days (approximately 2 foraging trips) and attached a TDR (Wildlife Computers, model MK7, Redmond, Washington) and RT in the same way as described for RTs above. TDRs weighed 25.4 g, measured 1.1 × 2.0 × 8.5 cm (2.2 cm^2 maximum cross section, $\leq$1% of the penguins' cross-sectional area), and were streamlined. They were mounted directly behind small, rounded RTs (ATS model PN5, similar but slightly smaller than the PN7 model described earlier), as low on the back as possible without interfering with preening. The two instruments' total length was ca. 13.3 cm. This style RT also had a 29 × 0.1 cm whip antenna pointed toward the tail and parallel with the back. We calculated foraging trip duration using the dive data collected by the TDRs to determine when birds entered and left the water, and adding 0.8 hr for birds at Crozier and 0.5 hr for Royds based on observed walking times from the beaches to the subcolonies. We confirmed these trip durations with data collected by the scanning receiver when available, but due to logistical difficulties we did not always

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use the receiver for monitoring TDR birds. Once birds had made at least 2 foraging trips, the instruments were removed, downloaded, and moved to new individuals.

NEST CHECKS

All nests (WB, RT, and TDR) were checked on a daily basis, weather permitting. Sample size for the WB subcolonies, totaled for the three field seasons, was 216 individuals among 215 nests (108 at Royds, 97 at Crozier). Sample size for RT birds was 108 individuals and 101 nests (40 at Crozier, 61 at Royds). Transmitters were removed when nests or transmitters failed.

As controls, nests in similar-sized subcolonies were monitored for outcome in each season at each colony. The control subcolonies were within 50 m of WB subcolonies. In both control and study subcolonies, observations were made of eggs and chicks at regular intervals by standing quietly 10 m away and waiting for birds to rise to show nest contents.

STATISTICAL ANALYSIS

We only included trips made by birds that had chicks within at least two days of a trip's completion to ensure that we measured trips of adults that were provisioning chicks. Trip duration data tended to be skewed toward shorter trips and were normalized using log-transformation. Foraging trips as measured by WB or RT were only included if they were between 6 and 120 hr long to help avoid artifacts from the sampling methodologies. For example, the WB often recorded short trips (<6 hr), which were primarily made by birds carrying rocks for nests (pers. obs.). Additionally, the RT receiver was susceptible to incorrectly logging a bird as present when radio interference bled onto a penguin's frequency, or as absent when birds were present in the colony for <90 min and missed a ''listening'' session. In general the RT data were less accurate than either WB or TDR, despite the low sample size of the latter. Although we checked the scanner/data logger as often as possible and adjusted the sensitivity and tuning as required, it still missed birds and incorrectly registered their presence. These effects could be mitigated by scanning for a larger range of frequencies around the target to capture drift, and by scanning more frequently (e.g., every 30 min). While these problems are artifacts of our study design, we believe the accuracy is far greater than checking the nests in person once per day or even as often as every 4 hr, protocols used in previous studies.

We tested for differences in trip duration between shoulder-mounted and lower-back attachments for RT animals, controlling for individual and colony, using a 3-way ANOVA. There were no differences in trip duration for the two attachments ($F_{1,236} = 2.5$, $P > 0.1$). Therefore, no differentiation between the attachments was made for any of the following analyses.

Because several individuals in each WB subcolony were present in multiple seasons, we evaluated the effect of season, colony, interaction between season and colony, and individual in the WB subcolonies using a 4-way repeated measures ANOVA. For this model we included only the individuals present as breeders in each year. Because all terms were highly significant, we followed this with a 5-way ANOVA that included treatment (RT versus WB) type and all birds. Wald tests were used to evaluate the significance of differences in trip duration between different colony-season combinations.

Since there was significant variation in trip length with season and season-colony interaction, we tested the effect of two within-season factors: 5-day period and cumulative trip number. Five-day period is the day in the season an individual returned from a foraging trip, binned into 5- or 6-day groups: 1 = 21–25 Dec, 2 = 26–30 Dec, 3 = 31 Dec–4 Jan, 4 = 5–10 Jan). We calculated cumulative trip number of each trip for each individual, and then placed this into one of four categories: 1 = 1–4, 2 = 5–6, 3 = 7–9, 4 = 10–31 trips. Though these variables are usually highly correlated, we treated them separately because RTs were occasionally installed after the beginning of the study (when nests or instruments failed). Because individuals were repeated in period and cumulative-trip-number categories, we used 3-way (treatment, individual, and period or trip number) repeated-measures ANOVAs for each season in order to obtain corrected significance estimates.

For the third season we included the TDR group in the comparison. We used 4-way ANOVA to compare the effects on trip duration of treatment (RT, WB, or TDR), colony, 5-day-period, and individual. Estimates of significance for 5-day-period were biased due to repeated individuals. We left cumulative trip number out of this model because we did not have the data for

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TABLE 2. Untransformed foraging trip duration as calculated by weighbridge (WB) and radio-transmitters (RT) compared for Cape Royds and Cape Crozier, December 21 to January 11. Periods are 5-day intervals. All birds had young within at least 2 days of a trip.

		Crozier					Royds				
Season	Period	WB (hr)	*n* (trips)	RT (hr)	*n* (trips)	Difference (hr)	WB (hr)	*n* (trips)	RT (hr)	*n* (trips)	Difference (hr)
1997–1998	all	30.0	367	24.0	187	6.0	16.0	571	23.3	207	–7.3
	1	24.1	61	19.2	43	4.9	17.3	87	19.1	62	–1.8
	2	24.9	103	22.1	51	2.8	15.9	148	25.7	55	–9.8
	3	27.3	99	24.6	46	2.7	15.3	148	24.6	46	–9.3
	4	41.0	104	29.8	47	11.2	14.8	188	25.1	44	–10.3
1998–1999	all	32.6	259	31.6	135	1.0	26.3	375	29.0	135	–2.7
	1	32.0	53	22.8	33	9.2	33.3	57	27.6	33	5.7
	2	30.5	63	31.9	36	–1.4	25.4	97	25.9	39	–0.5
	3	33.7	60	33.5	29	0.2	28.9	84	31.7	26	–2.8
	4	33.8	83	37.8	37	–4.0	22.4	137	31.8	37	–9.4
1999–2000	all	32.5	312	32.9	127	–0.4	23.0	495	23.2	131	–0.2
	1	31.3	61	28.8	30	2.5	31.9	71	no data		
	2	31.9	76	31.6	32	0.3	22.3	94	17.1	21	5.2
	3	29.0	90	29.1	33	–0.1	23.0	142	25.0	46	–2.0
	4	37.4	88	41.9	32	–4.5	20.0	192	24.1	63	–4.1

the TDR birds (which were only measured for two foraging trips each and at different times within the season).

To test the effect of sex on foraging-trip duration, we compared trip lengths among the 80 known-sex birds in the WB group at Cape Crozier using a 5-way ANOVA that included season, 5-day period, cumulative trip number, and individual. Five-day period and cumulative trip number were treated as continuous variables. We then used body mass as a surrogate for sex in order to test for the effect of sex at Cape Royds (as well as at Cape Crozier). To do this, we first included body mass and the interaction between sex and mass in the above model. Finding no significant effect of either term (mass: $F_{1,\ 710} = 6.1$, $P > 0.1$; mass × sex: $F_{1,\ 710} = 0.02$, $P > 0.8$), we compared the mass of males to females using a two-tailed *t*-test and found that males were heavier than females ($P < 0.001$). We then substituted mass for sex in the above ANOVA, and added colony to the model to test if heavier birds (mostly males) made shorter trips than lighter birds (mostly females).

We classified nests (pairs) as successful when they produced at least one crèched chick. We summarized the number of successful and total nests for each study season at each colony for each treatment type (WB, RT, and control). We then used logistic regression to test whether nest success varied among these treatments.

Means are presented ± SE. Significance for all tests was assumed at the 0.05 level and tendencies at the 0.10 level, unless corrections for repeated terms were necessary (see above). Residuals from ANOVAs were normally distributed. All statistical calculations were performed using STATA (Stata Corp. 1999).

RESULTS

TRIP DURATION

Mean trip durations ranged from 16–42 hr (Table 2). Among the 14 WB birds that bred in each season (9 at Royds, 5 at Crozier), chick-provisioning trips varied significantly with individual, season, colony, and the interaction of season and colony (Table 3A). The combined model was surprisingly robust (adjusted $R^2 = 0.89$, $P < 0.001$). WB trips were longer at Cape Crozier than at Cape Royds ($\beta = 0.36$, $P < 0.02$).

WB trips did not differ from RT trips (using all breeders, despite between-season repetition of some) when compared using season, colony, season × colony interaction, and individual (Table 3B). Season, individual, and the interaction of season and colony were all highly significant in this model (all $P < 0.001$, combined adjusted $R^2 = 0.37$). The significance of the season × colony interaction was mostly due to trips being longer at Cape Crozier and shorter at Cape Royds in the first season than in other seasons ($P < 0.001$).

Within-season results were similar (Table 4).

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TABLE 3. Factors affecting foraging-trip duration in Adélie Penguins. (A) Weighbridge birds only: results from 4-way repeated-measures ANOVA (using log-transformed data); adjusted R^2 = 0.89, residual df = 24. (B) Weighbridge and radio-transmitters compared: results from 5-way ANOVA (using log-transformed data); adjusted R^2 = 0.37, residual df = 2957.

Term	df	*F*	*P*
A. Season[1]	2	41.3	<0.001
Colony	1	12.2	<0.002
Season × Colony[1]	2	6.0	0.03
Individual	12	7.8	<0.001
B. Treatment	1	1.5	>0.2
Season[2]	2	69.2	<0.001
Colony	1	2.1	>0.1
Season × Colony[2]	2	20.1	<0.001
Individual	324	3.4	<0.001

[1] *P*-values for these terms adjusted using Box's conservative epsilon.

[2] *P*-values for these terms are possibly inaccurate due to some repeated individuals (see text).

Individual was always the most significant factor affecting trip lengths (Fig. 1). In some seasons there were some significant or marginally significant but irregular effects of 5-day period and cumulative trip number. For example, trip durations grew consistently longer as the 1997–1998 season progressed (i.e., trip duration was significantly correlated with 5-day period, $P < 0.001$, Table 2) and as individuals made more trips ($P < 0.09$). However, cumulative trip numbers tended to be negatively correlated with trip duration in 1999–2000 ($P < 0.07$).

In 1999-2000, trip durations for birds carrying TDRs did not differ from those in WB or RT groups ($F_{2,\ 980} = 2.3$, $P > 0.1$). Trip durations for all three groups did vary with individual ($F_{179,\ 980} = 2.8$, $P < 0.001$) and tended to vary with 5-day period ($F_{1,\ 980} = 3.7$, $P < 0.06$). This was primarily due to trips getting shorter later in the season at Cape Royds.

We next turned our attention to factors affecting trip duration for which we only had data from the WB group: sex and body mass. Males' trips were shorter than females' ($F_{1,\ 712} = 5.8$, $P < 0.02$) among the known-sex birds at Cape Crozier. Males were also heavier than females (4.0 ± 0.03 kg vs. 3.8 ± 0.03 kg, $P < 0.001$), so we substituted mass for sex and used all WB birds (at both colonies) to see if there was a correlated effect on trip duration. For all birds combined and for Cape Crozier alone, mass did not play an important role. However, when we fit the model for Cape Royds only, heavier birds made shorter trips ($F_{1,1122} = 3.86$, $P = 0.05$), suggesting that males in this colony also made shorter trips than females.

BREEDING SUCCESS

There was no significant difference in nest success between treatments ($P > 0.6$), seasons ($P > 0.3$), or colonies ($P > 0.2$; Table 5). In 1999, we also monitored 22 nests while one adult from each carried a TDR (12 at Crozier, 10 at Royds). None of the TDR nests failed during the study, but these birds were only observed for approximately 2 foraging trips (usually 4–6 days).

DISCUSSION

Differences in trip duration between individuals, seasons, and colonies masked any effects of the instruments measuring the trips in this study. The seasonal and colony-related differences in trip duration were consistent with comparisons of Royds and Crozier birds in the 1996–1997 season using different methods (Ainley et al.

TABLE 4. Within-season trip durations related to treatment, individual, and (A) cumulative trip number or (B) 5-day period using 3-way repeated-measures ANOVA.

	1997–1998			1998–1999			1999–2000		
Term	df	*F*	*P*	df	*F*	*P*	df	*F*	*P*
A. Treatment	1	0.3	>0.5	1	4.2	<0.05	1	0.0	>0.8
Trip number[1]	3	2.3	<0.09	3	1.6	>0.1	3	2.5	<0.07
Individual	68	2.9	<0.001	35	3.1	<0.001	51	2.8	<0.001
Residual df	207			147			211		
B. Treatment	1	0.0	>0.9	1	0.1	>0.8	1	2.5	>0.1
5-Day period[1]	3	6.3	<0.001	3	1.5	>0.2	3	1.1	>0.3
Individual	98	4.4	<0.001	87	1.3	<0.05	80	2.3	<0.001
Residual df	399			355			327		

[1] *P*-values for trip number and 5-day period adjusted using Huynh-Feldt epsilon.

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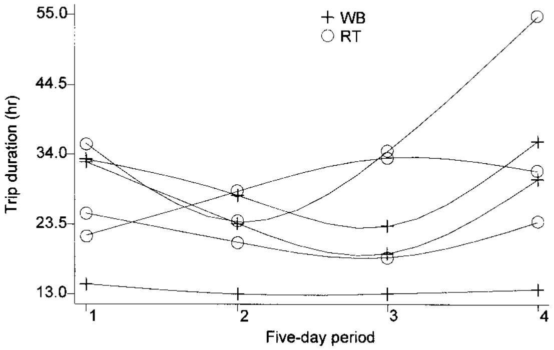

FIGURE 1. Individual variation in foraging-trip duration by 5-day period for six individuals in 1999–2000 at Cape Crozier. Three individuals carried radio-transmitters (RT) and three were in the weighbridge (WB) group.

1998). That study concluded that varying sea-ice conditions near the colonies probably had the greatest effect on foraging-trip duration.

Measuring foraging-trip duration may not be the best way to discern instrument effects on penguins. Kooyman et al. (1992) noted that instrumented Emperor Penguins (*Aptenodytes forsteri*) stayed with the same group of birds during foraging, but dove less often. Similarly, Ropert-Coudert et al. (2000) found that King Penguin (*Aptenodytes patagonicus*) foraging trip durations did not vary with instrumentation, but that there were subtle differences in diving behavior which the authors attributed to drag caused by TDRs (although they were unable to statistically control for individual variation). Watanuki et al. (1992) found stronger effects of instruments on Adélie Penguins by looking at feeding efficiency and chick survival, both of which decreased with instrumentation more obtrusive than used in our study, particularly later in the chick-rearing period. In contrast, Wilson et al. (1997) did not note an effect on breeding success of Adélie Penguins and other penguin species that were carrying relatively large packages. Regardless, some authors have found that instruments lengthened foraging trips of penguins, and that larger packages had bigger effects (Wilson et al. 1989, Hull 1997).

Although we did not see an effect on chick survival, it is possible that birds carrying TDRs and RTs made fewer or shallower dives and carried less food to their chicks, while staying at sea approximately the same amount of time as the WB birds. Also, it is possible that any effects on the chicks' well-being could have been expressed after completion of our study in each season (although we did remove instruments when we finished our measurements). On the other hand, the instruments that we deployed were smaller compared to almost all those used in the various studies we have referenced. Moreover, we applied them with much less obtrusive means than most other studies; we used a single layer of tape instead of surgical implantation or attachment by Velcro® and cable ties to epoxy-glued feathers.

Though trip durations for birds carrying the TDR/RT combinations did not vary from the other treatments, the comparison is not entirely valid because of the relatively short period these instruments were attached. Hull (1997) found

TABLE 5. Proportion of nests from which at least one chick reached crèche for controls, weighbridge (WB), and radio-transmitter (RT) groups in each colony and season. Number of nests monitored for each group shown in parentheses.

	Controls		WB		RT	
Season	Crozier	Royds	Crozier	Royds	Crozier	Royds
1997–1998	0.93 (30)	0.97 (35)	0.93 (30)	0.97 (35)	1.00 (19)	0.95 (21)
1998–1999	0.96 (28)	0.78 (37)	0.91 (23)	0.96 (28)	0.83 (12)	0.83 (18)
1999–2000	0.97 (29)	0.90 (30)	0.93 (44)	0.93 (45)	1.00 (9)	0.91 (22)

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that Royal Penguins carrying TDRs made longer foraging trips than controls, and occasionally failed to return from foraging trips at all, especially during the incubation stage when foraging trips were relatively long (20–25 days). However, the TDRs in that study were approximately twice as wide and twice as heavy as ours, and were not tapered for hydrodynamics. Still, a proper comparison of TDRs with RTs and controls would require treatments of equal duration, especially since other researchers have found a cumulative effect, particularly with larger instruments.

Since Wilson et al. (1989) found that effects of instruments were not manifest until later in the season (e.g., after 19 days), we compared later trips to earlier trips. We found no consistent pattern using date or cumulative trip number as the metric. It is possible that this effect is only noticeable in seasons or populations experiencing more difficult conditions (e.g., less food or less access to open water). The positive effect of date on trip duration during the 1997–1998 season may be related to prey-depletion near the colonies, or to late-season movement of pack ice (sea ice not connected to land) farther offshore (Ainley et al. 1998). Adélie Penguins are noted pack-ice obligates when at sea (Ainley et al. 1994) and may make long trips to reach the pack ice when necessary. Prey availability and proximity of pack ice may be highly correlated (Daly and Macaulay 1988). Conversely, fast ice (sea ice connected to land) was more extensive in 1999–2000 than in the other seasons in front of Cape Royds (pers. obs.), meaning that penguins had to walk several kilometers to reach open ocean early in the season. As the season progressed, the ice broke up and walking distances (and foraging trip durations) decreased accordingly.

Some of the differences in trip duration between colonies were no doubt due to different amounts of walking between the water and the study subcolonies. At Cape Crozier, the RT and TDR subcolonies were approximately 500 m from the sea, and Adélie Penguins had to negotiate several other subcolonies and a steep hill to reach their nests. Similarly, the Crozier WB colony was about 300 m from the ocean. At Cape Royds, all study subcolonies were closer to the ocean (approximately 100 m) and there were fewer intervening subcolonies. Adélie Penguins walk at 2.6–3.9 km hr^{-1} (Taylor 1962), but perhaps a little slower when climbing steep hills, so these differences could account for 10 to 30 min of the differences in foraging trips between colonies.

Regardless of instrumentation, some individuals made consistently shorter or longer trips than others (Fig. 1). Birds may revisit the same foraging areas once they find a reliable food supply (Ainley et al., unpubl. data). Most other studies of trip duration have not looked for individual variation, or even sexual variation, though Clarke et al. (1998), using data provided by relatively large satellite transmitters, found that males made shorter trips than females. In that study, the females fed beyond the continental shelf, while the males foraged over the shelf. We found a similar sexual differentiation in trip duration at Cape Crozier, where birds averaged longer trips than at Royds, but both males and females fed only over the continental shelf (unpubl. data). However, sexual differences in trip lengths did not explain all the variation; individual variation remained important.

In the two study seasons that Cape Royds RT birds made their longest trips, RTs were attached to 15 individuals from different nests (rather than to their mates). It is possible that at the time of attachment these individuals were mostly females, since timing of colony attendance can be skewed by sex, particularly during incubation or near peak hatching time (Ainley et al. 1983). It is also possible that attaching a RT to both members of pairs would cause longer trips than if only one member were treated. This would be more of a concern in studies including the incubation period (when trips are longer and one mate must attend the nest). We do not believe that a bird's foraging-trip duration during chick rearing is affected by the presence of a RT on its mate, except under the extreme circumstance that a mate's prolonged absence causes the guarding bird to undergo abnormal food deprivation, requiring more time at sea to recover. If there were such an effect, it should show up in the early part of the study period (while adults are guarding chicks) rather than later (when chicks are crèched and adults do not wait for each other to return). Since foraging trips in this study were relatively short ($<$6 days) and since we did not see any instrument effect at any time during the study, it does not seem likely that there was such an impact.

We conclude that a careful telemetry or time-

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depth recorder study, using the latest, smallest possible, streamlined instruments, applied using methods now well worked out (attached using tape to the centerline of the lower back), can result in data not seriously affected by the instruments. All instruments in this study were 1% or less of the cross-sectional area of the study species, which may provide a useful guideline. Antenna angle and length are variables that still need further consideration, and we have yet to properly look for effects of flipper bands on trip durations or foraging effort. The fact that banded birds sometimes made shorter and sometimes longer trips than instrumented birds indicates that no simple result will follow, but with the WB in place we do have the means to pursue this question. We encourage researchers investigating foraging-trip duration to consider a wide range of variables when interpreting their results, particularly those relating to individual variation. Clearly, more effort is still required to understand the factors that cause variation in trip duration among penguins, but it appears that these factors can now be addressed with confidence using the methods described here.

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USING A COLLISION RISK MODEL TO ASSESS BIRD COLLISION RISKS FOR OFFSHORE WINDFARMS

MARCH 2012

Bill Band

This guidance has been prepared for The Crown Estate as part of the Strategic Ornithological Support Services programme, project SOSS-02. It provides guidance for offshore wind farm developers, and their ecological consultants, on using a collision risk model to assess the bird collision risks presented by offshore windfarms.

The guidance has been extended in this March 2012 version to make use of flight height distribution data, where that data is available and robust; and to include a methodology for considering birds on migration, for which survey data on flight activity may be limited.

The guidance is accompanied by

- a Collision Risk Spreadsheet, which enables the calculations required to be undertaken and presented in a standardised manner
- a Worked Example, to illustrate the process
- a Tidal Variation spreadsheet, for use only when tidal effects may be significant

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The SOSS steering group includes representatives of regulators, advisory bodies, NGOs and offshore wind developers (or their consultants). All SOSS reports have had contributions from various members of the steering group. However the report is not officially endorsed by any of these organisations and does not constitute guidance from statutory bodies. The following organisations are represented in the SOSS steering group:

SOSS Secretariat Partners: The Crown Estate
British Trust for Ornithology
Bureau Waardenburg
Centre for Research into Ecological and Environmental Modelling, University of St. Andrews
Regulators: Marine Management Organisation
Marine Scotland
Statutory advisory bodies: Joint Nature Conservation Committee
Countryside Council for Wales
Natural England
Northern Ireland Environment Agency
Scottish Natural Heritage
Other advisors: Royal Society for the Protection of Birds
Offshore wind developers: Centrica (nominated consultant RES)
Dong Energy
Eon (nominated consultant Natural Power)
EdF Energy Renewables
Eneco (nominated consultant PMSS)
Forewind
Mainstream Renewable Power (nominated consultant Pelagica)
RWE npower renewables (nominated consultant GoBe)
Scottish Power Renewables
SeaEnergy/MORL/Repsol (nominated consultant Natural Power)
SSE Renewables (nominated consultant AMEC or ECON)
Vattenfall
Warwick Energy

PURPOSE OF GUIDANCE

1. Offshore windfarms may have a number of effects on bird populations:
   - **Displacement** – birds may partially or totally avoid a windfarm and hence be displaced from the underlying habitat.
   - **Barrier effects** – birds may use more circuitous routes to fly between, for example, breeding and foraging grounds, and thus use up more energy to acquire food.
   - **Habitat effects** – birds may be attracted or displaced by changes in marine habitats and prey abundance as a consequence of the windfarm.
   - **Collision risk** – birds may be injured or killed by an encounter or collision with turbines or rotor blades.

   This guidance relates to the last of these, collision risk.

2. **An environmental statement for an offshore windfarm should include a quantitative estimate of collision risk for all bird species present on the site for which the level of risk has the potential to be important. The environmental statement should provide a view on the significance of that collision risk on the respective bird populations.**

3. The aim of this guidance is to promote a standardised approach to collision risk assessment for offshore windfarms, to increase the transparency of calculations, and hence promote greater confidence in the results; to enable estimates from different windfarms to be more easily compared and combined so as to facilitate cumulative assessment; and hence enable collision risk assessment to be used as a tool in selecting the best areas for offshore windfarm development.

4. The guidance describes the information needed, and how to use that information, to arrive at an estimate of collision risk. It is accompanied by a spreadsheet which enables the necessary calculations to be performed in a standardised way.

INFORMATION NEEDED

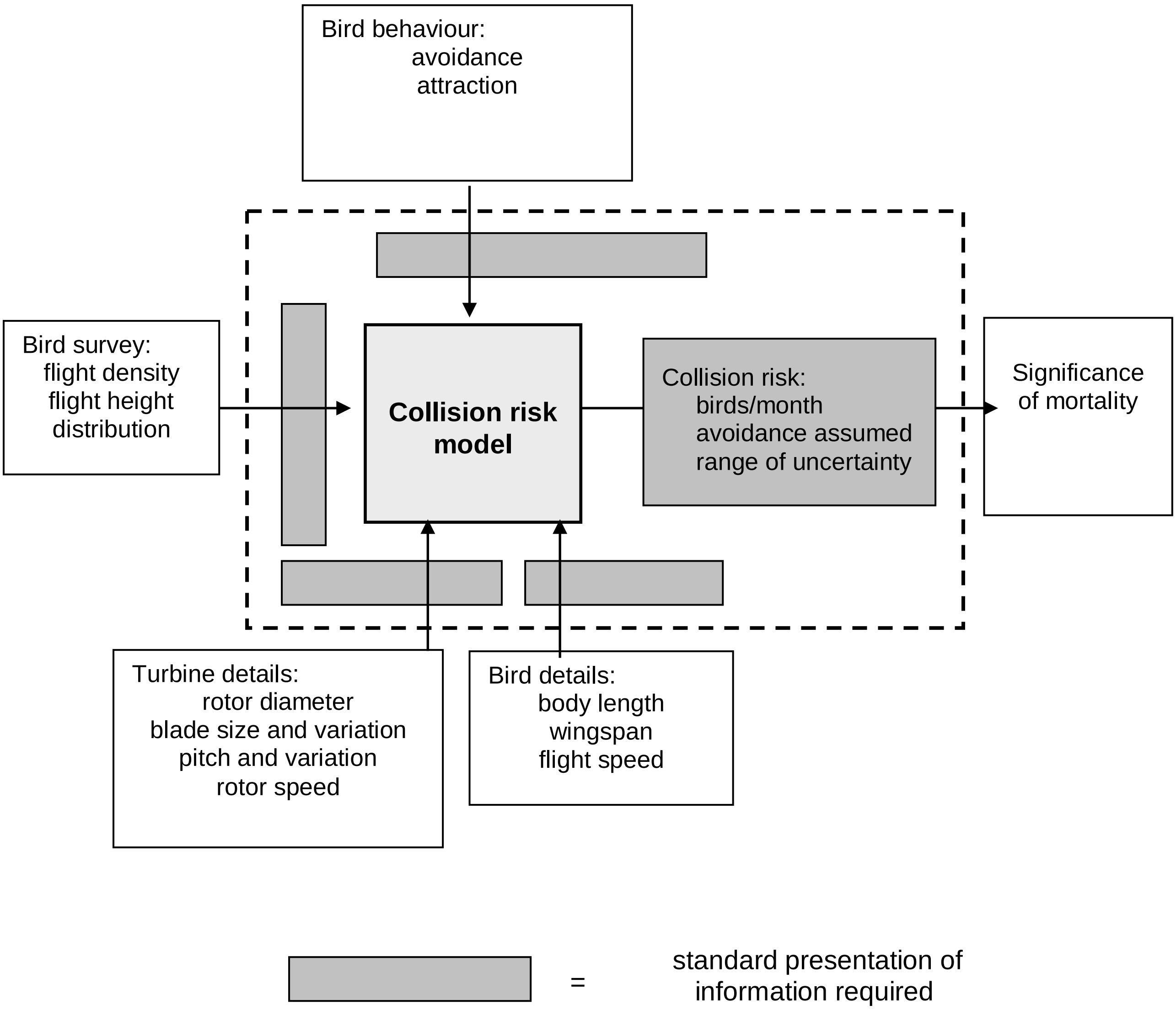
5. Figure 1 shows the information needed to estimate collision mortality:
   - **information derived from bird survey** - on the number of birds flying through or around the site, and their flight height
   - **bird behaviour** - prediction of likely change of behaviour of birds, eg in avoiding, or being attracted to, the windfarm
   - **turbine details** - physical details on the number, size and rotation speed of turbine blades
   - **bird details** - physical details on bird size and flight speed

6. This guidance sets out how that information should be presented and used within a collision model, and how the outputs from that model should be expressed – ie the components in the dashed ‘box’ in Figure 1. The guidance does not cover:
   - bird survey methods - for which there are various advisory sources.
   - bird behaviour - while it outlines how an avoidance rate factor should be used in the collision risk calculation, the guidance leaves it to other sources, where possible based on actual monitoring of bird collisions at windfarms, to advise on what avoidance rates should be used.

   Figure 1 also indicates the key outputs from the collision model – the collision risk, expressed in terms of the likely number of birds per month or per year which will collide with the windfarm, and the range of uncertainty surrounding that estimate. These should be accompanied by a clear statement of the assumptions on avoidance made in arriving at that estimate, as such assumptions are often be critical to the magnitude of the collision estimate. This guidance includes advice on how these outputs should be presented.

7. Note that the collision risk model stops at an assessment of collision risk. Where collision risk is not negligible, a developer will need to further consider the significance of the predicted mortality - which will depend on the sensitivity of the bird population, and the degree of protection afforded by legislation and any protected sites in the vicinity which may be designated for that species.

Fig 1: Role of collision risk model



COLLISION RISK MODEL

8. The approach adopted follows in general terms that developed by Band (2000)[i] and Band et al (2007)[ii] and promoted in guidance published by Scottish Natural Heritage, but it has been updated to facilitate application in the offshore environment. The offshore approach differs from onshore mainly in the methods used to gather and present information on flight activity, given that direct observations of birds from key vantage points are not usually possible in the marine environment. The approach is described below in six stages:

Stage A	assemble data on the number of flights which, in the absence of birds being displaced or taking other avoiding action, or being attracted to the windfarm, are potentially at risk from windfarm turbines;
Stage B	use that flight activity data to estimate the potential number of bird transits through rotors of the windfarm;
Stage C	calculate the probability of collision during a single bird rotor transit;
Stage D	multiply these to yield the potential collision mortality rate for the bird species in question, allowing for the proportion of time that turbines are not operational, assuming current bird use of the site and that no avoiding action is taken;
Stage E	allow for the proportion of birds likely to avoid the windfarm or its turbines, either because they have been displaced from the site or because they take evasive action; and allow for any attraction by birds to the windfarm eg in response to changing habitats; and
Stage F	express the uncertainty surrounding such a collision risk estimate.

9. The basic model has recently (March 2012) been extended to make use, where it is available, of data on the distribution of bird flight heights; in particular to enable use of the data on flight heights of birds at sea compiled for SOSS by Cook et al[iii]. This 'extended model' is described following Stage D, as within that model Stages B, C and D become merged in a single calculation. Another addition is Annex 6, which describes use of the model when assessing the collision risk to birds on migration, where there may be limited bird survey information on flight activity.

General features

10. Risk is turbine-based. Risk in this model is calculated directly from the rotor parameters and the flight activity in the airspace surrounding each turbine. Some practitioners have used an approach which considers the risk to each bird passing through a windfarm, taking account of the layout and spacing of turbines to calculate the likelihood of encountering one or more turbines and the resulting risk. This is unnecessary if one focuses, as in this guidance, on the risk resulting from each turbine operating within its own airspace within which there is a known (or projected) level of flight activity.

11. Relationship to previous guidance. The approach to quantifying and expressing flight activity in this guidance differs from that set out in the earlier Band papers. These papers offered two alternative approaches for calculating the likely number of flights through turbines: the first using observations of bird flux passing through a vertical 'risk window' enveloping the turbines; and the second assessing the 'bird occupancy' of the volume of airspace occupied by the windfarm as a whole. Both these methods are mathematically equivalent to the method described below and in the attached spreadsheet, in which the core measures of flight activity used are the density of flying birds per unit horizontal area of the windfarm, and the proportion flying at turbine height. The current approach leads to the same results and avoids the need to identify arbitrary risk windows or to define an arbitrary windfarm boundary. The basic model and spreadsheet used to calculate the risk for a single bird flight through a rotor are also as in the earlier papers (though subject to minor refinement). Thus, collision

risk estimates resulting from application of the basic model in this guidance should not differ substantively from those deriving from correct application of the earlier Band papers.

12. Oblique approach simplified. There is a simplification involved in separating out Stages B and C, in assuming that the probability of collision for any bird passing through a rotor is the same regardless of the direction of flight. In fact, the collision risk depends to some extent on a bird's angle of approach, determined by the direction of its flight and the orientation of the turbine blades. A bird approaching a turbine at an oblique angle is exposed both to a reduced probability of flying through the rotor, because the rotor presents an elliptical rather than circular cross-section, and an increased risk of collision if it does so. The model adopted for use here assumes that these two factors exactly offset each other, such that all bird transits can be treated as if making perpendicular approach to the rotor. This enables Stages B and C to be undertaken sequentially. A more exact approach would require estimating the number of flights from each direction, applying the collision probability for that direction, and summing the probability over all directions. Annex 1 provides a fuller explanation of this issue and the justification for adopting the simplified approach. It should be recognised that this simplification leads to some underestimation of collision risk, which may be as much as 10% for large birds.

13. Taking account of bird flight height distribution. Seabirds mostly fly at relatively low heights over the sea surface. The height distribution varies from species to species and may depend on the site and its ecology and related bird behaviour. The basic model considers the risk only to birds flying at risk height (above the minimum and below the maximum height of the rotors) and of these, only those which pass through the rotors. However within these limits it assumes a uniform distribution of bird flights. There are three consequences of a skewed distribution of flights with height:

- the proportion of birds flying at risk height decreases as the height of the rotor is increased;
- more birds miss the rotor, where flights lie close to the bottom of the circle presented by the rotor; and
- the collision risk, for birds passing through the lower parts of a rotor, is less than the average collision risk for the whole rotor.

This guidance now includes, in addition to the basic model, an extended model (March 2012) which enables flight height distributions to be incorporated in the calculation, for use in circumstances where flight height data is available and adequately robust.

14. Best estimate not worst-case. This guidance does not recommend use of 'worst case' assumptions at every stage. These can lead to an overly pessimistic result, and one in which the source of the difficulty is often concealed. Rather, it is recommended that 'best estimates' are deployed, and with them an analysis of the uncertainty or variability surrounding each estimate and the range within which the collision risk can be assessed with confidence. In stating such a range, the aspiration should be to pitch that at a 95% confidence level, that is, so that there is 95% likelihood that the collision risk falls within the specified range. However, given the uncertainties and variability in source data, and the limited firm information on bird avoidance behaviour, it seems likely that for many aspects the range of uncertainty may have to be the product of expert judgement, rather than derived from statistical analysis.

15. Spatial exploration of risk. While this guidance, and the attached spreadsheet, is written around quantifying the collision risk from an entire windfarm, it can equally well be applied at the level of a subgroup of turbines or even an individual turbine. If the data on flight activity is sufficiently robust to allow such discrimination, this facilitates the examination of risk on a spatial basis. Collision risk is directly proportional to flight activity which is dependent on bird density at rotor risk height. Siting windfarms, or groups of turbines, in areas of lower bird density is likely to yield a proportionately lower collision risk.

16. Use for onshore windfarms. The approach described here could equally well be applied to onshore as to offshore windfarms, using vantage point or other land-based survey or radar to generate the required data on bird density (see paragraph 19).

STAGE A - FLIGHT ACTIVITY

17. The aim of this stage is to estimate the number of flights which, in the absence of birds being displaced or taking other avoiding action, or being attracted to the windfarm, would potentially be at risk from the windfarm turbines. This requires field data to determine levels of flight activity within the proposed windfarm.

How flight activity is expressed

18. Flight activity may be expressed in a variety of ways.

- Bird density is a measure of how many birds (of any given species) are in flight at one time. It may be expressed in terms of birds per m^3 (cubic metre) of air space (the 'true density' D_v). However, more commonly, reflecting the use of boat-based or aerial survey techniques, it may be expressed on an area basis as the total number of birds in flight at any height at a given point of time, per m^2 (square metre) or per km^2 (square kilometre), as viewed from the air, D_A.

- Bird occupancy applies to a given volume of airspace, and is simply the number of birds on average occupying that volume. Thus, in a volume of air for which the bird density is uniform, bird occupancy (birds) = true density (birds/ m^3) x volume (m^3). The concept of 'bird occupancy' is not used in this guidance, but is referred to here to facilitate comparison with the Band (2000) model[1].

- Bird flux is the number of birds crossing an imaginary surface within the airspace, expressed as birds/sec or birds/sec per m^2 of that surface. It is commonly measured in the field in terms of a Mean Traffic Rate which is the number of birds flying per hour across an imaginary horizontal line of length 1km. If all birds crossing that imaginary line, as viewed from above or below, are recorded at any flight height up to height h metres, then the Mean Traffic Rate is the total number of birds N birds/km/hour crossing that line. MTR must be divided by 3600 (seconds in an hour) and 1000 (metres in a km) to express bird flux in birds/sec per metre of baseline, and divided further by the height h to get the bird flux in birds/ sec /m^2.

  Bird flux is directly related to bird density, but depends on the speed of the birds (if they were stationary, there would be no flux). If the total bird flux (flights at any height, in either direction) across the baseline is F_L birds/sec per metre of baseline, then the bird density D_A per m^2 is

$$D_A = (\pi/2)\, F_L \,/\, v$$

  where v is the speed of the birds in m/sec: see Annex 2 for the derivation of this formula and fuller information on converting between flux and bird density.. Flux is directional – for a given density of birds moving in random horizontal directions, a vertical 'window' will intercept more birds flying perpendicular to the area than birds flying at an oblique angle, to which the window will appear narrower. The (π/2) factor takes account of this angle-dependence.

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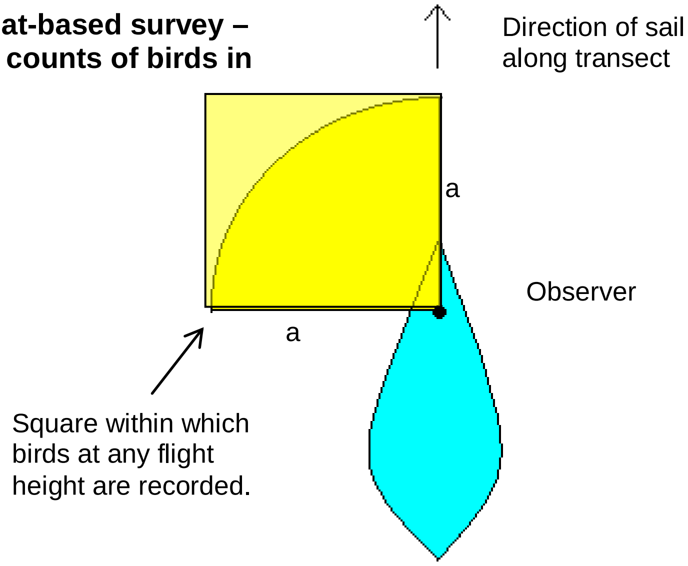
[1] In the Band (2000) model, bird occupancy is expressed in 'bird-seconds per year' as a convenient way of expressing low levels of bird occupancy. An occupancy of 31.6 x 10^6 bird-seconds per year means that on average, within the specified volume, there is one bird throughout the year, 31.6 x 10^6 being the number of seconds in a year.

Fig 2: Boat-based survey – snapshot counts of birds in flight



19. How flight activity is expressed in output from surveys often reflects the type of survey method deployed:

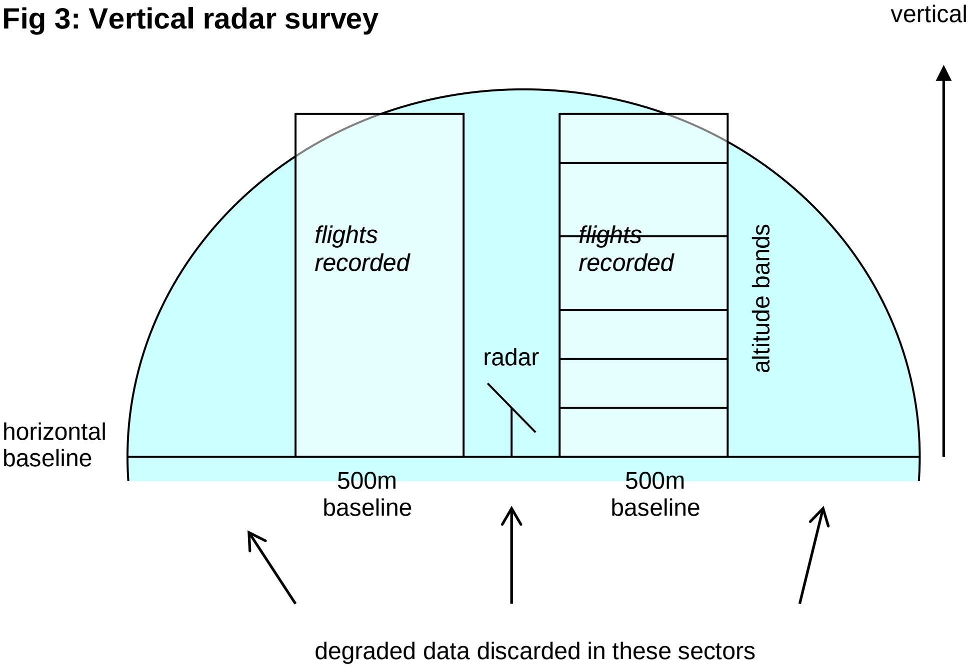
- Boat-based surveys, where the boat follows a transect through the site, and records are taken at intervals of birds in flight, provide a 'snapshot' of the number of birds in flight within the range of observation (see diagram) which is usually 300m. If a snapshot has N birds (at any flight height) within an observation square of side a from the boat then the bird density per unit area of sea is N / a^2 (see Fig 2). Some surveyors record flights on both sides of the boat, thus covering two such squares, such that the density is $N / (2 a^2)$. Other surveyors record flights over a quadrant area of sea of radius a, in which case the density is $N / (\pi a^2/4)$.

  Boat-based survey can also provide information on flight heights, such as to enable an estimate of the proportion of flights which fall within the rotor risk height (from the lowest point to the highest point of a rotor, a height equal to twice the rotor radius. Cowrie guidance on boat-based survey methods is provided in Camphuysen et al (2004)[iv].

- Aerial survey methods, whether photographic or not, provide a direct sampling measure of the density of birds in flight per unit area of sea, provided that birds in flight can be discriminated from those on the sea surface, and that species can be identified at an adequate level.

- Radar survey methods which observe bird transits across a radar platform provide a measure of bird flux, ie the number of birds crossing an imaginary vertical surface, defined by a horizontal line between two points and the vertical surface extending from the sea upwards through that line. In practice, vertical radar typically allows most effective scanning of birds crossing two vertical windows of base around 500m, which may be divided into altitude bands (see diagram). Observations both at close range and at large distances, where detection rates degrade, are discarded. Adding the birds crossing each of these windows gives the bird flux across an imaginary baseline of 1km length (eg see report for Bureau Waardenburg, Krijgsveld et al. (2008)[v]).

Fig 3: Vertical radar survey



- Vantage point survey methods which record all bird flights in a defined volume of the windfarm airspace from a key vantage point lead to a measure of bird occupancy in that volume. Such survey is not normally practicable at sea unless a semi-permanent observation platform is installed, or if the relevant sea area can be observed in its entirety from shore. Bird occupancy is readily converted to bird density (per m^2) by dividing by the area scanned from the vantage point (see paragraph 18).

Density of birds in flight and at risk

20. For the purpose of estimating collision risk, this guidance starts from measurements, derived from survey information, of bird density, and of the proportion of birds flying at risk height (ie between the lowest and highest points of the rotors) or, if more detailed observations are available, of the distribution of bird density with height. The calculations set out later use that information to calculate the flux of birds through each rotor (using the simplifying assumption that flight direction is perpendicular to the rotors).

21. The most useful way to present information on bird density is on an area basis, ie the total number of birds in flight at any height at a given point in time, per square kilometre (km^2). Stating the bird density per unit area provides a better basis for comparison of risk assessments, and for cumulative risk assessment, than would be the case if only bird flight density at rotor height were stated. It also provides a level of data which can be re-interpreted

in the future, for example if a new generation of larger turbines came available. Such overall bird density information does not embody assumptions or uncertainties relating to flight height distribution. Where survey information is based directly on measurements of flux (eg from use of radar survey methods) then these should be translated, using the formula in paragraph 18, to estimates of bird density.

22. **An Environmental Statement should clearly state the bird density used in collision calculations, expressed in terms of birds per km^2 across the site, counting birds flying at all heights. It should also state the proportion of birds estimated to be flying within the risk height band – ie between the lowest and highest points of the rotors. Where a bird flight height distribution is used in the calculation, the Environmental Statement should state the distribution used and its source.** Where survey information leads to a range of perspectives on bird density (eg including or excluding data for buffer areas), the Environmental Statement should make clear which survey data has been used, and why. Paragraphs 25-31 describe how information on flight heights should be presented.

23. The number of birds of any one species passing through a rotor is, among other factors, proportional to the density of flying birds in the vicinity of the rotor, and hence so too is the collision risk to which they are exposed. Therefore, where one of the aims of a collision risk assessment is to choose a windfarm location and design so as to minimise bird collision risks, the starting point should be to select those areas with the lowest density of the bird species vulnerable to collision. For large sites, or for consideration of collision risks at a strategic level, it may be possible to discriminate between different zones of the site or areas with different bird densities. Such information will be helpful in identifying preferred zones for development. However care should be taken to ensure that any differences are statistically significant. For most development sites, the statistical variation in the data derived from survey is likely to mask any within-site variations in bird density.

24. While the approach to collision risk in this guidance does not require definition of a windfarm boundary, and the area of the windfarm area does not feature in the calculations, it is important to be clear as to the boundary within which an estimate of bird density applies. Survey recommendations usually recommend survey wider than the windfarm itself so as to ensure that any bird density estimates for the wind farm site are adequately representative of the marine area as a whole.

Flight heights

25. There is only a risk of collision with turbine blades at flight heights between the lowest and highest points of the rotors, a total height 2R, twice the length of a blade. Therefore an important parameter to estimate is the proportion Q_{2R} of birds flying within that risk height band. The data on bird density should be accompanied by an estimate of the proportion of birds flying within the risk height band for the proposed windfarm.

26. If data is available on the distribution of bird flight density with height, that enables the calculation to be refined to allow for the fact that most flights within this risk height are at a height where the chance of passing through the rotor is low, and the actual risk of collision if they do is also lower than for an average rotor transit. Most seabirds spend a high proportion of their flight time quite close to the sea surface, and therefore any collision risk tends to be concentrated in the lower parts of the rotor[vi].

27. Accurate data on flight heights is difficult to capture. In boat-based surveys, it relies on observers being able to estimate flight heights, and the accuracy of such estimates decreases with height. While aerial survey in the past has not normally yielded flight height information, high definition digital photography systems are now available which provide increasingly accurate information on flight height.

28. For some species, survey information at a site may be insufficient to provide a reasonably precise figure for the proportion of birds flying at risk height. Where this is the case, it may be

better to use a generic view of flight height behaviour, obtained by combining flight height information gathered from surveys at different sites – for which a detailed report has been compiled by Cook et al (BTO) for SOSS[iii]. In combining results from different surveys, care is needed to place greatest weight on those with the most robust data, which may imply discarding data with poor levels of precision. The generic information should be reviewed, assessing whether it provides more precise information than the site-based data, and whether the site-based data, if limited, is nonetheless compatible with the generic information. If so, then the generic information should be used. Care must however be taken not to mask any feature of flight behaviour at the site in question which could reflect a genuine difference of behaviour due to environmental variables or the specific use of the site made by the birds. For some species typical flight heights are dependent on the season, and in such a case it will be best to use seasonally dependent typical flight heights in assessing collision risk for each month, rather than average flight heights across the year.

29. Often, at the time of undertaking field survey, the actual turbines to be used have not been selected, and turbine models may vary in their risk height. Estimates of the proportion of birds flying at risk height should reflect the range of turbine heights which potentially may be used. Survey methods should be designed to ensure that data are available to inform all potential turbine options. Guidance on the extent to which the details of a scheme may be kept flexible during the environmental assessment process is published by the Infrastructure Planning Commission (2011)[vii].

30. The central estimate of the proportion of birds flying at risk height should be based on a straightforward analysis of flight height survey data, without any 'margin of uncertainty' added to the risk height range. In addition, alternative +/- estimates should also be presented, reflecting the possibility of a higher or lower proportion of birds flying at risk height. Confidence intervals on flight height data should be used where these are available from the survey information. Otherwise, a realistic view should be taken of the potential for mis-estimation and error in flight height observations by field observers. Confidence intervals should be aimed at around 95% confidence that the true result lies within that range. In some circumstances, this may be no more than an expert view based on an understanding of the limitations of the survey techniques.

31. **For the purpose of estimating collision risk, the ES should state**

- **the proportion of birds estimated to be flying within the risk height band – ie between the lowest point of the rotors and the highest point of the rotors – based on survey information at the site;**
- **any flight height distribution derived from combining wider survey data for the species in question, and the proportion of birds thereby assumed to fly at a height exposed to collision risk;**
- **which of the above is used in the collision risk estimate, and why.**

Daylight hours and nocturnal activity

32. For obvious reasons, most bird survey is undertaken by day, and it is generally assumed that such sampled levels of flight activity persist throughout daylight hours. Daylight hours depend both on time of year and on latitude. Forsythe et al (1995)[viii] provide a ready reckoner for daylight hours which is reproduced in Sheet 7 (Daylight and night hours) of the attached spreadsheet. Input of the latitude of the site in Sheet 1 (Input data) triggers the calculations in Sheet 7 (Daylight and night hours) which in turn populates Sheet 2 (Overall collision risk) with the appropriate number of daylight and night hours in each month.

33. There is considerable uncertainty about levels of bird flight activity by night. Garthe and Hüppop (2004) [ix]offer an expert view on levels of nocturnal flight activity for a range of marine bird species, expressed in terms of a 1-5 ranking of the likely level of nocturnal activity in comparison with observed levels of daytime activity. A rating of 1 represents hardly any flight activity at night, and 5 much flight activity at night. King et al (2009) (Appendix 7)[x] provides a

more comprehensive table with rankings on a similar expert basis for a wider range of seabirds.

34. Figures used in the collision model should take both day and night flights into account. Where there is no night-time survey data available, or other records of nocturnal activity, for the species in question, (or for other sites if not at this site), it should be assumed that the Garthe and Hüppop/ King et al 1-5 rankings apply. These rankings should then be translated to levels of activity f_{night} which are respectively 0%, 25%, 50%, 75% and 100% of daytime activity. These percentages are a simple way of quantifying the rankings for use in collision modelling, and they may to some extent be precautionary. For some species, there are no such expert rankings available. Levels of activity may vary from season to season, and activity at sea may in any case differ from the levels of activity in breeding colonies for which the rankings have been formulated. Some species are particularly active during dawn and dusk or extended twilight periods, or in locations where there is ambient windfarm lighting. When expressing the output of the collision risk assessment, the uncertainty surrounding flight activity should reflect the degree of confidence (or lack of confidence) in the flight activity information.

35. **Flight activity estimates should allow both for daytime and night-time activity. Daytime activity should be based on field survey. Night-time flight activity should be based if possible on night-time survey; if not on expert assessment of likely levels of nocturnal activity.**

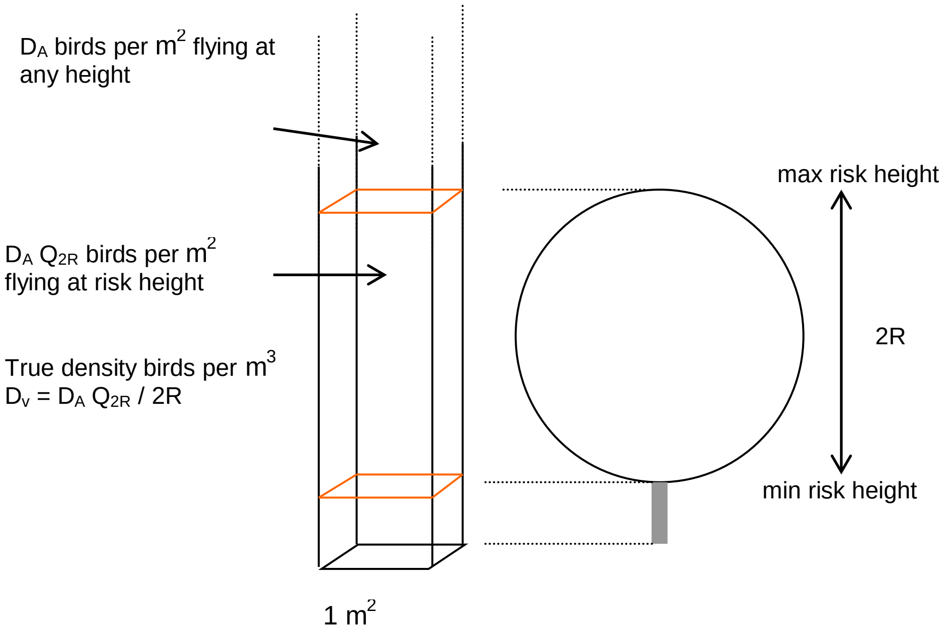
STAGE B - ESTIMATING NUMBER OF BIRD FLIGHTS THROUGH ROTORS

36. In the basic model, this stage is straightforward, but one which often causes some difficulty. It can be addressed in the following steps:

    (i) Start with the observed bird density on an area basis, expressed per unit area, D_A. Convert if needed to units of birds/ m^2. If the survey data is expressed in birds/km^2 then divide by 10^6.

    (ii) Multiply by the proportion Q_{2R} of birds flying at risk height to get only those birds at risk in a column of air of unit area base and 2R high (ie from bottom to top of the rotor) – see Figure 4.

Fig 4: Birds flying at risk height

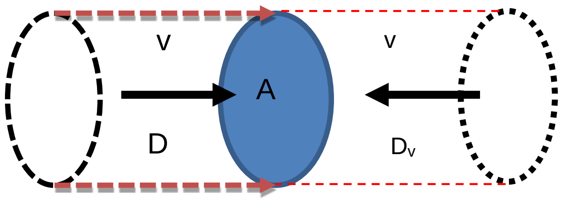


(iii) Calculate the true bird density per unit volume $D_V = (D_A Q_{2R})/2R$, expressed in birds per m^3 (birds per cubic metre).

(iv) Now calculate the flux of birds through a rotor within an airspace of true bird density D_V, noting that we are making the simplifying assumptions that all birds are flying perpendicular to the rotor, and that they are all flying with a single flight speed v. Also, the rotor may be assumed to face the wind at all times. It is also, for simplicity, assumed that there are equal numbers of birds flying upwind as are flying downwind, which is important as the collision risk when flying upwind is greater than for downwind flight[2].

Consider the area of the rotor $A = \pi R^2$. If the birds fly at speed v m/sec, then within one second, all birds within a distance v on one side and flying towards the rotor will pass through the area A. At any one time, half the birds will be travelling upwind and half downwind. Thus, referring to Figure 5, at any time there will be $\frac{1}{2} D_v A v$ birds flying downwind towards the rotor and, on the other side of the rotor, $\frac{1}{2} D_v A v$ birds flying upwind towards the rotor.

Fig 5: Bird flux due to bird density



Thus bird flux $F = \frac{1}{2} D_v (\pi R^2)\, v$ upwind plus $\frac{1}{2} D_v (\pi R^2)\, v$ downwind

$= v\, D_v (\pi R^2)$ in total $= v\, (D_A/2R)\ (\pi R^2)\ Q_{2R}$ .. (1)

This is expressed in birds/second passing through the rotor.

(v) Now multiply by the appropriate number of seconds during which the birds are potentially active – usually the daylight hours in the month t_{day} plus an allowance if appropriate for nocturnal activity $f\, t_{night}$, multiplied by 3600 to convert to seconds.

(vi) Multiply by the number T of turbines. Each turbine in a windfarm, if it is surrounded by an airspace with the same bird density, and if all turbines are of the same size, will experience the same number of bird transits and will therefore contribute the same collision risk to the overall total. If the windfarm includes turbines of different sizes, or zones of differing bird densities, then the calculation should be broken down into subgroups of wind turbines where turbine size and bird density is constant within each subgroup.

37. The result is an estimate of the total number of bird transits through rotors of the wind farm in the specified period. In the spreadsheet provided, the entry for ‘bird transits’ calculates the total number of bird transits for each month, taking account of the proportions of flights deemed to be upwind and downwind. It calculates the result on the basis of the values entered for D_A, Q_{2R}, R, v, T , time for which birds are active, ie the calculation includes all of stages (i) to (vi) above.

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[2] If the collision model is applied specifically to migration flights, or to flights in adverse weather conditions, it may be that a majority of flights will be downwind, in which case the proportions of bird flux should be altered as appropriate from the ½ upwind and ½ downwind assumption made here.

Total number of bird transits =

$$\boxed{v\ (D_A / 2R)\ \ (T\ \pi R^2)\ \ (t_{day} +\ f_{night}\ \ t_{night})} \quad x \quad \boxed{Q_{2R}} \qquad ..\ (2)$$

flux factor x proportion at risk height

38. **A key output within the collision risk assessment should be a clear statement of the potential number of bird transits per month, and per year, through the windfarm turbines, assuming birds take no avoiding action. The collision risk is directly proportional to the potential number of bird transits.**

Box 1: Converting from bird density to rotor transits (basic model)

<u>Worked example:</u>

v	Bird flight speed	10.5	m/sec
D_A	Bird density per unit area 50% upwind, 50% downwind =	0.1128 $0.1128\text{x}10^{-6}$	birds/km^2 birds/ m^2
R	Rotor radius	63	m (metres)
T	Number of turbines	150	
$T\pi R^2$	Frontal area of all rotors	1870345	m^2
t	Hours active in June $(t_{day} + f_{night}\ t_{night})$ =	480 $1.728\text{x}10^6$	hours seconds
F	Flux factor $v\ (D_A / 2R)\ \ (T\ \pi R^2)\ \ t$	30380	
Q_{2R}	Proportion flying at risk height	28.1%	
	Total bird transits through turbines in June	8537	50% upwind, 50% downwind

STAGE C – PROBABILITY OF COLLISION FOR A SINGLE ROTOR TRANSIT

39. This stage begins with the model described in the earlier Band (2000) and Band et al (2007) papers which uses information on the size and speed of the turbines, and physical details on the size and speed of the bird, to compute the risk of collision for a bird flying through a rotating rotor. Annex 3 is an extract from Band (2000) outlining the core of the model and its derivation.

40. A bird is simplified in shape to a flying cross with length, wingspan, and speed, and always flying perpendicularly towards the rotor. A bird may be 'gliding' ie with the arms of the cross fixed, or 'flapping' ie with the arms of the cross flapping so as to occupy a space similar to that of a spinning top, with the length of the bird being the axis of spin. 'Gliding' flight has a marginally lower collision risk than 'flapping' flight – notably for passage at points level with the rotor hub, where the wings lie parallel with potentially colliding blades. However the difference is rarely sufficient to warrant detailed consideration of different bird behaviours; the flight type used should be that which best typifies most flights for the species in question.

41. Rotor blades are assumed to be laminar (ie with zero blade thickness) but they have length, a chord width which varies along the length of the blade tapering towards the tip, and a pitch angle (the angle between the blade and the rotor plane) which also varies along the length of the blade. Due to commercial sensitivities by blade manufacturers, some of this detailed information may not be readily available for each make/model of blade and hence generic information may have to be used.

42. With these simplifications, the model calculates the risk of actual collision between the bird and the rotor blades. Such a model has a number of important limitations:

- Stationary infrastructure - it is assumed that birds can avoid stationary infrastructure, so no account is taken of the turbine towers, nor the blades when stationary; While this may be a valid assumption in clear daylight conditions it may not be wholly true at night or in conditions of poor visibility. Onshore, for example, there are records of gamebird species colliding with turbine towers. In this respect, the model may underestimate collision risk.

- Turbulence - no account is taken of the effects on a bird's flight of turbulence in the wake of a blade. Observers have seen birds 'knocked out of the sky' by turbulence, and there is potential for this to increase mortality through disorientation or impact with the sea surface. The model only takes account of the potential for physical contact between the bird and the turbine blades. In this respect, the model may underestimate collision risk.

- Slipstream - however, it is also the case that the model does not take account of any 'slipstream' effects whereby the air rushing over the surface of a blade may carry a bird clear of the blade when otherwise it was on a collision course. In this respect, the model may over-estimate collision risk.

- Bird shape - real birds are larger than represented by a flying cross, though a cross should represent the main extremities. In this respect, the model may underestimate collision risk.

- Flight height distribution - the basic collision model evaluates the probability of a bird colliding if it passes at random at any point through the rotor disk on a flight path perpendicular to the rotor plane. In practice, the points of passage of seabirds through the rotor are not distributed uniformly across the rotor. Survey data for seabirds has made clear that typical flight heights for many species are relatively low, such that much of the bird flux through a rotor, and the associated collision risk, will relate to the lower parts of the rotor plane. Since it averages risk over the entire rotor including higher-risk areas close to the hub, the basic model will overestimate the collision risk for seabirds whose flight passages are more concentrated towards the lower part of the rotor plane. Where data are available on the distribution of bird density with height, an extended calculation may be undertaken which takes account of this variation with height. This extended model is described following stage D, in paragraphs 61-75.

- <u>Perpendicular approach assumption</u> – as outlined in Annex 1, the model used assumes that the collision probability for oblique angles of approach is the same as for perpendicular approach. In fact, some increase in collision risk should be expected, which, taking account of both upwind and downwind flight, may be of order 10% for large birds. In this respect, the model may underestimate collision risk.

43. The model uses a probability p of collision for a bird flying through a rotor, at a point in the rotor plane defined by coordinates r, φ :

$$p(r, \varphi) = (b\Omega/2\pi v) \left[\quad | \pm c \sin\gamma + \alpha\, c \cos\gamma | \quad + \quad \max (L,\ W\alpha F) \quad \right] \qquad \ldots \qquad (3)$$

where

r = radius of point of passage of bird
φ = angle within rotor plane (relative to vertical) of point of passage of bird
ie $\varphi=0$ is top, $\varphi=\pi$ is bottom, etc

b = number of blades in rotor
Ω = angular velocity of rotor (radians/sec)
c = chord width of blade
γ = pitch angle of blade
R = outer rotor radius

L = length of bird
W = wingspan of bird
β = aspect ratio of bird ie L / W
v = velocity of bird through rotor
α = $v/r\Omega$
F = 1 for a bird with flapping wings (no dependence on φ); $F = \cos \varphi$ for a gliding bird

This probability is then averaged, by integrating over the entire rotor area, to yield the average collision risk for a bird making a single flight through the rotor at any point through the rotor.

44. By way of explanation, there are three terms in equation (3) within the square brackets.

- The first [$c \sin\gamma$] relates to the time taken for the bird to clear the depth of the blade, which increases with pitch γ.
- The second [$\alpha\, c \cos\gamma$] relates to the probability of the bird striking the front face of the blades. Note that the appearance of α cancels any dependence of this term on rotor angular velocity Ω and bird speed v.
- The final term [the greater of L, or $W\alpha F$] relates to the time taken for the full length and wingspan of the bird to clear the sweep of the rotors, for which the geometry depends on the relative speed of bird and blade. Where the bird's aspect ratio $\beta > \alpha$, the bird length is the limiting parameter. However if $\beta > \alpha$ the wingspan is the limiting parameter. For a flapping bird, $p(r)$ not dependent on φ and F is set to 1. For a gliding bird, the effective wingspan depends on φ, reducing to zero at $\varphi = \pi/2$ or $3\pi/2$ where the wings lie parallel to the rotor blade; thus $F = \cos \varphi$.

45. Because of the geometry of the blades in relation to the flight direction, the collision risk for upwind flight is higher than for downwind, even if the bird's flight speed v relative to the ground is taken to be the same. This is expressed in the alternate sign in the first term, which is + for upwind flight, - for downwind. In practice, birds will fly more slowly in upwind flight

than downwind, further widening the difference in risk between upwind and downwind flight (see paragraph 51). If both upwind and downwind flights are equally likely, it is appropriate to take an average of upwind and downwind collision probabilities.

46. The basic model assumes that bird flights may occur with equal probability at any point through the rotor disc. Having ascertained the collision risk p(r,φ) at different points r,φ of the rotor, the basic model then calculates an average of p(r,φ) over the entire area of the rotor disc, firstly summing over φ, then summing (integrating) over successive concentric rings, taking account of the area of each ring which increases with radius (= ring circumference $2\pi r$ times thickness of ring dr). Finally this sum is divided by the overall disk area to get the average collision probability:

$$p_{average} = \int_0^R p(r)\,(2\pi r)\,dr \,/ \int_0^R (2\pi r)\,dr \quad = \int_0^R p(r)\,(2\pi r)\,dr \,/\, \pi R^2 \;=\; 2\int_0^1 p(r)\,(r/R)\,d(r/R) \quad \ldots(4)$$

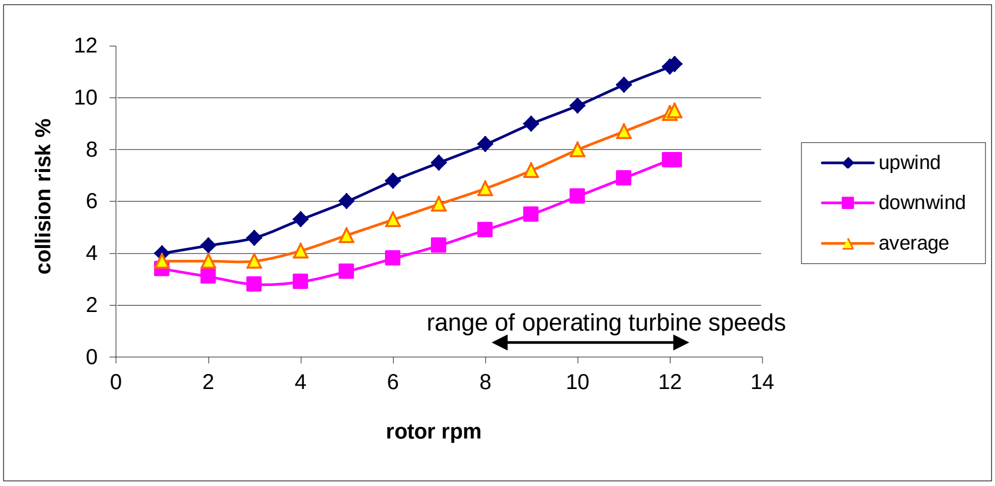
47. Sheet 3 (Single transit collision risk) of the spreadsheet accompanying this guidance provides a collision risk calculator for a single passage through the rotor, evaluating p(r) for a series of twenty radii from r/R=0.05 to r/R=1, and undertaking the above integration numerically to evaluate $p_{average}$, the average collision risk for a passage at any point across the rotor. This is essentially the same as the spreadsheet referred to in Band (2000)[i] but with refinements to the numerical integration[xi].

Wind turbine speed

48. Wind turbines currently available are designed to operate at a range of speeds. Typically they do not operate below a cut-in speed (usually between 3 and 4 m/sec), then increase in speed with wind speed up to an operating wind speed (which may be around 12 m/sec). Thereafter, they maintain a constant operating speed by altering the pitch of the blades until, in extreme conditions, the turbine is shut down for safety.

49. **Collision risk should be evaluated using the turbine rotational speed for an operating turbine. Where turbines operate with a range of rotational speeds, the calculation should be done using a mean operational turbine speed. The mean used should be a mean over time, using an analysis of wind data to enable the likely frequency distribution of turbine speeds to be determined.** Allowance is made elsewhere in the calculation (at Stage D) for the proportion of time that a turbine is non-operational, either because of low wind speeds or for maintenance. The mean turbine speed should thus be a mean over operational time only, not including times when the turbine is idling or stationary. Within the typical range of operating turbine speeds, collision risk varies almost linearly with turbine speed, so that use of a mean turbine speed is adequate in order to yield a mean collision risk – see Fig 6 for a turbine with a maximum operating speed of 12.1rpm. If a frequency distribution of turbine speeds is not available, then collision risk may be evaluated using the maximum operating turbine speed, but acknowledging that this will result in a collision risk which is an upper bound rather than a mean.

Fig 6: No-avoidance collision risk as a function of turbine speed for a 5MW turbine and bird (gannet)



Accuracy of model

50. Having regard for the various simplifications in the model, and the potential sources of under- and over-estimation described above, it is judged that this stage of the model, calculation of no-avoidance collision risk for a single transit, should be regarded as indicative of collision probability within around ±20%. If the flight height distribution is strongly skewed towards the low edge of the rotor, the basic model is likely to overestimate collision risk by more than this margin, while there should be no such overestimation if the extended model is used. These uncertainties are in addition to any uncertainty due to variance in flight activity and other input data (Stage A), or due to uncertainties in avoidance rates (Stage E).

Possible refinements

51. The spreadsheets are set up so that the average collision risk from the ‘Single transit collision risk’ calculation is copied over to the ‘Overall Collision Risk’ sheet and used, as described in the next section, to calculate projected collision mortality. However two refinements may be made at this stage.

- The ‘Single transit collision risk’ sheet assumes that the bird speed for both upwind and downwind flight is the same, derived from standard references. In fact, it is likely that ground speed downwind will be greater, and ground speed upwind, less than this value. If good data are available, either from field survey or from the literature, to support the use of different up/downwind ground speeds, then this spreadsheet may be run once for each, taking the average of the respective ‘upwind’ and ‘downwind’ outputs to copy over to the ‘Overall Collision Risk’ sheet.
- In taking an average for upwind and downwind flights, the ‘Single transit collision risk’ sheet uses the relative proportion of upwind and downwind flights to weight the respective collision probabilities. By default the proportion should be set to 50% upwind (and thus 50% downwind). However there are some circumstances, eg migration flights, in which downwind flights may dominate, though flight directions are often far from regular. If field data support the use of differing proportions of upwind and downwind flight, then the proportions may be changed by altering the ‘Proportion of flights upwind’ field in the Input Data sheet.

STAGE D – MULTIPLYING TO YIELD EXPECTED COLLISIONS PER YEAR

Basic model – assuming uniform flight density

52. If the basic model is used, multiplying by the number of bird flights through the rotor is nearly trivial. Stage A has estimated the level of flight activity at potential risk; Stage B has estimated the likely number of flights through rotors across the windfarm; Stage C has calculated the risk of collision for a single bird transit through a rotor. In the present stage, Stage D, these are multiplied together to yield an estimate of total potential collision risk, including a factor to allow for the proportion of time that the wind turbines are operational (before considering avoidance behaviour, which is stage E).

Expected collisions =

$$\underset{\text{No of transits}}{\boxed{\text{Flux factor} \times Q_{2R}}} \times \underset{\text{Single transit collision risk}}{\boxed{\text{Average probability of collision}}} \times \underset{\text{Proportion of time operating}}{\boxed{Q_{op}}} \qquad \ldots(5)$$

Units

53. Whichever model is used, there is a need for care with units. In the spreadsheet, flight activity becomes expressed as rotor transits per month and hence the collision risk is in predicted collisions per month.

Non-operational time

54. Turbines do not operate all of the time. Typically a turbine may be at rest or idling for a considerable proportion of time, eg 20%, because the wind is too weak to generate power, or (exceptionally) because the turbines have been closed down to avoid damage in high wind. There is also a requirement for some downtime for maintenance. This non-operational time is accounted for in equation (5) by the factor Q_{op} representing the proportion of time the turbine is operational. If data is available, this factor may be stated on a monthly basis to reflect the different proportions of non-operational time at different times of year – for example reflecting differing wind conditions across the year and increased access for maintenance during the summer.

Large turbine arrays

55. The model assumes that risks are additive, ie that a windfarm with 200 turbines will have 200 times the risk of a single turbine. Where a bird passes successively through two or more turbines, it is exposed to the same risk for each rotor transit. While it is possible that a bird encountering its first turbine may deviate so as to pursue a safer course through (or above or around) the windfarm, this is avoidance behaviour and therefore properly taken into account at Stage E rather than here. Stages A - D simply work out the consequences of birds taking no avoiding action[3]. Thus, if two turbines 'overlap' in the sense that the bird passes through both turbines in a single passage, no allowance is made for that overlap, the collision risk is the sum of the risk from each rotor passage.

56. More strictly, for large windfarms where the overall probability of a bird colliding is relatively high, it may be appropriate to take account of the fact that a declining proportion of the birds will survive passage through early rows of turbines and will thus be exposed to collision risk in later rows. This adjustment is only likely to be of any significance for large arrays of turbines.

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[3] This position was somewhat confused by a reference in Band et al (2007) to making a 50% allowance for overlapping turbines. It is now preferred that any amendment to collision risk resulting from avoidance behaviour should be built into the avoidance rate applied at the end of the calculation.

57. Annex 4 sets out how such a correction may be made for a windfarm with approximately n rows of turbines. Very often the layout of a windfarm is not known at the time of collision risk assessment, so an exact value for n is not known; and in any case the collision risk has to account for birds entering the windfarm from all directions. Sometimes the layout of the windfarm is irregular, lacking in clearly defined rows; but the principle remains that a declining number of birds will be exposed to collision risk if a proportion have already been killed by collision with earlier rotors as they pass through the windfarm. A reasonable and simple approximation is to use $n = \sqrt{T}$ ie the square root of the total number of turbines.

58. If the probability of collision for a single bird passage through the windfarm is C, based on the purely additive approach elsewhere in this guidance, then it may be adjusted to allow for depletion of bird density in later rows of the windfarm by multiplying by a ‘Large array correction factor’

$$C_{LA} / C = 1 - ((n-1)/ 2n)\, C + ((n-1)(n-2) / (6 n^2)\, C^2 \ldots \qquad \ldots(6)$$

plus further negligible terms of powers of C

59. If realistic avoidance rates have been taken into account in the collision model, such ‘large array corrections’ are typically small and can be ignored; typically it is only worth making corrections for values of $C > 0.1$.

60. See Annex 4 for a derivation of this ‘large array factor’, and a worked example. Sheet 8 – ‘Large array correction’ in the spreadsheet provides a calculator for this factor. The spreadsheet applies this correction factor to the output of Sheet 2 – ‘Overall collision risk’ by multiplying each projected collision rate, for each of the various avoidance rates, by the correction factor. In most circumstances it will be evident that the difference is minimal.

EXTENDED APPROACH TAKING ACCOUNT OF FLIGHT HEIGHTS

Effects of taking flight height into account

61. Seabirds tend to fly at relatively low altitude over the sea surface. If the flight height distribution is skewed towards low heights in this way, there are three ways in which taking account of flight height is important to the calculation of collision risk:

    (i) The proportion Q_{2R} of birds flying at risk height will decrease with the height of the rotor above the sea surface. This is accounted for in the basic model if the parameter Q_{2R} is adjusted, but the way in which Q_{2R} changes with height can only be known if a flight height distribution for the species in question is available.

    (ii) If most of the birds flying at risk height (ie above the minimum level of the rotor) do so at a level not far above the bottom edge of the rotor, the probability of passing through the rotor disc is relatively small, simply because the rotor circle occupies less width at that level than, for example, at the midpoint of its diameter. Therefore the expected number of rotor transits is reduced. For some species the reduction may be 50% or more, reducing the collision risk in proportion.

    (iii) Finally, if the birds flying through the rotor do so close to the extremity of the blades, the single-transit probability of collision there is rather less than for passages closer to the hub. This is a smaller effect, but may typically account for a reduction of around 10%.

    For these reasons, if the data is adequate to support an extended analysis taking account of flight heights, it is well worth doing so.

When to use generic flight height distribution data

62. Normally, the bird survey data available for a particular site is insufficient to provide a full flight height distribution. However it may provide some insight into typical flight heights at the site, and it should provide information on the proportion of birds flying at risk height ie above minimum rotor height. The Crown Estate SOSS group has commissioned a compilation of flight height data from windfarm sites across the UK (Cook et al 2012[iii]). That paper contains generic flight height distributions for a number of seabird species.

63. Caution is needed in deploying this generic data. It is entirely possible that the ecological circumstances of a particular site differ from those in the sites used to generate the generic data, and hence bird behaviours and flight heights may not be well represented by the generic data. Before using generic data, consideration should be given to whether

- is the site survey data compatible with the generic data? Does it indicate that the generic data reasonably represents the observations at this site?
- are there particular ecological circumstances which might be expected to lead to non-standard behaviour, eg proximity to breeding sites?

64. A collision risk assessment for a specific site should not be based solely on the use of generic data. Where generic data is used, it is recommended that the collision risk for three different options is stated:

    - Option(i) - using the basic model, ie assuming that a uniform distribution of flight heights between lowest and highest levels of the rotors; and using the proportion of birds at risk height as derived from site survey.
    - Option (ii) - again using the basic model, but using the proportion of birds at risk height as derived from the generic flight height information.
    - Option (iii) - using the extended model, using the generic flight height information.

    The spreadsheet supporting this guidance provides for the calculation of all three options. If site survey information is sufficient to generate a flight height distribution, this should be used as an Option (iv) as well.

    Supporting text should then discuss and justify which of the options is most likely to characterise the collision risks at this site.

The hard stuff (ie maths)

65. This section extends the basic model, and the calculations in Stages B-D, to enable the distribution of flight heights to be taken into account. The basic model calculates the number of transits through rotors, then multiplies these by the average collision probability for a single transit (see equation (5) in paragraph 52):

    No of collisions = number of transits x probability of collision

    The extended approach is underlain by this same equation. However, in this extended model, both bird flux and the probability of collision may vary over the area of the disc, such that their product must be summed over the whole area of the rotor disc.

66. The bird flux through an element of rotor area δA is

    $$v \, D_v \, \delta A$$

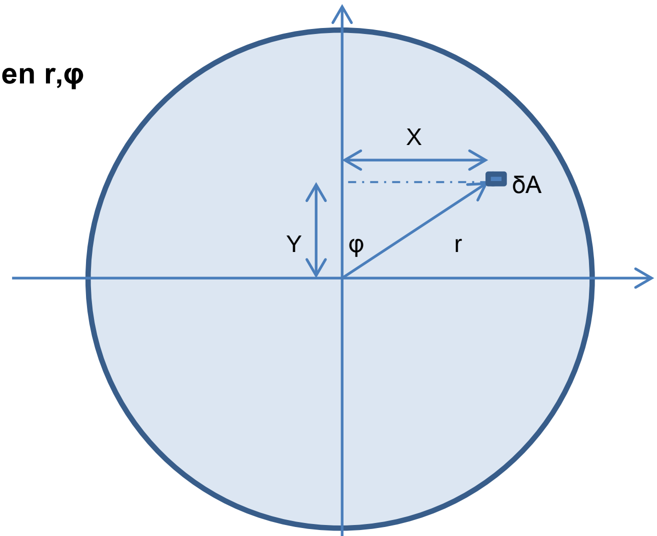
    as in equation (1) in paragraph 36, but applying it to a small area δA rather than the full rotor area A. As before there is a need to consider the proportions of flights upwind and downwind; we shall assume (for example) 50% upwind, 50% downwind.

    In this extended model, D_v may vary with height Y – this is the flight height distribution $D_v(Y)$ in birds/m^3 at height Y metres.

67. The collision risk for a single transit through this element δA is $p(X,Y)$, which is the same as $p(r,\varphi)$ except that X-Y coordinates, with origin at the rotor hub, are used to reference the point of transit instead of r-φ coordinates; the relationship between these two coordinate sets are

$$X = r \sin \varphi, \quad Y = r \cos \varphi \quad \text{or conversely} \quad r = \sqrt{(X^2+Y^2)}\,, \quad \varphi = \tan^{-1}(X/Y)$$

Fig 7: Relationship between r,φ and X,Y coordinates



The collision rate through this small element δA (take it as a small rectangle of width dX and height dY) is thus

$$v \; D_v(Y)\; p(X,Y)\; dX\, dY$$

The total collision rate for flights through the whole rotor disc is then obtained by integrating this over the whole area of the disc:

$$\text{Collision rate } = v \int_{\text{Min rotor height}}^{\text{Max rotor height}} D_v(Y) \int_{-\sqrt{(R^2-Y^2)}}^{+\sqrt{(R^2-Y^2)}} p(X,Y)\; dX\, dY \qquad \dots (7)$$

The limits $\pm\sqrt{(R^2-Y^2)}$ to the integration over X define the outer limits of the rotor circle, and the limits to the integration over Y are the minimum and maximum rotor heights respectively.

68. With this approach, it is not easy to think in terms of there being a defined bird flux, and an average probability of collision, which are then multiplied. The bird flight density varies with height Y, the breadth of the circle (and therefore the number of birds flying through the circle) varies with height Y, and the collision risk too depends on height Y, as it varies with both r and φ. Hence all these factors are expressed and multiplied within the integral, and the integration yields the collision rate.

69. As with the basic model, to translate this into collisions per month in the windfarm, this must be multiplied by the number of seconds the birds are active, and the number of turbines, and by the factor making allowance for non-operational time.

70. For computational purposes, it is best to translate the factors into dimensionless units, within which the rotor has a radius of 1, by using the parameters $x = X/R$, $y=Y/R$; and using a dimensionless flight height distribution $d(y) = R\, D_v(Y)/D_A$. Using these factors, and adding in the other factors (number of turbines, etc), equation (6) becomes

$$\text{Collisions} = v\, D_A\, R \int_{-1}^{1} \int_{-\sqrt{1-y^2}}^{+\sqrt{1-y^2}} d(y)\;\; p(x,y)\, dx\, dy \;\times\; \text{No of turbines } T \;\times\; \text{Time active } t \;\times\; \text{Proportion of time operational} \qquad \ldots (8)$$

$$= \underbrace{\boxed{v\,(D_A/2R)\, T\pi R^2\, t}}_{\text{Flux factor}} \;\times\; \underbrace{\boxed{(2/\pi) \int_{-1}^{1} \int_{-\sqrt{1-y^2}}^{+\sqrt{1-y^2}} d(y)\;\; p(x,y)\, dx\, dy}}_{\text{Collision integral}} \;\times\; \underbrace{\boxed{Q_{op}}}_{\text{Proportion of time operational}} \qquad \ldots (9)$$

It is written in this way for comparability with equation 5 above; the ‘flux factor’ and Q_{op} are the same as used in the basic model. The ‘Collision integral’ is a dimensionless quantity. If we apply this to the earlier scenario in which a proportion Q_{2R} of birds fly at risk height, and are distributed uniformly at all heights within that zone, we then have $d(y) = Q_{2R}/2$, a constant. The Collision integral is then Q_{2R} times the average of $p(x,y)$ over the rotor disc; in that case equation (9) reproduces equation (5).

71. The total bird flux passing through the rotors is similar to equation 9 but with $p(x,y)$ set to 1, ie

$$\text{Flux} = \underbrace{\boxed{v\,(D_A/2R)\, T\pi R^2\, t}}_{\text{Flux factor}} \;\times\; \underbrace{\boxed{(2/\pi) \int_{-1}^{1} \int_{-\sqrt{1-y^2}}^{+\sqrt{1-y^2}} d(y)\;\; dx\, dy}}_{\text{Flux integral}} \;\times\; \underbrace{\boxed{Q_{op}}}_{\text{Proportion of time operational}} \qquad (10)$$

72. The average collision probability is just the ratio Collisions /Flux. However it should be noted that this ‘average probability’ is conditioned both by the shape of the circle (more flux at greater height) and by the skewed distribution of flights (ie more flux at lower height), so it is not a very meaningful parameter.

73. Note that the factor Q_{2R} does not appear explicitly in the above equations, as the proportion of birds flying at various levels is included within the distributional data $d(y)$. However, for comparison with the basic model, a value Q'_{2R} is readily calculated from the distribution data, as

$$\int_{-1}^{+1} d(y).\ dy = Q'_{2R}$$

The symbol Q'_{2R} is used to differentiate this calculated figure from the figure for Q_{2R} input earlier based on bird survey data.

Annex 5 provides a more detailed derivation of these equations.

The easy stuff (how to do the calculation)

74. Calculating a collision estimate using equation (9). and the number of transits through rotors using equation (10), can be done simply using Sheet 4 ‘Extended model’ which computes both the Collision integral and the Flux integral, if an appropriate flight height distribution is input. The flux factor remains as calculated in Stage B for the basic model, and Q_{op}, the proportion of time turbines are operational, as in Stage E.

(i) Start, as in Stage B of the basic model, with the observed bird density on an area basis, expressed per unit area, D_A. Convert if needed to units of birds/ km^2; the spreadsheet divides this by 10^6 so as to work in birds/m^2. As with the basic model, multiply by the total cross-sectional area of the rotors $T\pi R^2$, and the number of seconds t during which birds are active, to get the Flux factor. There is no need however to deploy Q_{2R}.

(ii) Data on the flight height distribution must be available as a table showing the relative frequency of bird flights at different heights. This data should be normalised, that is the sum of all the relative frequencies across all heights should be 1. Relative frequency is $D_v(Y) / D_A$, and the sum of $D_v(Y)$ across all heights is just D_A, the total bird density per km^2 ,so the sum of all relative frequencies is 1. Frequency is in units of 'per metre of height'.

(iii) Sheet 5 of the spreadsheet 'Flightheights' contains generic data from Cook et al[iii] for a number of species. These give flight height relative frequencies at 1m intervals; only the data up to 150m height is shown in the spreadsheet. Columns A and B are the 'master data' ie these columns contain the data which are used in the calculations of Sheet 3. To use a new data table (eg for other species, copy the appropriate flight height column for this species and paste the column into column B (note, don't cut and paste, just copy, so as to leave intact a copy of the data outwith the master columns. The entire column should be copied and pasted, as it includes the name of the species and the number of points in the table, as well as the table of frequencies itself.

(iv) Normally, the hubheight of wind turbines is measured from Highest Astronomical Tide (HAT), to help ensure navigational clearance requirements are satisfied. However, bird flight heights are measured relative to sea level, which may be 2-3 metres or more lower. Mean sea level (Z_0) and HAT are normally stated relative to Chart Datum (CD). The calculation allows for a tidal offset to be added to the hubheight, to allow for this additional height above mean sea level. The tidal offset should be entered in the Input Data sheet. This offset can make a substantive difference to the calculated collision risk, reducing the estimate of risk by 25-30% for some species.

(v) Sheet 4 'Extended model' then does the necessary work in calculating the Collision and Flux integrals. The sheet undertakes a numeric integration of p(x,y), first across x for each horizontal chord of the rotor, and secondly across all heights y, factoring in the flight distribution d(y).

(vi) Following equation (9), multiply the Collision integral by the Flux factor and by the proportion of time Q_{op} for which the turbines are operational, to get the expected collisions assuming no avoidance. Sheet 2 'Overall collision risk' draws on the Collision integral calculated in Sheet 4, and does this multiplication. It also draws on the Flux integral in Sheet 4, to provide a view on the total number of rotor transits in each month. These calculations are presented as 'Option 3'

(vii) In this extended model, the distribution of bird flights with height already includes the information on the proportion flying at risk height. It is valuable nonetheless to evaluate Q'_{2R} from the flight height data and check that it is consistent with survey findings and other sources of data. Sheet 4 shows the value of Q'_{2R} derived in this way directly from the flight height distribution, using the formula

$$Q'_{2R} = \int_{-1}^{+1} d(y)\, dy$$

75. Adding a tidal offset as at stage (iv) takes account of the height of the rotors above mean sea level, but not of the variation of the tides. If the distribution of bird flight heights relative to the sea surface is independent of the level of the tide, then at times of high tide there will be an increased bird density at rotor level, and reduced at times of low tide. As the flight height distribution is non-linear with height, these two effects do not balance out. The 'tidal asymmetry correction' factor is generally small and may be ignored, but a method of calculating it is nonetheless provided, in Annex 7, for use at sites with a particularly large tidal range (eg > 5metres).

STAGE E – AVOIDANCE AND ATTRACTION

Avoidance

76. The preceding stages of the model assume that birds take no avoiding action whatsoever in response to wind turbines. In reality, birds mostly do take effective avoiding action so as to avoid collision with wind turbines. Birds may avoid the area of the windfarm altogether, or they may use more indirect flight routes to bypass the windfarm – referred to as 'macro' or 'far-field' avoidance or 'displacement'. Alternatively, birds may continue to fly within or close to the windfarm, but exhibiting 'micro' or 'near-field' or 'behavioural' avoidance in which birds choose routes which pass between rotors; or fly higher or lower to avoid the rotors; or take emergency action in-flight to escape an approaching blade.

77. Monitoring of windfarms onshore is generating some useful information on levels of avoidance of some land-based bird species. Some of that data derives from collision monitoring, based on regular site scans for bird corpses, and some of it from observations of habitat use in the vicinity of windfarms. For many bird species, avoidance rates of 98% or higher have been observed, implying that the collision risk is less than 2% of that calculated from stages A-D alone. **Avoidance is included in the collision risk model simply by multiplying the before-avoidance collision estimate by (1 - A) where A is the appropriate overall avoidance rate (see Scottish Natural Heritage 2010[xii] for a review).**

78. In general the information for onshore species is not sufficient to discriminate in a quantitative way between macro avoidance (ie displacement or far-field avoidance) and micro (near-field) avoidance, though some Dutch studies are yielding useful data. Offshore, a number of studies have examined macro and micro avoidance behaviour for some seabirds (see Cook et al (2012)[iii]). As monitoring data builds up from constructed offshore windfarms, it may be possible to make more definitive predictions than at present on rates of both macro and micro avoidance. The overall avoidance rate $A_{overall}$ is simply related to macro and micro avoidance rates:

$$(1 - A_{overall}) = (1 - A_{macro}) \times (1 - A_{micro})$$

To obtain an overall avoidance rate in this way, information is needed on both macro and micro avoidance rates, each of which will be less on its own than the overall avoidance rate. In particular, if information on likely displacement is used to conclude that a proportion of birds will not use the windfarm site, that is in effect an application of the $(1 - A_{macro})$ factor. The avoidance rate then applied to those birds not displaced would then have to be a micro-avoidance rate A_{micro}, derived from monitoring observations solely of birds actually flying through windfarms. A micro-avoidance rate will be considerably lower than a rate for overall avoidance which includes displacement effects.

79. Where detailed information on macro and micro avoidance is not available then overall avoidance rates are best estimated by using monitoring data from existing windfarms, comparing actual mortality to that predicted if pre-construction levels of flight activity were maintained:

$$A_{overall} = 1 - \left\{ \frac{\text{Actual collision rate}}{\text{Predicted collision rate if pre-construction levels of flight activity were maintained}} \right\}$$

Care should be taken to ensure that the data on which such avoidance rates are based are on a consistent basis, having regard for example to the potential for changes in turbine model and flight risk heights as between those modelled in a collision risk assessment at the time of preparing an environmental statement, and those actually built.

80. In particular, if the extended model taking account of flight height distribution is used, **it is important that the calculations on which avoidance rates are based also start with a no-avoidance collision rate derived using the extended model.** Where the bird flight

density is skewed towards low altitude, a greater proportion of birds above the minimum risk height will miss the rotor, simply because, at a level close to rotor minimum height, the rotor circle intercepts relatively few flights. This is taken into account through the limits to the x integration in equations (9) and (10). This propensity to miss the rotor must not be confused with avoidance, which requires a behavioural response by a bird. Put another way, if an avoidance rate is calculated by comparing collision rate observations with a calculated avoidance rate using the basic (uniform flight density) model, then that avoidance rate will already include for the fact that low-flying birds will more often miss the rotor. Using such an avoidance rate in conjunction with the extended model would double-count that factor.

81. **All current flight activity should be included within a windfarm collision risk estimate, and the avoidance rates used for collision risk estimates should be characteristic of overall avoidance, ie they should include both macro avoidance (displacement or far-field avoidance) and micro (near-field or behavioural) avoidance.** In particular the likelihood of displacement should be included as an aspect of overall avoidance. Elsewhere in the bird impact assessment the potential direct impact of displacement on the bird population, in terms of reduction in available habitat, should also be assessed.

82. The lack of firm evidence surrounding avoidance rates will almost certainly dominate the uncertainty inherent in the collision risk estimate. For a few land-based bird species there is now substantial international experience on levels of avoidance from long-standing monitoring studies, such that some confidence can be placed in the assumption of high levels of avoidance. However for marine species there is limited firm data as yet on which to base predictions. It should be noted that avoidance behaviour may vary seasonally, and between groups of birds of the same species.

83. **The collision risk estimate should conclude with a table showing potential collision mortality using a range of assumed avoidance rates.** The text relating to this table should point to any evidence from existing post-construction monitoring on the respective or similar bird species which might indicate what levels of avoidance are best supported by evidence. As a default in the absence of specific avoidance information for the species in question, it is recommended that collision risks be evaluated assuming avoidance rates of 95%, 98%, 99% and 99.5%.

Attraction

84. Offshore windfarms may create new habitat which encourages aggregation of fish, and as a result birds may be attracted into the windfarm for foraging. Lighting on wind turbines may also have an effect in attracting birds at night. Where such attraction occurs, it follows that collision risk may be enhanced as a result of increased flight activity through the windfarm. Attraction is in effect a form of ‘negative displacement’ and could in principle be included in the collision risk assessment by including an appropriate negative component in macro avoidance. However, in most circumstances there is not enough definitive evidence to make quantitative predictions on attracting birds with any certainty.

85. **Where, as part of an overall bird impact assessment, attention is drawn to the potential for a wind farm to attract birds, the potential for additional collision risk should also be considered.**

STAGE F - EXPRESSING UNCERTAINTY

86. In a collision risk estimate following the above method, there are a large number of sources of variability or uncertainty in the output. The main sources of uncertainty are:

    - survey data is sampled, often both in time and space, and usually exhibits a high degree of variability. Mean estimates can only be representative of flight activity
    - survey data is unavailable for certain conditions, including night time and storm conditions
    - natural variability in bird populations, over time and space, for ecological reasons
    - flight height information may be subject to observer bias
    - the collision risk model uses a simplified geometry for turbine blades and bird shape
    - it does not include any risk of collision with turbine towers
    - details of blade dimension and pitch may be unavailable at the time of making the estimate
    - turbines deployed may differ from those used in the collision risk analysis
    - bird parameters (length, wingspan, flight speed) have a distribution, they are not fixed
    - bird speed is not a constant but is dependent on wind speed
    - insufficient knowledge about bird displacement and attraction effects
    - there is limited firm information on bird avoidance behaviour at sea

87. Perhaps the most important issue is to keep these uncertainties in proportion. For some of these uncertainties (eg bird density from survey data) the range of variability may be fairly clear from the variability between different survey days. Observer bias in flight height estimates may be tested, for example, by duplicating observers on occasion and comparing results. There are uncertainties in using the collision model itself, for example in using a single bird speed, or if the calculation is made for only one turbine speed rather than deriving an average over all turbine speeds. However these uncertainties are probably less significant than the errors introduced by variability in the survey data input.

88. Then there is uncertainty over avoidance behaviour. At present there is only a handful of bird species for which collision mortality at onshore windfarms has been sufficiently monitored to enable an avoidance rate to be used with confidence. For marine bird species, there is as yet limited information upon which to base a judgement on an appropriate avoidance rate to use. The uncertainty here ranges over an order of magnitude. If an avoidance rate of 98% is used, for example, that may be judged subject to uncertainty covering a range from 95% to 99.5%, representing non-avoidance behaviour between 5% and 0.5%. For the foreseeable future, it seems likely that the uncertainties surrounding bird avoidance behaviour are likely to dwarf the errors and uncertainties arising from an inexact collision model or variability in survey data.

89. A similar position relates to the extent to which birds may respond to habitat changes caused by the windfarm. Here also there is insufficient experience yet to be able to predict with confidence likely levels of displacement or attraction in response to new habitats, or indeed whether these patterns of behaviour will persist or change over time.

90. For these reasons it is proposed that uncertainty due to avoidance behaviour, and uncertainty over response to habitat changes, should be handled differently from uncertainties elsewhere in the calculation.

91. **The output should convey the uncertainty in the collision risk estimate, by indicating, in addition to a 'best estimate', a range of confidence around that estimate. Though it is unlikely (with the exception of the survey data) that these can be subject to detailed statistical analysis, the aim should be to express the range of uncertainty at around the 95% confidence level.**

92. The range of uncertainty should reflect

- uncertainty or variability in flight activity data (including imprecision on flight height estimates and lack of knowledge about night-time behaviour)
- uncertainty due to the limitations of the collision model, including the variability of bird dimensions and flight speed, the simplification in shape of a bird and turbine blades. As an expert guesstimate, the uncertainties arising from the collision model, if all required turbine parameters are fully available, may be of order ± 20**%.**
- uncertainty arising from turbine options yet to be decided, in number, size and speed, where that is consistent with the 'Rochdale envelope' flexibility described in guidance by the Infrastructure Planning Commission (2011)[vi] . These options should include a 'worst case' in terms of the option likely to present greatest bird collision risk.

The range of uncertainty due to each of these three sources should be separately identified and, as the three uncertainties are of independent origin, they may be combined to give an overall uncertainty of $\sqrt{(u_1^2 + u_2^2 + u_3^2)}$ where u_1, u_2 and u_3 are respectively the percentage uncertainties from each of these sources.

Box 2: Example of presentation of uncertainty

(Note that the asterisked figures are chosen for example only and should be derived or judged from detailed consideration of the accuracies and uncertainties inherent in the input data.)

Best estimate of annual collision risk (birds per annum)

assuming 98% avoidance rate	147 *	

Range of uncertainty

due to variance and uncertainty in flight activity	± 50% *	
due to simplifications in collision model	± 20%	
due to design options yet to be finalised	± 15% *	
overall $\pm\sqrt{(0.5^2 + 0.2^2 + 0.15^2)} = 0.56$	± 56%	range 65 - 230

93. Where the extended model is applied using the generic height data from Cook et al[iii], that paper provides confidence intervals around the median data points. The range of uncertainty relating to flight height can be estimated by replacing the median set of data (as shown in Sheet 5: Flightheight) by, respectively, the upper and lower 95% confidence levels, and noting the corresponding uncertainty in the collision risk.

94. **Finally, the output should state the effect on the collision risk of a range of assumptions on avoidance. This should be covered by a statement conveying the status of current information on avoidance behaviour of the bird species in question, noting any variability in this behaviour, and drawing conclusions about the likely collision risk.**

Box 3: Example of presentation of uncertainty on avoidance

Species: XXXXXXX

Best estimate of annual collision risk (birds per annum)

assuming	95% avoidance	367	}	
	98% avoidance	147	}	all subject to ± 56%
	99% avoidance	73	}	
	99.5% avoidance	37	}	
	99.75% avoidance	18	}	

Information on avoidance for this species suggests 99% is most appropriate (refer to text in ES) but the lack of data means that the confidence interval may extend from 95% to 99.9%.

95. The collision risk estimate should also outline qualitatively the possible likelihood and scale of any further collision risks which might result from the wind farm attracting birds (see paragraphs 84/85).

FOOTNOTE

96. One risk of prescriptive guidance is that it could stifle innovation in improved methods. Developers and their advisors are encouraged where appropriate to go beyond the core requirements set out in this guidance; but where they do so, the standard approach of this guidance should also be pursued so as to make clear how the results of any improved methods differ from that of the standard approach.

NOTES ON USING THE SPREADSHEET

The Excel spreadsheet which accompanies this guidance is intended to take the user easily through the first five stages of the process.

Sheet 1: Input data is provided so that all input data is input on this sheet. There are no calculations or calculated fields on this sheet. The user should not input data on any of the three following sheets 2-4, other than (if desired) to replace the blade profile in Sheet 3 with a more specific one for the actual turbine blades used. Sheet 1 is organised with blocks of input data on the bird species; on flight activity from bird survey; on migrant birds (to be used if relevant); on the windfarm; on the turbines to be used in the windfarm; and finally on the avoidance behaviour used in presenting the results.

The source data used for each input should be identified for easy reference on the spreadsheet, and the sources should be listed in full within the Environmental Statement.

Sheet 2: Overall collision risk is the master sheet bringing together all the calculations of Stages A through E, and concluding with overall collision estimates, given a range of assumptions on rates of avoidance:

- Stage A states the information on the density of flying birds, the proportion flying at risk height, and the time over which such bird activity persists.
- The sheet then presents the basic model (Option 1), giving
  - Output from Stage B - the estimated number of potential bird transits through rotors of the windfarm.
  - Output from Stage C - the probability of collision during a single bird rotor transit.
  - Output from Stage D - the potential collision mortality for the bird species in question, assuming current use of the site and no avoiding action is taken
- The sheet then re-applies the basic model, only using the value of Q'_{2R}, the proportion of bird flying at risk height derived from the flight height distribution (Option 2). For this purpose flight height distribution data must be loaded in the first two columns of the Flightheight sheet.
- Finally the sheet applies the extended model allowing flight height distribution to be taken into account (Option 3).
- Output from Stage E is the potential collision mortality for the bird species in question, taking avoidance and other likely behaviour change into account. The user must choose to which of the above set of results (Options 1, 2 or 3) the avoidance factors should be applied.

Sheet 2 draws in turn from Sheets 3-8. Sheet 2 will not display the results from the Extended model until Sheet 4 (Extended model) has been activated by clicking on that sheet, when it will automatically calculate. Once it has done so, Sheet 2 will display the appropriate results.

Sheet 3: Single transit collision risk. This sheet covers stage C of the process, calculating the probability of collision for a bird making a single passage through a rotor at each radius r, in increments from r/R=0.05 out to r/R=1. The collision probability is then averaged over the entire area of the rotor disc, by summing the probability over successive concentric rings each of width 0.05R, multiplying by the area of the successive concentric rings, and dividing by the total area πR^2 of the rotor disk (see paragraphs 46/47). The method used is essentially a trapezoidal numerical integration. The calculation is undertaken separately for upwind and downwind flight, and an average taken.

Sheet 4: Extended model. It should be noted that this sheet requires macros to be enabled, as much of the functionality of this sheet is based on function routines programmed in Visual Basic. If at any stage calculations are not triggered automatically, press Shift-F9 to force recalculation.

This sheet repeats, in the panel at top right, the calculations of the basic model[4]. In the lower panel, this sheet carries out the calculations of the extended model, based on the flight height distribution data in the following sheet 'Flightheight'. The key calculated outputs are Q_{2R}, the flux integral, and the collision integral (for upwind and downwind flight and average of both). The sheet allows input of xinc and yinc, the increments used in the numerical integration. By default these are set to 0.05, ie one twentieth of a rotor radius. For increased precision these may be set to a smaller value like 0.01, but the worksheet calculation time may become significant.

This sheet also shows a table and a set of graphs derived from the table. The table shows the height y from rotor minimum to rotor maximum; the corresponding bird density d(y) (interpolated from the Sheet 5 data); the contribution of that horizontal strip of rotor at height y to risk (up and downwind), and the product of bird density and contribution to risk (up and downwind). The chart then presents these as line graphs. These calculations are all based on a value of 0.05 for xinc and yinc, the increments used in integration. The graphs are included to provide a live illustration of the effect of a skewed flight distribution. They show how the reduction of collision risk towards the rotor minimum height at y=-1, and the reduction in bird density due to the rapidly falling bird density with height, combine to squeeze the zone in which most collisions occur to an area just above y=-1.

Note that the table and graph are calculated entirely separately from the calculation of the Collision and Flux integrals, which make use of the user-input values of xinc and yinc.

Sheet 5: Flight Height. This sheet contains, in the first two columns, the flight height distribution used by Sheet 4 to calculate collision risk. Data is also shown for a number of other species, simply for ease of copying the data and pasting in to the first two columns. The standard form for this flight height distribution data is in 1m height intervals, with values of $D_v(Y)/D_A$ such that the column totals to 1.0. A frequency distribution with a wider height interval may be used, but then D(Y) must be divided by the interval, such that the values properly represent relative bird density per metre of height, and the column will total to (1.0/interval).

Sheet 6: Migrant collision risk. This sheet undertakes a similar calculation of collision risk to Sheet 2, but makes use of information in a different form on the density of birds passing through the windfarm, such as may be available for migrating bird species - see Annex 6 for a full description.

Sheet 7: Daylight and night hours. Given the input latitude, this sheet computes the daylight and night hours in each month within which there could potentially be bird activity.

Sheet 8: Large array correction. This is an add-on, which enables a correction to be made for large arrays where the collision rate is such that bird density might significantly decline as birds pass through the windfarm. These correction factors are then applied to the collision rate estimates in Sheet 2 'Overall collision risk'. In most circumstances the results will demonstrate that a large array correction is not significant and can be ignored.

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[4] There are small differences in the output values for gliding flight, as this sheet avoids a simplification in the earlier model

Notes on input data

Care is needed throughout to use the correct units as specified below and in the spreadsheet. In the main standard SI units are used. However some of the inputs (eg bird density) use units which developed in use as a matter of common practice – as indeed are the outputs in terms of collisions per month.

All the following input data should be entered using Sheet 1 – 'Input data', unless information is available to use a blade chord profile specific to the turbine being used in the relevant columns of Sheet 3 – 'Single transit collision risk'.

Bird data			
Symbol	**Description**	**Units**	**Notes**
	Species name		to help identify this spreadsheet
L	Length of bird	m (metres)	these should be drawn from standard reference works, eg Cramp & Simmons (1983)[xiii] or from BTO Bird Facts[xiv].
W	Wingspan of bird	m (metres)	
v	Flight speed	m/sec	
F	Flight type		'flapping' or 'gliding' - the spreadsheet then applies the relevant factor F = 0 for flapping flight, or +1 for gliding flight
	Nocturnal activity factor	1-5 ranking from Garthe and Hüppop/ King et al	the spreadsheet converts this factor to 0%/ 25%/ 50%/ 75%/100% daytime activity

For flight speed, usually a typical mean flight speed as given in such standard references will be adequate. However, where there is a need to explore the collision risk arising from different types of bird behaviour involving very different flight speeds (eg pursuit, or foraging), then the collision risk calculation should separate out the risk for those birds engaged in each behaviour, and sum the collision risk, as this varies with flight speed in a non-linear way.

Flight activity data			
Symbol	**Description**	**Units**	**Notes**
D_A	Bird density (day)	birds/km^2	Average number of birds in flight in daytime at any height, per square kilometre, as derived from field observation
Q_{2R}	Proportion at rotor height	%	% derived from bird survey, in the light of the projected rotor diameter and rotor hub height. The extended model also computes a figure for this, termed Q'_{2R} to distinguish it
	Proportion of flights upwind	%	This should be set to 50% unless survey indicates a predominant direction relative to wind, eg for large-scale migration flights

Flight activity data – additional for migrants see Annex 6 for details

Windfarm data			
Symbol	**Description**	**Units**	**Notes**
	Latitude of windfarm	degrees latitude (including decimal places)	include degrees and minutes in degrees with decimal places; this data is used to work out daylight hours in each month
T	Number of turbines		
Q_{op}	Proportion of time turbines are operational	%	This includes down-time for maintenance as well as time inactive because of low-wind or storm conditions
	Width of windfarm		optional; this is used only in the large array correction

Turbine data			
Symbol	**Description**	**Units**	**Notes**
R	Rotor radius	m (metres)	measured from the axis of rotation to blade tip. (This differs from the blade length, which is the length of the blade itself from where it is attached to the hub to the blade tip.)
H	Hub height	m (metres)	This is the height in metres of the rotor hub, ie the axis around which it rotates, above the sea surface taken as the Highest Astronomical Tide. In conjunction with the rotor radius and tidal offset, this determines the flight altitudes at risk. In the basic model this parameter is not used in the calculation but it is desirable to state it, as the proportion of birds flying at risk height is strongly dependent upon it. It is however a key parameter in the extended model.
	Tidal offset	m (metres)	This is the difference in metres between HAT (from which hub height is measured) and mean sea level Z_0. The difference is typically 2-3m but may be up to 5m or more in estuarine locations
Ω	Rotation speed	rpm (revolutions per minute)	The spreadsheet converts to radians/sec as required in the underlying formulae
c	Blade chord width (along length)	m (metres)	see below
γ	Average blade pitch	degrees relative to rotor plane	see below

Rotation speed when generating of most contemporary turbines is variable within a pre-determined range. A time-averaged mean of operational rotor speeds should be used, taking account of the expected frequency of different wind speeds and the resulting projected operational speeds (see paragraphs 48-49).

Note that the Band 2000 version of this spreadsheet requires input of the Rotation Period, ie the time required for one full rotation of the rotor, which is the inverse of Rotation Speed: *Rotation period = 1/ (Rotation speed in rpm)*

The underlying formulae make use of rotation speed Ω expressed in radians per second. One complete revolution is 2π radians, and there are 60 seconds in a minute, so Ω = (rpm /60) x 2π, a conversion undertaken by the spreadsheet.

Chord width. The model considers a blade to be a twisted lamina, ie of zero thickness. It has a chord width, which varies along the length of the blade as it tapers towards the tip. The chord profile in the spreadsheet is typical of a modern 5MW turbine used for offshore generation.

Pitch. The blade also has a pitch angle – the angle between the blade surface and the axis of the rotor. Pitch angle varies along the length of the blade, from a high angle close to the hub, to a low pitch angle towards the blade tips, ie the blade is twisted. Pitch angle also varies as the pitch is controlled to alter the rotation speed of the turbine. In the model, an average angle is used, representing an average pitch along the blade length. 25-30 degrees is reasonable for a typical large turbine.

Note that it is the total cross-sectional area of all the rotors ($T\ \pi R^2$) which is used to calculate the number of bird transits through a rotor. If the size and number of turbines is not known, a figure may be entered directly in Sheet 2 (Overall collision risk) for the ‘total rotor frontal area’: which may be amenable to a better estimate than either the turbine number or size.

Avoidance data

These are the range of avoidance rates to be used when presenting the collision risk conclusions (see paragraphs 76-83). Use avoidance rates if possible which have been established from previous monitoring studies for this species, and an appropriate range to cover the uncertainties involved.

Spreadsheet protection

To protect against unintentional overwriting of formulae, or the entry of input data other than in the ‘Input data’ sheet, each of the worksheets is ‘protected’, and the spreadsheet is fully usable in this state. Should there be a need to change or add to the spreadsheet, the protection can be turned off for any worksheet by going to ‘Tools’ – ‘Protection’ and setting to ‘off’ - there is no password protection in place.

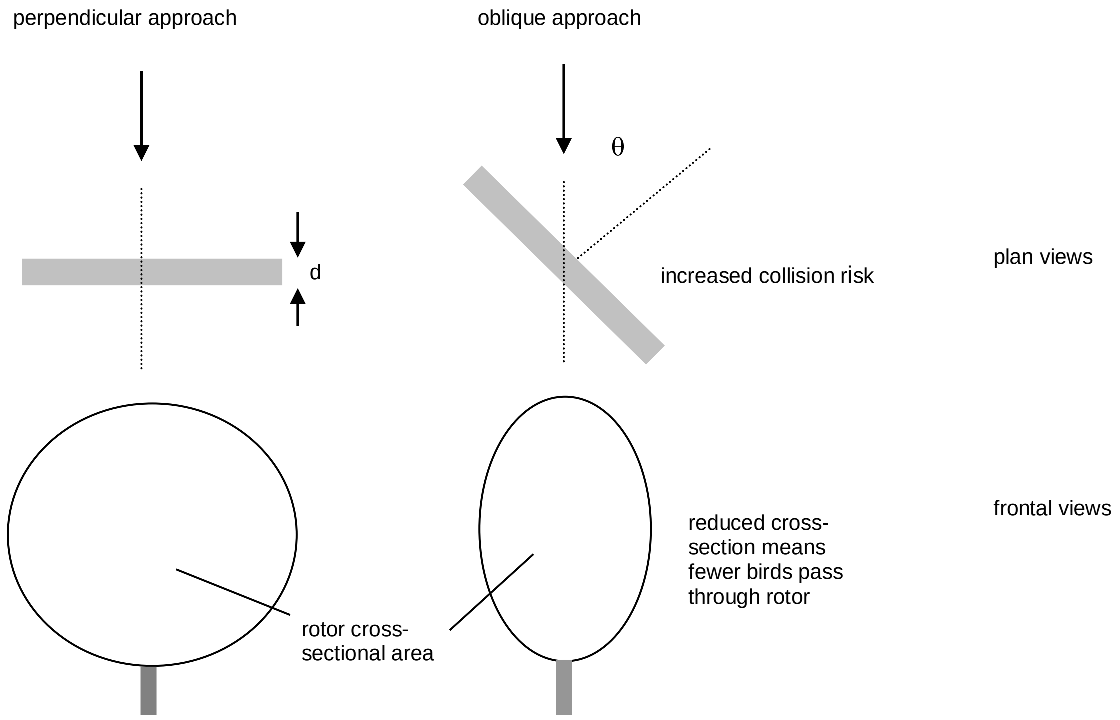
Notes on spreadsheet programming

The functionality of Sheet 4 (Extended model) is entirely based on computations programmed as user-defined functions in Visual Basic. Macros must be enabled. The programme code may be viewed using the ‘Developer’ tab and ‘Visual Basic’ icon. The user-defined functions are listed for reference in Annex 8.

Annex 1 Oblique approach

The collision risk of a bird passing through a turbine is dependent on the angle of approach. If it approaches obliquely, the cross-sectional area presented by the rotor disk will be reduced, as the rotor presents an elliptical rather than circular cross-section to the bird.

Fig A1-1: Effect of oblique approach



If the rotor has radius R and cross-sectional area πR^2 , then to a bird incoming at an oblique angle θ with respect to perpendicular approach, it presents a cross section $\pi R^2 \cos \theta$, thus reducing with $\cos \theta$.

However, if a bird approaching obliquely does pass through a rotor, its collision risk is greater than for a bird approaching perpendicularly, in part because of the increased time the bird takes to clear the full depth of the rotors from back to front, and in part because of the changes in speed of blade approach relative to a bird flying across (as well as towards) a rotor.

In relation to the first of these, a bird making a perpendicular approach has to clear a distance $d + L$ in order not to collide with the blades, d being the depth of the rotor from front to back, and L being the length of the bird. A bird making an oblique approach has to clear a distance $(d+L) / \cos \theta$. The collision risk thus increases, in a first approximation, with $1/\cos \theta$. This 'oblique factor' thus cancels the reduction with $\cos \theta$ due to the reducing cross-section presented by the rotor.

On this basis this guidance considers all bird flights as if they were perpendicular to the rotor plane, and uses the collision risk relevant to flights perpendicular to the rotor. With this simplification stages B and C can be followed sequentially.

However, this does not take account of the second of the above factors, the changes of blade approach relative to a bird flying across a rotor. This leads to a dependence on θ of the collision risk for a bird making an oblique transit which is more complex than $1/\cos \theta$. In particular, an oblique approach leads to the wingspan rather than the length of the bird becoming the dominant element in determining the time it takes for the bird to pass through the rotor plane. Holmstrom et al (2011)[xv] have explored the dependence of collision risk on angle of approach, using a bird modelled as a flat rectangle, building on the original analysis by Tucker (1996a and 1996b)[xvi]. They demonstrate that for large raptors flying downwind through a rotor, collision risk increases with an increasingly oblique angle of approach, reaching a maximum at around 30 degrees from perpendicular approach, then tailing off as the effect of the reduced cross-sectional area presented by the rotor begins to dominate. At the maxima, the collision risk is calculated to be between 10% and 31% higher than for perpendicular approach, dependent on bird parameters and wind speed. Averaged across all angles of approach, the increases for downwind flight may be of order 10-15%, though likely to be less for upwind flight. It is also probable that at values of θ close to $\pi/2$ (ie for flight nearly parallel with the rotor) collision risk rises steeply for birds passing through the rotor, though the likelihood of such an encounter is low because of the edge-on cross-section presented.

The spreadsheet approach accompanying this guidance does not deal with the complexity of oblique angled approaches. If a model for oblique approach were to be used, a stricter approach would require calculation of the number of flights *from each direction* passing through the swept area of the windfarm turbines, applying the probability of collision applying *for that direction*, and *summing these probabilities* for birds flying in all directions.

This guidance makes the simplifying assumption that all flights can be treated as perpendicular to the rotor plane (ie parallel to the rotor axis). This is equivalent to assuming a $1/\cos\theta$ dependence of collision risk for a bird flying through a rotor at angle θ, thus exactly cancelling the $\cos\theta$ dependence of the number of birds flying through the rotor. In the light of the Holmstrom et al (2011) results, it should be recognised that this simplification may underestimate collision probabilities by a factor which, taking account of both upwind and downwind flights, may be of order 10% for large birds.

Annex 2 Relationship between bird flux and bird density

There is a direct relationship between bird density and flux, which is involves a dependence on the speed of the birds (if they were stationary, there would be no flux).

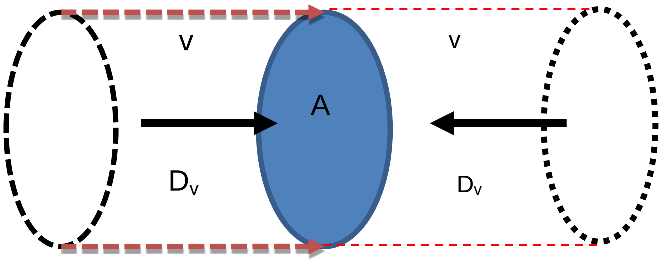
Simplified approach – treating all flights as perpendicular to rotor

First, take a simplified approach in which birds fly either downwind or upwind through a circular rotor area A, but not at oblique angles. Within one second, all birds within the cylinder of base area A and length v will pass through the area A. So the flux F is

$$F = \tfrac{1}{2} D_v \; A \; v \quad \text{downwind} \quad \text{and} = \tfrac{1}{2} D_v \; A \; v \quad \text{upwind}$$

where F is the bird flux per unit area, D_v is the bird density (true density) per m^3 and v is the speed of the birds.

Fig A2-1: Bird flux due to bird density (copy of Fig 5)

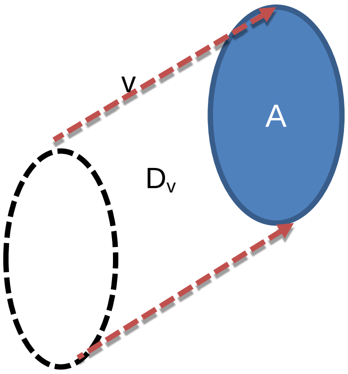


General approach - random horizontal directions

More generally, if one assumes that the birds fly in a horizontal plane, but may fly in random horizontal directions, the flux is

$$F = (1/\pi)\, D_v \; A \; v \quad \text{downwind} \quad \text{and} = (1/\pi)\, D_v \; A \; v \quad \text{upwind}$$

Fig A2-2: Bird flux due to bird density – oblique approach



This takes account of the fact that at an approach angle θ, the area A now looks like an ellipse, not a circle, and thus the volume of the squashed cylinder of length v containing the birds which will reach area A within one second is now $A\, v \cos\theta$ rather than simply A v for the perpendicular approach. The proportion of birds flying at an approach angle between θ and $\theta + d\theta$ is ($d\theta/2\pi$). Total flux from this upwind side is then

$$F = \int_{-\pi/2}^{\pi/2} (D_v / 2\pi)\; A\, v \cos\theta \; d\theta \quad = \quad (1/\pi)\, D_v\, A\, v \qquad \ldots \text{(A2-1)}$$

where F is the bird flux across the area A, D_v is the bird density (true density) per m^3 and v is the speed of the birds.

It should be noted that a flux measurement is directional – for a given density of birds moving in random horizontal directions, a unit area will intercept more birds flying perpendicular to the area than birds flying at an oblique angle, to which the unit area will appear narrower. The $(1/\pi)$ factor accounts for this angle-dependence.

Total bird flux, counting both upwind and downwind flights, is then

$$F_{tot} = (2/\pi)\, D_v\, A\, v \qquad \text{(A2-2)}$$

To convert from a bird flux measurement to a measurement of bird density, use the converse expression

$$D_v = (\pi/2)\, F_{tot} \;/\; (A\, v) \qquad \text{(A2-3)}$$

Using areal bird density

The above refers to bird flux crossing an area such as a rotor disk, and relates it to the bird density D_v surrounding the rotor.

Flux is often referred to as the number of birds F_L flying across a horizontal line, per metre length of that line, at any altitude (as observed, for example, in vertical radar surveys). Taking an aerial view, that is the sum of birds crossing in each 1m band of height, for which the flux is given by equation A2-3:

$$F_L = \sum_{h=0}^{h\ =\ \text{max height}} (2/\pi)\, D_v\, v$$

$$= (2/\pi)\; v\; \Sigma\; D_v$$

But summing the bird density within each successive metre height gives the areal bird density $D_{A.}$. So we have

$$F_L = (2/\pi)\, D_A\; v \qquad \text{birds/sec (per metre length of horizontal line)} \qquad \text{(A2-4)}$$

This equation is the equivalent, using areal density, of equation (A2-2) which uses true density.

The converse is the equivalent of equation (A2-3):

$$D_A = (\pi/2)\, F_L / v \qquad \text{birds / m}^2 \qquad \text{(A2-5)}$$

Annex 3 - Probability of bird being hit when flying through the rotor

The following text is extracted from the Band (2000) guidance published on the Scottish Natural Heritage website. Text in italics has been updated to reflect changes in the accompanying spreadsheet.

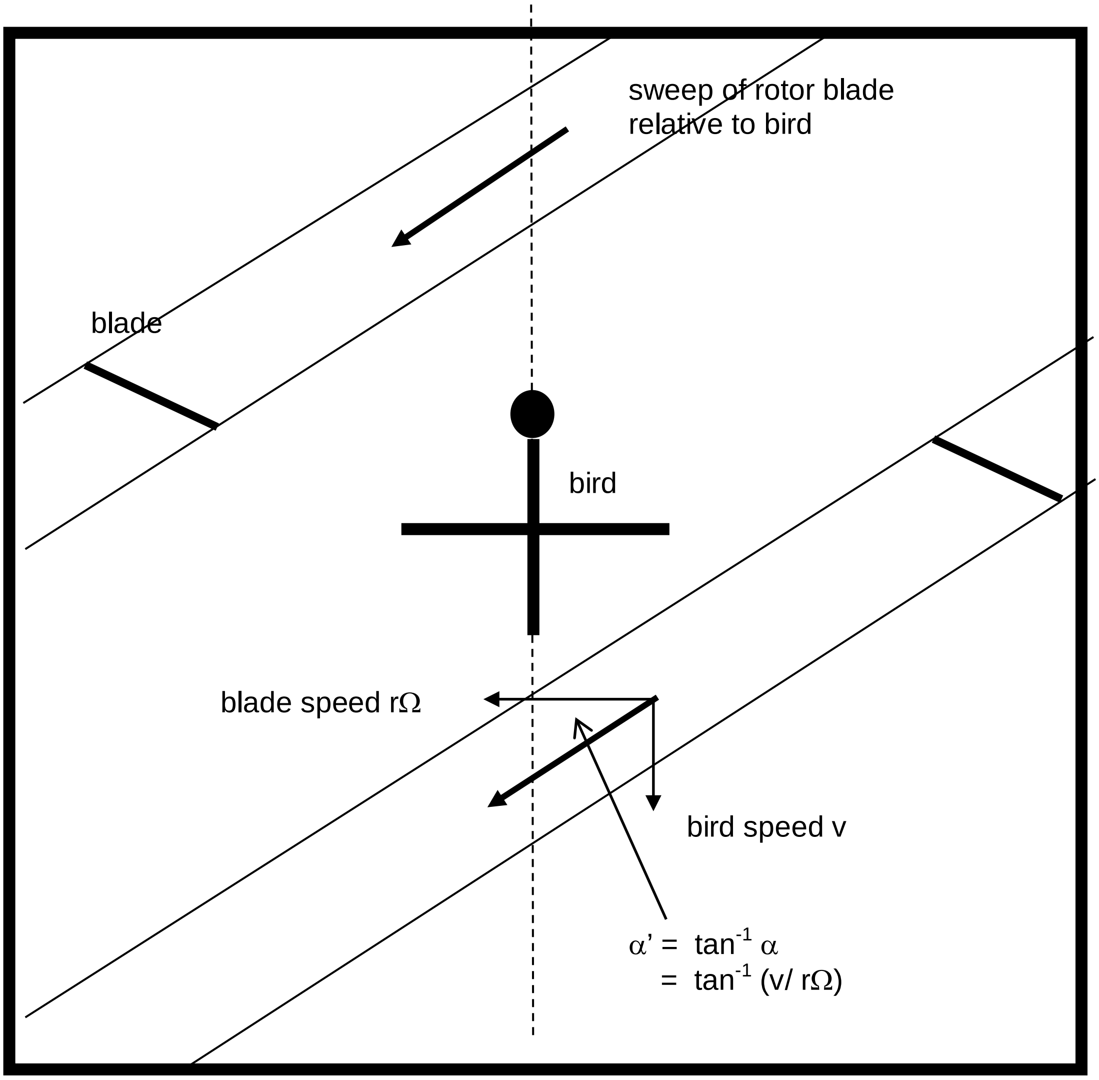
This stage computes the probability of a bird being hit when making a transit through a rotor. The probability depends on the size of the bird (both length and wingspan), the breadth and pitch of the turbine blades, the rotation speed of the turbine, and of course the flight speed of the bird. *The calculation assumes that a bird has an equal probability of passing at any point through the rotor.*

To facilitate calculation, many simplifications have to be made. The bird is assumed to be of simple cruciform shape, with the wings at the halfway point between nose and tail. The turbine blade is assumed to have a width and a pitch angle (relative to the plane of the turbine), but to have no thickness.

It is best to visualise this as in Fig A3-1, looking vertically down on the flying bird in a frame which is moving with the bird. In this moving frame, each rotor blade is both moving from right to left (say) and also progressing towards the bird. Each blade cuts a swathe through the air which depends both on the breadth of the blade and its pitch angle. Successive blades cut parallel swathes, but progressively closer to the bird. The angle of approach of the blade α, in this frame, depends on both bird speed and blade speed. At the rotor extremity, where blade speed is usually high compared to bird speed, the approach angle α' is low, ie the blades approach the bird from the side. Close to the rotor hub, where the blade speed is low and the bird is therefore flying towards a slow-moving object, the approach angle α' is high.

The probability of bird collision, for given bird and blade dimensions and speeds, is the probability, were the bird placed anywhere at random on the line of flight, of it overlapping with a blade swathe (since the bird, in this frame, is stationary). It may therefore be calculated from simple geometric considerations. Where the angle of approach is shallow, it is the length of the bird, compared to the separation distance of successive swathes, which is the controlling factor. Where the angle of approach is high, it is the wingspan of the bird compared to the physical distance between blades, which is the controlling factor.

Fig A3-1: Collision risk from flying through the rotor



The calculation derives a probability $p(r, \varphi)$ of collision for a bird at a radius r from the hub, and at a position along a radial line which is an angle φ from the vertical. It is then necessary to integrate this probability over the entire rotor disc, assuming that the bird transit may be anywhere at random within the area of the rotor disc:

Total probability

$$= (1/\pi R^2) \int\int p(r, \varphi)\, r\, dr\, d\varphi$$

$$= 2 \int p(r)\, (r/R)\, d(r/R) \qquad \ldots\ldots \quad \textbf{(A3-1)}$$

where $p(r)$ now allows for the integration over φ.

Probability p of collision for a bird at a radius r from hub

$$p(r) = (b\Omega/2\pi v) \left[K \; | \pm c \sin\gamma + \alpha\, c \cos\gamma | \; + \begin{array}{ll} L & \text{for } \alpha < \beta \\ W\alpha F & \text{for } \alpha > \beta \end{array} \right] \qquad \ldots\ldots \quad \textbf{(A3-2)}$$

where b = number of blades in rotor
Ω = angular velocity of rotor (radians/sec)

c = chord width of blade
γ = pitch angle of blade
R = outer rotor radius

L = length of bird
W = wingspan of bird
β = aspect ratio of bird ie L / W
v = velocity of bird through rotor

r = radius of point of passage of bird
α = $v/r\Omega$

F = 1 for a bird with flapping wings, or = $(2/\pi)$ for a gliding bird

K = 0 for one-dimensional model (rotor with no zero chord width)
= 1 for three-dimensional model (rotor with real chord width)

The chord width of the blade c and the blade pitch γ, ie the angle of the blade relative to the rotor plane, vary from rotor hub to rotor tip. The chord width is typically greatest close to the hub and the blade tapers towards the tip. The pitch is shallowest close to the tip where the blade speed is highest. The apparent width of the blade, looked at from the front, is $c \cos\gamma$, and the depth of blade from back to front is $c \sin\gamma$.

The factor F is included to cover the two extreme cases:

(i) F=1: where the bird has flapping wings. In this case ($p(r, \varphi)$ has no dependence on φ); or

(ii) F = $2/\pi$: where the bird is gliding, $p(r, \varphi)$ is dependent on φ, with a maximum above and below the hub, and a minimum at the sides when the wings are parallel with a passing rotor blade.

The sign of the $c \sin\gamma$ term depends on whether the flight is upwind (+) or downwind (-).

The factor K is included to give a simple option of checking the effect of real blade width in the result: K=0 models a one-dimensional blade with no chord width.

As α, c and γ all vary between hub and rotor tip, a numerical integration is easiest when evaluating equation (A3-1).

For ease of use these calculations are laid out on a spreadsheet. (*This is reproduced in an updated form in Sheet 3 'Single transit collision risk' in the spreadsheet accompanying this guidance. However the input data must now be entered through Sheet 1 'Input data'.*)

The spreadsheet calculates p(r) at intervals of 0.05 R from the rotor centre (ie evaluating equation (A3-2)), and then undertakes a numerical integration from r=0 to r=R (ie evaluating equation (A3-1). The spreadsheet is set out as follows:

1 The input parameters are in the first two columns. Bird aspect ratio β is calculated.

2 Collision probabilities are then calculated for radii at intervals of 0.05 R from the hub to the tip. Each radius is represented by a row in the table, with the value of the radius r/R in the first column..

3. The second column of the table is the chord width at radius r as a proportion of the maximum chord width. The taper will differ for different turbine blades. *The taper profile in the updated spreadsheet circulated with this guidance is based on the blade of a typical 5 MW turbine used for offshore generation.*

4. Factor α is calculated.

5. The 'collide length' is the entire factor within square brackets within equation (2) above, using the upwind case.
6. p(collision) is p at radius r, as calculated by equation (A3-2). It is however limited to a maximum value of 1.
7. 'contribution from radius r' is the integrand of equation (A3-1) (including the factor 2) prior to integration.
8. The total risk is then the sum of these contributions.
9. The calculation is then repeated for the downwind case.

10 The spreadsheet then shows a simple average of upwind and downwind values. (Note that in a real case it may be important to add in the effect of wind to the bird's ground speed, and flight patterns may not be such that upwind and downwind flights are equally frequent.)

The result is an average collision risk for a bird passing through a rotor.

Note that there are many approximations involved , for example in assuming that a bird can be modelled by a simple cruciform shape, that a turbine blade has width and pitch but no thickness, and that a bird's flight will be unaffected by a near miss, despite the slipstream around a turbine blade. *Thus the calculated collision risks should be held as an indication of the risk - say to around ± 20%, rather than an exact figure*[5]*.* It is also simplistic to assume that bird flight velocity is likely to be the same relative to the ground both upwind and downwind. Ideally, separate calculations should be done for the upwind and downwind case, using typical observed flight speeds.

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[5] In the 2000 version, the uncertainty was judged to be ±10%. In the light of the possible effect of skewed flight distributions and the effects of oblique angle approach, as well as the various simplifications in the model, this advice is updated to ±20% in the present guidance.

Annex 4 - Large turbine arrays

The overall approach in this guidance calculates the rate of collision arising from each turbine independently operating in an airspace with a projected density of flying birds, and sums up the risk from all T turbines in the windfarm. In this approach, the size and layout of the windfarm are unimportant, if the density of flying birds is the same for all turbines.

For large turbine arrays where the overall probability of a bird colliding is relatively high, it may be appropriate to take account of the declining proportion of the birds surviving passage through early rows of turbines and thus exposed to collision risk in later rows. In effect, the density of flying birds surrounding turbines in later rows may be reduced as a consequence of collisions in earlier rows. (While it is convenient to think in terms of successive rows of turbines, the same principle applies within any array of turbines, even if located in a disordered array.)

For this, the overall size and layout of the windfarm are relevant. Here we need to consider the risk to a bird flying through the windfarm as a whole, which depends on how widely spaced the turbines are. Again maintaining the assumption of perpendicular approach to rotors, the collision risk for a single bird due to any one turbine (ie disregarding the risks to the bird presented by other turbines) is

$$c = (\pi R^2 / 2Rw)\ p\ Q_{op}\ (1\text{-}A)$$

where πR^2 is the cross-sectional area of a single turbine, 2Rw is the overall cross-sectional area of the windfarm of width w and risk height 2R, p is the collision risk for a bird passing through a rotor, Q_{op} is the proportion of time the turbine is operational, and A is the avoidance rate assumed.

Imagine an array of turbines with n rows of t turbines, each of which on its own would present a collision risk c. The overall collision risk for a single bird passage, if bird density depletion effects are ignored, would be simply C = ntc.

To take account of depletion, consider that the probability of incoming birds surviving a passage across the first row is (1-tc), and the proportion attempting to pass through row 2 is therefore (1-tc). The proportion surviving row 2 is $(1\text{-}tc)^2$ and so on until:

$$\text{after row } n \text{ the proportion surviving is } (1\text{-}tc)^n \qquad \ldots \text{(A4-1)}$$

which may be expanded as a convergent binomial series

$$(1\text{-}tc)^n = 1 - ntc + (n(n\text{-}1)/2)\,(tc)^2 - (\,n(n\text{-}1)(n\text{-}2)/6\,)\,(tc)^3 + \ldots\ldots$$

where the terms are successively smaller.

The 'large array collision risk' C_{LA} is (1 – proportion surviving) ie

$$C_{LA} = ntc - (n(n\text{-}1)/2)\,(tc)^2 + (\,n(n\text{-}1)(n\text{-}2)/6\,)\,(tc)^3 - \ldots\ldots$$

The first term here is ntc = C, the risk from a single turbine multiplied by the number of turbines. The subsequent terms provide a correction to that value which takes account of bird density depletion.

Dividing throughout by C we get

$$C_{LA} / C = 1 - ((n\text{-}1)/2n)\,C + (\,(n\text{-}1)(n\text{-}2)/6n^2\,)\,C^2 - \ldots\ldots \qquad \ldots \text{(A4-2)}$$

Thus a first order correction to the value C given by the collision model can be made by subtracting ((n-1)/2n) C. The C^2 and subsequent terms are most likely to be insignificant.

Box 4: Example of large array correction

Take an array of $T = 144$ turbines, rotor radius 50m, in an array of width 6km.

Assume input data

Probability of collision for single rotor transit	= 0.15
Proportion of time operational	= 90%
Avoidance rate assumed	= 97.5%

$$C = T\,(\pi R^2 / 2Rw)\;\; p\, Q_{op}\, A$$
$$= (\pi \times 50 \times 50)/(2 \times 50 \times 6000) \quad \times 144 \;\times\; 0.15 \;\times 0.9 \;\times 0.025 = 0.00636$$

Take number of rows $n = \sqrt{T} = 12$

$$C_{LA} / C \;=\; 1 - ((n-1)/ 2n)\, C + ((n-1)(n-2) / 6\, n^2)\; C^2 \;-\; \ldots\ldots$$
$$=\; 1 - 0.0029 + 0.0000051 \ldots\ldots.$$
$$=\; 0.997 \quad \text{ignoring terms of order } C^3 \text{ and higher}$$

Thus ‘Large array correction factor’ = 99.7%

Very often the layout of a windfarm is not known at the time of collision risk assessment, so an exact value for n is not known; and in any case the collision risk has to account for birds entering the windfarm from all directions. A rough approximation is to use $n = \sqrt{T}$ ie the square root of the total number of turbines. If a more analytic approach is necessary, with discrimination between flight directions, then the model of Bolker et al (2006)[xvii] may be used.

If realistic avoidance rates have been taken into account in the collision model, such ‘large array corrections’ are likely to be small and can be ignored. However if the overall risk to a single bird passage is of order 0.1 or above, the large array correction will be significant. A spreadsheet is provided at sheet 8 ‘Large Array Correction’ to enable the correction to be calculated easily. The output from this sheet is then applied in the final set of collision estimates in the ‘Overall Collision Risk’ spreadsheet.

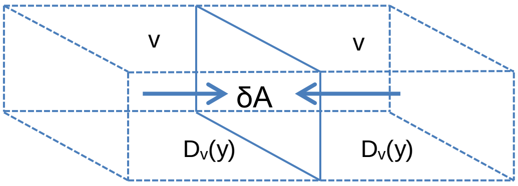
Annex 5 - Using flight height distributions – derivation of equations

Take a rotor disc of radius R, perpendicular to incoming bird flights at various heights Y. Assume that bird density $D_v(Y)$ (in birds per m^3) is a function of flight height; $D_v(Y)$ is the 'flight height distribution'.

Using the same approach as in Figure 5 and equation (1) (paragraph 36) of the main text, consider the bird flux through a small element δA of the disc. Within one second all birds within a distance v on one side and flying towards the rotor will pass through the area δA, as in Figure A5-1. At any time there will be ½ v $D_v(Y)$ δA flying towards the rotor in each direction. Total bird flux is

$v\, D_v(Y)\, \delta A$ where v is the bird flight speed
and $D_v(Y)$ is the bird flight density, per m^3, at this height Y

Fig A5-1: Bird flux through small element of rotor disc



Take δA to be a small rectangle of width dX and height dY. If p(X,Y) is the probability of collision for a bird flying through the rotor at point (X,Y), the collision rate through this small element δA at that point is

$$v\ D_v(Y)\, p(X,Y)\, dX\, dY$$

The total collision rate for flights through a single rotor disc (while the turbine is operational) is then obtained by integrating this over the whole area of the disc:

$$\text{Collision rate} = v \int_{\text{Min rotor height}}^{\text{Max rotor height}} D_v(Y) \int_{-\sqrt{(R^2-Y^2)}}^{+\sqrt{(R^2-Y^2)}} p(X,Y)\ dX\, dY \quad \ldots. \qquad \text{(A5-1)}$$

This is equation (7) (paragraph 67) of the main text. The limits $\pm\sqrt{(R^2-Y^2)}$ to the integration over X define the outer limits of the rotor circle, and the limits to the integration over Y are the minimum and maximum rotor heights respectively.

Translate the factors into dimensionless units, within which the rotor has a radius of 1, by using the parameters x = X/R, y=Y/R; thus dX =Rdx, dY=Rdy. Use the dimensionless relative frequency flight height distribution

$$d(y) = R\, D_v(Y)/D_A$$

D_A, the areal bird density, is just the sum of D_v over all flight heights from sea level upwards, ie

$$D_A = \int_{\text{sea level}}^{\infty} D_v(Y)\, dY$$

Hence d(y) is normalised, ie

$$\int_{\text{sea level}}^{\infty} d(y)\, dy = \int R\, D_v(Y)/\, D_A\ (dY\,/\,R) = \int D_v(Y)\, dY\,/\,D_A = 1$$

Using these factors, equation (A5-1) becomes

$$\text{Collision rate} = v\,(D_A/R) \int_{-1}^{+1} d(y) \int_{-\sqrt{(1-y^2)}}^{+\sqrt{(1-y^2)}} p(x,y)\ R\,dx\ R\,dy$$

$$= v\,D_A\,R \int_{-1}^{+1} d(y) \int_{-\sqrt{(1-y^2)}}^{+\sqrt{(1-y^2)}} p(x,y)\,dx\,dy \qquad \text{(A5-2)}$$

which when multiplied by the total number of turbines T, the time birds are active in a month t, and the proportion of time the turbines are operational Q_{op}, is equation (8) (paragraph 70) of the main text.

This can be rearranged in the form of equation (9) (paragraph 70) of the main text, so as to use the same 'flux factor' as in the basic model:

$$\textbf{Collisions} = \underbrace{\boxed{v\,(D_A/2R)\,T\pi R^2\,t}}_{\text{Flux factor}} \times \underbrace{\boxed{(2/\pi) \int_{-1}^{1} \int_{-\sqrt{(1-y^2)}}^{+\sqrt{(1-y^2)}} d(y)\ \ p(x,y)\,dx\,dy}}_{\text{Collision integral}} \times \underbrace{\boxed{Q_{op}}}_{\text{Proportion of time operational}} \qquad \ldots\ \text{(A5-3)}$$

The total count of birds passing through the rotors is given by the same equation but with p(x,y) set to 1, ie such that every bird is counted, as in equation (10) (paragraph 71) of the main text:

$$\textbf{Flux} = \underbrace{\boxed{v\,(D_A/2R)\,T\pi R^2\,t}}_{\text{Flux factor}} \times \underbrace{\boxed{(2/\pi) \int_{-1}^{1} \int_{-\sqrt{(1-y^2)}}^{+\sqrt{(1-y^2)}} d(y)\ \ dx\,dy}}_{\text{Flux integral}} \times \underbrace{\boxed{Q_{op}}}_{\text{Proportion of time operational}} \qquad \ldots\ \text{(A5-4)}$$

Comparison with basic model

In the case where flight heights are assumed to be uniformly distributed across the risk height, ie from lowest to highest point of the rotor, then d(y) is a constant over the range y=-1 to y=+1 and can be taken in front of the integrals. Moreover, if all flights take place within this height band then d(y) takes the value ½, because d(y) is normalised, ie $\int_{-1}^{+1} d(y)dy = 1$. The Flux integral then reduces to

$$(2/\pi)\ (½) \int\!\!\int dx\,dy \;=\; (2/\pi)\ (½)\ (\pi) \;=\; 1$$

as the integral is just the area π of a circle of unit radius. The Collision integral is simply the average of p(x,y) over the area of the disc.

More generally, if a proportion Q_{2R} of flights take place between minimum and maximum rotor heights, and the distribution is uniform within these limits, d(y) takes the value $Q_{2R}/2$, the Flux integral = Q_{2R}, and the Collision integral is Q_{2R} times the average of p(x,y) over the area of the disc.

The average of p(x,y) over the area of the disc is the 'single transit collision risk' in the basic model. Hence equation (A5-3) above becomes

$$\text{Collisions} = \text{Flux factor} \times Q_{2R} \times \text{Single transit collision risk} \times Q_{op}$$

This reproduces equation (5) (paragraph 52) of the main text, which describes the collision rate in the basic model.

Annex 6 – Assessing collision risks for birds on migration

(DRAFT Extension to Collision Risk Guidance – Bill Band March 8 2012)

Birds on migration are often of particular interest in collision risk assessment, as the birds may be coming from or be heading for a distant site with conservation designations which imply special legal responsibilities in avoiding adverse impacts on the bird population. It will therefore be important to understand the impact of a given windfarm on such a bird population if its migration routes are through the windfarm. Report SOSS-05 by BTO on 'Assessing the risk of offshore wind farm development to migratory birds designated as features of UK Special Protection Areas'[xviii] describes the issues and uncertainties involved in such an assessment.

Calculating collision risk for migrants is little different from the process for other birds, and may make use of the Collision Risk Spreadsheet provided with this Guidance[xix]. The main difference arises in estimating the number of migrant birds passing through the windfarm, and how that data is input to the spreadsheet. The data is usually in terms of the number of birds passing through a migration corridor, rather than starting with bird density, as does the normal process in following the Collision Risk Guidance. To facilitate this, an additional sheet 'Migrant Collision Risk' has been added to the suite of spreadsheets, and to make use of this sheet, additional data on migrants is required in the 'Input Data' sheet.

Estimating total bird flux over the migration period

Report SOSS-05 outlines a number of different methods which may potentially be used to estimate the number of birds flying through a windfarm. Each of these leads directly to information on bird flux density F – the number of birds passing through a tall window of unit width (a metre, or a kilometre) during each migration period.

- In the simplest approach, it may be assumed that an entire bird population uses a migratory corridor twice each year. Report SOSS-05 provides data on the total GB (also international) populations of a range of migratory species. Documentation for individual conservation sites often provides information on the typical occupancy of the sites by species during migration. The maps in the SOSS-05 report may then be used to estimate the width W (km) of the corridor used for migration – the 'migratory front', and the assumption may be made that the entire population of N birds passes through this migratory front, with an even distribution across the front. Thus the bird flux density is N/W birds km^{-1}.
- Instead of assuming an even distribution of birds over the migratory front, tracking studies can help indicate the proportion of a bird population likely to cross a wind farm (or different parts of a wind farm) during an average migration period.
- Migrant birds may be counted along with other birds in the snapshot counts in boat-based surveys. As boat based surveys are usually undertaken on a 1- or 2-days a month sample basis, they are generally unsatisfactory as a means of counting birds on migration: whether or not a flock of migrating birds is observed on sampling occasions, and the size of that flock, is likely to be a matter of chance. However, where the sampling is sufficiently frequent it may be used to generate an estimate of the total number of birds flying across the site during the migration period.
- Finally, the flux of migrant birds may be recorded by visual observation from shore or from a sea platform, or by radar, where the observation period covers a high proportion of the possible migration period. Such data will be measured directly in birds crossing an imaginary baseline, eg of 1km length, ie in birds km^{-1} $hour^{-1}$, and can be grossed up for the complete migration period. If this measurement is of birds approaching the baseline from all directions, the result should be multiplied by $\pi/2$ to convert to the equivalent 'perpendicular flux' (see Annex 2). This allows for the fact that the 'tall window' through which birds may pass – defined by the baseline and extending to all heights - presents a reduced cross-sectional area to birds approaching from an oblique angle.

Calculating the Flux factor

Equation (2) (paragraph 37) of this Guidance indicates:

Total number of bird transits =

$$\underbrace{\boxed{v\,(D_A / 2R)\ \ (T\,\pi R^2)\ \ (\text{time active})}}_{\text{flux factor}} \quad x \quad \underbrace{\boxed{Q_{2R}}}_{\text{proportion at risk height}}$$

where v is the bird flight speed, D_A the areal density of birds (ie flying at any altitude), $T\pi R^2$ is the total cross-sectional area of all rotors, and Q_{2R} is the proportion of birds flying at risk height.

The data assembled above for birds on migration is on flux density F - the number of birds passing through a tall window of unit width (a metre, or a kilometre) during each migration period. To relate that to bird density, note that if all birds were flying perpendicularly to the window[6], in any second the number flying through the window would be those within a distance v where v is the flight speed. As the total density of birds, summing over all heights, is D_A birds m^{-2}, the number within distance v of a window one metre wide, and hence about to pass through that in the next second, is just $D_A v$. Over a period of time t, the total number of birds passing through this window is thus $D_A v t$. So, over a full migration period, v D_A (time active) may be replaced in the above equation by F, in units of birds m^{-1}:

Total number of bird transits through rotors

$$\underbrace{\boxed{F\ \ (T\,\pi R^2) / 2R}}_{\text{flux factor}} \quad x \quad \underbrace{\boxed{Q_{2R}}}_{\text{proportion at risk height}} \qquad .. \quad \text{(A6-1)}$$

Note that F is commonly expressed in birds km^{-1} and if so must be divided by 1000 (as the spreadsheet does) for use in this formula.

Flight height

Cook et al 2012[iii], in their SOSS-02 report, have documented the typical flight heights of many bird species at sea, as recorded in wind farm surveys around the UK and elsewhere. However, data on flight heights of birds on migration is patchy, as described by Wright et al in their SOSS-05 report. The proportion of birds on migration $Q_{2R\text{-}m}$ flying at risk height is likely to be different from the Q_{2R} proportion for non-migrants. Table 3 of the SOSS-05 report makes recommendations on the values to be used for $Q_{2R\text{-}m}$ for various species groups, ranging from 100% for raptors to 50% for passerines. For seabirds, divers, gulls and terns use is recommended of the values listed in the SOSS-02 report.

Flight speed

The single transit collision risk uses the bird flight speed as a factor, so if a different flight speed v_m is available for migrating birds, this should be used.

Migration period

Usually, the collision risk of interest will be over a full year, ie over outward and return migration periods. The spreadsheet is arranged, like the ‘Overall collision risk’ sheet, on a monthly basis. If data is available to support a monthly subdivision, the migration fluxes should be apportioned out over the relevant months. As the proportion of time that the wind farm is operational varies from month to month, this is the most accurate approach. However, if the distribution of migration passages over months is not known or highly variable, any two convenient months (eg April and September) may be used as the assumed migration periods.

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[6] in accord with the assumption of perpendicular approach – see paragraph 12 of main text

Calculating collision risk

Subject to the modified approach outlined above in calculating the Flux factor, the calculation of collision risk follows exactly the same methodology as for other birds. As in the usual approach, the Collision Risk Spreadsheet offers three options for calculating collision risk:

(1) Option 1 - using the assumption that flights at risk are evenly distributed across all rotor heights;

(2) Option 2 - ditto, but using the proportion of birds flying at risk height as derived using flight height distribution data; and

(3) Option 3 - making use of the flight height distribution data to calculate risk in each part of the rotor, and summing that risk.

For some species groups, Table 3 of the SOSS-05 report indicates that a simple percentage should be entered for Q_{2R-m} . This indicates that the flight height distributions documented in Cook et al[iii] are not likely to be characteristic of migrating birds. Only Option (1) should be used, unless good data is available indicating the flight height distribution of migrating birds.

For those species groups where Table 3 indicates the Cook et al data may be used, then Options (2) and (3) may be used. As in the usual approach, it is recommended that for these species the calculations for all three Options should be presented, so as to note the effect of taking an assumed flight height distribution into account.

Use of options to take account of flight height distribution

		SOSS-05 Table 3 recommendation	
Calculation option		Percentage	Use figure from Cook et al
Option 1	assume flights uniformly distributed across risk height	●	o
Option 2	use species flight height distribution to generate Q_{2R-m}		●
Option 3	use species flight height distribution in full to calculate collision risk		●

Uncertainties

One of the main uncertainties is likely to be the uncertainty in flight activity, due to uncertainty and year-to-year variation in the number of birds migrating, and in the precise flight corridor used. Realistic assessments should be made, even if this is no more than an expert view, on the limits within which 95% confidence can be assured for the value of flux density input to the model.

Supplementary notes on using the spreadsheet

The ‘Input data’ sheet now includes:

- ‘bird survey data’, which includes data on bird density. This drives the ‘Overall collision risk’ sheet which provides the overall collision risk calculation for the birds described in terms of bird density.
- ‘birds on migration data’, which includes the number of migration passages, the width of the migration corridor, the proportion of migrants flying at risk height, and the proportion of migratory flights which are upwind. This drives the ‘Migrant collision risk’ sheet which provides the collision risk calculation for the birds included in this ‘birds on migration’ block.

Therefore, to avoid double-counting collisions, the ‘bird density’ figures should exclude any migrants for which collision risk is calculated using the ‘Migrant collision risk’ sheet.

The spreadsheet does not add the two collision elements together, as they are likely to be used for different purposes.

The ‘Migrant collision risk’ spreadsheet only differs from the ‘Overall collision risk’ spreadsheet in the data used on flight activity (as above) and in the resulting calculation of the Flux factor. All other parameters – Bird data, Windfarm data, Turbine data and Avoidance rates – are common to both spreadsheets.

Notes on additional input data

Flight activity data – additional for migrants			
Symbol	**Description**	**Units**	**Notes**
N	Bird population	birds	This is the total number of birds migrating through the migration corridor in question. May be subdivided by month if there is data to support that.
W	Width of migration corridor	km	
$Q_{2R\text{-}m}$	Proportion at rotor height	%	Based on recommendations in Table 3 of Report SOSS-05, unless bettered by new data.
	Proportion of migratory flights upwind	%	This is set at 50% by default, but for migration flights it may be appropriate to assume some bias towards downwind.

Annex 7 - Taking account of tidal variation

This section considers how to take account of changing tidal levels in calculating bird collision risks. It is assumed that the extended collision model – taking account of flight height distribution – is being used.

The flight height distribution D(Y) describes the relative density of bird flights at different heights above the sea surface. However (other than for floating wind turbines) the height of the rotor above the sea surface varies with the tide. The issue to be addressed is how to take account of that variation in the calculation of collision risk.

Height above Mean Sea Level

In order to satisfy navigational clearance requirements, turbine hub heights are usually expressed in metres above Highest Astronomical Tide (HAT), which is the maximum sea height theoretically possible, excluding waves and surges and other sea conditions due to meteorological conditions. To use bird flight height distributions, these heights need to be adjusted to the height above actual sea level.

Tidal information is normally presented in metres above Chart Datum (CD), with mean tidal level Z_0 and a tidal variation which oscillates around that level. If turbine height is H relative to HAT, then it becomes $H + (HAT – Z_0)$ relative to mean sea level. Thus a tidal offset has been added to the height:

Tidal offset = Highest Astronomical Tide (HAT) – Mean Sea Level (Z_0)

Typically this offset is in the range 2.5 - 4 metres. A new 'tidal offset' field has been included (in the extended version Mar 2012) as an input field in the 'Input Data' sheet in the Collision Risk Spreadsheet. The extended model then includes this adjustment to rotor heights when making use of a bird flight height distribution.

This adjustment in expressing turbine height can make a significant difference to collision risk, for some species reducing the estimated risk by around 25% to 30%. The size of the change depends on both species and turbine details, depending on the rate at which the flight height distribution curve varies around the minimum height of the rotor.

Allowing for sea level rise

Current predictions on sea level rise due to climate change are described in UKCP09[xx] . By 2060 the predictions are typically for a rise of order 0.25 – 0.3 metres for a global high emissions scenario.

The aim as far as possible should be for bird collision risk assessment to be valid for the full operational period of the project. Therefore the height of the rotor relative to sea level should be reduced by an amount to take account of the likely increase in sea level over the lifetime of the windfarm. It is recommended that this reduction should be of order 0.25 – 0.3 metres. This should be done by amending the tidal offset, so that it becomes

Tidal offset = Highest Astronomical Tide – Mean Sea level – Climate change adjustment

Tidal variation

The above takes account of the height of the rotors above mean sea level, but it does not take account of the variation of the tides. Assuming that the distribution of bird flight heights relative to sea level is independent of the state of the tide (which may not be the case in estuarine or near-shore locations), at times of high tide there will be increased bird density at rotor level, and at low tide decreased. If the flight height distribution were linear with height, then the increases at high tides would exactly offset the decreases at low tides. But flight height distributions are typically highly non-linear, and there is a 'second-derivative' effect, dependent on the degree of curvature in the flight height distribution, with the increases at high tides more than outweighing the decreases at low tides. Only the section of the flight height distribution above rotor minimum height is relevant to collision risk, so it is the curvature of the distribution at those heights which matters.

This non-linear effect – the 'tidal asymmetry correction' - is in general small, but a method for calculating it is set out here.

Calculation of tidal asymmetry correction factor

Take all heights Y as measured with respect to mean sea level. At height Y above mean sea level, the flight density takes the value D(Y) only briefly, twice each tide as the tidal level passes the mean sea level. More generally, the flight density is D(Y-h) when the tide is h metres above mean sea level. The time-averaged flight density is

$$D{\sim}(Y) = \Sigma\, f(h)\, D(Y-h) \qquad \text{(A7-1)}$$

where the sum is over all tidal height bands from lowest to highest, and f(h) is the proportion of time that the sea level is within each height band h.

Figure A7-1 shows the frequency of sea levels f(h) at one site (Cromer in East Anglia), ranging from --2.3m to + 2.3m, and banded within 0.2m height bands. Commercial tidal prediction software is available, such as the POLPRED Offshore tidal computation software available from the National Oceanographic Centre, which can generate such a sea level frequency chart with a high level of accuracy for any point in and around the UK [xxi]. For coastal sites near to ports, the 'Notes on using the spreadsheet' below describe how an approximate frequency chart can be generated, given basic tidal data published by the National Oceanographic Centre[xxii] on their Website, using the 'Sea Level Frequency' spreadsheet provided with this guidance. For Figure A7-1, tide level was calculated at 12 min intervals over 1 year and allocated to 0.2m wide bins. The curve shows symmetrical peaks at around mid-tide levels ±0.9m – not only do all tides pass through that level, but neap tides have their 'high tide' turning point in mid-range. In contrast, relatively few tides approach the maximum of the tidal range. Tides are changing most rapidly as they pass the mean sea level, so the curve is characterised by a dip in the middle.

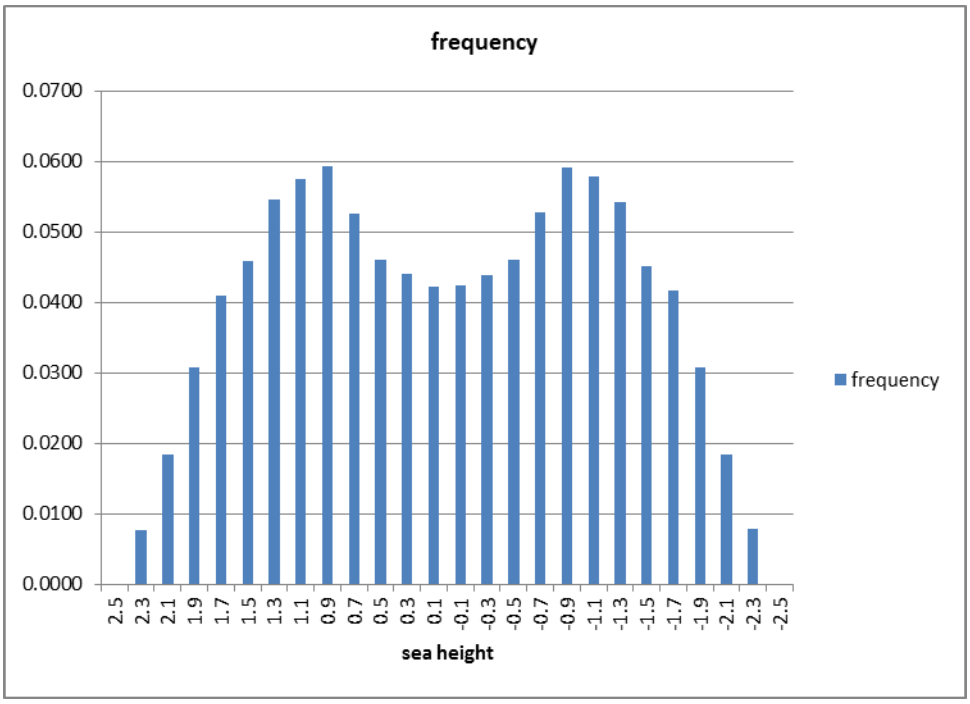

Figure A7-1: Sea level frequency at Cromer, East Anglia

The effect of applying the distribution of tides f(h) to the flight height distribution, ie applying equation (1), is to 'smear' the flight height distribution, drawing from a range within ±2.3 metres (for this site) higher or lower, to yield a time-averaged flight height distribution D~(Y). D~(Y) may now be used in place of the original flight height distribution D(Y) in the collision calculation, pasting it in place of D(Y) as the ‘current data’ in column B of the Flightheight sheet of the Collision Risk Spreadsheet.

Table A7-2 shows sample output for the time-averaged flight density of kittiwake, under the tidal regime at Cromer. The original flight height distribution is from the work by Cook et al SOSS[iii], and the sea level frequency distribution is that in Figure A7-1.

Effects of smearing height distribution

Table A7-1 shows the effects of applying such smearing to flight height data for gannet, kittiwake and fulmar at a sample of five sites around the coast of the UK. The tidal data given is for ports, that for offshore sites may differ.

Table A7-1: Effects of using a tidally-smeared flight height distribution

Base tidal information:

	tidal range (springs)*	HAT	Z_0	tidal offset
Stornoway	4.14	5.53	2.893	2.64
Aberdeen	3.62	4.85	2.557	2.29
Heysham	8.49	10.76	5.176	5.58
Cromer	4.23	5.74	2.920	2.82
Avonmouth	12.27	14.65	6.955	7.69

The data below is calculated using the same 4MW turbine scenario as in the Worked Example: 3 blades, 9.9rpm, 57.5m rotor radius, 80m hub height, 4.21m max chord, 15 degree pitch.

***** tidal range in metres, taken as difference between mean high water spring tides and mean low water spring tides

Gannet	**tidal range**	**collision integral x 10^3**		
		without tidal smear	**with tidal smear**	**change**
Stornoway	4.14	**1.288**	**1.309**	**+1.6%**
Aberdeen	3.62	**1.347**	**1.363**	**+1.2%**
Heysham	8.49	**0.931**	**0.989**	**+6.2%**
Cromer	4.23	**1.259**	**1.284**	**+2.0%**
Avonmouth	12.27	**0.784**	**0.860**	**+9.7%**

bird length 0.94, wingspan 1.72, flight speed 14.9, flight style flapping, 50% upwind

Kittiwake	**tidal range**	**collision integral x 10^3**		
		without tidal smear	**with tidal smear**	**change**
Stornoway	4.14	**1.131**	**1.139**	**+0.7%**
Aberdeen	3.62	**1.176**	**1.184**	**+0.7%**
Heysham	8.49	**0.816**	**0.847**	**+3.8%**
Cromer	4.23	**1.108**	**1.118**	**+0.9%**
Avonmouth	12.27	**0.649**	**0.697**	**+7.4%**

bird length 0.39, wingspan 1.08, flight speed 14.9, flight style flapping, 50% upwind

Fulmar	**tidal range**	**collision integral x 10^3**		
		without tidal smear	**with tidal smear**	**change**
Stornoway	4.14	**0.059**	**0.059**	-
Aberdeen	3.62	**0.061**	**0.060**	-
Heysham	8.49	**0.049**	**0.049**	-
Cromer	4.23	**0.058**	**0.058**	-
Avonmouth	12.27	**0.043**	**0.043**	-

bird length 0.48, wingspan 1.07, flight speed 14.9, flight style flapping, 50% upwind

The effects of the smearing are highly dependent on the species. This is to be expected as the concave-upwards curvature of the flight height distribution, in the lower height range of range of the rotors, differs markedly for different species. Both gannet and kittiwake distributions have strong curvature in this height range, while the fulmar height distribution has flattened off at these heights above the sea surface.

The effects also depend on the tidal range, reflecting in particular the separation of the two peaks in the sea level distribution curve. The effects are generally small (less than 5% of collision risk) except at the two high-tidal range sites, Heysham and Avonmouth. The latter has among the most extreme tides in the UK. For gannet at Avonmouth, the effect is 9.7% of collision risk. That means that the collision risk is increased by 9.7% due to the asymmetry of the flight height distribution. (It should be stressed that these are proportional changes – ie if predicted collisions were 50 per month this effect would raise that estimate to 54.85.)

While for most potential offshore windfarm sites such effects may be judged minimal, at sites with tidal range in excess of 5 metres it may be sufficiently significant to warrant incorporation of use of a ‘tidal asymmetry correction’.

The correction increases with tidal range, more than just linearly. As it depends on the curvature of the flight distribution curve, ie its second derivative, it should be expected to depend on the square of the breadth of the distribution (which is characterised by the tidal range). Making this assumption and using the data in Table A7-1 yields very approximate ‘rule of thumb’ factors:

Correction factors (percentage adjustment of collision risk):

Gannet	0.08 x (tidal range)^2
Kittiwake	0.05 x (tidal range)^2
Fulmar	0

Thus, for gannet at Cromer where the collision integral is 1.259 x 10^{-3}, one should apply a tidal asymmetry correction of 0.08 * (4.23^2) = 1.43%, raising the collision integral to

Adjusted collision integral = 1.259 x 1.0143 x 10^{-3} = 1.277 x 10^{-3}

Where species other than the above three are involved, there will be a need to undertake a comparable analysis to establish the 'rule of thumb' factors.

It should be noted that these rule-of-thumb factors have been evaluated for one particular (fictitious) turbine model. However it may be expected to apply to any large turbine with a similar height clearance above the sea surface: the crucial factor is the degree of curvature of the flight height distribution curve for the species in question, in the vicinity of the lower reaches of the rotor.

Conclusion

Given the additional data processing required to take account of this adjustment, it is not recommended that the effects of tidal asymmetry should be taken into account routinely in collision risk assessment. However, where the tidal range exceeds 5m, the adjustment is significant enough to warrant use of a correction, using the 'rule of thumb' factor if the species is one for which such a factor has been established, and if not, by undertaking the analysis outlined above.

Summary of recommendations

The following recommendations only apply to turbines which are fixed relative to the seabed (ie not floating turbines)

1. For the purposes of collision risk assessment, turbine hub and blade heights should be adjusted so they are relative to mean sea level, by including the height of Highest Astronomical Tide above Mean Sea Level as a 'tidal offset'.
2. A reduction of around 0.25 – 0.3 metres in that offset should be made to allow for the likelihood of increasing sea levels over the period to 2060.
3. The skewed distribution of seabird flight heights means that tidal variation affects bird densities in an asymmetric way, ie the increases at higher sea levels are greater than the decreases at lower sea levels. The changes to collision risk are typically small (<5%). However at sites with a high tidal range (> 5 metres) the effects for some species may be significant. A 'rule of thumb' correction factor is provided for gannet, kittiwake and fulmar. For other species there will be a need to apply the methodology outlined above to establish the correction. Tools are provided in spreadsheet form to assist this process.

Spreadsheet support

A spreadsheet 'Tidal smear' is provided which contains a routine to 'tidally smear' data, ie using D(Y) as input and calculating D~(Y) as output.

Two ancillary spreadsheets 'Tidal height' and 'Sea level frequency' are also included which enable an approximate sea level frequency distribution to be generated for near-coastal sites, if software such as POLPRED is not available.

These are intended for users conversant with spreadsheets and with an understanding of the transformation required; the process involves cutting and pasting data between worksheets. Notes on using these spreadsheets are provided below.

Notes on using the spreadsheets

These spreadsheet tools involve some cut-and-pasting and use of macros so should be undertaken by someone with adequate spreadsheet skills. There are three sheets in the 'Tidal variation' workbook.

Tidal height uses published tidal data to generate tidal predictions. Tidal data for ports around the UK is published by the National Oceanography Centre at http://www.pol.ac.uk/ntslf/tidalp.html.

Sea level height, ignoring any meteorological effects such as surges or waves, is governed by a series of cycles with different frequencies, relating to the position of the moon and sun in relation to the earth and the location in question. Sea level height is given, where t is the time in hours elapsed from a reference start time, by the formula

$$\Sigma H_i \cos(\sigma_i \; t - g_i)$$

where for each cyclical component i, H_i is the amplitude, σ_i its angular frequency, and g_i its phase.

Based on observations over the period 1989 – 2007, the National Oceanography Centre publishes information on amplitude H_i and phase g_i – the 'harmonic constants' - for the four largest cyclical constituents, termed M_2, S_2, K_1 and O_1 respectively. Their associated frequencies σ_i are drawn from a description of the Doodson numbers http://en.wikipedia.org/wiki/Arthur_Thomas_Doodson (see w0, w1, w2 and w3 in the sample programme).

The spreadsheet uses these four principal harmonic constants and their associated phases and frequencies to calculate sea level at times t which increases in steps in successive rows. 'Step' sets the period in hours between successive rows.

This tidal calculation is not used directly, but provides a graph showing alternation of tides and springs and neap tides, which will help explain the shape of the sea level frequency distributions produced next.

It is stressed that this is a very approximate tidal series. More precise prediction involves the addition of a long series of harmonic components, not just four. If greater precision is required, then tidal predictions from various commercial systems may be used. However, these four harmonic components are sufficient to generate the broad pattern of spring and neap tides, and the daily alternation of tide heights, which should be adequate as a basis for a sea level frequency distribution.

Sea level frequency runs exactly the same routine as a time series. As it runs, it categorises each output in a tide height bin, building up a frequency distribution of sea level heights. As input it requires the same table of tidal constants for the location in question as the Tidal height sheet. The programme is initiated as a macro 'Sealevel frequency' - click on 'Developer' then 'Macros' and 'Run' the macro 'Sealevelfrequency'. The programme requires three further inputs:

bin width – use 0.2 for east coast or north coast, use 0.4 or 0.5 for estuarine locations. The distribution matrix is 13 times this bin width both + and -, so 0.2 bin width runs from -2.6m to + 2.6; 0.5 bin width runs from -6.5m to +6.5m.

interval – a value of 0.2 (meaning 0.2 hours or 12 minutes) seems satisfactory,. remembering that the aim is to sample sea level heights.

number of data points – the system should be tested with only 100 or 1000 points, but once working, run it for 45000 which at 12 minute intervals is a little over a year.

The output is a sea level frequency table, which is then normalised in the next column (divided by the total to give a frequency set which adds to 1). This normalised frequency distribution can then be copied then pasted into the Tidal smear spreadsheet.

Tidal smear uses the sea level frequency data as input, and applies it to the flight height distribution (eg that in the SOSS report by Cook et al[iii]), as described above, to produce a 'smeared' output, in which D~(Y) is the time-averaged value of the bird density at height Y. The programme uses two named ranges 'tidefreq' which contains the sea frequency data, and 'gannetdata' (for example) which contains the bird flight height distribution. Both ranges must be two columns wide, the left one with the height in metres, and the right one with the normalised frequency data. The ranges must start at the first data point (ie not including column titles). The sea level data ranges must be 26 rows deep, and the bird data tables 150 rows deep. The output column then uses the function 'tidesmear' to compute the result for each height y. Note that if the sea level frequency distribution runs from say -5m to +5m, then at height y metres the programme will draw from distribution data from y-5 to y+5 metres. So omit the output formula for heights 0-5m and 145-150 metres to avoid the programme going out of range.

The output tide-smeared distribution may then be copied and pasted into the main Collision Risk Assessment spreadsheet, in the 'Flightheights' sheet.

Table A7-2: Sample output of tide-smeared flight distribution Kittiwake, using tides at Cromer

height (m)	original flight height distribution	tide-smeared flight height distribution
0	0.08571	
1	0.07850	
2	0.07175	
3	0.06526	
4	0.05987	0.06039
5	0.05499	0.05548
6	0.05095	0.05100
7	0.04680	0.04686
8	0.04263	0.04299
9	0.03907	0.03938
10	0.03590	0.03606
11	0.03293	0.03302
12	0.02997	0.03022
13	0.02747	0.02763
14	0.02505	0.02530
15	0.02305	0.02317
16	0.02118	0.02122
17	0.01929	0.01940
18	0.01765	0.01760
19	0.01587	0.01584
20	0.01398	0.01419
21	0.01247	0.01264
22	0.01115	0.01127
23	0.00999	0.01009
24	0.00895	0.00902
25	0.00801	0.00805
26	0.00710	0.00718
27	0.00631	0.00639
28	0.00565	0.00568
29	0.00496	0.00504
30	0.00444	0.00447
31	0.00391	0.00395
32	0.00345	0.00350
33	0.00305	0.00309
34	0.00271	0.00273
35	0.00238	0.00242
36	0.00213	0.00214
37	0.00185	0.00189
38	0.00164	0.00166
39	0.00145	0.00147
40	0.00128	0.00130
41	0.00113	0.00115
42	0.00101	0.00103
43	0.00092	0.00092
44	0.00081	0.00081
45	0.00071	0.00072
46	0.00063	0.00063
47	0.00055	0.00056
48	0.00048	0.00049
49	0.00042	0.00043
50	0.00038	0.00038
51	0.00033	0.00034
52	0.00030	0.00030
53	0.00026	0.00027
54	0.00023	0.00024
55	0.00021	0.00021
56	0.00018	0.00019
57	0.00016	0.00017
58	0.00015	0.00015
59	0.00013	0.00013
60	0.00012	0.00012
61	0.00010	0.00010
62	0.00009	0.00009
63	0.00008	0.00008
64	0.00007	0.00007
65	0.00007	0.00007
66	0.00006	0.00006
67	0.00005	0.00005
68	0.00005	0.00005
69	0.00004	0.00004
70	0.00004	0.00004
71	0.00003	0.00003
72	0.00003	0.00003
73	0.00003	0.00003
74	0.00003	0.00003
75	0.00002	0.00002
	etc	

This bracket shows the range of data drawn upon in calculating the smeared distribution for height 4m

Annex 8 Notes on spreadsheet Visual Basic functions

The functionality of Sheet 4 (Extended model) is entirely based on computations programmed as user-defined functions in Visual Basic. Macros must be enabled. The programme code may be viewed using the ‘Developer’ tab and ‘Visual Basic’ icon to view ‘Module 1’. The user-defined functions are as follows:

interpolate (N,a,y)

Assumes a set of points and associated values in a two-column named range A. It compares y with the set of points and performs a linear interpolation to provide an appropriate intermediate value. It is used twice in the programme: once to extract intermediate values of the chord c/C, using the data table in the Single Transit Risk sheet; and to extract appropriate values of bird density using the table of flight height data in sheet ‘Flightheights’. If N is greater than the length of the named range A, an error message appears, but N is allowed to be less than the range length.

pcoll (r, φ, updown)

Calculates the single transit collision risk at point (r, φ) in the rotor, using equation (3). The parameter updown may be either ‘up’ or ‘down’. r is in dimensionless form, ie r= actual radius/rotor radius. φ is in degrees, where φ=0 is the top of the rotor.

pcoll_rav (r, updown)

Calculates the average of pcoll (r, φ, updown) over angles φ, in 10-degree increments.

pcollxy (x,y,updown)

Calculates the single transit collision risk at point (x,y) in the rotor, by converting (x,y) to (r,φ) and calling pcoll (r, φ, updown) . x and y are in dimensionless form ie x=X/R, y=Y/R (see Fig 7).

xareasum (y)

Calculates the length of a horizontal chord at height y

xrisksum (y,xinc,updown)

Integrates the collision risk times bird density along a horizontal chord at height y, using the interpolate function to evaluate the bird density at this height. The parameter xinc is the increment used for integration along the x-axis.

ydistsum (xinc,yinc,updown,flag)

When flag=0, integrates the collision risk times bird density over all heights from y= -1 to y= +1. This is the double integral within the ‘collision integral’ box in equation (9). The Collision integral is (2/π) ydistsum.

When flag=1, integrates bird density only over all heights from y=-1 to y=+1. This is the double integral within the ‘Flux integral’ box in equation (10). The Flux integral is (2/π) ydistsum .

The parameter yinc is the increment used for integration along the y axis.

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[xxii] National Oceanography Centre http://www.pol.ac.uk/ntslf/tidalp.html

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The avoidance rates of collision between birds and offshore turbines

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EXECUTIVE SUMMARY

1. The selection of appropriate avoidance rates for use in collision risk models at offshore windfarms is often a key part of the Environmental Impact Assessment process. Ideally, these avoidance rates should reflect the behavioural responses of birds to turbines. However, they are often used as a 'fudge-factor' to incorporate aspects of model error. The situation is further complicated by a lack of data for marine birds and offshore windfarms. As a consequence, present guidance is based on values that have been derived for terrestrial species at onshore windfarms. This study reviewed data that have been collected from offshore windfarms and consider how they can be used to derive appropriate avoidance rates for use in the offshore environment. Aims of the study were five-fold:

   - To produce definitions for the types and scales of avoidance;
   - To review current use of avoidance rates;
   - To review and critique existing avoidance behaviour studies and any derived rates;
   - To provide summary avoidance rates and a total avoidance rate for each priority species/species group based on the evidence available at present;
   - To undertake an assessment of the sensitivity of the conclusions reached to inputs and conditions under which they were collected.

   The study focussed on five priority species – northern gannet, black-legged kittiwake, lesser black-backed gull, herring gull and great black-backed gull – whose behaviour and distribution make them particularly prone to collision with offshore turbines.

Definitions (section 3)

2. A key hurdle to defining appropriate avoidance rates for use in the offshore environment has been a lack of clear, agreed definitions of avoidance behaviour. Therefore, the first step of this review was to define the different scales at which avoidance behaviour may occur. Three categories of behaviour were initially defined – macro-, meso- and micro. Micro-avoidance refers to 'last-second action taken to avoid collision, which is considered to occur within 10 m of the turbine rotor blades. Meso-responses reflect all responses to individual turbines occurring between the base of each turbine and the windfarm perimeter (defined as 500 m from the base of the outermost turbines). Macro-responses reflect all behavioural responses to the presence of the windfarm that occur at distances greater than 500 m from the base of the outermost turbines. Avoidance rates are typically derived by comparing observed collision rates to the number of collisions that would be expected in the absence of avoidance behaviour, considering all bird movements within the perimeter of the windfarm. Consequently, calculations do not usually consider whether any avoidance action takes place at the meso- or micro-scale. It was thus also necessary to consider a fourth category, within-windfarm avoidance, which combines micro-avoidance and meso-responses.

Current use of avoidance rates (section 4)

3. The avoidance rates used with collision risk models have shown substantial variation over time. Initially, very high values, often based on incorrect interpretations of data, were used. Since the earliest environmental impact assessments, there has been a broad tendency to follow standard guidance with avoidance rates of 0.95 and more recently, 0.98 used. However, in light of recent evidence from both on- and offshore windfarms these values are coming under increasing scrutiny from developers and their consultants.

Macro-responses (section 5.1)

4. As with micro-avoidance and meso-responses, the evidence for macro-responses to the presence of a windfarm was typically inconsistent for gulls. Studies designed to look at potential displacement effects reported both evidence for attraction and for displacement and others no significant response at the limited number of sites which were available for consideration. Thus, for gulls, the balance of evidence suggests a macro-response of 0 (i.e. no attraction to or avoidance of the windfarm). However, the response of northern gannet to the presence of windfarms appeared to be more consistent, with strong avoidance evident at several sites, although again it was not always clear whether the macro-response was a result of barrier effects or displacement. Based on the evidence currently available, it is suggested that a macro-response rate of 0.64 is a suitable precautionary value for northern gannet.

Micro-avoidance (section 5.2) and meso-responses (section 5.3)

5. Data for micro-avoidance and meso-responses were extremely limited. No clear and consistent patterns were evident for any of our five priority species. For this reason, it was not possible to derive micro-avoidance or meso-response rates for these species.

Within-windfarm avoidance (section 5.4)

6. A total of 20 sites were identified as having sufficient data to derive within-windfarm avoidance rates by comparing observed collision rates to those expected in the absence of avoidance behaviour. Of these, nine were considered to have data of sufficient quality to estimate robust within-windfarm avoidance rates to be calculated using the Band (2012) collision risk model. Within-windfarm avoidance rates were derived for use with both the basic Band model (Options 1 and 2), that assumes that birds are distributed evenly within the rotor-swept area of a turbine, and with the extended Band Model (option 3) that uses a continuous flight height distribution to estimate collision risk at different points within the turbines rotor-swept area. Based on these data within-windfarm avoidance rates of 0.9959 (± 0.0006 SD) and 0.9908 (± 0.0012 SD) were derived for herring gull for use with the basic Band model and extended Band model respectively. Similarly, within-windfarm avoidance rates of 0.9956 (± 0.0004 SD) and 0.9898 (± 0.0009 SD) were derived for large gulls for use with the basic Band model and extended Band

model respectively, and rates of 0.9921 (± 0.0015 SD) and 0.9027 (± 0.0068 SD) derived for small gulls also for use with the basic Band model and extended Band model respectively. Within-windfarm avoidance rates of 0.9893 (± 0.0007 SD) for the basic Band model and 0.9672 (± 0.0040 SD) for the extended Band model were derived for all gulls. Insufficient data were available to calculate a within-windfarm avoidance rate for northern gannet. (Note, where we report the standard deviation around the derived within windfarm avoidance rates, this relates variability between sites and not to uncertainty in the model input parameters. Estimating the contribution of the model input parameters to the uncertainty associated with the derived avoidance rates requires a more detailed understanding of the real range of values associated with each parameter than is available currently.)

Sensitivity of derived within-windfarm avoidance rates (section 6)

7. The sensitivity of within-windfarm avoidance rate values to model input parameters was also assessed and it was found that the final derived values were most sensitive to assumptions about the proportion of birds at collision risk height. However, it was also found that sensitivity to input parameters declined as the number of flights through a windfarm increased.

Recommended total avoidance rates (section 7)

8. Whilst we have estimated within-windfarm avoidance rates to four decimal places, current guidance from SNH is that expressing avoidance rates to more than three decimal places is unwarranted (SNH 2013). Given the inherent uncertainty in the data we feel that this is a sensible approach to apply to total avoidance rates. For this reason, we round within-windfarm avoidance rates down to three decimal places when deriving recommended total avoidance rates. For gulls the balance of evidence suggests a macro-response of 0 (i.e. no consistent attraction to or avoidance of the windfarm). Consequently, the recommended total avoidance rates for these species are equal to the within-windfarm avoidance rates. Therefore, avoidance rates of 0.995 for herring gull, lesser black-backed gull and great black-backed gull and 0.992 for black-legged kittiwake are recommended for use with the basic Band model. Based on the evidence available, it is suggested that the total avoidance rate for northern gannet is unlikely to be lower than that for all gulls. Assuming a macro-avoidance rate of 0.64, this would reflect a within windfarm avoidance rate of 0.9703. We acknowledge that this is precautionary, but in the absence of more species-specific data, we feel it is appropriate. Hence, an avoidance rate of 0.989 for northern gannet is recommended when using the basic Band model. For the extended Band model, avoidance rates of 0.990 for herring gull and 0.989 for lesser black-backed gull and great black-backed gull were recommended. Based on the evidence available at present, it was not possible to recommend an avoidance rate for use with the extended model for either black-legged kittiwake or northern gannet.

Transferability of avoidance rates between models (section 8)

9. Whilst the basic and extended Band models are the most widely used collision risk models at present, there are a number of alternatives. Based on our assessment of the alternative models which we were able to obtain descriptions of, the definitions and values we present in this report are likely to be broadly applicable to other models.

RECOMMENDATIONS AND LIMITATIONS

Definitions (Section 3)

- **Micro avoidance** should be defined as ‘last-second’ action taken to avoid collision, occurring within 10 m of the rotor blades.

- **Meso-response** should be defined as all behavioural responses, including attraction, in flight deflection and functional habitat loss, to the presence of a turbine occurring more than 10 m from the rotor blades and within the perimeter of the windfarm (500 m from the base of the outermost turbines).

- **Macro-response** should be defined as all behavioural responses, including attraction, displacement, and barrier effects, to the presence of a windfarm occurring beyond its perimeter (> 500 m from the base of the outermost turbines).

- Where an avoidance rate has been derived by comparing observed collisions to those expected in the absence of avoidance, this should be referred to as **within-windfarm avoidance**, it is a combination of meso-responses and micro-avoidance.

Recommended avoidance rates

- A **macro-avoidance rate of 0.64** is recommended for **northern gannet** (section 5.4). However, no data were available to derive a within-windfarm avoidance rate for this species (section 5.3). Based on the evidence available, there is no reason to suppose that the total avoidance rates for **northern gannet** should be less than those for all gulls. A **total avoidance rate** of **0.989** is thus recommended for use with the basic Band (2012) collision risk model. This would reflect a within windfarm avoidance rate of 0.970. We acknowledge that this is precautionary, but in the absence of more species-specific data, we feel it is appropriate. It was not possible to recommend an avoidance rate for use with the extended Band (2012) collision risk model based on the evidence available at present.

- No consistent evidence of macro-avoidance was found for **black-legged kittiwake** (section 5.4). As it was not possible to derive species-specific within-windfarm avoidance rates for **black-legged kittiwake**, the within-windfarm rates derived for the small gulls group were considered appropriate for use for this species (section 5.3). A **total avoidance rate** of **0.992** is thus recommended for the basic Band model. It was not possible to recommend an avoidance rate for use with the extended Band (2012) collision risk model based on the evidence available at present.

- No consistent evidence of macro-avoidance was found for **lesser black-backed gull** (section 5.4). Whilst it was possible to derive species-specific within-windfarm avoidance rates for lesser black-backed gull, these were based on limited data and thus the within-windfarm avoidance rates for large gulls were considered more appropriate for use for this species (section 5.3). A **total avoidance rate** of **0.995** is thus recommended for use with the basic Band model and a **total avoidance rate** of **0.989** for use with the extended Band model (section 7).

- No consistent evidence of macro-avoidance was found for **herring gull** (section 5.4) and thus total avoidance rates reflect species-specific within-windfarm avoidance rates. A species-specific **total avoidance rate** of **0.995** is thus recommended for use with the basic Band model and a **total avoidance rate** of **0.990** for use with the extended Band model (section 7).

- No consistent evidence of macro-avoidance was found for **great black-backed gull** (section 5.4). As it was not possible to derive species-specific within-windfarm avoidance rates for **great black-backed gull**, the within-windfarm rates derived for the large gulls group were considered appropriate for use for this species (section 5.3). A **total avoidance rate** of **0.995** is thus recommended for the basic Band model and a **total avoidance rate** of **0.989** for use with the extended Band model (section 7).

- Given the multiple ways in which data can be interpreted, it is vital that future studies in which avoidance rates are derived are completely transparent and present their workings as a step-by-step process. Appendix 7 enables the reader to go back to the original source material and fully understand how the values presented in this report have been derived. This also offers an indication of the uncertainty present in the derived values.

- Based on the available data, it was not possible to derive species-specific avoidance rates for three of the five priority species. Of particular concern is the lack of within-windfarm avoidance data for northern gannet given that it is taxonomically distinct from the other four species, all of which are gulls. Future projects should focus on collecting data for northern gannet as a priority. Given the limitations in the data we identified for macro-responses, especially for gulls, there is also a need to collect further data on barrier effects and displacement/attraction rates.

1. INTRODUCTION

The Scottish Government has a target for 100% of Scottish demand for electricity to be met from renewables by 2020 by creating a portfolio of both onshore and offshore technologies (Marine Scotland 2011). However, concern over the environmental impacts of these developments in the UK, and in particular the risk of birds colliding with wind turbines, has contributed to the delay and cancellation of some projects. In order to quantify the risk of birds colliding with wind turbines, a number of collision risk models have been developed (Band 2012, Smales *et al*. 2013). These include an update to the Scottish Natural Heritage (SNH) collision risk model, originally developed for onshore windfarms (Band 2000, Band *et al.* 2007), redeveloped to better reflect data collected in relation to impact assessments for offshore windfarms (Band 2012). This work was undertaken as part of one of the projects undertaken through the Strategic Ornithological Support Service (SOSS) programme, a joint initiative involving industry, statutory nature conservation bodies (SNCBs) and the RSPB. These models combine a series of parameters describing the turbine design and operation with estimates of a bird's size and behaviour in order to predict the number of birds that would be expected to collide with a turbine over a given time period. Of these parameters, detailed analysis has suggested that these models are highly sensitive to variation in the avoidance rate, the proportion of birds which take action to avoid colliding with a turbine (Chamberlain *et al.* 2005, 2006). Despite this, there has been relatively little empirical evidence put forward to support avoidance rates for offshore windfarms, which are likely to vary according to species and weather conditions, in particular visibility.

Whilst avoidance rates can be determined from observed mortality rates or actual observations of birds' behaviour, defining robust values for use in collision risk modelling can be extremely challenging. However, there are concerns that avoidance rates derived from observed mortality rates may act as a 'fudge-factor', incorporating observer biases and model error, as opposed to the actual behaviour of the birds (May *et al.* 2010, Douglas *et al.* 2012). Current guidance from SNH (2010) is that, in the absence of species-specific empirical data, a default avoidance rate of 0.98 should be used for most species in onshore windfarm assessments and this value has been widely used in the offshore environment as well. However, in light of recent evidence (e.g. Everaert & Stienen 2007, Krijgsveld *et al.* 2011) the validity of this approach has been questioned and concerns have been raised by developers that it will lead to an over-estimate of the likely number of collisions (Moray Offshore Renewables Limited 2012, Trinder 2012, Smartwind/Forewind 2013) and, as a consequence, potentially contribute to the delay and cancellation of key projects. In a policy environment where there is limited evidence on which to base decisions it is important to reflect uncertainty, but not to apply unrealistic levels of precaution which will make it difficult to reach informed decisions about where and where not to build windfarms.

There is a strong need for a consensus on the appropriateness of recommended avoidance rate values given the influence they have on collision estimates and, therefore, consenting decisions. However, at present, there is a lack of clarity over the interpretation of studies of avoidance behaviour and the applicability of the resultant avoidance rates to different collision risk models, study sites and species. As a result, details presented in reviews of avoidance behaviour of birds in the marine environment (e.g. Maclean *et al.* 2009, Cook *et al.* 2012) have been subject to confusion. A key reason for this is the lack of consistency in the terminology applied to different spatial scales of avoidance, and the widely varying interpretation of the types of avoidance behaviour occurring. There is therefore, an urgent need for a review of avoidance behaviour in offshore windfarms in order to provide a clear appraisal of the existing evidence base, provide a robust critique of the data available with which to refine recommendations on avoidance rates and offer clear guidance as to how they should be used in future collision risk modelling scenarios. Whilst the focus of this review will be on collision risk modelling and species relevant to the UK context, it will draw on evidence from Europe and beyond.

This work aims to reduce the current level of uncertainty around appropriate avoidance rates for seabird species within collision risk modelling by providing a thorough review of the existing evidence base. The scope of this review is broader than those previously undertaken (e.g. Cook *et al.* 2012) and includes quantitative and qualitative analyses of the data identified with a view to identifying representative avoidance rates for five priority species – northern gannet, black-legged kittiwake, lesser black-backed gull, herring gull and great black-backed gull. The review identifies current knowledge gaps and aims to ensure that future strategic work is targeted at addressing the most appropriate issues. Due to the sensitivity of the work and the importance of its conclusions, the work has been overseen by a steering group of key stakeholders and experts, with a view to gaining widespread acceptance of its conclusions.

2. OBJECTIVES

2.1 Produce definitions for the types and scales of avoidance rates that will be used throughout the review document

It is important to make a distinction between avoidance rates, as used in collision risk models, and avoidance behaviour. Avoidance behaviour refers specifically to the behavioural response of birds to wind turbines. However, at present, in addition to accounting for avoidance behaviour, avoidance rates are often used as a ‘fudge-factor’ to account for error in the model itself and in its input parameters (see May *et al.* 2010, Douglas *et al.* 2012). Whilst SOSS guidance (Band 2012) sets out how these uncertainties should be accounted for in the collision risk modelling process, in practice, this is rarely done. The purpose of this review is to identify suitable avoidance rates for use in collision risk models; these rates will be informed, where appropriate, by recorded estimates of avoidance behaviour.

A lack of clear, working definitions for different avoidance rates has hampered attempts to come up with standardised measures. Present definitions of avoidance rates rely on an ability to collect empirical data with which to compare predicted and observed collision rates (SNH 2010). As this is impractical for the offshore environment, Band (2012) proposes combining estimates of micro- (or near-field) avoidance, where a bird takes action to avoid collision at a point close to the turbine, and macro- (or far-field) avoidance, where a bird takes action to avoid collision at a point distant from the turbine, to generate an estimate of total avoidance. However, the empirical data underpinning such definitions is currently inconsistent and difficult to interpret.

A key problem is often the lack of detail over what spatial scale data have been collected at. For example, radar monitoring has shown that birds may take action to avoid entering a windfarm at distances of up to 6 km (Christensen *et al.* 2004), far further than could be observed by eye. As a result, by relying on visual observations, avoidance rates may be under-estimated as a significant proportion of birds will have taken action to avoid the windfarm before they are visible. Similarly, at present, it is not possible to identify birds to species level on the basis of radar echoes; consequently, by relying on radar, it will not be possible to derive species-specific avoidance rates. This is further complicated by evidence that avoidance can occur in a three-dimensional space, with horizontal avoidance, where a bird alters its heading to avoid collision, and vertical avoidance, where a bird alters its altitude to avoid collision (Krijgsveld *et al.* 2011, Plonczkier & Simms 2012). Such alterations may be relatively subtle and difficult to detect by eye. Where radar is utilised to monitor movements in response to turbines, it requires the use of both horizontal and vertical radar. Evidence describing three-dimensional avoidance behaviour, if it exists, is likely to be extremely limited. In defining different avoidance behaviours, the review therefore gives careful consideration to the methodologies used to collect the necessary data.

Wind turbines are most typically in the order of seven rotor diameters apart (Meyers & Meneveau 2012), based on existing turbine designs, this may vary from 480 m to 1.5 km, depending on the capacity used. Given the variable distances between turbines and the difficulties in obtaining consistent estimates of avoidance behaviour over the relevant spatial scales, the review considers whether it is possible to define micro-and macro-avoidance with reference to distance to turbines, or whether a more pragmatic approach, basing definitions on whether a bird is inside or outside a windfarm would be more appropriate. The review considers whether these definitions are appropriate to all species and groups, or whether a more flexible approach is necessary. This may depend on what evidence is available for different species. For example, avoidance rates for terns have often been derived from observed collision rates (Everaert 2008), whilst for other species, such as northern gannets, avoidance rates may be more reliant on radar data (Krijgsveld *et al.* 2011). The review then considers evidence for avoidance behaviour occurring over horizontal and vertical planes.

The review provides clear and concise definitions for micro-horizontal avoidance, micro-vertical avoidance, macro-horizontal avoidance and macro-vertical avoidance. Definitions are produced based on the behaviour of the birds as opposed to the requirements of a model and offer guidance about how final values can be adapted for use in different models.

Defining the different forms of avoidance behaviour represents a major step forward in collision risk modelling. These definitions are central to the rest of the project, and, as such, have been agreed through discussion with the project steering group of key stakeholders and experts.

2.2 Review the current use of avoidance rates

In order to provide context to this work, it is important to consider how avoidance rates are currently used. With this in mind, the review considers published EIAs and identifies what avoidance rates have been used within the collision risk modelling process and what justifications have been put forward for their selection. This will help us determine how consistently existing guidance has been interpreted and applied, and help refine future guidance in order to minimise discrepancies in its application.

2.3 Review and critique existing avoidance behaviour studies and any derived rates

Avoidance rates have been derived from both observed mortality rates and actual observations of birds' behaviour (Cook *et al.* 2012, Trinder 2012, Moray Offshore Renewables Limited 2012, Smartwind/Forewind 2013, Everaert 2014). In Belgium, at Zeebrugge port breakwater, onshore collision rates in terns and gulls have been used to derive avoidance rates based on recorded movement patterns and assumptions about turbine design (Everaert & Stienen 2007 Moray Offshore Renewables Ltd. 2012, Everaert 2014). However, the difficulties in directly recording collisions in the marine environment mean that studies of avoidance at offshore windfarms have relied on observing behaviour (Desholm *et al.* 2006, Blew *et al.* 2008, Krijgsveld *et al.* 2011). These studies have varied both in the species they have investigated, and also in the potential form of avoidance behaviour reported.

Recognising that appropriate data may be extremely limited, we initially take a broad approach to our review, reviewing evidence for avoidance behaviour in marine birds generally. We demonstrate how this evidence relates to the definitions set out in the previous section of the report. Having done this, we assess whether sufficient evidence exists to draw conclusions about avoidance behaviour in five priority species – northern gannet, black-legged kittiwake, lesser black-backed gull, herring gull and great black-backed gull. If this is not possible, we will consider how to combine evidence within groups of species, on the basis of the ecology of the species concerned. Where this is necessary, we clearly state which species are in each group.

In order to make an assessment of the level of confidence in the reported avoidance rates for each species or species group, we make a detailed qualitative critique of each study. Key questions include:

i. How have avoidance rates been derived?

We consider first whether the avoidance rates reported have been determined from observed mortality rates or actual observations of birds' behaviour. The data collection methods used are summarised, and the limitations of each method discussed. Where avoidance rates have been back-calculated from observed collisions at reference windfarms, they may incorporate error associated with model input parameters including population estimates, flight heights and turbine operational characteristics in addition to the actual avoidance behaviour of the birds. In contrast, direct observations of birds' behaviour in relation to turbines will not incorporate model error. However, these observations may still need careful interpretation given methodological constraints over how data may be collected, for example, the distances over which birds can be observed in comparison to the distances over which they may take avoidance action.

ii. How comparable are the different datasets?

Avoidance rates based on behaviour have typically been derived from a series of visual or radar observations (Desholm & Kahlert 2005, Blew *et al.* 2008), or through a combination of both (Krijgsveld *et al.* 2011, Plonczkier & Simms 2012). The range of distances over which data can be collected varies markedly between these platforms (Cook *et al.* 2012) and it is important to consider whether estimates – particularly of macro-avoidance – are comparable between different studies.

It is also important to consider how and when data have been collected. For example, visual observations from land, or an offshore platform, may differ from those obtained during a boat-based survey, where the movement of the boat may mean that surveyors have a less stable platform or because birds may exhibit a behavioural response to the presence of a boat (although following standard guidance should help to minimise the influence of these factors: Camphuysen *et al.* 2004). Visibility may also strongly influence results from visual observations. Seasonality may influence the results from both radar and visual observations as foraging birds may respond very differently to migrating birds (Blew *et al.* 2008, Krijgsveld *et al.* 2011). This may be particularly important for radar studies, where it is not possible to identify radar echoes to species level and, as a result, it is more difficult to separate observations of migrants from those of local, foraging birds during periods of passage.

iii. Are reported avoidance rates affected by any special factors?

The location of the windfarm may have a strong impact on reported collision rates. If these collision rates are then used to calculate avoidance rates, it may lead to an erroneous assessment of avoidance behaviour. For example, a Belgian study has reported collision rates at a windfarm in Zeebrugge for terns (Everaert & Stienen 2007). The results from this study have been widely used to calculate micro-avoidance rates for terns (e.g. Whitfield 2008). However, as this windfarm was located on a seawall, next to a breeding tern colony, it is unclear whether behaviour around the turbines would be consistent with that of foraging terns, further out to sea. In addition, the size of turbines planned for offshore windfarms is significantly greater than those installed at many of the sites for which collision data are available. For this reason, we will consider whether there is any evidence for a relationship between turbine size and the avoidance rates derived from mortality data.

2.4 Provide summary avoidance rates and a total avoidance rate for each priority species/species group based on the evidence available at present

Based on the information compiled from the above review, we derive avoidance rates based on published evidence for each of the five priority species – northern gannet, black-legged kittiwake, lesser black-backed gull, herring gull and great black-backed gull, and other species as relevant. Where necessary, this involved going back to the source material of the studies concerned and back-calculating avoidance rates following the methodology set out by Band (2000). Where insufficient data were available to make recommendations for individual species, we combine estimates within species groups, based on the ecologies of the species concerned. Based on our critique of the studies from our review we then indicate where our confidence in each reported value is affected by the quality of the data it is based on.

Where possible, we combine avoidance rates collected at different scales, in order to calculate a total avoidance rate for each species. Estimates of micro-avoidance and macro-response can be combined to give an overall avoidance rate following equation 1, if sufficient data are available, we will extend this equation to include horizontal and vertical avoidance, as detailed in equations 2 and 3. Given the limited evidence available, it may be necessary to draw in data from closely related species and derive avoidance rates based on a group, rather than species-specific basis. Where this is necessary, we will clearly state what we have done and indicate our confidence in the derived rate accordingly.

$$A_{\text{rate}} = 1 - [(1 - A_{\text{micro}}) \times (1 - A_{\text{macro}})] \text{ [Eq. 1]}$$
$$A_{\text{micro-}} = 1 - [(1 - Mi_{\text{horiz}}) \times (1 - Mi_{\text{vert}})] \text{ [Eq. 2]}$$
$$A_{\text{macro-}} = 1 - [(1 - Ma_{\text{horiz}}) \times (1 - Ma_{\text{vert}})] \text{ [Eq. 3]}$$

Where A_{rate} is the total avoidance rate, $A_{micro-}$is the micro-avoidance rate, $A_{macro-}$is the macro-avoidance rate, Mi_{horiz} is the micro-horizontal avoidance rate, Mi_{vert} is the micro-vertical avoidance rate, Ma_{horiz} is the macro-horizontal avoidance rate and Ma_{vert} is the macro-vertical avoidance rate. Note that the ability to combine horizontal *and vertical movements in this way will depend on how data have been collected. It is likely that some birds will make horizontal and vertical movements concurrently, and therefore, it would not be appropriate to combine data in this way.*

This summary is used as the basis for a gap analysis based on our earlier definitions of avoidance behaviour. In combination with the above critique of avoidance rate studies, this gap analysis will help provide a target and possible methodologies for future research on avoidance behaviour of birds in relation to offshore windfarms, for example the Offshore Renewables Joint Industry Project (ORJIP), due to get underway in summer 2014 (Davies *et al.* 2013).

2.5 Undertake an assessment of the sensitivity of the conclusions reached to inputs and conditions under which they were collected

The final avoidance rates are likely to be sensitive to both factors which are directly parameterised within the collision risk model, such as species' flight heights, turbines' operational time and rotation speed, those parameterised in collecting collision data such as corpse collection, and also those which are not directly parameterised, such as seasonality, weather conditions and whether data have been collected during the day or night. Whether estimates of avoidance behaviour have been derived from behavioural observations or recorded collision rates, they are likely to be influenced by the factors which are not directly parameterised. For this reason, we assess how such variables are likely to have influenced the final avoidance rate in each study. For example, avoidance rates based on data only collected during conditions with better than average visibility may be expected to differ from those based on data collected during periods of poor visibility, a potential source of model error. Where avoidance rates have been derived from collision data, there is the also potential for the model input parameters to influence the final values.

These methodologies have typically been restricted to turbines at onshore locations (Everaert & Stienen 2007), where corpse collection is practical. There are concerns that this may lead to an over-estimate of the avoidance rate as some corpses go undetected and correction factors to account for this (Winkelmann 1992, Bernardino *et al.* 2013) may not be correctly applied. With this in mind, we focus on the best quality studies, but also consider how undetected corpses may influence the avoidance rate we derive.

Where a collision rate is available for a site, the avoidance rate (A_{rate}) can be calculated as follows:

$$C_{pred} = (\text{Flux rate} * P_{coll}) + \text{error} \qquad \text{[eq. 4]}$$
$$A_{rate} = 1 - (C_{obs}/C_{pred}) \qquad \text{[eq. 5]}$$

Where C_{pred} is the predicted number of collisions in the absence of avoidance action, C_{obs} is the observed number of collisions, flux rate is the total number of birds passing through the rotor swept area and P_{coll} is the probability of a bird colliding with a turbine. The probability of collision, P_{coll} can be calculated following the formula set out in Band (2012). However, this highlights a second area where the conclusions about avoidance rates may be sensitive to the inputs as values of P_{coll} will be specific to the design of turbines (Cook *et al.* 2011). Consequently, knowledge of rotor speed, radius, chord width and pitch, for the turbine concerned, are required before estimating an avoidance rate from a collision rate. These characteristics can vary considerably, even between turbines of a similar generating capacity (http://www.4coffshore.com). As a result, error is likely to be introduced into the calculation through inaccuracies in estimates of the flux rate and also through inaccuracies in the estimation of P_{coll}.

As detailed in Cook *et al.* (2012), failing to account for turbine design correctly when deriving avoidance rates as described above can lead to erroneous estimates of P_{coll} and, therefore, the avoidance rate. For this reason, where a study reports a collision rate, rather than an avoidance rate, we have attempted to obtain data on these parameters. Where we are unable to obtain this information, we calculate a value of P_{coll} based on the parameters from a range of turbines of a similar size. We then consider whether avoidance rates derived from collision estimates are more sensitive to variation in turbine design or to correction factors that account for failure to detect corpses.

2.6 Applicability of avoidance rates to different collision risk models

We finally consider how the total avoidance rate, and its constituent elements, reflect the values necessary for collision risk modelling. At present, the collision risk model formulated by Band (2012) for use in the offshore environment has three different options which can be used to estimate the total number of birds at risk of collision. These options reflect different ways in which estimates of the proportion of birds at collision risk height can be incorporated into the collision risk modelling process. Band model option 1 assumes that birds are distributed evenly within the rotor-swept area of a turbine. It bases estimates of the proportion of birds at risk of collision on data collected during pre-construction surveys of the site in question. Band model option 2 is mathematically identical to the first option, also assuming an even distribution of birds within the rotor-swept area of the turbine. However, the proportion of birds at collision risk height is estimated from continuous distributions derived from data collected across multiple sites (Cook *et al.* 2012, Johnston *et al.* 2014a,b). Options 1 and 2 of the Band model are collectively referred to as the basic model. In practice, birds are unlikely to be evenly distributed across the rotor-swept area of a turbine (Johnston *et al.* 2014a). Band model option 3, often referred to as the extended Band model, accounts for this by using a continuous flight height distribution to estimate collision risk at different points within the turbines rotor-swept area.

As birds are typically clustered to the lower edges of the rotor-swept area (Johnston *et al.* 2014a), option 3 often results in lower estimates of collision rates. As a consequence, there is intense interest in its use within EIAs for offshore windfarms. However, avoidance rates currently in use that are derived for the onshore environment by combining collision rates with estimates of P_{coll} from the basic Band model are not suitable for use in the extended model, as accounting for a heterogeneous flight height distribution will result in a lower number of collisions predicted in the absence of avoidance. (Although, note that this difference may be partially offset as avoidance rates derived in this way do not account for changes in flight altitude in response to the presence of a windfarm.) As a result estimates of avoidance behaviour based on the basic model are likely to be higher than is appropriate for the extended model (equations 4 and 5) – this is considered as part of the review.

Where estimates of avoidance rates have been derived from behavioural observations, for example displacement from offshore windfarms, rather than recorded collision rates, the applicability to different models is less clear. We consider how our final avoidance rates have been derived and what implications this has for how they are incorporated in collision risk models.

We also offer guidance not just on the applicability of avoidance rates to the basic and extended Band models, but also their transferability of avoidance rates to alternatives including the Biosis model (Smales *et al.* 2013).

The data necessary to derive avoidance rates suitable for use with option 3 of the Band model following the formula given by equation 6 are often unavailable. However, a suitable avoidance rate can be derived by estimating the ratio of P_{coll} from option 2 of the Band model to P_{coll} from option 3 of the Band model and applying this to the inverse of the avoidance rate used for option 1. For the rationale and a full description of this approach see the supplement to the guidance on ‘Using a collision risk model to assess bird collision risks for offshore windfarms’ (Band 2012) provided by Bill Band as Annex 1 to this report.

3. DEFINITIONS OF AVOIDANCE BEHAVIOUR

3.1 Introduction

Chamberlain *et al.* (2005, 2006) demonstrated that, of the parameters used in the Band collision model (Band 2006), the avoidance rate used was among those that the predicted collision rates were most sensitive to. Subsequently, the identification of appropriate avoidance rates has been subject to widespread debate. Guidance produced by Scottish Natural Heritage (SNH 2010) has been largely accepted in the UK for the terrestrial environment, subject to revision as additional data become available (e.g. Pendlebury 2006). Whilst this document references some seabird species, its guidance for offshore windfarms is limited to the suggestion that a range of avoidance rates should be presented. Country agencies have provided advice to developers as necessary, but the lack of guidance produced specifically for the offshore environment, and for the updated Band model for use in the offshore environment (Band 2012), has led to uncertainty amongst developers, regulators and other stakeholders as to what values reflect realistic avoidance rates (e.g. MacArthur Green 2012, MORL 2012) and for which collision risk models they are appropriate. Previous studies have attempted to review avoidance behaviour in offshore species (e.g. Maclean *et al.* 2009, Cook *et al.* 2012) but a failure to gain widespread consensus about the values presented has meant the situation remains largely unresolved.

Deriving avoidance rates for terrestrial windfarm developments has been based largely on the ability to estimate the numbers of birds killed by collisions. Every bird flying through the rotor-swept area of a turbine has a probability of colliding with the turbine blades (P_{coll}), typically in the range of 5-10% for seabirds, depending on species and the design of the turbine concerned (Cook *et al.* 2011). By multiplying the total number of birds expected to pass through the rotor-swept area of a turbine by P_{coll} it is possible to predict the number of collisions that would be expected, should birds take no action to avoid collision. In the case of terrestrial windfarms estimates of the total number of collisions actually occurring, once turbines are operational, can be made by using corpse searches around the windfarm to assess actual mortality rates, or observed collision rates[1]. Band (2000) therefore suggests that the avoidance rate can be thought of as equation 6, where the collision rate expected in the absence of avoidance is the total number of birds (Flux rate) passing through the rotor-swept area of a turbine, multiplied by P_{coll}. However, in practice both P_{coll} and the flux rate are likely to be subject to error – P_{coll} in relation to the model input parameters and flux rate in relation to estimates of the total number of birds passing through the windfarm. Of the two, the error associated with the flux rate is likely to be greatest as a result of the difficulty in recording the number of birds passing through a site over an extended period of time and the need to extrapolate from, often brief, observation periods to estimate a flux rate for the study period as a whole. As a result of the need to incorporate this error, it may be better to think of this in terms of an avoidance correction factor, as opposed to an avoidance rate, which implies it is solely dependent on the behavioural responses of birds:

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[1] Subject to some carcass recovery factor (i.e. the potential to miss carcasses, removal by predators, etc.).

$$Avoidance = 1 - \frac{Observed\ collision\ rate}{Pcoll \times Flux\ Rate} \quad \text{(eq. 6)}$$

However, in the case of offshore windfarms, recording actual collisions, or mortality rates, is not currently practical, although the forthcoming Offshore Renewables Joint Industry Project (ORJIP) will aim to provide additional data to inform avoidance rates using behavioural observations (Davies *et al.* 2013). Therefore, at present, guidance on appropriate avoidance rates for use in the offshore environment draws on the experiences gained in the terrestrial environment, as well as being informed by studies of bird movements, where suitable data are available (e.g. Desholm & Kahlert 2005, Petersen *et al.* 2006, Masden *et al.* 2009, Blew *et al.* 2008, Krijgsveld *et al.* 2011). Where studies have sought to use movement data to inform values for avoidance rates, this has often led to confusion due to uncertainty over the spatial scales involved. Birds have been shown to alter their flight paths in order to avoid entering an offshore windfarm at distances of up to 6 km (Christensen *et al.* 2004). As a result, where avoidance rates have been derived from human observations they may represent a substantial under-estimate of total avoidance, as many birds will have taken action to avoid the windfarm before they become visible to observers. The difficulties caused in attempting to draw firm conclusions from such disparate data sources has led to a variety of terms being used to sub-divide avoidance behaviour at different spatial scales.

At a simple level, Cook *et al.* (2012) and Band (2012) suggest that the total avoidance rate for an offshore windfarm could be considered as (eq. 7):

$$Total\ Avoidance = 1 - (\ 1 - Macro \ \times\ 1 - Micro\ ) \quad \text{(eq. 7)}$$

We use this definition as the basis for discussion relating to the different types of avoidance that need to be quantified in order to derive an estimate of total avoidance, and extend it to incorporate meso-avoidance (eq. 8), as defined below.

$$Total\ Avoidance = 1 - (\ 1 - Macro \ \times\ 1 - Meso \ \times\ 1 - Micro\ ) \quad \text{(eq. 8)}$$

3.2 Defining appropriate spatial scales of avoidance

This section aims to define appropriate spatial scales of avoidance; for detailed review of the evidence for avoidance at these defined scales, see section 5.

A bird may respond to a fixed object, such as a turbine, at any point between the time at which it first observes the object and the time at which it passes or collides with the object, or based on previous experiences of the site. As such, attempts to subdivide avoidance behaviour with reference to spatial scale are largely arbitrary and the different behaviours should be seen as part of a continuum. Nevertheless, such divisions are necessary given the spatial scales over which these behaviours can be recorded. Band (2012) focusses on macro- and micro- avoidance, with a third category, meso-avoidance, fitting in the gap between the two also suggested (Pendlebury, *pers. comm.*). We consider these scales in turn, with each reflecting an increasing distance between the bird and the turbine blades (Figure 3.1). However, the distances over which these categories of behaviour occur are more difficult to define.

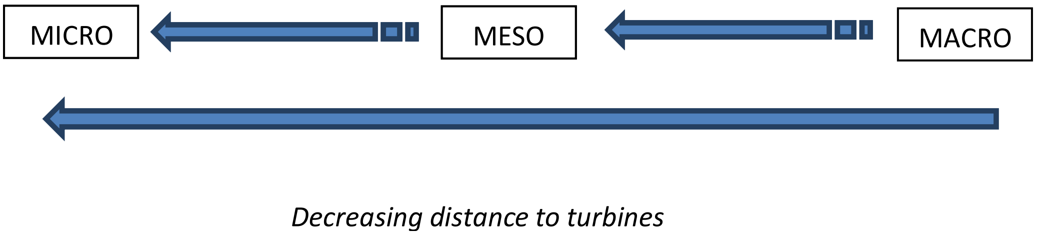


Figure 3.1 Spatial scales over which avian responses to turbines have been recorded

It is also necessary to consider how avoidance rates are applied within the collision risk modelling framework. Expected collision rates (as per eq. 7) are typically derived using estimates of the numbers of birds flying through the windfarm area prior to construction. Therefore, overall avoidance rates need to account for birds no longer entering the windfarm area post-construction (i.e. birds exhibiting displacement and barrier effects) in addition to avoidance of the turbines themselves. As a result, it is necessary to consider how other effects, such as displacement and barrier effects, may contribute to the overall avoidance rates, as part of macro-avoidance.

We consider how each of these scales may be used to inform collision risk modelling below:

Macro- Band (2012) gives the example of displacement as one impact which may contribute to macro-avoidance. Displacement is typically assessed by comparing numbers of birds in the area of the windfarm to those recorded in a baseline period. However, difficulties in quantifying displacement rates – numbers may vary for many reasons in addition to the development of the windfarm, and it is important that this is considered in an appropriate survey design, for example using a BACI-approach (Masden *et al.* 2010) – mean that interpreting these data must be undertaken with caution and careful consideration of the survey design (Maclean *et al.* 2013). Furthermore, published displacement rates can refer to the numbers of birds displaced from the windfarm plus a significant (species-dependent) buffer distance around the windfarm. Consideration must also be given as to whether displacement rates reflect all birds within the windfarm area and buffer, or just those on the water. As collision risk modelling relates only to birds in flight, if displacement rates refer only to birds on the water, they may not reflect macro-avoidance. Relying solely on displacement, as often reported in Environmental Impact Assessments, may therefore underestimate the true scale of macro-avoidance because 1) estimates may not account for birds in flight; and 2) estimates do not account for birds that are displaced from the windfarm area, but remain within the buffer surrounding the windfarm.

In addition to measuring displacement rates, a number of offshore windfarm post construction monitoring studies have used radar to assess the proportion of birds which enter a windfarm area (e.g. Petterson 2005, Petersen 2006, Krijgsveld *et al.*

2011). The potential for windfarms to act as a barrier to birds in this way has been widely discussed, mostly in the context of migrants (e.g. Desholm & Kahlert 2005, Masden *et al.* 2009), although it may also be of relevance to seabirds commuting between breeding colonies and feeding areas – an area of study that needs addressing with some urgency. Such studies would illustrate changes in flight trajectory amongst birds approaching windfarms and would help to determine the spatial scale over which such responses may occur.

In addition to displacement and the windfarm acting as a barrier, several studies have suggested that some species, notably gulls and cormorants, may be attracted to the area of offshore windfarms (e.g. Lindeboom *et al.* 2011, Leopold *et al.* 2011). The macro-avoidance rate needs to capture the change in bird numbers within the windfarm area resulting from the development of the windfarm site. Consequently, the term 'macro-avoidance', may lead to confusion as, conceptually, the idea of a negative macro-avoidance rate (i.e. birds being attracted to a windfarm) may be difficult to communicate to stakeholders. For this reason, use of the more neutral term, **macro-response**, may be preferable as it implicitly covers both attraction and avoidance (Figure 3.2).

Figure 3.2 Range of proportional responses to the presence of an offshore windfarm as they would be incorporated in eq. 2 (above), i.e. a response of -0.1 would reflect an increase in the number of birds present within the windfarm of 10% in comparison to baseline numbers, whilst a response of 0.1 would reflect a decrease of 10% in comparison to baseline numbers, which are sensitive to survey design due to the extent of year on year variation in seabird abundance.

The macro-response of birds to the presence of a windfarm should be defined as the behavioural response taking place outside the windfarm perimeter. It is important that the perimeter of the windfarm is clearly defined. Definitions could be based on characteristics such as turbine rotor diameter, or the inter-array turbine spacing. However, such definitions would vary between sites in relation to the layout and size of turbines used, meaning values for the macro-response rate would be less directly comparable between sites. For this reason, defining the perimeter as extending a fixed distance from the base of the outermost turbines is preferable. The review will define of the perimeter as the boundary of a minimum convex polygon encompassing an area extending from a distance of **500 m from the base of the outermost turbines** (see Figure 3.3).

The term macro-response will be used to refer to changes in bird numbers within the windfarm area resulting from the development of the windfarm site, through processes including, but not limited to, attraction, displacement and barrier effects. Where displacement is used to infer a macro-response rate, it is important to be clear whether this reflects displacement from the windfarm only, or displacement from the windfarm plus a buffer. Buffers considered in the assessment of displacement effects typically extend beyond the 500 m around the windfarm perimeter considered here as some birds may respond to the presence of the windfarm at distances greater than this. Measures of displacement that use such buffers may thus underestimate the macro-response rate considered here. As collision risk models refer to birds in flight only, when using displacement rates to estimate a part of macro-avoidance behaviour, it is also important to lend more weight to studies that distinguish the displacement rates of birds in flight and on the water, or those for which it is possible to estimate the number, or proportion, of birds in flight.

Micro- Blew *et al.* (2008) suggests that micro-avoidance reflects a 'last-second' alteration to a flight path in order to avoid collision with a turbine. Petterson (2005) and Blew *et al.* (2008) both suggest that birds adjust their flight paths to avoid entering the rotor-swept zone of a turbine and that, therefore, birds may only rarely need to take last second action to avoid collision, possibly as a result of adverse conditions, such as poor visibility. This is borne out by empirical evidence presented in Desholm (2005) and Krijgsveld *et al.* (2011) (see section 5.3).

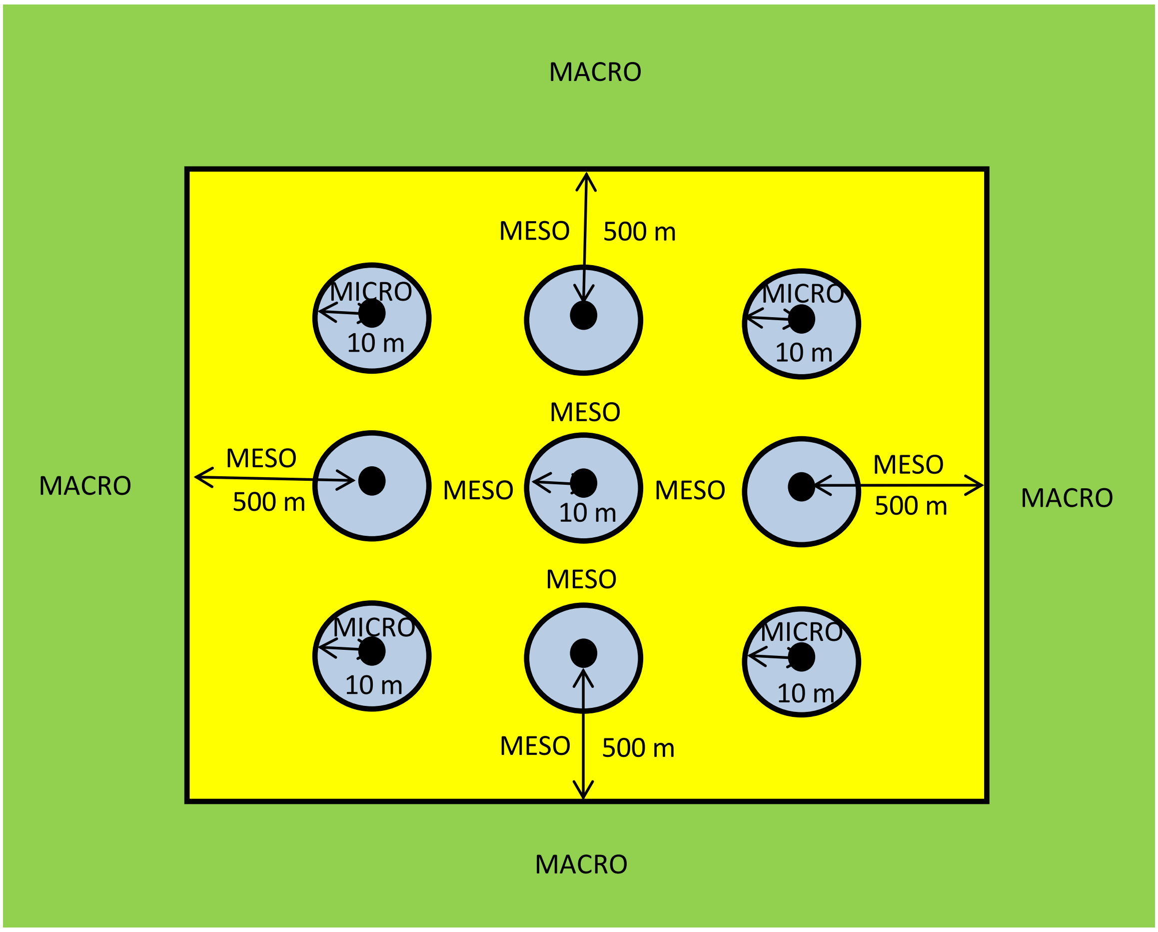
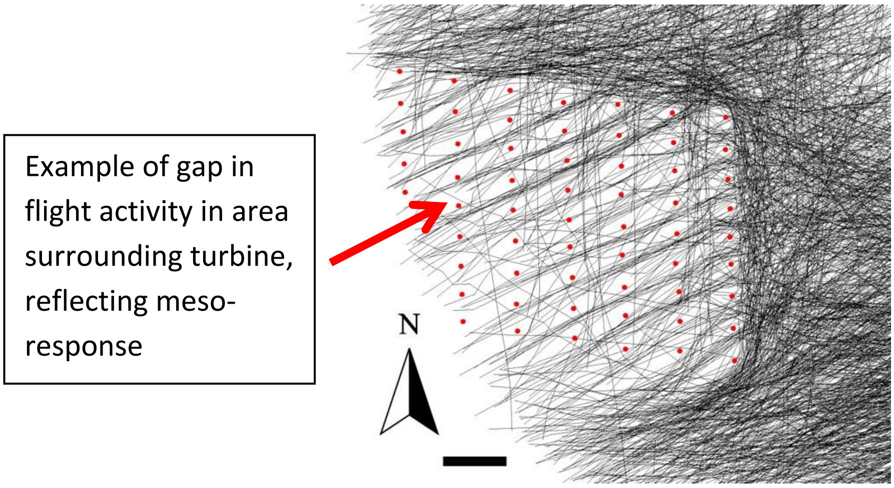


Figure 3.3 Schematic illustrating the spatial scales over which micro-avoidance, meso- and macro- responses operate. Dots refer to turbine tower locations (not to scale).

Therefore, it would seem reasonable to define **micro-avoidance** as a last-second alteration to a bird's flight path in order to avoid collision. For the purposes of observational studies, such last-second avoidance would be expected to occur in a 3-dimensional space **within 10 m of the turbine blades** (i.e. at distances of 10 m horizontally or vertically from edges of the turbine blades) – though note that this distance (and consequently the appropriate definition of micro-avoidance) may be refined based on future advances in the techniques used to collect the necessary data (see Figure 3.3). Such behaviour is likely to be recorded relatively rarely.

Meso- Whilst macro-responses reflect behaviour outside the windfarm and micro-avoidance reflects last-second action taken to avoid collision, there is a need to consider a third category, reflecting species responses to turbines within a windfarm (Figure 3.4). Both Desholm & Kahlert (2005) and Krijgsveld *et al.* (2011) demonstrated that the majority of birds do not pass within 50 m of a turbine. However, some, such as cormorants, may be attracted to structures, which offer potential roosting sites (e.g. Leopold *et al.* 2011). For this reason, as in the case of macro-response, it may be more straightforward to talk about a **meso-response** to turbines than meso-avoidance. The term meso-response should be used to refer to all behavioural responses to the **turbines beyond the 10 m buffer around the rotor blades, covered by micro-avoidance, and within the perimeter of the windfarm** (see Figure 3.3). This may include, for example the attraction of cormorants to turbine bases as a roosting site, as the base of the turbine would be beyond the 10 m buffer around the rotor blades.



Example of gap in flight activity in area surrounding turbine, reflecting meso-response

Figure 3.4 Flight trajectories of migrating waterbirds within an offshore windfarm, red dots indicate locations of turbines. Reproduced with permission from Desholm & Kahlert (2005) Avian collision risk at an offshore windfarm. *Biology Letters* 1: 296-298.

At present, the scale at which data are collected may make it difficult to differentiate between a meso-response and micro-avoidance. Therefore, it is recommended that the term **macro-response** is used to refer to a response outside the windfarm and within-windfarm response, covering both the meso- and micro-scale, is used to refer to a response occurring inside a windfarm. In response to technological advances, a fuller separation of meso-responses from micro-avoidance is likely to be possible in the near future. For example, it may be possible in future to combine radar monitoring of flight paths through offshore windfarms to capture meso-responses (as in Desholm & Kahlert 2005) with images captured from turbine mounted cameras to capture micro-avoidance (as in Desholm *et al.* 2006).

3.3 Defining the appropriate 3-D level of avoidance

This section aims to define appropriate 3-D scales of avoidance; for detailed review of the evidence for horizontal and vertical meso-avoidance, see section 5.2.

In addition to occurring over a range of different spatial scales, avoidance behaviour may occur in both the horizontal and vertical planes. Below, we describe how observations of horizontal and vertical avoidance may be collected and the spatial scales which may be relevant to each. This distinction is important given that some methodologies for recording avoidance behaviour, such as radar, may not detect both horizontal and vertical movements, meaning that where only one is recorded, the derived avoidance rate is likely to be an underestimate, which may be offset by an inability to record horizontal and vertical movements occurring concurrently. There is also a need to consider the relationships between avoidance and other effects of offshore windfarms on birds, for example barrier effects and displacement.

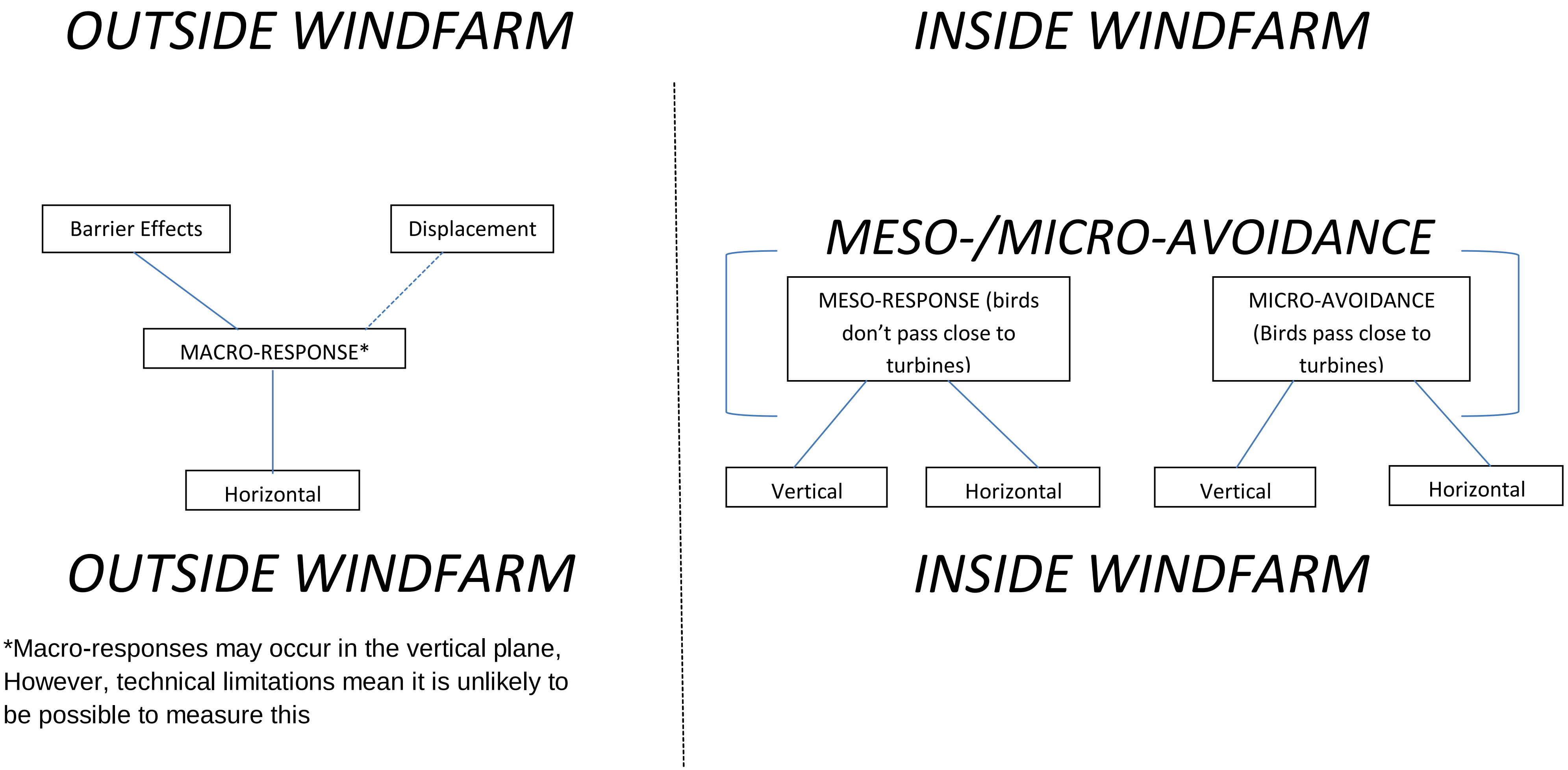
Horizontal Avoidance Much of the research into the avoidance behaviour of seabirds in relation to offshore windfarms has focussed on horizontal avoidance, whereby birds alter their flight paths so that they fly around turbines or do not enter the perimeter of the windfarm (i.e. Desholm & Kahlert 2005, Masden *et al.* 2009). These data have been collected using a variety of methodologies, notably visual observations (i.e. Krijgsveld *et al.* 2011) and radar observations (i.e. Petersen *et al.* 2006). We consider that all 3 spatial scales defined here are relevant in the context of horizontal avoidance.

Vertical Avoidance As technologies and survey protocols for monitoring collisions become more developed (e.g. Desholm *et al.* 2006, Collier *et al.* 2011a, 2011b) monitoring of both horizontal and vertical movements around turbines should become more feasible. For radar, however, at greater distance this may be more challenging as detecting both horizontal and vertical avoidance requires the use of both x- and y-band radar. At present, radar monitoring of bird movements in and around offshore windfarms typically focuses on horizontal avoidance behaviour, using horizontal radar (e.g. Petersen *et al.* 2006). Where changes in flight height amongst birds entering the windfarm have been estimated (e.g. Blew *et al.* 2008) this has been at too coarse a resolution to inform vertical avoidance. However, recent developments in radar technology (e.g. http://www.robinradar.com/3d-flex/) may make this a more practical solution to investigate vertical avoidance behaviour amongst birds approaching offshore windfarms.

Krijgsveld *et al.* (2011) demonstrate that a number of species may fly at lower altitudes within-windfarms than outside windfarms and incorporate vertical avoidance behaviour in their estimation of micro-avoidance rates using a combination of visual and radar observations. Their results suggest that a substantial proportion of birds may alter their flight altitudes in order to avoid collision. Given the development of technologies capable of monitoring the movement of birds close to turbines, such as the Thermal Animal Detection System (Desholm *et al.* 2006), these results suggest that focussing on vertical avoidance at a micro-meso, as opposed to macro, scale may be worthwhile. At a micro-scale, it is likely that vertical avoidance would be captured as part of an evasive manoeuvre.

3.4 Total avoidance rates

In this section, we have produced definitions that are considered to work within the constraints of our current understanding of avoidance behaviour and data collection limitations. It is clear, given the multiple potential components of avoidance behaviour that we have identified (Figure 3.5), that equation 7 is an over-simplification of overall avoidance rates. In future studies it is important to consider how each of these components can be quantified. As technological capabilities advance, the definitions outlined above may become obsolete. However, any refinement to these definitions should be based on the behaviour of the species concerned, rather than artificially induced by methodological constraints, for example, the distance over which observations can be made with the use of binoculars or telescopes.



*Macro-responses may occur in the vertical plane, However, technical limitations mean it is unlikely to be possible to measure this

Figure 3.5 Schematic detailing how different behavioural responses to offshore windfarms may combine to give a total avoidance rate. At each different level birds may respond either vertically or horizontally. Outside a windfarm, both displacement and barrier effects are likely to contribute to the macro-response rate. However, the contribution of displacement to macro-avoidance may be hard to quantify as a result of uncertainty associated with estimating its effects. Avoidance behaviour inside a windfarm is often termed micro-avoidance, however, it may be appropriate to split this term further by considering a meso-response, where birds enter a windfarm but do not pass close to turbines, and micro-avoidance, where birds take last minute action to avoid collisions.

3.5 Recommended Definitions

For the purposes of this review, the definitions we will use for bird behaviour in response to offshore windfarms and turbines are (Figure 3.3):

> **MACRO-RESPONSE –** The response of birds to the presence of the windfarm outside its perimeter, defined as a 500 m buffer surrounding the outermost turbines. Responses may include attraction to the windfarm, displacement from preferred foraging habitat or an alteration to flight paths as a result of seeing the windfarm as a barrier. These may occur in either horizontal or vertical planes, although at present technological limitations mean that it is not possible to measure vertical macro-responses. For this reason, for the purposes of this review, we consider only horizontal macro-responses.
>
> **MESO-RESPONSE –** A redistribution of birds, or alteration of flightpaths within a windfarm in response to the presence of the turbines. This may encompass both horizontal and vertical responses. These responses are in contrast to micro-avoidance, see below.
>
> **MICRO-AVOIDANCE** – Last-second action taken by birds flying at rotor height to avoid collision, encompassing both horizontal and vertical movements, within a 10 m buffer surrounding turbine rotor-swept areas.

Due to current methodological difficulties in distinguishing micro-avoidance behaviour from meso-response behaviour, a fourth category is defined for the purposes of this review to act as a proxy for responses to windfarms at these scales:

> **WITHIN-WINDFARM AVOIDANCE –** Encompassing both meso-responses and micro-avoidance to describe how birds within a windfarm respond to the presence of a turbine.

The review focuses on data relating to macro-responses and within-windfarm avoidance. Distinctions between responses at the meso- or micro-scale and horizontal or vertical responses have not been made at this stage as insufficient data are available to support them. Future studies should aim to be able to make such distinctions to improve our understanding of avian avoidance behaviour at offshore windfarms.

4. REVIEW OF AVOIDANCE RATES USED IN COLLISION RISK MODELLING FOR OFFSHORE WINDFARMS

We reviewed the use of avoidance rates in collision risk modelling as part of the impact assessment process for 35 consented or proposed offshore windfarms (Table 4.1). There was considerable variation between assessments in the rates selected, which were as low as 0.87 and as high as 0.9999. In the majority of cases, a single avoidance rate for all species, ranging from 0.95 to 0.99, has been used in the collision risk modelling process to assess the potential impacts for all species considered. However, in some instances, developers and their consultants have felt that sufficient evidence exists to consider higher rates for some species, notably terns, although these values have not always been accepted within the decision process.

The species assessed during the collision risk modelling process vary on a site by site basis. This typically reflects the distribution of these species, for example, with Manx shearwater likely to be assessed at sites on the west coast of the UK. However, some species, such as northern gannet, black-legged kittiwake, lesser black-backed gull and great black-backed gull, are considered in most assessments, reflecting the broad scale distributions of these species. The flight height of birds is also an extremely important factor in determining the likely risk of collision (Johnston *et al.* 2014a). In several early assessments, a screening process was also carried out whereby species for which only a small proportion of individuals (typically <1%) were recorded flying at heights placing them at a risk of collision were excluded from the collision risk modelling process (Table 4.1). As a result of this screening process, the collision risk of some species, such as auks and divers, was assumed to be negligible and therefore not assessed using collision risk models.

In early assessments, the avoidance rates used in collision risk modelling were often very high, typically in excess of 0.99. The use of these rates was largely founded on collision rates reported at onshore windfarms (e.g. Winkelman 1992, Everaert 2003). However, these do not reflect true avoidance rates as they do not account for birds which pass safely through the rotor swept area of the turbines without taking avoidance action, or indeed those which pass through the windfarm without entering the rotor-sweep of the turbines.

In 2005, SNH issued guidance for sensitive bird species commonly identified in (onshore) windfarm environmental statements (SNH 2010) that a default avoidance rate of 0.95 should be used. This figure was based on expert opinion (Andy Douse *pers. comm.*) and acknowledged as being precautionary. It was felt that, as evidence became available, this rate would be revised upwards. Of the 13 assessments for offshore windfarms published between 2005 and the revision of this guidance in 2010 (SNH 2010), seven followed this guidance (see Table 4.1). The remaining assessments which argued that higher avoidance rates were more appropriate, cited as part of their justification empirical data of collision rates collected from sites in Belgium (see Everaert 2003, Everaert and Stienen 2006, Everaert 2008) or assessments of species' manoeuvrability as determined by Garthe and Hüppop (2004) and Maclean *et al.* (2009).

Following evidence obtained from onshore windfarms suggesting avoidance rates were likely to be significantly higher than 0.95 (Fernley *et al.* 2006, Pendlebury 2006, Whitfield and Madders 2006, Whitfield 2009) the default values were revised by SNH (2010). A default rate of 0.98 was recommended for all species considered in this guidance which included gull spp., tern spp, skua spp and diver spp. Exceptions to the default value included geese, hen harrier and golden eagle, for which sufficient evidence was available to support a 0.99 avoidance rate, and kestrel and white-tailed eagle, for which the 0.95 avoidance rate was retained as it was felt they were particularly susceptible to collisions. Again, a significant proportion (12 out of 18) of environmental impact assessments for offshore windfarms published since 2010 follow this guidance. The remaining studies cite evidence from Belgium (Everaert 2003, Everaert and Stienen 2006, Everaert and Kuijken 2007, Everaert 2008) and the Netherlands (Leopold *et al.* 2011, Krijgsveld *et al.* 2011), or again base avoidance rates on assessments of species' manoeuvrability as determined by Garthe and Hüppop (2004) and Maclean *et al.* (2009) in support of higher avoidance rates. As part of our review, we consider the strength of the quantitative evidence put forward in these studies and how qualitative information may be used to support these conclusions.

The evidence base for the revised avoidance rates is largely based on collision mortality observations at onshore / coastal windfarms – although recent behavioural avoidance evidence from Egmond aan Zee (Krijgsveld *et al.* 2011) is also being used – and there are uncertainties around the applicability of these values to offshore windfarms (Trinder 2012). First, whilst some seabird species may be attracted to offshore windfarms, others such as northern gannet show evidence of macro-avoidance (e.g. Krijgsveld *et al.* 2011, Vanermen *et al.* 2013) (see section 5.1). In contrast, while some terrestrial species, such as geese, may also show strong macro-avoidance of offshore windfarms (Plonczkier & Simms 2012), macro-avoidance is often less likely at terrestrial windfarms (e.g. Devereux *et al.* 2008, Garvin *et al.* 2011, Pearce-Higgins *et al.* 2012). As a result, avoidance rates in relation to offshore windfarms need to capture not just avoidance of the individual turbines, as is the case for species at terrestrial sites, but also of the windfarm itself.

Secondly, estimates of avoidance derived from collision mortality rates (rather than direct observations of avoidance – 'behavioural avoidance') follow the formula given in SNH (2010), whereby observed mortality is divided by the mortality expected in the absence of avoidance based on the flux of birds through the rotor-swept area (equation 6).

Surveys for terrestrial windfarms are usually carried from vantage points within 2 km of the area to be observed, ensuring that all observations are within 2 km. However, these methodologies rarely employ distance correction which means that the flux rates of birds (or population estimates) are likely to be underestimated. If the numbers of birds passing through the rotor-swept area of a turbine, and therefore the expected numbers of collisions, are underestimated, the derived avoidance rate will also be an underestimate. In contrast, population sizes within offshore windfarms of each of the five priority species considered as part of this review may potentially be over-estimated, given the attraction of each to boats (e.g. Garthe & Hüppop 1994, Skov & Durinck 2001). Even where population data have been collected from other platforms, for example, by digital aerial survey (e.g. Buckland *et al.* 2012), the potential for underestimating population size is considerably less than for surveys of onshore windfarms. As populations within offshore windfarms are unlikely to be underestimated, it has been argued (Trinder 2012) that an avoidance rate suitable for estimating collisions at an onshore windfarm will lead to underestimation of avoidance behaviour if used for estimating collisions at an offshore windfarm.

This review highlights the reliance of offshore windfarm developers, and their consultants, on guidance from Statutory Nature Conservation Bodies (SNCBs) about the use of appropriate avoidance rates. Of the 35 studies we identified, 19 cited the SNH guidance from either 2005 or 2010 in support of the avoidance rates selected for some, or all of their study species. Of these studies, several have suggested that these avoidance rates are potentially overly-precautionary, citing evidence from Belgium (Everaert 2003, Everaert and Stienen 2006, Everaert 2008), and the Netherlands (Winkelman 1992, Krijgsveld *et al.* 2011). The use of avoidance rates in excess of 0.98 in a number of recent environmental statements may reflect an increasing concern amongst developers that the SNH guidance is overly precautionary and posing an unnecessary risk to the consenting process. Many of the early developments were relatively small scale and consequently, collision risk estimates, even with an avoidance rate of 0.95, were extremely low. However, the scale of many of the developments proposed more recently is significantly greater, with commensurate increases in estimated collision rates. Consequently, it is important the subsequent review of avoidance rates can clarify the situation for developers and SNCBs alike.

Table 4.1 Avoidance rates considered during the collision risk modelling undertaken in assessments for proposed offshore windfarms and the justification for their use. All avoidance rates were used in conjunction with the basic (option 1) Band model and were taken from the final submitted environmental statements.

Offshore windfarm	Year	Avoidance rate(s) used	Species considered	Justification
Kentish Flats	2002	0.9998	Red-throated diver	Collision rate of 0.02% presented in Winkelman (1992)
Burbo Bank	2002	No Collision Risk Modelling	Red-throated diver, common scoter, common tern, wader sp., great cormorant, red-breasted merganser, little gull, common guillemot/razorbill	Sensitive species flew below rotor height and, therefore, were not at risk of collision
North Hoyle	2002	No Collision Risk Modelling	Red-throated diver, great cormorant, common scoter, tern sp., European shag, common guillemot, razorbill	Sensitive species flew below rotor height and, therefore, were not at risk of collision
Teesside	2004	0.9962 for all species	Red-throated diver, northern gannet, great cormorant , waders, Arctic skua, herring gull, great black-backed gull, black-legged kittiwake, Sandwich tern, common tern, common guillemot, geese sp.	Based on calculations from Blyth Harbour (citing Still *et al.* 1996, Painter *et al.* 1999)
Beatrice Demonstration Site	2005	0.95 for all species	Black-legged kittiwake, great black-backed gull, northern fulmar, northern gannet, auk spp, herring gull, tern spp	Follows SNH guidance from 2005 (SNH 2010) and is acknowledged as a conservative value.

Offshore windfarm	Year	Avoidance rate(s) used	Species considered	Justification
Thanet	2005	0.99 for all species	Red-throated diver, northern fulmar, northern gannet, common tern, Sandwich tern, black-legged kittiwake, common gull, herring gull, lesser black-backed gull, gull spp, auk spp	In line with previously published estimates of avoidance (Percival 1998, Everaert *et al.* 2002, Henderson *et al.* 1996, Winkelman 1992, Winkelman 1990, Percival 2001, Still *et al.* 1996)
London Array	2005	0.995 and 0.999 for gull spp, tern spp and Northern gannet, and 0.99 and 0.995 for diver sp.	Red-throated diver, black-throated diver, herring gull, lesser black-backed gull, great black-backed gull, common tern, northern gannet, Sandwich tern	Based on vulnerability to collision as assessed by Garthe & Hüppop (2004) and observed collision rates for gulls and terns presented by Everaert (2003)
Greater Gabbard	2005	High (0.9999), Medium (0.9982) and Low (0.87) for all species	Red-throated diver, lesser black-backed gull, great skua	High and Medium rates calculated from data presented in Winkelman (1992) based on total collisions numbers for gulls (High) and nocturnal collisions for gulls (Medium), Low avoidance rate derived from lowest reported avoidance rate of 0.87 found in American kestrel and considered highly unrealistic
Gwynt Y Mor	2005	No Collision Risk Modelling	Diver sp., northern fulmar, Manx shearwater, Leach's petrel, northern gannet, common scoter, small skua spp, great skua, black-legged kittiwake, Sandwich tern, 'comic' tern, common guillemot/razorbill	Sensitive species flew below rotor height and, therefore, were not at risk of collision

Offshore windfarm	**Year**	**Avoidance rate(s) used**	**Species considered**	**Justification**
Sheringham Shoal	2006	0.98 for all species	Sandwich tern, common tern, northern gannet, little gull, lesser black-backed gull	SNH guidance from 2005 (SNH 2010) guidance felt to be over-precautionary
West of Duddon Sands	2006	0.999	Lesser black-backed gull	Based on vulnerability to collision as assessed by Garthe & Hüppop (2004) and observed collision rates for gulls presented by Everaert (2003)
Humber Gateway	2007	0.95 for all species	Red-throated diver, northern gannet, great skua, Arctic skua, little gull, black-headed gull, common gull, black-legged kittiwake, herring gull, great black-backed gull, lesser black-backed gull, Sandwich tern, common tern, Arctic tern	Follows SNH guidance from 2005 (SNH 2010) and is acknowledged as a conservative value
Lincs	2007	0.95 for all species	Pink-footed goose, red-throated diver, northern gannet, little gull, common gull, lesser black-backed gull, common tern, Common guillemot	Follows SNH guidance from 2005 (SNH 2010) and is acknowledged as a conservative value
Westernmost Rough	2009	0.95 for all species	Northern gannet, black-legged kittiwake, common gull, lesser black-backed gull, herring gull, great black-backed gull, common tern	Follows SNH guidance from 2005 (SNH 2010) and is acknowledged as a conservative value

Offshore windfarm	**Year**	**Avoidance rate(s) used**	**Species considered**	**Justification**
Race Bank	2009	0.996 for Sandwich tern, 0.95 for all other species	Sandwich tern, common tern, northern fulmar, little gull, northern gannet, lesser black-backed gull, black-legged kittiwake, common guillemot, razorbill	Sandwich tern avoidance rate based on data from Zeebrugge, SNH guidance from 2005 (SNH 2010) for other species, but also discussion as to whether higher avoidance rates may be appropriate in some cases (northern gannet and lesser black-backed gull)
Dudgeon	2009	0.996 for Sandwich Tern, 0.99 for lesser black-backed gull, 0.97 Northern gannet	Sandwich tern, lesser black-backed gull, northern gannet	Evidence presented in Everaert & Stienen (2006) & Everaert (2008) for Sandwich tern and recommendations in Maclean *et al.* (2009) for northern gannet and lesser black-backed gull
LID6	2010	0.95 for all species	Black-throated diver, great northern diver, northern gannet, dark-bellied brent goose, little gull	Follows SNH guidance from 2005 (SNH 2010) and is acknowledged as a conservative value
Triton Knoll	2011	0.98 for all species	Northern fulmar, little gull, black-legged kittiwake, Sandwich tern, northern gannet, common guillemot, Arctic skua, lesser black-backed gull, great black-backed gull, common tern	Follows SNH guidance from 2005 (SNH 2010) guidance
Galloper Offshore Windfarm	2011	0.99 for gulls, 0.98 for other species	Red-throated diver, northern gannet, Arctic skua, great skua, common gull, lesser black-backed gull, herring gull, great black-backed gull, black-legged kittiwake	Evidence from ‘vantage point surveys’ for gulls, follows SNH (2010) guidance for all other species

Offshore windfarm	**Year**	**Avoidance rate(s) used**	**Species considered**	**Justification**
Rampion	2011	0.995 for Northern gannet, Gulls sp. , skuas spp and Auks, 0.99 for terns sp. and waterbirds	Brent goose, common scoter, northern gannet, bar-tailed godwit, Eurasian curlew, great skua, Mediterranean gull, common gull, lesser black-backed gull, herring gull, great black-backed gull, black-legged kittiwake, Sandwich tern, common guillemot, barn swallow, meadow pipit	Follows Maclean *et al.* (2009)
Aberdeen Offshore Windfarm	2012	0.98 for all species	Common guillemot, razorbill, Atlantic puffin, northern fulmar, common tern, Sandwich tern, herring gull, black-legged kittiwake, great black-backed gull, common gull, common scoter, common eider, European shag, great cormorant, northern gannet, red-throated diver, Arctic skua	Follows SNH (2010) guidance
Blyth Offshore Demonstration Project	2012	0.98 for all species	Northern gannet, common gull, herring gull, great black-backed gull, little gull, black-legged kittiwake, common tern	Follows SNH (2010) guidance

Offshore windfarm	Year	Avoidance rate(s) used	Species considered	Justification
Hornsea Project One	2012	0.98 for all species	Northern fulmar, northern gannet, black-legged kittiwake, little gull, common gull, great black-backed gull, lesser black-backed gull, herring gull, common tern, Arctic tern, common guillemot, razorbill, Arctic skua, great skua	Follows SNH (2010) guidance
Irish Sea	2012	0.98 for all species	Manx shearwater, great black-backed gull, lesser black-backed gull, herring gull, black-legged kittiwake, northern gannet, Greenland white-fronted goose	Follows SNH (2010) guidance
East Anglia One	2012	0.98 for all species	Northern fulmar, northern gannet, black-legged kittiwake, common gull, lesser black-backed gull, herring gull, great black-backed gull	Follows SNH (2010) guidance
Firth of Forth Alpha and Bravo	2012	0.98 for all species	Northern gannet, black-legged kittiwake, lesser black-backed gull, herring gull, great black-backed gull	Follows SNH (2010) guidance
Beatrice Offshore Windfarm	2012	0.99 for all species	Arctic skua, Arctic tern, northern fulmar, great black-backed gull, northern gannet, herring gull, black-legged kittiwake, great skua, common guillemot, razorbill	Review of micro-and macro-avoidance rates and criticism of the transferability of avoidance rates between onshore and offshore windfarms in MacArthur Green (2012)

Offshore windfarm	**Year**	**Avoidance rate(s) used**	**Species considered**	**Justification**
Dogger Bank Creyke Beck A and B	2012	0.99 for northern gannet, 0.98 for all other species	Northern fulmar, northern gannet, great skua, Arctic skua, black-legged kittiwake, lesser black-backed gull, great black-backed gull, common guillemot, razorbill, little auk, Atlantic puffin	Evidence from Egmond aan Zee(Krijgsveld *et al.* 2011) and elsewhere supporting 0.99 for northern gannet and following SNH (2010) guidance for all other species
Moray Firth Offshore Windfarm	2012	0.995 for northern gannet, 0.985 for lesser black-backed gull, 0.99 for black-legged kittiwake	Northern gannet, black-legged kittiwake, herring gull, great black-backed gull	Consideration of micro-and macro-avoidance rates presented for Dutch and Belgian windfarms (Everaert 2008, Krijgsveld *et al.* 2011)
Nearth na Gaoithe	2012	0.998 for northern gannet, 0.995 for gulls spp., 0.98 for Arctic tern	Northern gannet, little gull, lesser black-backed gull, herring gull, great black-backed gull, black-legged kittiwake, Arctic tern	High macro-avoidance rates for northern gannet presented in Leopold *et al.* (2011) suggest that avoidance rates presented in both SNH (2010) guidance and MacLean *et al.* (2009) are likely to be over precautionary for northern gannet. Tern and gull avoidance rates follow Maclean *et al.* (2009)
Bligh Bank Windfarm (Belgium)	2013	0.976 micro-avoidance rate for all species	Common gull, lesser black-backed gull, herring gull, great black-backed gull, black-legged kittiwake	Based on rates estimated at Egmond aan Zee by Krijgsveld *et al.* (2011)
Walney Extension Offshore Windfarm	2013	0.98 for all species	Whooper swan, pink-footed goose, lesser black-backed gull	Follows SNH (2010) guidance

Offshore windfarm	Year	Avoidance rate(s) used	Species considered	Justification
Burbo Bank Extension	2013	0.98 for all species	Red-throated diver, Manx shearwater, common scoter, little gull, black-headed gull, herring gull, lesser black-backed gull, common tern, Arctic tern, Sandwich tern, great cormorant, northern gannet, Arctic skua, great skua, black-legged kittiwake	Follows SNH (2010) guidance
Atlantic Array	2013	0.98 for all species	Manx shearwater, northern gannet, black-legged kittiwake, lesser black-backed gull, herring gull, great black-backed gull, common guillemot	Follows SNH (2010) guidance
Inch Cape	2013	0.99 for northern gannet, 0.98 for all other species	Northern gannet, Arctic skua, pomarine skua, great skua, black-legged kittiwake, great black-backed gull, herring gull	Evidence presented from Egmond aan Zee to justify 0.99 for northern gannet, follows SNH (2010) guidance for all other species

5. REVIEW OF PUBLISHED EVIDENCE FOR AVOIDANCE RATES OF MARINE BIRDS

This section provides a review of published evidence for macro-response (section 5.1), meso-response alone (section 5.2), micro-avoidance alone (section 5.3) and overall within-windfarm avoidance (i.e. combined micro-/meso-avoidance; section 5.4). For macro response rates (section 5.1) we consider data collected from the offshore environment only. The more limited evidence base for meso-response and micro-avoidance rates (sections 5.2 and 5.3) meant that it was necessary to include some evidence from the onshore environment. The difficulties in obtaining estimates of collision rates in the offshore environment mean that the majority of the evidence that relates to within-windfarm avoidance rates (section 5.4) originates from the terrestrial environment.

5.1 Review of Published Evidence for Macro-response Rates of Marine Birds

Here we consider macro-responses as including: (i) barrier effects for migrating birds or those commuting between breeding colonies and foraging areas; (ii) displacement effects from the windfarm area leading to an effective loss of habitat; and (iii) attraction. Each of these responses may result in a change in the numbers of birds in flight present within the perimeter of the windfarm between the pre- and post-construction periods. As collision risk modelling is usually based on the number of birds present during the pre-construction period, these changes must be accounted for as part of the collision risk modelling process. All of the studies we consider in this section originate from the offshore environment.

5.1.1 Causes of barrier, displacement and attraction effects

The term barrier effects describes the behavioural response of flying birds to the presence of the windfarm. The windfarm acts as a physical barrier, impeding the most direct route to a bird's destination, necessitating a change in flight direction in order to avoid entering the windfarm. This will ultimately reduce the numbers of birds recorded in flight within the windfarm area.

The effects of displacement are harder to classify since the habitat within the area of the windfarm may have been used by birds for a variety of purposes, notably foraging, but potentially other essential maintenance behaviours, such as moulting, preening or forming rafts. The availability of alternative foraging habitat may be more restricted, however, and hence for the purpose of this review we consider displacement as the inability of a bird to forage in a particular area due to the presence of the turbines. This may be manifested as a reduction in the number of birds flying into the area of the windfarm to look for food but this does not necessarily mean that birds will no longer enter the windfarm. It is possible, for example, that some species may land outside the windfarm and swim into the windfarm area. Studies of displacement, however, have tended to report the changes in all observed birds within the windfarm's perimeter relative to the areas outside and have not differentiated between the numbers flying and those recorded on the water. To better inform both studies of displacement and macro-avoidance, it would be prudent in future studies to separate flying birds from birds on the water when reporting displacement rates. Another important consideration relates to the flight height information that may be collected during surveys. This is primarily used to inform collision risk, but could potentially be used to inform on the vertical avoidance of birds over or under the rotor swept area.

Attraction is defined as an increase in numbers of birds within the windfarm area post-construction and can arise though several means. The monopiles of the turbine can act as a useful platform for birds to dry their feathers, rest, and socialise (e.g. great cormorant, Lindeboom *et al.* 2011). There is also evidence that structure of the turbines may also provide feeding opportunities through changes in local hydrography, seabed morphology or by acting as an artificial reef (Inger *et al.* 2009, Wilson & Elliot 2009, Maar *et al.* 2009, Lindeboom *et al.* 2011). Whilst there is the potential for collision risk to increase, as a result of attraction into the windfarm area, this will only occur if birds utilise the space covered by the rotor swept area.

5.1.2 Overall approach to assessing evidence for barrier, displacement and attraction effects

In reality, the ability to differentiate between birds exhibiting barrier and displacement effects may not always be possible since both are manifested as a decrease in the numbers of birds within the windfarm area (as defined both horizontally and vertically). For the purpose of this review, however, we will critique studies carried out at windfarms according to the type of effect they were designed to look at. For each example we present the relevant methods, key results and an overall assessment of the appropriateness of their use in looking at the effect they were designed to measure. Although our brief was to examine the evidence for five key species being considered in this review, we have also included several examples which have been cited as providing supporting evidence of macro-avoidance for seabirds in general (e.g. Desholm & Kahlert 2006 study on common eider and geese spp). We have not included studies carried out solely on migrating terrestrial species, e.g. such as pink-footed geese at Lynn and Inner Downing windfarm (Plonczkier & Simms 2012).

5.1.3 Studies of barrier effects

5.1.3.1 Methodologies used to look at barrier effects

Barrier effects have been measured mostly using (horizontal) radar and/or visual observations from fixed observation points (see Table 5.1 for summary). Radar technology has been used to measure barrier effects directly by quantifying the percentage of bird tracks that are deflected away from the windfarm, and also to look at the distance at which the deflection occurs (e.g. Peterson *et al.* 2006). However, due to technological constraints of horizontal radar (see below), this has been limited to quantifying horizontal macro-responses only.   Radar has also been used to look at barrier effects indirectly by comparing the number of flight paths (tracks) inside and outside the windfarms (e.g. Krijgsveld *et al.* 2011), to look at the densities of tracks in relation to distance from the windfarm (e.g. Skov *et al*. 2012) or to look at percentages of flight paths flying towards, away from and parallel to the windfarm (Blew *et al.* 2008). Such indirect measures may not necessarily be able to differentiate between barrier and displacement effects however. Visual observations, whilst also critical for the validation of the results of the radar, in terms of providing species identification and relative abundance, have also been used independently to compare numbers of birds in flight inside and outside the windfarm (e.g. Krijgsveld *et al.* 2011) although again, these methods may not necessarily preclude the possibility of inadvertently measuring displacement effects. Emerging technology in the form of laser range finders has also recently been used (e.g. Skov *et al.* 2012), and there may be scope to apply this approach in the context of barrier effects. There have been examples, notably in the UK, where data collected from boat based surveys have been used to look at barrier effects but this methodology is not considered to adequately provide the quantification needed here (MMO 2014).

There are a number of limitations associated with the use of radar (for further discussion see Krijgsveld *et al.* 2011 and Peterson *et al.* 2006) in terms of deriving avoidance rates: (i) Identification to the species level is not possible without visual validation and even then this information is generally only available as the species composition of birds passing through in a comparable time period – hence the values cited may be considered relevant only to the most commonly recorded birds species; (ii) There can be problems with distinguishing between flocks or individual birds – tracks recorded by radar may therefore not necessarily correspond to individual birds and corresponding avoidance rates could be more representative for flocks (which are likely to vary in size); (iii) Detection issues exist with picking up individual birds or flocks of small birds; (iv) Detection of birds can be affected by environmental conditions such as wave height and rain; (v) Seabirds such as the northern gannet, tubenose spp, sea duck spp and alcid spp tend to fly in the troughs between waves (as a means of flying in the most energetically efficient manner). In conditions where the wave height is sufficiently high, the total number of these birds is likely to be underestimated; (vi) The relative orientation of the radar beam to the flight direction of the birds can also affect detection (flying head on into the beam is the best) – this can present challenges when considering the optimum position for the radar; (vii) Whilst the range of detection for radar exceeds that of visual observations, there is a risk that birds could start to change their flight orientation beyond the range of the radar which would result in birds not being detected at all and hence the relative contribution of barrier effects to macro-responses is underestimated; (viii) Detection rates have been shown to be lower inside the windfarm due to interference caused by the presence of the windfarms (this is covered more extensively under the site accounts). Another considerable limitation of radar is that horizontal radar can only be used to record horizontal displacement (sometimes referred to as lateral displacement) as no information on altitude is collected. It is possible, therefore, that birds may fly over the windfarm at altitudes higher than the rotor swept area but this would not be picked up as avoidance behaviour (Blew *et al.* 2008). In contrast, vertical radar can only be used to determine flight height (altitude) and densities of birds in passage (flux) directly above the radar itself and provides insufficient information either on horizontal change or vertical avoidance that takes place outwith the windfarm perimeter. Radar has been useful, however, in demonstrating the importance of time of day (day versus night time), wind direction (head versus tail wind), season (spring versus autumn) for avoidance rates (e.g. Peterson *et al.* 2006 and Krijgsveld *et al.* 2011).

In terms of data collection issues for visual observations, there are also limitations when compared to radar: (i) Sampling is limited to daylight with reasonably calm conditions and good visibility. Although, under some circumstances, observations at night (e.g. moon watching) or auditory observations (based on bird calls) have been used, these have limited use; (ii) The range of detection is smaller; (iii) Individual observers may differ in assessing the distance and altitudes of birds, although there may be scope to reduce such differences through calibration with other techniques (Mateos *et al.* 2010; Norman *et al.* 2005).

5.1.3.2 Results of studies on barriers effects

Overall there is very little species-specific evidence for the five priority species for macro-avoidance as a consequence of barrier effects (see Appendix 1 for detailed site accounts) as radar was the most commonly used method. Of the studies reviewed all but one study looked at barrier effects during the post-construction period only – the exception being Nysted (Desholm & Kahlert 2005; Peterson *et al.* 2006) which also looked at the pre-construction period. Arguably comparison of the pre- and post-construction periods provides the most robust evidence for barrier effects rather than focussing solely on the post-construction period. Avoidance rates were only derived for three windfarms (see Table 5.1): (i) Egmond aan Zee (Krijgsveld *et al*. 2011); (ii) Nysted (Desholm & Kahlert 2005; Peterson *et al.* 2006); and Horns Rev (Peterson *et al*. 2006). The latter two are not considered further here since the derived values are likely only to be relevant to common eider (and geese) and common scoter respectively. Whilst there has been some additional work carried out at the Alpha Ventus test site to look at barrier effects (BSH 2011 and Mendel *et al.* 2014), the data have not been presented in such a way that would allow the derivation of a macro-avoidance rate and are hence not considered further here.

The only study which has specifically looked at barrier effects for northern gannet was that of Krijgsveld *et al.* (2011) at Egmond aan Zee which derived a macro-avoidance rate of 0.64. This was derived from indirect measure of barrier effects using visual observations made during panoramic scans to calculate the number of birds in flight within, at the edge and outside the windfarm (and by using a factor to correct for relative surface area – see Appendix 1, section A1.1). It is therefore not possible to discount the possibility that the apparent decreases within the windfarm could have included displacement due to the methodology used. These data were based on a total of 405 panoramic scans from spring 2007 to the end of 2009 (see Table 4.2 Krijgsveld *et al.* 2011) with particular emphasis on the spring and autumn periods as a total of 140 and 121 scans were carried out respectively compared to 71 and 73 scans in the summer and winter respectively. Overall, the sample sizes of the numbers of flying birds observed for northern gannet and common scoter were 282 and 123, although these figures were not broken down on a seasonal basis. However, it is also worth highlighting that northern gannets' use of the area – based on the density of flying birds – was highest during the spring, autumn and winter with an order of magnitude less use during the summer (mean density or numbers of birds per km^2 per scan for the periods of spring, summer, autumn and winter were 0.03, <0.005, 0.05 and 0.02 respectively – see Table 8.3 Krijgsveld *et al.* 2011). The extent to which the derived macro-avoidance rate is representative of breeding birds is thus questionable due to the relatively low use of the Egmond aan Zee site at this time and the lower sampling frequency. Therefore until such time that data are collected on northern gannet flights around OWFs specifically during breeding, this value should be applied with caution when considering the breeding season. It is also worth noting that Krijgsveld *et al.* (2011) reported a deflection rate of 0.89 for northern gannet based on the assessment of visual observations of flight paths. However, this result was based on a sample size of 38 birds and these observations were not based on systematic recording methods (c.f. the panoramic scans, which were based on strict protocols and recorded all birds seen). Consequently the authors do not recommend that these values be used as macro-avoidance rates (Karen Krijgsveld *pers. comm.*). Note, however, that these deflection rates have been cited as evidence for macro-avoidance rates by industry (e.g. Natural Power 2013).

There are no species-specific macro-avoidance rates, relating to barrier effects, for any of the four priority gull species of this review. Arguably, the most relevant study is that of Krijgsveld *et al*. (2011) which derived a macro-avoidance rate of 0.18 for the generic group of gull spp. These data were based on the indirect measure of barrier effects of the relative percentage of tracks that were outside the windfarm in winter. This was justified on the grounds that the species composition of bird tracks was heavily dominated by gulls spp (and great cormorants) at that time of year. A deflection rate of 0.4 was reported (based on the flight paths for 78 birds recorded as gull spp) but, as before, this value is not derived from systematic recording methods and the authors do not recommend this as evidence as macro-avoidance.

Table 5.1 Summary of key studies of barrier effects, the stage of data collection, methods used, parameters measured and species or species groups reported. *Italics* indicates species for which values were based on averages of other species (see Appendix 1 for site accounts)

Windfarm site	Study	Stage of data collection	Method used/parameter measured	Species/spp groups (values of macro-avoidance are given in parentheses where available)	Time of year data collected
Egmond aan Zee	Krijgsveld *et al.* (2011)	Post-construction	Radar /Numbers of tracks inside and outside the windfarm	*Gull spp (0.18)* *Grebe spp (0.28)* *Tubenoses spp (0.28)* *Skua spp (0.28)* *Tern spp (0.28)*	Winter All year All year All year All year
			Visual observations /Numbers of birds inside and outside the windfarm	*Northern gannet (0.64)* *Seaducks/scoter (0.71)* *Diver spp (0.68)* *Alcid spp (0.68)*	All year All year All year All year
Horns Rev I	Peterson *et al.* (2006)	Post-construction	Radar / The percentage of tracks that were considered to have a theoretical chance of entering the windfarm Radar / The distance at which deflection occurs	Common scoter (range 0.71-0.86 based on inter-annual variation and the direction at which birds approach the windfarm)	Spring/autumn combined
Horns Rev I	Blew *et al*. (2008)	Post-construction	Radar - Orientation of tracks in relation to the windfarm (% flying towards, away or parallel to the windfarm)	All birds	Spring/autumn combined
Horns Rev I and II	Skov *et al*. (2012)	Post-construction	Radar / Densities of tracks in relation to the radar station and windfarm	Common scoter and all birds	Spring/autumn combined

			Laser range finders / The distance at which deflection occurs (based on peak densities of radar tracks)		
Nysted	Peterson *et al.* (2006) Desholm & Kahlert (2005)	Pre- and post-construction	Radar / The percentage of tracks that were considered to have a theoretical chance of entering the windfarm Radar / The distance at which deflection occurs	Common eider and geese spp (0.78 – inter-annual variation 0.63-0.83)	Autumn
Nysted	Blew *et al*. (2008)	Post-construction	Radar / Orientation of tracks in relation to the windfarm (% flying towards, away or parallel to the windfarm)	All birds	Spring/autumn combined

5.1.4 Studies of displacement/attraction

5.1.4.1 Methodologies (and survey design) used to look at displacement/attraction

Data used to look at displacement effects have tended to be derived from boat and/or visual aerial surveys. Whilst industry guidance (Camphuysen *et al.* 2004) tends to be cited as the basis of the methodologies used, the extent to which guidelines are followed may be unclear (Maclean *et al.* 2009). Although digital aerial surveys are becoming more commonly used by the offshore windfarm industry (MMO 2014, Mackenzie *et al.* 2013), there appears to be a lack of sites where this technology has been used during all phases of the development. Further consideration of the advantages and disadvantages of these methodologies are given in Mackenzie *et al.* (2013). Additional to these, and of relevance to the assessment of displacement, there are concerns that boat surveys may overlook birds flying at higher altitudes and that might, therefore, fly over windfarms (Hartman *et al.* 2012). As is true for most of the studies designed to target barrier effects which may not necessarily exclude displacements effects, the same is true for the reverse situation.

Studies of displacement effects carried out at offshore windfarm sites within the UK have largely been based on the Before and After Control Impact (BACI) design which was viewed as being best practice at the time these sites were being set up (based on Stewart-Oaten *et al.* 1986). The extent to which this approach has been successfully implemented has been hampered by inadequate survey design including: (i) Location of the reference site often situated immediately adjacent to the impact site of the windfarm area – thus any changes as result of the windfarm may be over- estimated e.g. displaced birds could move into the adjacent area resulting in higher numbers recorded than during the pre-construction period; (ii) Insufficient spatial coverage e.g. boat surveys often only covered the windfarm area and a buffer, hence any possible changes that may have occurred in the wider environment cannot be taken account of; (iii) Gaps in temporal coverage e.g. survey periods between the different phases of the development did not always correspond or visual aerial surveys having to be abandoned following construction of the windfarm, due to Civil Aviation Authority flight height restrictions; (iv) The ability to select control sites which are truly comparable to the area impacted by the windfarm area (e.g. in terms of hydrography, seabird populations) has been questioned. For further consideration of these issues see MMO (2014).

A further limitation of displacement studies in their survey design is that little consideration is usually given to the power to detect change, which is related to a number of factors including the frequency of surveys and their relative spatial and temporal coverage (Maclean *et al.* 2013; Vanermen *et al.* 2012; Pérez Lapeña *et al.* 2010). The distribution and relative abundance of seabirds show high levels of both spatial and temporal variability within and between years. Therefore the use of power analyses, particularly at the start of any offshore windfarm development, can be extremely helpful in determining the most appropriate survey design in order to be able to adequately test for whether a windfarm impacts birds through either displacement or attraction effects.

There is also the problem that the post-construction reports, notably those leading up to the final report, have tended not to provide formal statistical analyses and any assessments of changes in species abundance are often based on simple comparisons of changes in absolute numbers or are qualitative (e.g. visual inspection of maps: MMO 2014). Even in instances where the significance of change has been looked at, the focus has been on measuring differences in numbers or densities between the pre-construction and post construction periods and any changes in distribution within the study area may go undetected (MacKenzie *et al.* 2013). Recently, there have been developments in model-based approaches such as density surface modelling (Rexstad 2011) which allow the inclusion of covariates (e.g. environmental factors such as water depth, sea surface temperature) which can help better explain inherent spatial and temporal variability in the abundance and distribution of animal populations. The resulting distribution maps of relative abundance provide a more robust means of assessing whether changes have occurred as a result of the presence of an offshore windfarm. There also appears to be a shift away from using BACI survey design for looking at displacement, with a Before-After-Gradient approach being recommended (MMO 2014, Jackson & Whitfield 2011), and this is highly compatible with density surface modelling approaches.

It is also important to highlight that displacement studies to date have tended to focus on comparing numbers or densities of birds pre-construction and post-construction which, in general, do not distinguish between birds in flight and birds on the water (the former group being more likely to show displacement). Despite ship-based data collection methods being distinct for birds on the water and birds in flight, counts are generally combined and for most studies presented below are not considered separately. Similarly, whilst visual aerial surveys do differentiate between birds on the water and those in flight, estimates are usually collated.

5.1.4.2 Results of studies on displacement/attraction

Of the studies considered, comparisons of pre- and post-construction surveys were carried out in all cases with the exception of Egmond aan Zee (Leopold *et al.* 2011), where it was argued that this was not possible due to considerable annual variation in seabird presence (Appendix 2, section A2.1). Instead analysis of the effect of the windfarm was carried out based on individual surveys (e.g. species monthly counts which were converted into presence/absence data) for which there were sufficient data and the results should therefore be considered with caution (see Table 5.2).

It was only possible to calculate actual values of macro-avoidance for a single study carried out at the Blighbank and Thorntonbank windfarms for which the model co-efficients generated from the Generalised Linear Models were provided (Vanermen *et al.* 2013). Results for Thorntonbank are not considered here, however, as they relate either to the first post-construction phase when only six turbines were operational or during the second phase of construction which was still ongoing at the time of reporting. Other studies have reported evidence for displacement or attraction based on the results of Jacob's selectivity indices (Nysted and Horns Rev - Peterson *et al.* 2006) or density surface maps of the predicted distribution over the different phases of the development (Robin Rigg – Natural Power 2014).

For northern gannet there was strong evidence for displacement effects at Blighbank based on comparisons of pre- and post-construction data. From this study, therefore, it was possible to derive a macro-avoidance rate of 0.84 for northern gannet. Currently the vast majority of monitoring tends not to present a seasonal breakdown of displacement (macro-avoidance) values and this report does not differ in that respect. However, there is notable variation in the seasonal use of the windfarm and the surrounding area (termed the BPNS) by northern gannet – mean numbers across the period of 2001-2007 in winter and autumn were 1,799 and 4,990 respectively compared to spring and summer at 737 and 556 respectively (see Table 2 in Vanermen *et al*. 2013). Therefore, as for barrier effects, the extent to which these data are representative of northern gannet during the breeding season is debateable. It is also worth reflecting that further monitoring work has been carried out since the publication of Vanermen *et al*. (2013) and that these results should be considered as being provisional (Nicholas Vanermen *pers. comm*.). Potential corroboration that northern gannets are displaced by windfarms is also provided by results from Egmond aan Zee (Leopold *et al.* 2011) where it was shown that the presence of northern gannets was significantly negatively related to the presence of the windfarm in two of nine monthly post-construction surveys (no other significant effects were reported for the other seven surveys). However the strength of this evidence is relatively weak as the analyses were based on within survey (monthly) comparisons – a comparison of pre- and post-construction data was not considered feasible (see Appendix 2 – A2.1 for further details). The study at Robin Rigg found no response from northern gannet to the windfarm which tend to use the site mainly during the breeding season, though the pre- and post-construction densities were generally rather low e.g. across the entire study within which the windfarm is located, a total of 352 birds were recorded in flight for the entire pre-construction period compared to 397 in the post-construction period (up to and including year 3 - see Table 3.22 in Natural Power 2014). Similarly at Horns Rev, there were never any birds recorded within the windfarm itself either pre- or post-construction (although an increased avoidance was reported for both the 2 km and 4 km buffers post-construction based on Jacob's selectivity indices). More recent work carried out at Alpha Ventus was inconclusive as the overall abundance of northern gannet was very low e.g. a total of nine individuals were seen in the pre-construction period (BSH 2011 and Mendel *et al.* 2014).

Lesser black-backed gull was only considered by three of the studies reported in Table 5.2. There was strong evidence of very high levels of attraction at Blighbank (Vanermen *et al.* 2013) – with relative increases in numbers at the windfarm provisionally estimated in the order of 3.81 (see Appendix 2, section A2.3 for further details). Far weaker evidence to support lesser black-backed gulls being attracted to windfarms was provided from Egmond aan Zee (Leopold *et al.* 2011) where the presence of lesser black-backed gulls was significantly negatively related to the presence of the windfarm in at least one out of 12 possible monthly post-construction surveys. However, despite the results being suggestive of displacement, it was concluded by the authors that, given the strong association shown by lesser black-backed gulls to fishing vessels (based on anecdotal observations during the surveys) attraction to the windfarm was apparently being masked by their strong association with boats which were excluded from the windfarm in the post-construction period. Completely contradictory results were derived for Alpha Ventus where comparison of the distribution of birds pre- and post-construction showed a marked decrease in densities (based on maps of 1 km^2 cells) and statistically significantly lower abundances were reported for the 0-2 km, 2-6 km and 6-10 km distance classes from the windfarm (BSH 2011 and Mendel *et al.* 2014). There is also no consistent pattern in the studies summarised in Table 5.2 for either displacement or attraction being shown by herring gulls, great black-backed gulls and black-legged kittiwake (Table 5.2).

Furness *et al.* (2013) developed a scoring system to quantify the vulnerability of marine bird population to offshore windfarms with respect to collision and disturbance/displacement. Northern gannet, lesser black-backed gull, herring gull, great black-backed gull, and black-legged kittiwake all scored very highly with respect to collision risk (within the top seven of all the species considered) and this was largely a result of time spent flying at rotor height (other parameters considered included flight agility, % of time flying, night flight and an overall conservation score). In contrast, with respect to displacement, all five species scored very low (species concern index values were no more than 6 compared to the highest value of 32). This was a result of the species being little affected by the disturbance effects associated with ships/helicopters and not being particularly constrained by foraging habitat (the same overall conservation score used for collision risk was also used with respect to displacement). Given this, it is therefore unsurprising that the majority of priority gull species appeared to show no consistent pattern for displacement.

5.1.5 Evidence for an overall macro-response rate

In terms of assessing whether changes in numbers (e.g. from the pre-construction to post-construction periods) are statistically significant, this has only been possible for displacement /attraction studies and not for barrier studies. The notable exception to this is the work carried out at Nysted windfarm (Desholm & Kahlert 2005; Peterson *et al.* 2006) where it was possible to record the number of flight paths that changed their direction by comparison of the pre-and post- construction periods.

There are also considerable issues in how data are collected in terms of differentiating between barrier and displacement effects. Migratory species, which have a distinct passage period during spring and/or autumn and do not occur in the vicinity of the windfarm outwith these periods (e.g. geese spp and passerine spp), are likely to experience solely barrier effects. In contrast, species which are resident in the vicinity of the windfarm, may be subject to a combination of barrier effects or displacement /attraction effects (e.g. the vast majority of seabird spp, at least in the breeding season). This is certainly the case for all of the five priority species being considered here and to date, there has not been a single study which can be considered as exclusive evidence for either barrier or displacement effects.

It is also worth flagging up that the extent to which impacts of the windfarm actually affects bird populations is likely to be site specific. Therefore it would be reasonable to expect that the barrier effects for migrating birds are far more likely to be pronounced when offshore windfarms are located on major flyways. Similarly, an offshore windfarm that is located within the foraging ranges of breeding seabirds is more likely to be an issue in terms of barrier and displacement/attraction effects compared to one that is not (although the latter scenario is unlikely). Another consideration which has been picked up by this review occurs when the windfarm has relatively low numbers of certain species using the site pre-construction. This may give a misleading impression as to the extent of any changes pre- and post-construction. Whilst an increase or decrease in numbers between these periods may give the impression of a significant effect, the power to detect such a change is extremely low, and, as a consequence, we cannot have much confidence in these results.

Another important caveat related to all studies of barrier and displacement/attraction effects, is that there has been very little attention given to teasing out potential variation over the annual cycle and only a single value of relative change between pre- and post-construction is presented. Yet there may be significant time and energy constraints imposed by the breeding season when birds have to return repeatedly to the nest whereas at other times of year they can move more freely (Stephens *et al.* 1986). In addition to this shift due to the onset of the breeding season, notable changes in foraging behaviour within the breeding season have also been extensively documented in seabirds (e.g. black-legged kittiwake trip duration typically decreases from incubation to the chick rearing period due to the need to feed the young frequently Hamer *et al*. 1993). Hence, the response of foraging and commuting birds to the presence of a windfarm may vary according to the stage of their life cycle e.g. birds which are limited in terms of time or energy may be willing to take more risks by entering the windfarm when otherwise they would simply avoid the area. While, due to the absence of evidence, any such seasonal variation in birds' responses to the impacts of windfarms is hypothetical, when utilising derived macro avoidance rates, the extent to which these values are considered representative for all times of year should be given careful consideration, particularly if they contribute to the collision risk modelling. There may also be further scope in the future for investigating variation in macro-responses between the breeding and non-breeding seasons (although investigating within the breeding season differences may be more problematic).

Among the priority species considered by this review, there is limited evidence, however, to suggest that northern gannet may show a tendency towards a negative macro–response. The study of barrier effects at Egmond aan Zee, Krijgsveld *et al.* (2011) suggests a macro-avoidance rate of 0.64, while the study of displacement at Blighbank, Vanermen *et al.* (2013) suggests a macro-avoidance rate of 0.84. At this stage, the lower and therefore the most conservative of these values is assumed to be a reasonable macro-response rate.

In contrast, there is a lack of species- or even species group-specific evidence for barrier effects relating to gulls. With respect to displacement/attraction, the evidence is equivocal, with some studies suggesting evidence for attraction, others evidence for displacement, and others no significant response. For gulls, the balance of evidence thus suggests a macro-response of 0 (i.e. no attraction to or avoidance of the windfarm).

Table 5.2 Summary of key studies of displacement and attraction studies, the stages of development at which data were collected, main methods used, parameters collected, species reported and responses. Grey indicates species which were not covered by that particular study. For further information see Appendix 2 for site accounts.

Windfarm	**Study**	**Survey/s used**	**Modelling approach**	**Species**	**Response (values are given in parentheses where available)**		
					Displacement	**Attraction**	**None**
Blighbank	Vanermen *et al.* (2013)	Boat	Generalised linear models with a negative binomial distribution with count data as the response	Northern gannet	✓(0.84)^1		
				Lesser black-backed gull		✓(-3.81)^6	
				Herring gull		✓(-51.98)^6	
				Great black-backed gull			✓
				Black-legged kittiwake			✓
Egmond aan Zee	Leopold *et al.* (2011)	Boat	Presence/absence modelling of individual monthly surveys (Generalised Additive Modelling)^2	Northern gannet (10/2) ^{2}	✓		
				Lesser black-backed gull (12/1) ^{2}	✓		
				Herring gull (14/3) ^{2}	✓		
				Great black-backed gull (17/6) ^{2}	✓^{4}	✓^{5}	
				Black-legged kittiwake (5/1) ^{2}		✓	
Horns Rev	Peterson *et al.* (2006)	Aerial	Comparison of Jacob's Selectivity Indices Encounter rates per survey km (student's t-test)	Northern gannet			✓
				Lesser black-backed gull			
				Herring gull			✓
				Great black-backed gull			
				Black-legged kittiwake			

Windfarm	Study	Survey/s used	Modelling approach	Species	Response (values are given in parentheses where available)		
					Displacement	Attraction	None
Nysted	Peterson *et al.* (2006)	Aerial	Comparison of Jacob's Selectivity Indices Encounter rates per survey km (student's t-test)	Northern gannet			
				Lesser black-backed gull			
				Herring gull			✓
				Great black-backed gull			
				Black-legged kittiwake			
Alpha Ventus	BSH (2011) and Mendel *et al.* (2014)	Boat and aerial	Comparison of changes distribution patterns (1 km$^{2)}$ Generalised Linear Mixed Models with a Poisson error[7]	Northern gannet			✓
				Lesser black-backed gull	✓		
				Herring gull			
				Great black-backed gull			
				Black-legged kittiwake	✓		
Robin Rigg	Natural Power (2014)	Boat	Generalised Additive mixed effects mixture modelling within a Bayesian framework	Northern gannet -			✓
				Lesser black-backed gull			
				Herring gull	✓		
				Great black-backed gull	✓[3]		
				Black-legged kittiwake			✓

[1] See Appendix 2 for calculations

[2] The total numbers of post-construction monthly surveys for which there were sufficient data for modelling / the number of which the results were significant

[3] Between pre-construction and construction only

[4] Four surveys

[5] Two surveys.

[6] negative values for attraction;

[7] lesser black-backed gull only.

5.2 Review of Published Evidence for Meso-Response Rates of Marine Birds

5.2.1 Studies of meso-avoidance

Within a windfarm, birds may respond to the presence of a turbine either by altering the altitude at which they fly, termed a vertical meso-response, or by altering the flight path they take, termed a horizontal meso-response. This is distinct from micro-avoidance, which occurs as a ‘last-second’ reaction to avoid collision, as meso-responses may take place at some distance from the turbines but still within the windfarm site.

On entering a windfarm, birds may alter their horizontal flight path so that they fly around, or between, turbines, thereby lowering their risk of collision. Alternatively, they may make no response, or even be attracted to a turbine, as a potential roost or perch. In such circumstances, the risk of collision for each individual bird would remain the same, or increase. Such behaviours can be assessed by examining flight paths within the windfarm and considering whether these show a change in direction as they approach the turbines or considering whether birds approach turbines in the proportions that may be expected if they were randomly distributed within the windfarm.

Birds may also alter their flight heights in response to turbines. They may alter their flight heights so that they fly under, or above, the turbines in order to avoid collision. Alternatively, they may make no alteration to their flight height in response to encountering a turbine, meaning either they make a horizontal alteration to their flight path to avoid collision, or their risk of collision remains the same.

Avian flight heights are commonly assessed during surveys of onshore or offshore windfarms. However, concluding that a certain proportion of birds might fly below the rotor swept area of a turbine does not necessarily imply avoidance behaviour as seabirds commonly fly at low altitudes in the absence of turbines (Cook *et al.* 2012, Johnston *et al.* 2014a). In order to assess the scale of any vertical responses to turbines it is necessary to compare the proportion of birds flying at rotor height within the windfarm to data collected either prior to the windfarm construction, or to the proportion of birds flying at rotor height in control areas adjacent to the windfarm.

5.2.2 Horizontal meso-response conclusions

Evidence for the extent and direction of horizontal meso-responses to turbines is extremely limited (see Appendix 3). We identified two studies with relevant information from the onshore environment (Everaert 2008 and Janoska 2012) and two studies from the offshore environment (Skov *et al.* 2012 and Krijgsveld *et al.* 2011). At De Put in Belgium, no evidence of a response was recorded amongst either black-headed or common gulls (Everaert 2008). Similarly, the data presented for Horns Rev I and II in Denmark only support a meso-response for large gulls, with none of the 402 flight paths recorded passing within less than 50 m of a turbine (Skov *et al.* 2012). The data presented do not make it possible to determine whether meso-responses occur within northern gannet, common scoter or terns although, on average these species passed turbines at a greater distance than large gulls. Data from two terrestrial sites in Hungary also suggest a strong, meso-response for large gulls, with only 2.5% of birds flying within 75 m of a turbine (Janoska 2012). However, confidence in these data is extremely limited given the lack of detail available about the methodology of this survey. The strongest evidence for a meso-response rate from an offshore windfarm comes from Egmond aan Zee in the Netherlands. Here, the number of birds recorded by radar within 50 m of a turbine was 66% of those recorded elsewhere within the windfarm (Krijgsveld *et al.* 2011), reflecting a meso-response rate of 0.34, considerably lower than the meso-response rate reported in the Hungarian study.

However, it should be noted that measurements of the proportions of birds passing within a set distance of a turbine may not be an accurate reflection of the true meso-response rate. To estimate species' meso-response rates it is necessary to consider whether the proportion or density of birds in areas close to turbines is higher or lower than would be expected within the windfarm as a whole. This could, potentially, be assessed either through visual observations during surveys of the area, or with the use of remote tracking technologies, such as radar. At present, however, such data are too limited to reliably quantify the horizontal meso-response rates of birds within a windfarm.

5.2.3 Vertical meso-response rates conclusions

All evidence for vertical meso-response rates which we identified originated from the offshore environment (Table 5.3). The quality of evidence presented by each of these studies varies considerably (see Appendix 4). For example, at Blyth, there was a reported increase in the proportion of birds flying at altitudes of more than 9.1 m above mean sea-level between pre- and post-construction (Rothery *et al.* 2009). However, as the rotor sweep of turbines at this site is between 26.4 and 92.4 m above mean sea level, it is unclear as to whether, despite this apparent increase in flight height post-construction, there was a significant increase in the proportion of birds flying at rotor height. Similarly, data from Nysted and Horns Rev were collected by radar and cover all birds flying below 200 m above mean sea level and are also, therefore, likely to incorporate a significant number of birds flying outside the rotor sweeps at these sites (Blew *et al.* 2008). Due to the significant proportion of birds in both of these studies that are likely to fly outside the turbine rotor sweeps, it is not possible to obtain useful information about the level of vertical meso-responses from either. In addition, at Robin Rigg (Natural Power 2013) concerns have been raised about the power of the available data to detect changes in species' flight heights, and about the methodology used to collect data on species in flight which may have led to the double-counting of individuals. For these reasons, data from these sites are not considered further in this section.

Of the remaining sites, estimates of vertical meso-avoidance rates can be obtained from Barrow (Barrow Offshore Wind Farm Ltd) and Gunfleet Sands (NIRAS 2011, GoBe Consultants Ltd. 2012) by comparing the proportion of birds flying in different height bands pre- and post-construction, and at Egmond aan Zee (Krijgsveld *et al.* 2011) by comparing the proportion of birds at different heights inside and outside the windfarm. Of the species or groups for which data were available, only divers showed a consistent vertical response to turbines, in the form of a reduced proportion of birds at rotor height. Other species appear to show a full range of responses covering a strong vertical avoidance to a strong vertical attraction. For example, the proportion of northern gannet assessed to be flying at heights placing them at risk of collision increased by 59% between pre- and post-construction at Barrow, but the proportion at risk height at Egmond aan Zee within the windfarm was 49% lower than the proportion outside the windfarm. A similarly mixed picture is evident for each of the remaining four priority species. The differences in the methodologies used by each study and the inconsistency in the different results mean it is not possible to draw conclusions about the magnitude or direction of any vertical meso-response to turbines.

Table 5.3 Vertical meso-avoidance rates obtained from reviewed studies – see Appendix 4 for the origin/derivation of these figures. Values of 0 indicate no response, values <0 indicate an attraction response, values >0 indicate an avoidance response.

	Barrow	**Blyth Summer**	**Blyth Winter**	**Egmond aan Zee**	**Gunfleet Sands 2010/11**	**Gunfleet Sands 2011/12**	**Nysted / Horns Rev**	**Robin Rigg**
	Barrow Offshore Wind Farm Limited.	**Rothery *et al.* (2009)**	**Rothery *et al.* (2009)**	**Krijgsveld *et al.* (2011)**	**Niras (2011), GoBe Consultants Ltd, (2012)**	**Niras (2011), GoBe Consultants Ltd, (2012)**	**Blew *et al.* (2008)**	**Natural Power Consultants (2012)**
Diver spp					1.00	1.00		
Red-throated diver					0.39	0.86		
Northern gannet	-0.59	-27.00	*exp**	0.49				*exp**
Common scoter	-0.24	0.00	0.00					
Common eider		0.00	-2.00					
Great cormorant	*exp**	-1.38	-0.61	-0.38				*exp**
Arctic skua	-1.00							
Black-headed gull	0.56	0.07	-7.00	0.49	-10.70	1.00		
Common gull	*exp**			-0.20	-0.25	0.33		
Little gull				-0.65				
Black-legged kittiwake	-0.41	-0.14	-28.00	0.20	-0.47	0.05		-1.00
Small gulls				-0.26				
Lesser black-backed gull	0.72			-1.00	-0.44	0.00		
Herring gull	0.35	-1.16	-1.67	0.02	-0.02	0.11		-8.00

Great black-backed gull	0.28	-1.38	-3.50	0.17	-0.75	-0.53		-0.67
Large gulls				-0.01				
Gull spp	-0.85			0.45	-1.98	-1.13		
Sandwich tern	0.56	-1.94		0.35				
Common guillemot	-1.00							
Auks spp	-1.00							
Daytime migrants <200							0.17	
Night time migrants <200 m							0.18	

*Increase in the number of birds recorded from 0 in pre-construction surveys, meaning it is not possible to calculate a proportional increase.

5.2.4 Meso-response rates conclusions

Data quantifying meso-response rates to turbines within offshore windfarms are extremely limited and of variable quality. Overall, evidence describing horizontal meso-responses appears to be stronger than the evidence for vertical meso-responses. Data from one onshore (Janoska 2012) and one offshore site (Krijgsveld *et al.* 2011) appear to suggest a moderate, negative horizontal meso-response to turbines. Whilst there was a stronger meso-response rate at the onshore site, a lack of methodological detail made it difficult to understand the reasons for this difference. Furthermore, an additional two studies did not offer evidence of a horizontal meso-avoidance rate (Everaert 2008, Skov *et al.* 2012). As all four studies we identified had limitations at this stage it is not possible to be confident about the magnitude of any horizontal meso-response, particularly at a species specific level. Whilst a greater quantity of data were available describing vertical meso-responses to turbines, the variable nature of these data and limitations associated with each study, mean it is not possible to draw firm conclusions about either the magnitude or direction of any vertical meso-response. Particular concerns included the low power of some of the datasets, and a lack of overlap between the height bands assessed and the rotor-swept areas of the installed turbines.

However, some studies do indicate how meso-responses may vary within-windfarms. Data from Horns Rev suggest that as birds travel further into a windfarm, they respond more strongly to turbines, with a greater number of directional changes in response to the third or fourth turbine rows than to the first or second rows (Petersen *et al.* 2006). Similarly, the operational status of turbines may influence species responses. Again at Horns Rev, common scoter, Arctic skua, herring gull, great black-backed gull, kittiwake, common/Arctic tern and Sandwich tern were all found to be less likely to pass by operational than non-operational turbines. This response is even stronger when considering birds passing between two adjacent turbines which are both either operational or non-operational (Petersen *et al.* 2006). Similar results have been found at Alpha Ventus and Egmond aan Zee, where concentrations of birds were higher when turbines were non-operational than when they were operational (Krijgsveld *et al.* 2011, Mendel *et al.* 2014).

5.3 Review of Published Evidence for Micro-Avoidance Rates of Marine Birds

5.3.1 Studies of micro-avoidance

We consider micro-avoidance to be the 'last-second' action taken to avoid collision with a turbine. In practice, this can be difficult to measure given the effort required to generate meaningful data. Several strategies have been employed to collect such data including: direct observations of bird interactions with turbines, using radar to track birds as they approach turbines and fitting cameras to turbines to record interactions. Interpretation of these data may be challenging and necessitate subjective judgements in relation to whether a bird is at risk of collision and what behavioural responses reflect a reaction.

5.3.2 Micro-avoidance conclusions

Data describing the 'last-second' response of birds to turbines have been collected from 16 individual turbines, of which 14 were offshore and two were onshore, across four sites for in excess of 3,000 hours (Desholm 2005, RPS 2011, Krijgsveld *et al.* 2011, Wild Frontier Ecology 2013; see Appendix 5). Despite this effort, very few birds have been recorded passing close enough to turbine rotors to necessitate micro-avoidance action. In total, 45 birds (excluding those recorded at Nysted in Denmark, which were not recorded passing within less than 20 m of turbines, Desholm 2005) have been recorded passing close enough to turbines to necessitate some form of avoidance action, and at least 42 of these have been recorded as taking some form of avoidance action (RPS 2011, Krijgsveld *et al.* 2011, Wild Frontier Ecology 2013). The remaining three birds were tracked at Egmond aan Zee in the Netherlands, using radar and it is unclear whether or not these may also have taken some form of avoidance action, although they were not recorded as colliding with the turbines (Krijgsveld *et al.* 2011).

These data suggest that last-second action to avoid collision is an extremely rare event. This is not because birds do not respond to turbines, but because most avoidance action takes place at distances from the turbines beyond which the methodologies in the studies above could record (i.e. at the meso- and/or macro scales). Whilst only limited data are available describing micro-avoidance rates, the 45 flights considered in the studies described above suggest that a high proportion of birds, >0.93 based on the data described above, may take last second action to avoid collision.

5.4 Review of Published Evidence for Within-Windfarm Avoidance Rates of Marine Birds

5.4.1 Background

In addition to monitoring behavioural avoidance of birds at windfarms, as described in the micro-avoidance and meso-response sections above, a key part of the post-construction monitoring programmes at onshore windfarms is recording the incidence of collisions between birds and turbines. This is typically achieved through organised searches at regular intervals around turbine bases (e.g. Winkelman 1992, Thelander *et al.* 2003, Everaert 2008). Corrections are then applied to account for factors including searcher efficiency and the removal of corpses by scavengers (e.g. Winkelman 1992). These records are often presented as a collision rate per turbine per year (e.g. Winkelman 1992, Musters *et al.* 1996, Brown & Hamilton 2004, 2006, Grunkorn *et al.* 2009). Whilst such values may provide a useful comparison of collision risk between individual turbines within a windfarm, or between windfarms in general, they do not, by themselves provide useful information about the behavioural responses of birds to the presence of turbines.

In order to use collision rates to derive meaningful information about the behavioural responses of birds to the turbines, it is necessary to combine them with an estimate of the rate at which birds pass through the windfarm. Estimates of the rate at which birds pass through the windfarm can be derived by converting the total number of birds observed over a known period of time into an hourly, or daily rate. These flux rates can then be multiplied by the total length of the study period, taking care to correct for factors such as variable day length, to estimate the total number of birds passing through the windfarm during the period in question – for example, the months over which searches were made for collision victims. It may also be necessary to rescale these estimates, for example if only a proportion of the windfarm was covered during surveys. However, as movement data refer to the windfarm as a whole, it is not possible to separate the meso and micro elements of these mortality derived avoidance rates. For this reason, these are collectively referred to as within-windfarm avoidance rates.

5.4.2 Methodology

5.4.2.1 Deriving within-windfarm avoidance rates

We identified 20 sites at which data were available combining an estimate of the collision rate with an estimate of the rate of flux through the windfarm that made it possible to derive within-windfarm avoidance rates (see Appendix 6). Of these, 17 sites were onshore and three were offshore. Using the methodology set out in Band (2007) it is possible to calculate the number of birds expected to collide with turbines at each of these sites if no avoidance action is taken.

The first step of this process is to estimate the total number of birds likely to have passed through the windfarm during the period in which collisions were recorded. As surveys are not, typically, carried out continuously over the study period, the number of birds recorded must be converted to an hourly rate. The total number of birds passing through the windfarm is then estimated by multiplying the hourly rate by the total number of hours over the study period as a whole, with a correction applied to account for the nocturnal activity level of the species concerned, based on the figures reported in Garthe and Hüppop (2004). However, it should be noted that the difficulty of collecting data on nocturnal activity in seabirds means it is unclear how accurate these figures are.

The next step is to use this value to estimate the total number of birds likely to pass through the turbine rotor sweeps. The total number of birds flying through the windfarm is multiplied by the proportion estimated to fly at rotor height, based on the original survey data. This value is then converted to the number of flying birds per m^2 and multiplied by the total area occupied by the turbine rotors.

A significant proportion of the birds passing through the turbine rotors are likely to do so without colliding (Band 2007). Therefore, a correction, the Probability of Collision (P_{coll}), must be applied to the data to account for this. This is calculated based on the turbine specifications, design of the windfarm array and the flight behaviour and morphometrics of the species of interest and based on the methodology set out in Band (2007). Species morphometric and behavioural data used to estimate P_{coll} are given in Table 5.4, whilst turbine details for each site are given in Table 5.5.

Table 5.4 Bird parameters to estimate P_{coll} for each windfarm. Speed data taken from Pennycuick (1997) and Alerstam *et al.* (2007), morphometric data from Robinson (2005), where species groups are given, data come from a species likely to be representative of that group as a whole, within the offshore wind context.

Species	Length (m)	Wingspan (m)	Speed (m/s)	Flap/glide
Diver (red-throated diver)	0.61	1.11	14.50	flap
Grebe (great crested grebe)	0.48	0.88	18.65	flap
Northern gannet	0.94	1.72	14.90	glide
Arctic skua	0.44	1.18	13.30	flap
Great cormorant	0.90	1.45	14.50	flap
Common eider	0.60	0.94	18.65	flap
Common scoter	0.49	0.84	18.65	flap
Long-tailed duck	0.58	0.88	18.65	flap
Black-headed gull	0.36	1.05	9.50	flap
Common gull	0.41	1.20	9.50	flap
Black-legged kittiwake	0.39	1.08	13.10	flap
Franklin’s gull	0.41	1.20	18.65	flap
Ring-billed gull	0.41	1.20	9.50	flap
Little gull	0.26	0.78	11.50	flap
Lesser black-backed gull	0.58	1.42	13.10	flap
Herring gull	0.60	1.44	12.80	flap
Great black-backed gull	0.71	1.58	13.70	flap
Little tern	0.23	0.52	10.00	flap
Common tern	0.33	0.88	10.00	flap
Sandwich tern	0.38	1.00	10.00	flap
Auk (common guillemot)	0.40	0.67	19.10	flap

Table 5.5 Turbine data used for each site. Figures in red indicate that the parameter was not presented for site in question and had to be estimated from a turbine with a similar design. Row colours indicate confidence assigned to data collected at each site – green indicates highest confidence, where there was both spatial and temporal overlap in the collection of corpse and movement data; yellow indicates moderate confidence where there was temporal overlap in the collection of corpse and movement data, but incomplete spatial overlap, meaning that bird activity had to be extrapolated across the site; red indicates lowest confidence, sites where there was incomplete spatial and temporal overlap in the collection of corpse and movement data, meaning bird activity had to be extrapolated both spatially and temporally; grey indicates studies in which flights through the windfarm were recorded so that collisions could be directly recorded, such studies typically had very little power.

Windfarm	N turbines	Turbine capacity (MW)	Width of survey window (m)	Height of survey window (m)[1]	N blades	Blade width (m)	Rotor diameter (m)	Rotor speed (rpm)	Pitch (degrees)	Hub height (m)
Altamont	685	0.12	7713.624[2]	33.5	3	0.66	19	43.0[25]	10	24.0
Blyth	2	2.00	600[3]	92.4	3	4.40	66	18.0[26]	10	59.4
Blyth Harbour	9	0.30	925[4]	37.5	3	66[32]	25	43.0	10[32]	25.0
Boudwijnkanaal	5/7/14[5]	0.6	1040/1536[6]	79	3	1.10	48	43.0[25]	10	55.0
Bouin	8	2.5	4000[7]	100	3	4.40	80	18.0[26]	10	60.0
Buffalo Ridge	143	0.75	9600[8]	74	3	1.10	48	32.3	10	50.0
De Put	2	0.8	300[9]	100[26]	3	1.10	48	43.0[25]	T10	75.0
Gneizdzewo	19	2.00	3700[10]	120	3	4.40	80	18.0[26]	10	80.0
Greater Gabbard	7	3.6	4000[11]	180[27]	3	4.20	107	15.0[27]	10	77.5
Groettocht	5	1.65	1000[12]	140[28]	3	4.40	66	21.3	10	78.0
Haverigg	8	0.6	920[13]	66	3	1.10	42	43.0[25]	10	45.0
Kauwnee County	31		12247[14]	89	3	1.10	47	43.0[25]	10	65.0
Kessingland	2		800[15]	126	3	2.50	92	15.0	10	80.0

Kleine Pathoweg	7	1.8	1820[16]	120	3	4.40	70	18.0[26]	10	85.0
Oosterbierum	18	0.3	1430[17]	60[18]	3	0.66	30	43.0[25]	10	35.0
Waterkaaptocht	5	1.65	1000[12]	140[28]	3	4.40	66	21.3	10	78.0
Yttre Stengrund	5	2.0	20000[19]	250[29]	3	4.40	72	10.0	10	60.0
Zeebrugge	4/6/23/24/25[20]	0.4[21]	400/720[22]	65/80[23]	3	0.66	34	43.0[25]	10	34.0
Hellrigg	4	2.3	4000[24]	121	3	4.40	82	18.0[26]	10	80.0
Avonmouth	3	2.00	1300[30]	160[31]	3	4.40	83	17.5	10	79.0

[1]Maximum turbine height unless otherwise stated; [2]see Table 2 of Thelander *et al.* 2003, total survey area of 59.5 km^2, width of survey window assumed to be $\sqrt{59.5} * 1000$; [3]Rothery *et al.* (2009) state 600 m scan area; [4] Lawrence *et al.* (2007) state that observations were carried out between turbines 5-9, turbines separated by 200 m with a rotor diameter of 25 m and arranged in a single line; [5]Collisions were recorded under all 14 turbines in 2002-2006. In 2001, bird activity surveys were carried out around five turbines and avoidance rates derived from collisions around these turbines are also presented. In 2005, bird activity surveys were carried out around seven turbines and avoidance rates derived from collisions around these turbines are also presented; [6]In 2001, only five turbines were present with diameters of 48 m and spacing of 200 m, therefore, the total survey window in 2001 was 1,040 m wide (section 3.3.1 in Everaert *et al.* 2002, Table 27 in Everaert 2008). In 2005, 14 turbines were present, but activity was only monitored around seven of these, therefore in 2005 the total survey window was 1,536 m wide (Table 27 in Everaert 2008). Turbines were all arranged in a single line; [7]Observations carried out along four 1 km linear segments on the edge of the windfarm, see section 5.1 of Dulac (2008); [8] Raptor/large bird surveys carried out through point counts at six locations, each with a radius of 0.8 km, (page 7, Johnson *et al.* 2000); [9]Estimated from Figure 101 in Everaert (2008); [10]Estimated from Google Earth map of windfarm (https://www.google.co.uk/maps/place/Gnie%C5%BCd%C5%BCewo/@54.7467485,18.3525275,3643m/data=!3m1!1e3!4m2!3m1!1s0x46fdb3a54ca46bb1:0x5926557d4b8964d0); [11]Data collected within viewing arc with a radius of 2 km, covering seven turbines (Galloper Offshore Windfarm Environmental Statement, Appendix 4); [12]Data presented as number of birds/km/hr; [13]Table A.3.13 in Galloper Offshore Windfarm Environmental Statement; [14]Abstract of Howe *et al.* (2002) states that 150 km^2 were surveyed, width taken as $(\sqrt{150}) * 1000$; [15]Birds recorded were those passing within a 200 m radius around each turbine, Wild Frontier Ecology (2013); [16]Table 32 of Everaert (2008), activity monitored around turbines 3-7 which are each separated by 280 m, arranged in a single line and have a diameter of 140 m; [17]Section 2.1 of Winkelman (1992), turbines have a diameter of 30 m and are 250 m apart and arranged in three lines of six turbines; [18]Birds up to 60 m recorded (Tables 12a-d Winkelman 1992); [19]Movements monitored over four 5 km observation lines (Figure 3, Petterson 2005); [20]Based on Everaert (2008) – 23 turbines were operational and searched for corpses in 2001-2003, 25 turbines were operational and searched for corpses in 2004, and 24 turbines were

operational and searched for corpses in 2005-2007. In addition, collision data for the four turbines monitored for gull activity in 2000 and 2001 (Everaert *et al.* 2002) and the seven turbines monitored for tern activity in 2004 and 2005 (Everaert & Stienen 2007, Everaert 2008) are also analysed in this report; [21]While different turbine types have been used at Zeebrugge, the analysis in this report is based on the assumption that they share the characteristics of those on the eastern wall, where the greatest number of collisions are typically recorded (Everaert 2008); [22]Gull activity was monitored along a 400 m section of the eastern wall in 2000 and 2001 (Everaert *et al.* 2002) and tern activity was monitored along a 720 m section of the eastern wall in 2004 and 2005 (Everaert & Stienen 2007, Everaert 2008); [23]In 2000 and 2001, flight height was estimated up to a maximum of 65 m and in 2004 and 2005 flight height was estimated up to a maximum of 80 m; [24]States that standard SNH vantage point methodology with radius of 2 km from a single point used (Percival 2012, 2013);[25]Based on rotational speed of Blyth Harbour turbines;[26] Based on rotational speed of Enercon E-70 2.3 MW turbine;[27]Similar size to Kessingland turbines; [26]Birds up to 100 m recorded, see Table 37 of Everaert (2008); [27]Birds up to 180 m recorded, see section 1.11 of Appendix 4; [28]Radar monitoring of flight heights up to 140 m, see Krijgsveld *et al.* (2011); [29]Flights monitored up to altitude of 250 m, see figure 11 of Petterson (2005); [30]Estimated from Google Earth map of windfarm (https://www.google.co.uk/maps/search/Bristol+Port+Wind+Park/@51.5117476,-2.7031114,1372m/data=!3m1!1e3); [31]Paragraph 2.3 of The Landmark Practice (2013).[32]highlighted grey so red numbering shows up against red background.

The number of birds predicted to collide with the turbines in the absence of any avoidance action can be estimated by multiplying the total number of birds predicted to pass through the rotor sweep of the turbines over the course of the time period in which collision searches were carried out by the probability of those birds colliding with the rotor blades. An avoidance rate can now be derived from these data by dividing the observed collision rate by the predicted collision rate, as in equation 6.

Avoidance rates were derived, as described above, for each species-site combination for which sufficient data were available in the studies identified as part of our literature review. The quality of data presented in each of these reports was highly variable, in particular in the level of spatial and temporal overlap between the periods over which corpses were collected and bird movement data were collected. The feasibility of collecting movement data over the course of the study periods as a whole meant that some extrapolation was inevitable when calculating avoidance rates. However, we sought to minimise this extrapolation and sought to categorise the studies we identified accordingly (Table 5.5).

The first category (green) we identified, which we had greatest confidence in, was that in which activity data were collected at intervals throughout the period in which corpse data were collected, and from around all turbines which were searched for corpses. This meant that no spatial extrapolation was necessary to derive an avoidance rate, and the need for temporal extrapolation was minimised. The second category (yellow) we identified was similar to the first, with the exception that activity data were not collected around all of the turbines which were searched for corpses, for example at Kleine Pathoweg, where bird movements were only monitored around five of the seven turbine where corpse searches were carried out. This meant that spatial extrapolation of movement data was necessary, potentially leading to erroneous conclusions if flights were not to occur evenly throughout the site. The third category (red) also required spatial extrapolation of activity data. In addition, movement data were only collected for a portion of the time in which corpse data were collected, meaning that bird activity had to be extrapolated across seasons or years. Such extrapolation is extremely likely to give a misleading picture of the true level of bird activity at a site over the study period which is likely to vary seasonally, e.g. over breeding or migration periods. The final category (grey) relates to studies in which bird movements through windfarms have been monitored in order to directly observe collisions. Given the relative rarity of birds colliding with turbines, these studies typically have low power to detect a collision.

We consider how each of these categories influences the avoidance rates that are derived. We also consider the influence of other factors, such as turbine size, on avoidance rates in order to assess whether it is appropriate to apply avoidance rates from some of the smaller onshore turbines to the much larger turbines used in the offshore environment.

The estimation of predicted collisions requires assumptions to be made regarding the proportion of birds flying at collision risk height and their flight height distributions. Consequently, we derive avoidance rates appropriate for use with each of the three model options presented in the Band offshore collision risk model spreadsheet (Band 2012):

i. Option 1, where site specific flight height data are used to estimate the proportion of birds flying at collision risk height;
ii. Option 2, where modelled data are used to estimate the proportion of birds flying at collision risk height, based on the distributions presented in Johnston *et al.* (2014a) and the exact rotor dimensions presented in each report;
iii. Option 3, where modelled flight height distributions are used to account for collision risk not being distributed evenly within a turbine's rotor swept area.

It should be noted that different values would be expected for Band model options 1 and 2 because option 2 uses generic distributions from compiled data sources, which may not be directly comparable to data collected from some of the sites included in this review. In some cases, option 2 may also use a better defined risk window, as it reflects the actual turbine dimensions rather than a pre-defined window set during pre-construction surveys.

5.4.2.2

The aim of this review was to derive representative within-windfarm avoidance rates that can be used to inform a total avoidance rate for use in collision risk modelling for each of the priority species. Whilst the above methodology can give us a range of different values for marine birds in general, and some of the priority species in particular, combining them to get a single, representative figure is far from straightforward. This is further complicated as several studies report no collisions, suggesting an avoidance rate of 1 over the study period. However, were the study periods of these studies to be extended indefinitely, it is likely that the avoidance rate would drop to below 1 as some individuals will always fail to take action to avoid collision, given sufficient time and bird flux within the site. Whilst one approach would be to discard studies in which no collisions were recorded, this would be inappropriate as it would risk negatively biasing our dataset and, potentially, result in a within-windfarm avoidance rate which is overly precautionary.

We identified five methodologies – ratio estimators, meta-analysis, proportional hazard models and mark-recapture models, events-trials models and Poisson regression – that could potentially be used to combine collision records and flux rates across sites in order to derive representative avoidance rates (Table 5.6). We then considered the limitations and assumptions associated with each technique, before determining which was likely to be the most effective approach.

Meta-analysis is most appropriate when estimates of variance around effect sizes are available, which was not the case in this instance. The data available from the studies we reviewed fail basic assumptions about perfect detectability required for proportional hazard models. Similarly, as individual birds are recorded only upon their deaths, and not on their entry to the population, mark-recovery models were not appropriate. Collisions between birds and turbines are rare events. As event-trials models are most effective when the probability of an event is moderate, this methodology is also likely to be ineffective. Poisson regression models may be an effective approach. However, such an approach would require time to develop and test using simulated data. It may also be ineffective without access to raw survey data from each site. Whilst this approach may provide a useful framework for future studies it was not considered feasible within the framework of the current project.

Having considered each of the different approaches, we concluded that ratio estimators would be the most appropriate approach to combining the avoidance rate data. Given the limitations of the data, we felt that any of the more complex modelling approaches may result in undue confidence being assigned to the derived values. In the absence of raw data, we feel that any more involved modelling approach is likely to be less than robust and that, in this instance, a simpler approach, such as ratio estimators, is most appropriate.

Ratio estimators divide the total number of collisions across all sites by the total number of collisions predicted in the absence of avoidance behaviour across all sites (equation 9). By dividing the total number of collisions by the predicted collision rate, sites with greater levels of bird activity are given greater weighting than sites at which bird activity is relatively low. Arguably, this approach to weighting is more appropriate than weighting flux rate alone, as it accounts for the fact that a higher flux rate may not necessarily reflect a greater number of birds at risk of collision. For example, a site may have a relatively high flux rate, but only a relatively small proportion of these birds may be at a height which places them at risk of collision. Using equation 9, we derive representative avoidance rates for all species and groups for which sufficient data were available.

$$Within\ Windfarm\ Avoidance = 1 - \frac{Observed\ collisions}{Pcoll \times Flux\ Rate} \quad \text{(eq. 9)}$$

As data come from multiple sites, there is likely to be a degree of uncertainty associated with avoidance rates derived in this manner. The importance of incorporating uncertainty in the Environmental Impact Assessment process is receiving increasing recognition (Masden *et al.* 2014). The variance associated with the avoidance rates derived using ratio estimators can be calculated using the delta method (Powell 2007). The square root of this value will give an estimate of the standard deviation around the avoidance rates derived using ratio estimators (Batschelet 1976). It is important to note that this value will reflect variability between sites, as opposed to uncertainty in the input parameters. At present, many of the input parameters for the Band model are only available as single values (e.g. mean rotor speed), until a realistic range of values is available for the key parameters, quantifying uncertainty from these sources will be challenging.

As we are looking for representative values for the within-windfarm avoidance rates, it is important to ensure that the values we are deriving are not unduly influenced by a single data point (each data point reflecting a single site-year-species combination), or set of data points. For this reason we investigate how different factors may influence the final avoidance rates we derive. As a first step, we explore how much influence (leverage) each data point has on the final, representative avoidance rates. We identify sites which have a high leverage and determine whether there are any common factors linking them, for example, an unusually high or low flux rate or the presence of small turbines.

We then consider how bird flux and turbine size may influence the final derived avoidance rates using a stepwise approach. These analyses are not an essential part of deriving our final avoidance rates, instead, they help us to understand how reliant our values are on the inclusion of all of our data points. Ideally, as we drop data points from our calculations, the avoidance rates derived should remain fairly constant. In the first analysis, we drop sites based on their estimated flux rates. This helps to demonstrate whether our final avoidance rate is dependent on the inclusion of data from a handful of sites with high levels of bird activity. In our second analysis, we drop sites based on maximum turbine height, to identify whether sites with smaller turbines, less typical of the offshore environment are unduly influencing the values we derive. A more detailed analysis of the sensitivity of our derived values is carried out in section 6 (below).

Table 5.6 Methodologies considered for synthesising avoidance rates across multiple data sources.

Method	**Description**	**Used**
Ratio estimators	Ratio estimators provide a relatively simple approach that compares the mean of the number of collisions to the mean of the number of birds at risk of collision (Cochran 1977). The approach does this by combining data across sites prior to any calculation and, therefore, accounting for the differing levels of bird activity at each site. As the number of birds at risk of collision is proportional to the bird flux at a site, this approach effectively weights sites by the level of bird activity recorded. Depending on the data available, such calculations can be undertaken on a species, group or global basis. They have the advantage of offering a single, easily interpretable output. This approach has previously been used to derive avoidance rates for geese from multiple data sources (Pendlebury 2006).	✓
Meta-analysis	Meta-analysis provides a way of combining studies, which may have different uncertainties attached to them, to determine the size and statistical significance of a given effect. The units of meta-analysis are the independent results of studies, rather than the responses of individual subjects (Arnqvist & Wooster 1995), with a strong recommendation from statisticians that they should use weighted combination of effect sizes (Stewart 2010). Meta-analyses are most appropriate when studies present estimates of variance around the effect sizes (Gurevitch & Hedges 1999, Stewart 2010), which were not available from the studies we have reviewed.	✗
Proportional hazard models / mark-recovery models	We considered the possibility of using time to event style models such as proportional hazard or mark-recovery models. In the case of proportional hazard models, the data fail basic assumptions about perfect detectability necessary for such analyses. As each individual bird is recorded only on its death (and not on entry to the population, i.e. when it enters the turbine space), it was not possible to use mark-recovery type models to produce synthesised ARs from the various studies.	✗
Events-	Events-trials models involve combining the number of events	✗

trials models	(in this case, collisions) with the number of trials (in this case, birds passing through the turbines) within a binomial generalised linear model (GLM). However, collisions are rare events and binomial GLMs work best when the probability of an event is moderate (typically in the region of 0.2-0.8). We therefore feel such a methodology is inappropriate in this instance.	
Poisson regression	As collisions are rare events the mean across sites is likely to be low and may be expected to follow a Poisson distribution. We could use bird flux as an offset in such a model to account for the different abundance of birds at each site and incorporate a weighting factor to account for survey effort. Zero-inflation is also likely to be an issue (i.e. many sites record no collisions). Whilst this approach may be possible and provide a useful framework for future analyses, it would require some time to develop and test using simulated data and was thus outside of the scope of this project. It should also be noted that we are uncertain about how effective such an approach would be without access to the raw survey data from each site.	✗

5.4.3 Results

5.4.3.1 Derived within-windfarm avoidance rates

Data combining collision rates and passage rates through windfarms were obtained from 20 sites – see Appendix 6 for details of sites and species, and Appendix 7 for full results. However, based on the available data, it was only possible to derive within-windfarm avoidance rates for eider, gulls and terns. Whilst other species had been recorded within the windfarms, these were often present in relatively low numbers, and only a single collision event, involving a flock of four eider, was observed during visual observations of turbines. The range of species reflects the onshore or coastal locations of the study sites, and it should be noted that, among the priority species being considered in this review, no estimates could be derived for northern gannet.

The range of responses estimated from the available data runs from an apparent strong attraction whereby the proportion of birds within the rotor-swept area increases by >1000% in some cases, to strong avoidance, where close to 100% of birds avoid the rotor-swept area.

Multiple years’ data were collected from several onshore sites including Avonmouth and Hellrigg in the UK, Boudwijnkanaal and Zeebrugge in Belgium, Gneizdzewzo in Poland, and an offshore site at Yttre Stengrund in Sweden. Multiple years’ collision data were also available from Kleine Pathoweg in Belgium. However, bird activity data were not collected concurrently with data on collision rates for this site, meaning the results cannot be used for the purposes of this review. Whilst we have been able to derive a within-windfarm avoidance rate in both study years for this site, this approach is flawed as it involves extrapolating from one year’s activity data to the next. The same is true for some of the data collected for both Zeebrugge and Boudwijnkanaal. Whilst multiple years' data were also collected from sites such as Altamont Pass and Buffalo Ridge in the U.S.A. and Blyth Harbour in the UK, these data were summarised across years so it was not possible to examine year to year variation in derived avoidance rates and the resultant avoidance rates should be treated with a high degree of scepticism. Of the sites where it may be possible to compare values between years, only Hellrigg, Gneizdzewo, Boudwijnkanaal, Yttre Stengrund and Zeebrugge provide data that allow this.

We present within-windfarm avoidance rates from all sites for illustrative purposes only (Appendix 7). For the purposes of deriving representative values, we only use what we consider to be the highest quality data (green rows in Appendix 7) where there is both spatial and temporal overlap between the collection of corpses and the collection of bird activity data. Unless otherwise stated the within-windfarm avoidance rates presented in the text from this point refer to those derived using option 1 of the Band model, but these are applicable to option 2.

In the five years for which data were available for Gneizdzewo, only a single collision involving a gull species was recorded (Appendix 7). Similarly, in the years for which data are available from Hellrigg and Yttre Stengrund, collisions were only recorded in a single year at each site. At Boudwijnkanaal, the within-windfarm avoidance rate for herring/lesser black-backed gulls declined from 0.9903 in October 2001 to 0.9556 in October 2005. At Zeebrugge, it was possible to compare within-windfarm avoidance rates for herring and lesser black-backed gulls both between seasons and years. For herring gulls within-windfarm avoidance rates declined from 0.9861 in the 2000 breeding season to 0.9722 in the 2001 breeding season. For lesser black-backed gulls the equivalent figures were 1 in 2000 and 0.9706 in 2001. In 2001, activity data at Zeebrugge were collected in both the breeding season and autumn. The within-windfarm avoidance rates showed an increase for both species during the autumn, to 0.9976 in the case of herring gulls and 0.9990 in the case of lesser black-backed gulls. However, given the limited data available to explore these patterns, more data are required to make firm conclusions about aspects such as seasonal variation in avoidance rates.

Everaert (2014) presents within-windfarm avoidance rates for gulls derived from the same datasets for Zeebrugge, Boudwijnkanaal, Kleine Pathoweg and De Put, using the basic Band collision risk model. The results differ from those we present. The reason for this is likely to be that Everaert (2014) extrapolate bird activity data to cover broader spatial and temporal scales, whilst we focus only on the turbines and months in which bird activity data were specifically collected. The author highlights this extrapolation as a reason why his results should be treated with caution in his discussion of the results. For our purposes, we felt that focussing on the period when activity data were collected when deriving within-windfarm avoidance rates was more consistent with our approach at other sites. A similar issue has been raised in the past in relation to Sandwich tern within-windfarm avoidance rates derived from collision data at Zeebrugge, where rates derived from the same dataset have varied from 0.9664-0.9955 (see NE/JNCC note on subject). This highlights the importance of transparency in the calculations used to derive within-windfarm avoidance rates, enabling people to understand why differences may have arisen and come to an informed position about which values are likely to be most applicable to the situation at hand.

Our analyses of the tern data from Zeebrugge suggest that within-windfarm avoidance rates are likely to be towards the high point of this range. Using only collisions reported in June and around the seven turbines from which activity data were collected, we estimated a within-windfarm avoidance rate of 0.9944 for common tern in 2004 and 0.9948 in 2005. For Sandwich tern, we estimated within-windfarm avoidance rates of 0.9980 in 2004 and 0.9989 in 2005. No collisions involving little terns were recorded around these turbines in either year. These data suggest that tern within-windfarm avoidance rates are very high, and may be consistent year on year.

In addition to estimating collision rates from fatality searches, at four sites – Blyth Offshore Windfarm, Greater Gabbard Offshore Windfarm, Haverigg Windfarm and the Yttre Stengrund and Utgrunden Offshore Windfarms in Sweden – bird activity has been monitored with a view to directly observing collisions. In total, 646 hours of observations have been collected in this manner across the four sites and five windfarms. These surveys documented 3,167,238 bird movements within-windfarms, including at least 5,319 involving gulls. Despite this, these had relatively low power to detect a collision. In the absence of avoidance action, across these sites only 63 collisions would have been expected based on the basic Band model and only 45 based on the extended Band model (Appendix 7). In relation to the priority species covered by this report, 17 of the collisions predicted using the basic Band model and 13 predicted using the extended Band model would have involved gulls. A single collision involving a gull would reflect an avoidance rate of less than 0.95 for both the basic and extended Band model. Such an avoidance rate would be extremely conservative, and it is therefore, unsurprising that no collisions were recorded during visual observations. Indeed, over the course of these studies, only a single collision event, involving four common eider at a single turbine at Yttre Stengrund Offshore Windfarm was observed, reflecting a within-windfarm avoidance rate of 0.1861 for common eider in autumn 2003 or 0.9024 across all seasons and years. Consequently these studies do not provide strong evidence for the behavioural response of our five priority species to turbines.

Deriving within-windfarm avoidance rates using the different Band model options

Whilst the observed number of collisions remains constant, regardless of the model option used, the predicted number of birds at risk of collision varies. As avoidance rates are derived by dividing observed collisions by predicted collisions (eq. 6), avoidance rates derived using different model options will vary. Collision estimates produced using the different Band model options and option-specific avoidance rates will only be identical if the windfarm in question has the same specifications as used to derive those avoidance rates. However, this will not be the case when these avoidance rates are applied to a novel site as a result of differences in model input parameters (e.g. turbine specifications and site-specific estimates of the proportion of birds at collision risk height).

Within-windfarm avoidance rates derived using option 1 of the Band model are higher than those derived using options 2 and 3. The difference in values derived using option 1 and option 2 results from the use of site-specific data on the proportion of birds at risk in option 1, and the use of a generic flight height distribution to inform the proportion of birds at risk in option 2 – in other respects these options are mathematically identical. The difference between option 2 and option 3 lies in how the flux rate and probability of collision are applied across the turbines rotor-swept area. Using option 2, an average collision probability is multiplied by an average flux rate. This introduces error when a species' flight height distribution is not uniform. Option 3 accounts for the non-uniform flight height distribution, common to many species (Johnston *et al.* 2014a), by integrating the flux rate and collision probability over the turbines' rotor-swept area.

5.4.3.2 Representative within-windfarm avoidance rates

The within-windfarm avoidance rates data described above, and presented in Appendix 7 are of extremely variable quality. The final, derived within-windfarm avoidance rates are heavily dependent on the accuracy of the estimated flux rates at each site and on the accuracy of collision estimates. As continuous monitoring of bird activity at these sites was not feasible, some degree of extrapolation to estimate the total flux rate will be inevitable. However, it is desirable to keep this extrapolation to a minimum. For this reason, we only combine data from sites at which it was not necessary to make a spatial extrapolation in order to estimate a flux rate, and for which activity data were collected at intervals throughout the period in which collisions were monitored, to minimise the potential for inappropriate temporal extrapolation. The sites meeting these criteria were Avonmouth (Winter 2007/08, 2008/09, 2009/10, 2011/12), Boudwijnkanaal (October 2001 and October 2005), Bouin, De Put, Gneizdzewo (autumn 2007, 2008, 2009, 2010, 2011, 2012), Hellrigg (winter 2011, 2012), Kessingland, Oosterbierum (autumn 1990, spring 1991) and Zeebrugge (June-July 2000, June-July 2001 and September-October 2001). All of these sites were located onshore.

Across these sites, a total of 3,880,794 seabirds, of which the majority (66%) were gulls, were expected to have passed through the windfarms over the periods in which corpse searches were carried out. We determined that sufficient data were available to derive avoidance rates for four species – black-headed gull, common gull, lesser black-backed gull and herring gull – and four species groups – all gulls, large gulls (lesser black-backed gull, herring gull, great black-backed gull, Caspian gull, yellow-legged gull), small gulls (black-headed gull, common gull, little gull) and all terns.

Black-headed gull

A total of 746,668 black-headed gulls were expected to have passed through seven sites – Avonmouth (four studies), Boudwijnkanaal (one study), Bouin (one study), Gneizdzewo (three studies), Hellrigg (two studies), Kessingland (one study) and Zeebrugge (three studies) – over the course of their respective study periods. After adjustments were made to this total to account for the proportion of birds flying at rotor height, the size of the rotor swept area and the probability of birds passing through the rotor-swept area without colliding, this was predicted to result in 1,839 collisions based on option 1 and 582 collisions based on option 2, and 297 based on option 3. However, in total only 38 black-headed gull collisions were recorded across all sites during their respective study periods. This corresponds to within-windfarm avoidance rates of 0.9795 (± 0.0033 SD) using option 1 of the Band model, 0.9351 (± 0.0031 SD) using option 2 of the Band model and 0.8731 (± 0.0056 SD) using option 3 of the Band model.

We investigated the leverage that each study site had on the final within-windfarm avoidance rates derived for black-headed gull. We identified three sites which had high leverage for the within-windfarm avoidance rates derived using options 1 and/or 2 and/or 3 of the Band model (Figure 5.1). Of these, the exclusion of data from Bouin resulted in an increase in the value derived using option 1. This is likely to be due to the presence of the turbines on the edge of a black-headed gull breeding colony. This may have led to a greater number of flights through the rotor-swept area of turbines by adult birds returning to provision chicks and/or newly fledged chicks less experienced at flying. As a result of the relatively high collision rate at this site, including this site in our analysis reduced the overall within-windfarm avoidance rate derived using option 1. The leverage of data from Hellrigg in 2012/13 was of a similar magnitude, but in the opposite direction. Despite having the highest level of black-headed gull activity and a high proportion of birds flying at collision risk height, no collisions were recorded at this site, in this year. As a consequence, excluding these data from our analysis resulted in a reduction in the overall within-windfarm avoidance rate.

In contrast to option 1, the exclusion of data from Bouin led to a substantial decrease in the overall within-windfarm avoidance rates derived using options 2 and 3. The relative importance of data from Bouin is exaggerated using options 2 and 3 of the Band model as modelled flight height distributions suggest that other sites with high levels of bird activity, such as Hellrigg, should have very low proportions of birds flying at collision risk height. As such, the predicted collision rates at these sites are much lower than when using option 1 and they have much less influence when used to derive overall within-windfarm avoidance rates using ratio estimators. In addition, the within-windfarm avoidance rates derived for Bouin using options 2 and 3 are significantly higher than for other sites at which collisions were recorded meaning, given its importance relative to other sites, excluding data from Bouin from the analysis results in a significant decrease in the overall within-windfarm avoidance rate derived. Excluding data from Boudwijnkanaal from the analysis for options 2 and 3 results in an increase in the overall within-windfarm avoidance rate. As with Bouin, a relatively high number of collisions were recorded at this site. However, as observed data suggest a high proportion (69%) of birds flew at collision risk height, this site did not have particularly high leverage for the within-windfarm avoidance rates derived. However, using options 2 and 3, only 4.5% of birds were predicted to fly at collision risk height, meaning the final within-windfarm avoidance rate derived for this site was relatively low.

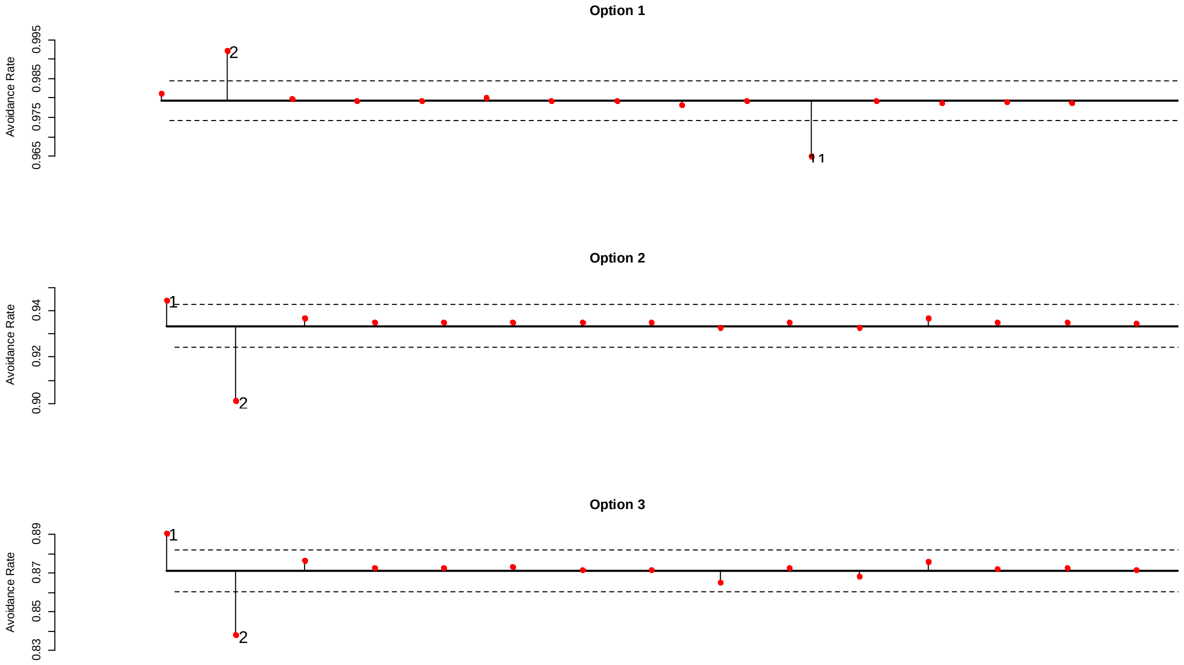


Figure 5.1 Leverage exerted by each site at which within-windfarm avoidance rates were calculated on the overall, mean within-windfarm avoidance rate derived for black-headed gull. Solid line indicates mean within-windfarm avoidance rate across all sites, broken line indicates mean within-windfarm avoidance rate across all sites ± 1 standard deviation, dots indicate mean within-windfarm avoidance rate with each site excluded from analysis. Sites are considered to have high leverage when their exclusion from the analysis leads to a change of more than 1 standard deviation in the overall mean within-windfarm avoidance rate. Sites with high leverage are: 1 – Boudwijnkanaal, 2 – Bouin and 11 – Hellrigg in 2012/13.

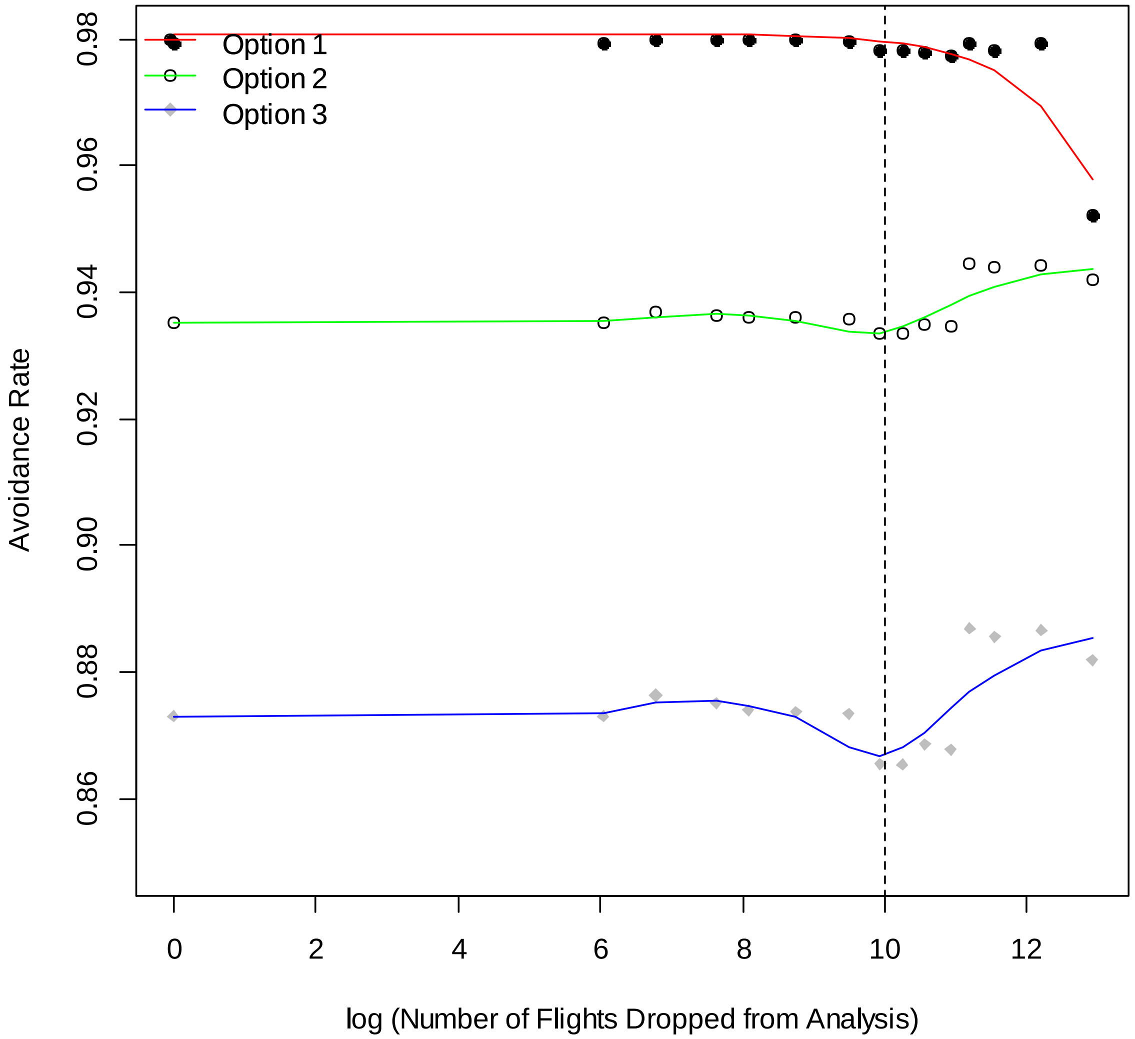


Figure 5.2 Impact of dropping data points (each site-year-species combination) on the within-windfarm avoidance rates derived using ratio estimators for options 1, 2 and 3 of the Band model for black-headed gull.

As might be expected, dropping sites from the analysis can influence the final within-windfarm avoidance rates. Only sites at which there is a relatively limited level of flight activity can be dropped from the analysis before the within-windfarm avoidance rates derived become less stable (Figure 5.1). In all three model options, this is noticeable after around 22,000 of the 746,668 flights through the windfarms have been removed (Figure 5.2).

Using option 1 of the Band model, the derived within-windfarm avoidance rate remains relatively stable at around 0.9795 until Bouin is the only site remaining in the analysis at which point it drops to around 0.9370. As discussed previously, this may reflect the fact that Bouin is located on the edge of a black-headed gull breeding colony, resulting in a higher number of collisions than were recorded elsewhere. In contrast, using options 2 and 3, within-windfarm avoidance rates start to increase after the first 22,000 flights have been dropped. Again, as discussed previously, this is likely to reflect the fact far fewer collisions were predicted at several key sites due to differences in the predicted proportions of birds at collision risk height. As a consequence, as more sites are dropped from the analysis the influence of Bouin, previously identified as having a strong influence on the final derived values for options 2 and 3, becomes stronger.

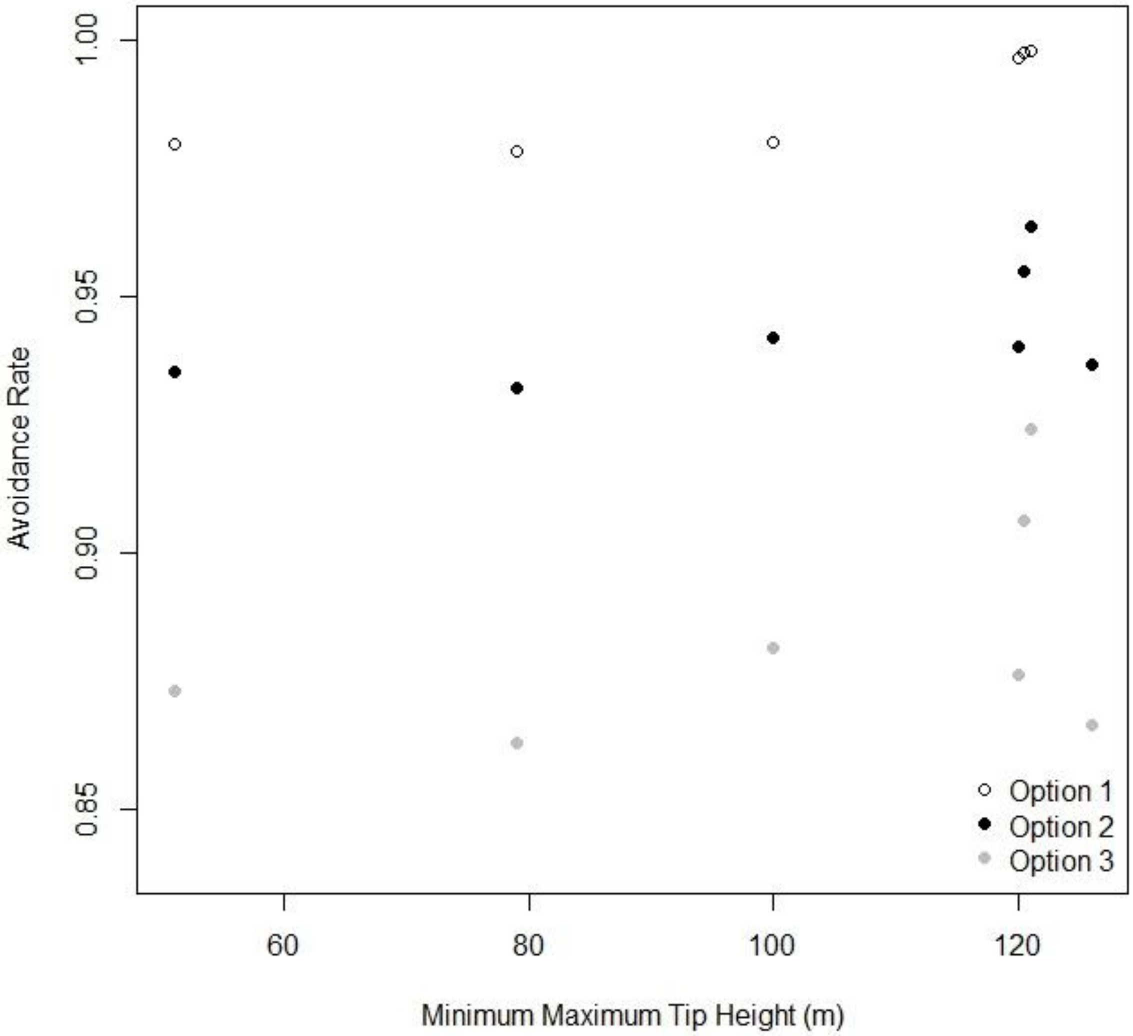


Figure 5.3 Impact of excluding sites with smaller turbines on the within-windfarm avoidance rates derived using ratio estimators for options 1, 2 and 3 of the Band model for black-headed gull.

The inclusion of sites with smaller turbines did not appear to strongly influence the final within-windfarm avoidance rates derived for black-headed gull using any of the three model options (Figure 5.3).

We consider within-windfarm avoidance rates of 0.9795 (± 0.0033 SD) for the basic Band model, and 0.8731 (± 0.0056 SD) for the extended Band model to be realistic, precautionary values given the data available. Whilst we identified several sites as having a strong influence over the final values derived, we do not feel there is sufficient reason to exclude these data from our analysis. It should be noted that the influence of these sites occurs in similar magnitudes in both positive and negative directions. The within-windfarm avoidance rates derived, especially for option 1, remain relatively stable regardless of which sites are included in the analysis. We did not identify any strong impact of turbine size on the final within-windfarm avoidance rate derived.

Common gull

A total of 841,008 common gulls were expected to have passed through three sites – Gneizdzewo (three studies), Kessingland (one study) and Hellrigg (two studies) – over the course of their respective study periods. After adjustments were made to this total to account for the proportion of birds flying at rotor height, the size of the rotor swept area and the probability of birds passing through the rotor-swept area without colliding, this was predicted to result in 3,405 collisions based on option 1 and 218 collisions based on option 2, and 129 based on option 3. However, in total only two common gull collisions were recorded across all sites during their respective study periods. This corresponds to within-windfarm avoidance rates of 0.9995 (± 0.0003 SD) using option 1 of the Band model, 0.9918 (± 0.0046 SD) using option 2 of the Band model and 0.9861 (± 0.0078 SD) using option 3 of the Band model.

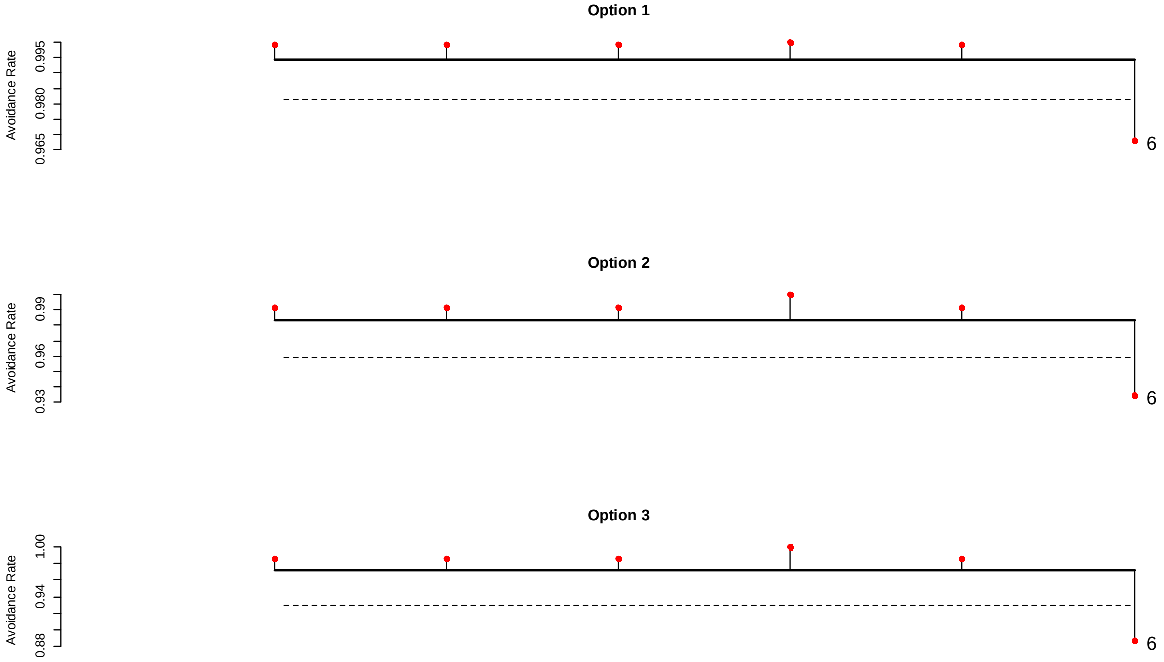


Figure 5.4 Leverage exerted by each site at which within-windfarm avoidance rates were calculated on the overall, mean within-windfarm avoidance rate derived for common gull. Solid line indicates mean within-windfarm avoidance rate across all sites, broken line indicates mean within-windfarm avoidance rate across all sites ± standard deviation, dots indicate mean within-windfarm

avoidance rate with each site excluded from analysis. Sites are considered to have high leverage when their exclusion from the analysis leads to a change of more than 1 standard deviation in the overall mean within-windfarm avoidance rate. Site with high leverage is 6 – Hellrigg in 2012/13.

For all three model options, Hellrigg in 2012/13 appears to have a strong influence over the final within-windfarm avoidance rate derived (Figure 5.4). This is likely to reflect the fact that of the total number of common gulls estimated to have flown through windfarms, over 94% were estimated to have flown through Hellrigg in this year. Despite this, no collisions were recorded involving common gulls at Hellrigg in 2012/13. As a result, excluding these data from our analyses results in an overall within-windfarm avoidance rate of 0.9680 for option 1 of the Band model, 0.9345 for option 2 of the Band model and 0.8865 for option 3 of the Band model. However, we do not feel it would be appropriate to exclude such a substantial portion of our data from the analysis in this way.

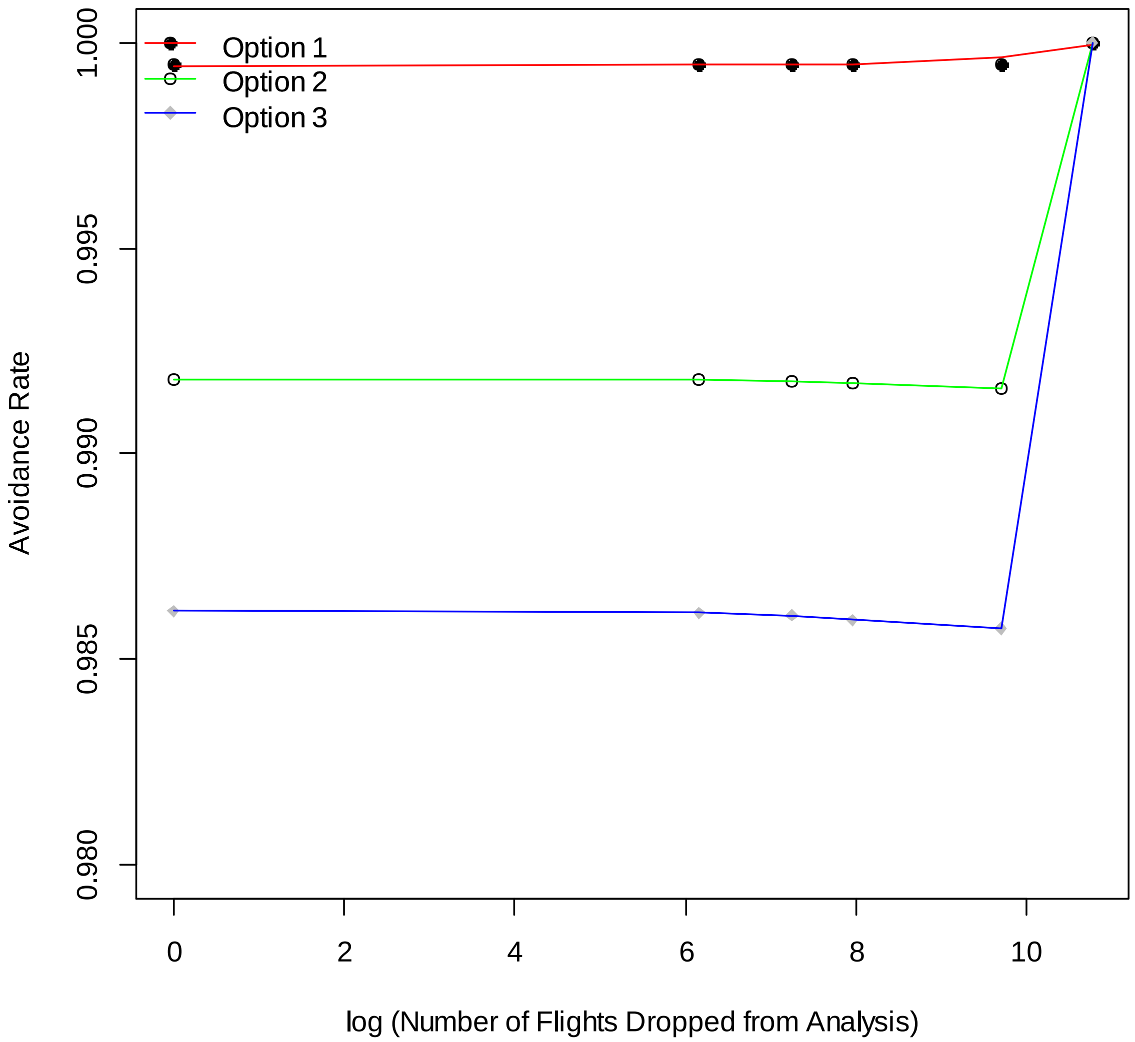


Figure 5.5 Impact of dropping data points (each site-year-species combination) on the within-windfarm avoidance rates derived using ratio estimators for options 1, 2 and 3 of the Band model for common gull.

For all three model options, the within-windfarm avoidance rate derived using ratio estimators remains stable until the only site remaining in the analysis is Hellrigg in 2012/13 (Figure 5.5). As stated above, this is likely to reflect the extremely high leverage of this data point.

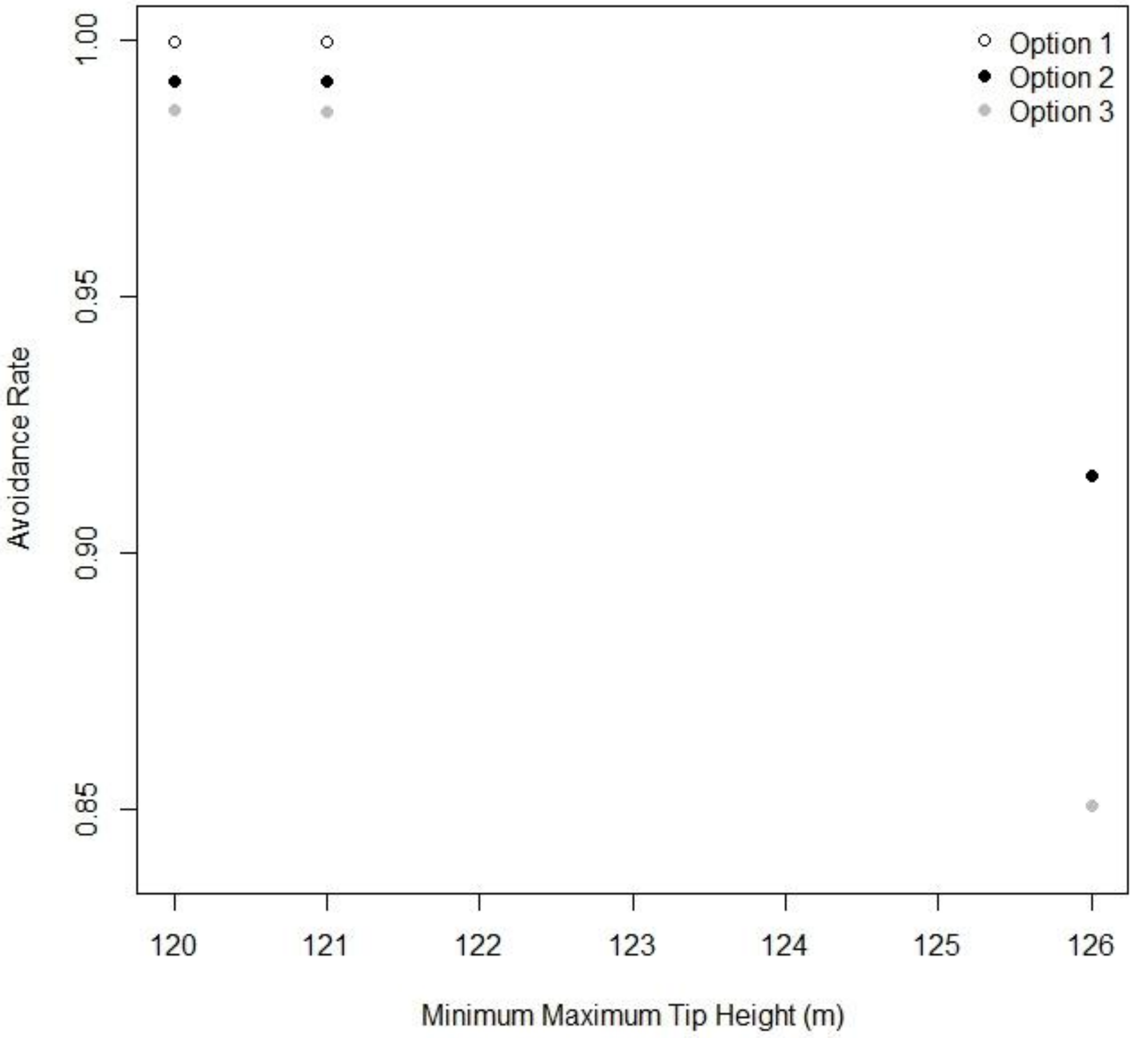


Figure 5.6 Impact of excluding sites with smaller turbines on the within-windfarm avoidance rates derived using ratio estimators for options 1, 2 and 3 of the Band model for common gull.

Maximum tip height appeared to influence the within-windfarm avoidance rates reported, with lower within-windfarm avoidance rates associated with the tallest turbines (Figure 5.6). In reality, this is likely to reflect the fact that collisions were only recorded at Kessingland, the site with the largest turbines, and may, therefore, be coincidence.

Whilst data from Hellrigg in 2012/13 have strong leverage, this must be considered in the context of the sheer number of flights that were estimated at the site in that year, and in combination with the fact that collisions involving common gulls were only recorded at one of the three study sites in a single year. We therefore feel that within-windfarm avoidance rates of 0.9995 (± 0.0003 SD) for the basic Band model and 0.9861 (± 0.0078 SD) for the extended Band model are likely to reflect realistic, precautionary within-windfarm avoidance rates for common gulls. Whilst we feel there is no valid reason to exclude the data from Hellrigg in 2012/13 from our analyses, we feel that its high leverage means that the final within-windfarm avoidance rates derived must be treated with caution.

Herring gull

A total of 526,047 herring gulls were expected to have passed through seven sites – Avonmouth (four studies), Boudwijnkanaal (one study), Bouin (one study), Gneizdzewo (one study), Hellrigg (two studies), Kessingland (one study), Zeebrugge (three studies) – over the course of their respective study periods. After adjustments were made to this total to account for the proportion of birds flying at rotor height, the size of the rotor swept area and the probability of birds passing through the rotor-swept area without colliding, this was predicted to result in 2,157 collisions based on option 1, 1,147 collisions based on option 2, and 957 based on option 3. However, in total only nine herring gull collisions were recorded across all sites during their respective study periods. This corresponds to within-windfarm avoidance rates of 0.9959 (±0.0006 SD) using option 1 of the Band model, 0.9924 (±0.0010 SD) using option 2 of the Band model and 0.9908 (±0.0012 SD) using option 3 of the Band model.

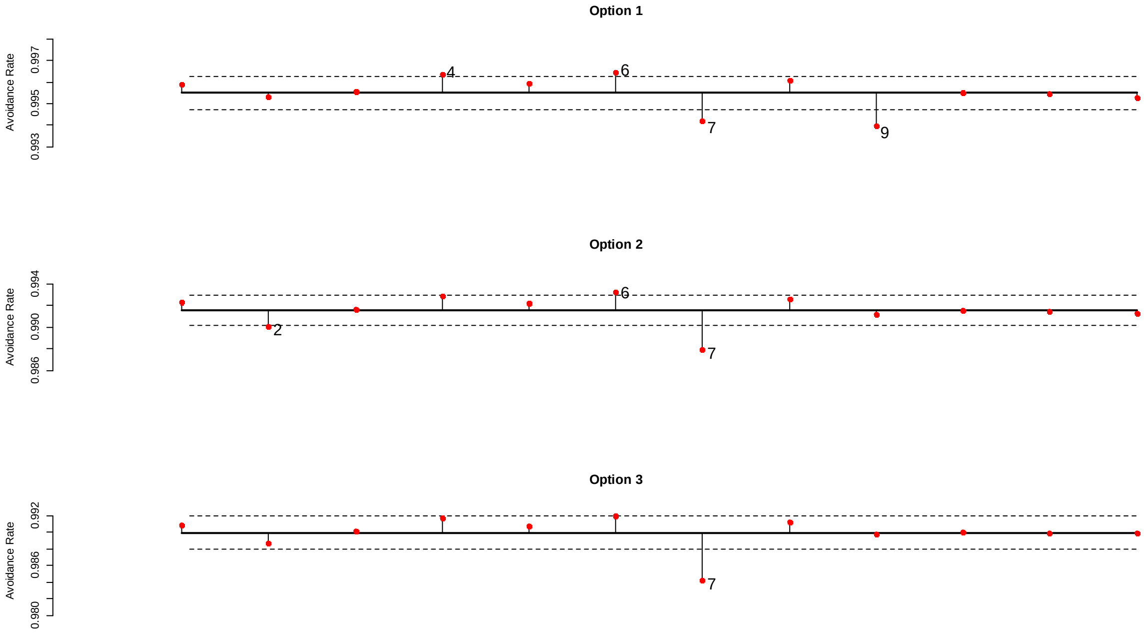


Figure 5.7 Leverage exerted by each site at which within-windfarm avoidance rates were calculated on the overall, mean within-windfarm avoidance rate derived for herring gull. Solid line indicates mean within-windfarm avoidance rate across all sites, broken line indicates mean within-

windfarm avoidance rate across all sites ± 1 standard deviation, dots indicate mean within-windfarm avoidance rate with each site excluded from analysis. Sites are considered to have high leverage when their exclusion from the analysis leads to a change of more than 1 standard deviation in the overall mean within-windfarm avoidance rate. Sites with high leverage are 2 – Bouin, 4 – Kessingland, 6 – Zeebrugge (June-July 2001), 7 – Zeebrugge (September-October 2001) and 9 – Hellrigg (2012/13).

No obvious patterns were evident amongst the sites with high leverage (Figure 5.7). The exclusion of data from Kessingland and Zeebrugge (June-July 2001) from the analysis led to an increase in the overall within-windfarm avoidance rates as both these sites recorded two collisions over the course of their respective study periods. Whilst these were amongst the highest collision rates at the sites we considered, there is no evidence that turbine size played a role. Whilst the turbines at Zeebrugge were the smallest among our study sites, those at Kessingland were the largest. The exclusion of Hellrigg (2012/13) and Zeebrugge (September-October 2001) led to a decrease in the overall within-windfarm avoidance rates. This is likely to reflect the fact that whilst these data points represented the greatest numbers of birds passing through the sites, only two collisions were recorded at Zeebrugge (September-October 2001). It is worth noting that the magnitude of the effect of removing data from Zeebrugge was similar whether data from June-July 2001 or September-October 2001 were removed, although the effect was in opposing directions. Based on these analyses, we did not feel it was appropriate to exclude any data points from our analysis when deriving an overall within-windfarm avoidance rate for herring gull.

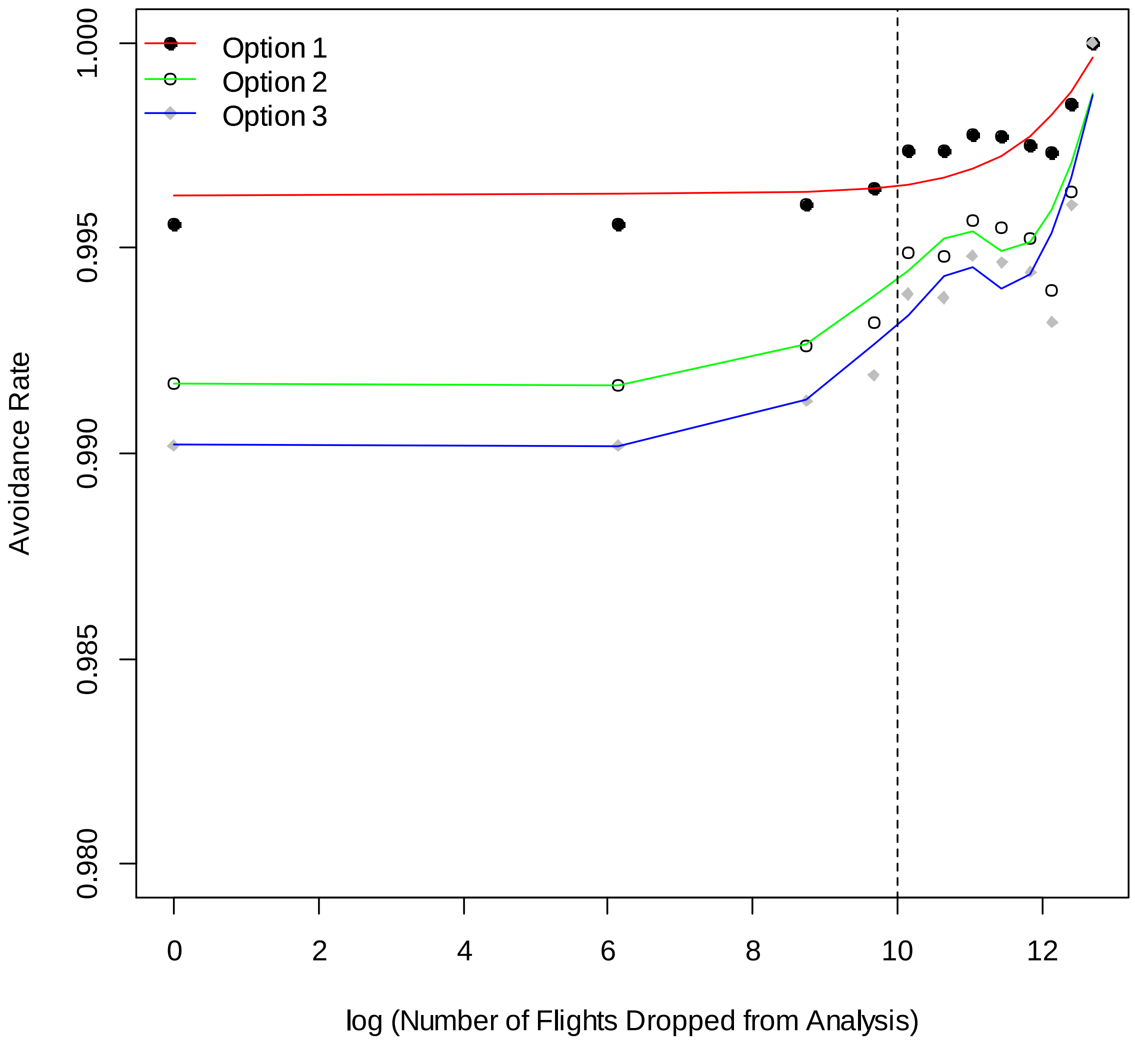


Figure 5.8 Impact of dropping data points (each site-year-species combination) on the within-windfarm avoidance rates derived using ratio estimators for options 1, 2 and 3 of the Band model for herring gull.

Dropping sites with lower levels of flight activity leads to an increase in the within-windfarm avoidance rates derived for herring gull using all three model options (Figure 5.8). Whilst ideally, within-windfarm avoidance rates would remain stable, regardless of the number of flights included in the analysis, it does suggest that the rates derived using the full dataset may be realistic, precautionary values.

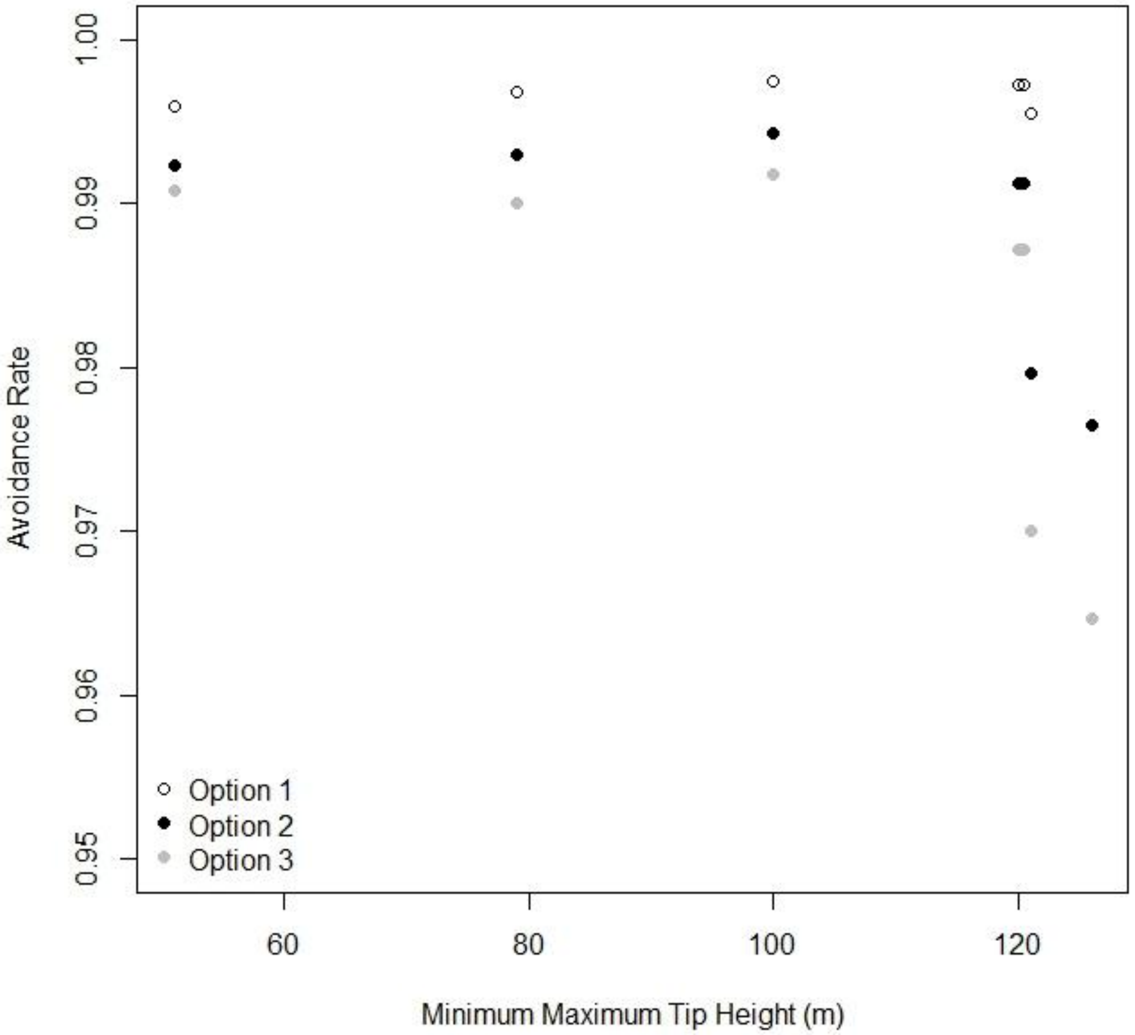


Figure 5.9 Impact of excluding sites with smaller turbines on the within-windfarm avoidance rates derived using ratio estimators for options 1, 2 and 3 of the Band model for herring gull.

Using option 1 of the Band model, there does not appear to be a relationship between turbine size and the within-windfarm avoidance rates derived using ratio estimators (Figure 5.9). However, in the case of options 2 and 3, there is a trend for lower within-windfarm avoidance rates with larger turbines. This apparent discrepancy is likely to reflect differences between the proportion of birds observed flying at collision risk height and the proportion of birds estimated to fly at collision risk height from generic distributions. The generic distributions estimated a lower proportion of birds flying at collision risk height for the larger turbines, meaning the predicted collision rate, and therefore overall within-windfarm avoidance rate, was reduced.

We consider within-windfarm avoidance rates of 0.9959 (±0.0006 SD) for the basic Band model, and 0.9908 (±0.0012 SD) for the extended Band model to be realistic, precautionary values given the data available. Whilst we identified several sites as having a strong influence over the final values derived, we do not feel there is sufficient reason to exclude these data from our analysis. It should be noted that the influence of these sites occurs in similar magnitudes in both positive and negative directions. We did not identify any strong impact of turbine size on the final within-windfarm avoidance rate derived.

Lesser black-backed gull

A total of 101,745 lesser black-backed gulls were expected to have passed through three sites – Hellrigg (two studies), Kessingland (one study) and Zeebrugge (three studies) – over the course of their respective study periods. After adjustments were made to this total to account for the proportion of birds flying at rotor height, the size of the rotor swept area and the probability of birds passing through the rotor-swept area without colliding, this was predicted to result in 1,110 collisions based on option 1,512 collisions based on option 2, and 473 based on option 3. However, in total only two lesser black-backed gull collisions were recorded across all sites during their respective study periods. This corresponds to within-windfarm avoidance rates of 0.9982 (±0.0005 SD) using option 1 of the Band model, 0.9960 (±0.0010 SD) using option 2 of the Band model and 0.9957 (±0.0011 SD) using option 3 of the Band model.

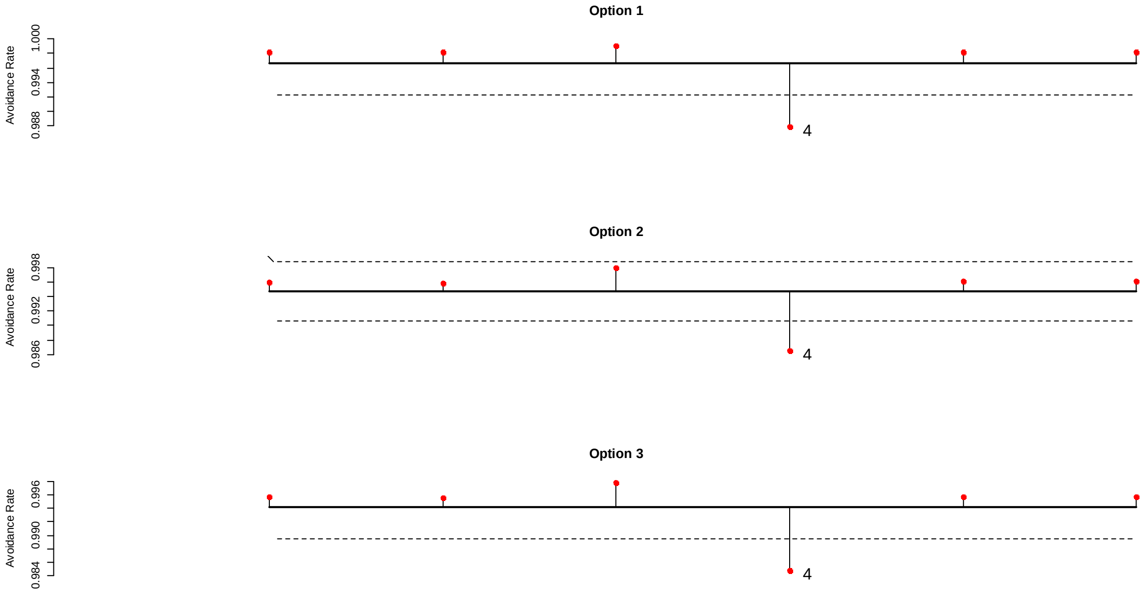


Figure 5.10 Leverage exerted by each site at which within-windfarm avoidance rates were calculated on the overall, mean within-windfarm avoidance rate derived for lesser black-backed gull. Solid line indicates mean within-windfarm avoidance rate across all sites, broken line indicates mean within-windfarm avoidance rate across all sites ± 1 standard deviation, dots indicate mean within-windfarm avoidance rate with each site excluded from analysis. Sites are considered to have high leverage when their exclusion from the analysis leads to a change of more than 1 standard deviation in the overall mean within-windfarm avoidance rate. Site with high leverage is 4 – Zeebrugge (September-October 2001).

Data from all three model options indicated that Zeebrugge in September-October 2001 had a relatively high leverage on the final within-windfarm avoidance rates derived using ratio estimators (Figure 5.10). This is likely to reflect the fact that Zeebrugge in September-October 2001 had the highest levels of bird activity by some distance. Despite this, only a single collision was recorded over the study period. Excluding these data from the analysis results in within-windfarm avoidance rates of 0.9878 using option 1, 0.9865 using option 2 and 0.9847 using option 3. However, we do not feel it is appropriate to exclude data in this way.

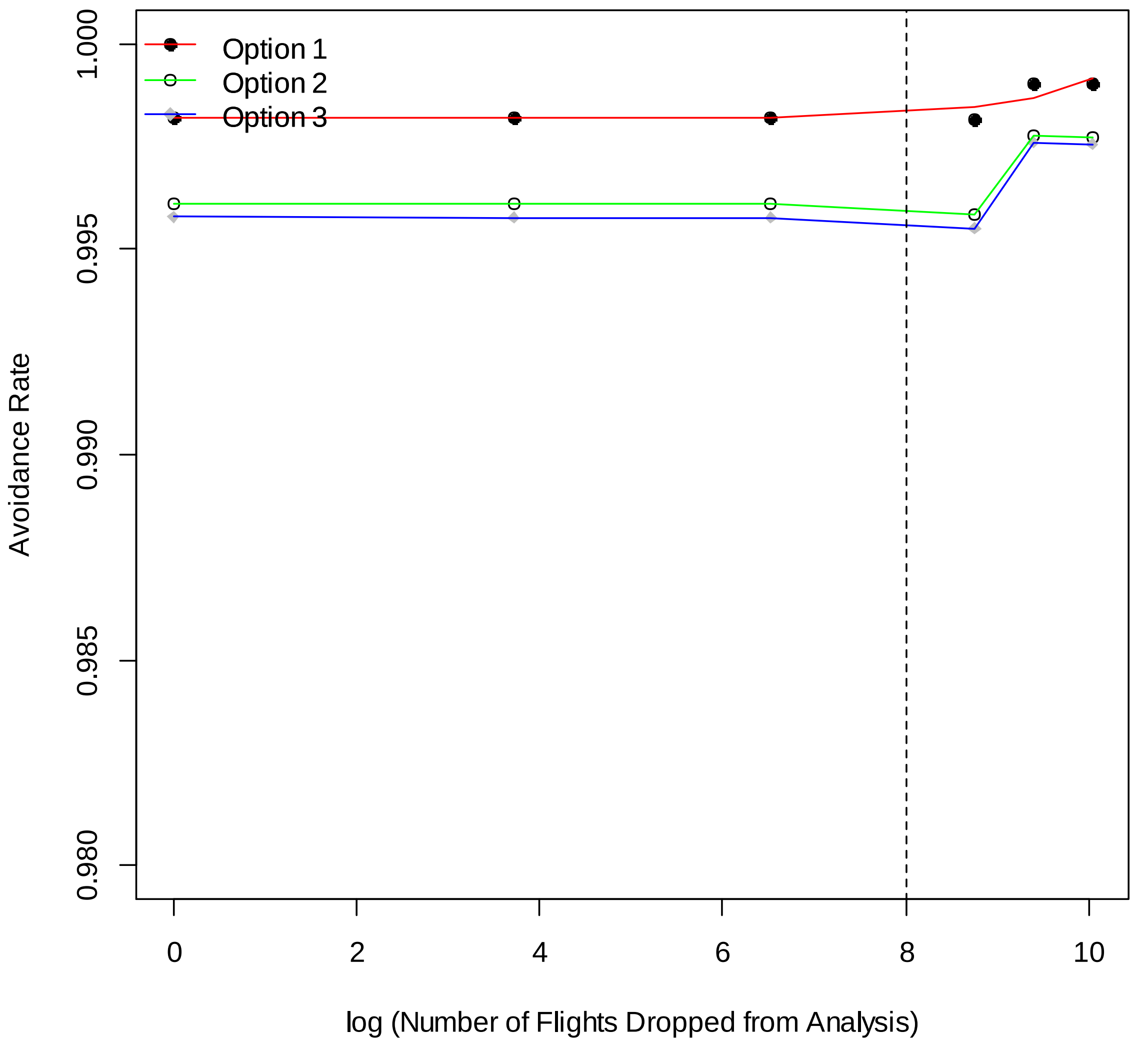


Figure 5.11 Impact of dropping data points (each site-year-species combination) on the within-windfarm avoidance rates derived using ratio estimators for options 1, 2 and 3 of the Band model for lesser black-backed gull.

Using option 1 of the Band model to derive within-windfarm avoidance rates, values remain fairly stable regardless of the number of birds recorded flying through the study sites (Figure 5.11). Using options 2 and 3 the final value remains relatively stable until the first 6,000 flights have been removed. This is likely to reflect that fact that whilst a relatively high number of birds were predicted to have flown through the final two sites, only a single collision was recorded.

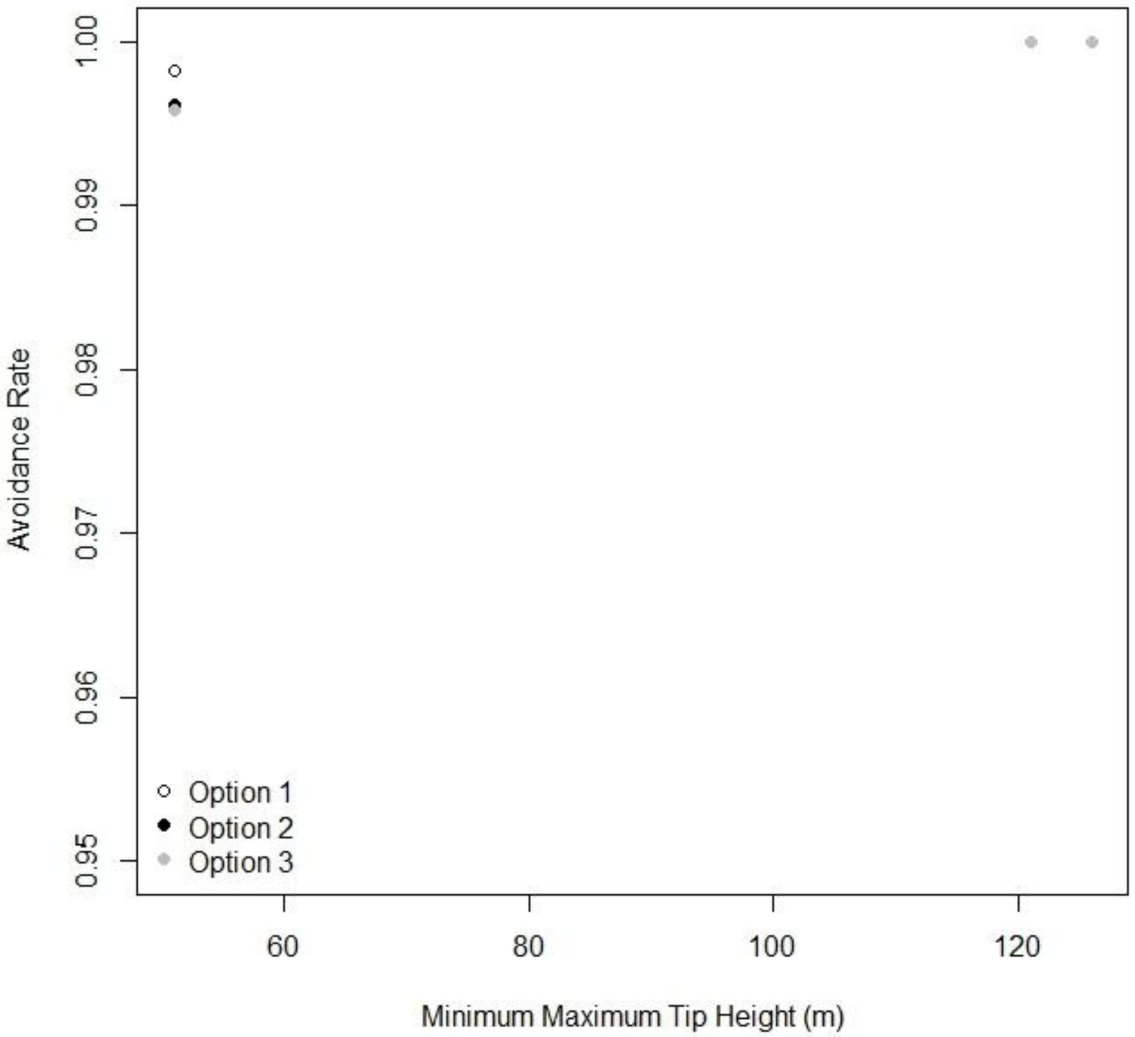


Figure 5.12 Impact of excluding sites with smaller turbines on the within-windfarm avoidance rates derived using ratio estimators for options 1, 2 and 3 of the Band model for lesser black-backed gull.

Excluding smaller turbines did not appear to have a significant impact on the final within-windfarm avoidance rate derived for lesser black-backed gull using any of the three model options (Figure 5.12).

Whilst data from Zeebrugge in September-October 2001 had a relatively high leverage on the final within-windfarm avoidance rates derived, we did not feel there was a compelling reason to exclude these data from our analysis. Based on the data available for lesser black-backed gull, we consider within-windfarm avoidance rates of 0.9982 (±0.0005 SD) for the basic Band model and 0.9957 (±0.0011 SD) for the extended Band model to be realistic, precautionary values given the data available. However, given the data come from only three sites and incorporate a relatively small number of flights through the windfarm, we feel these values should be treated with caution. Whilst we identified several sites as having a strong influence over the final values derived, we do not feel there is sufficient reason to exclude these data from our analysis. It should be noted that the influence of these sites occurs in similar magnitudes in both positive and negative directions. We did not identify any strong impact of turbine size on the final within-windfarm avoidance rate derived.

Small gulls

A total of 1,589,953 small gulls were expected to have passed through eight sites over the course of their respective study periods. After adjustments were made to this total to account for the proportion of birds flying at rotor height, the size of the rotor swept area and the probability of birds passing through the rotor-swept area without colliding, this was predicted to result in 5,263 collisions based on option 1,801 collisions based on option 2, and 427 based on option 3. However, in total only 42 small gull collisions were recorded across all sites during their respective study periods. This corresponds to within-windfarm avoidance rates of 0.9921 (±0.0015 SD) using option 1 of the Band model, 0.9481 (±0.0032 SD) using option 2 of the Band model and 0.9027 (±0.0068 SD) using option 3 of the Band model.

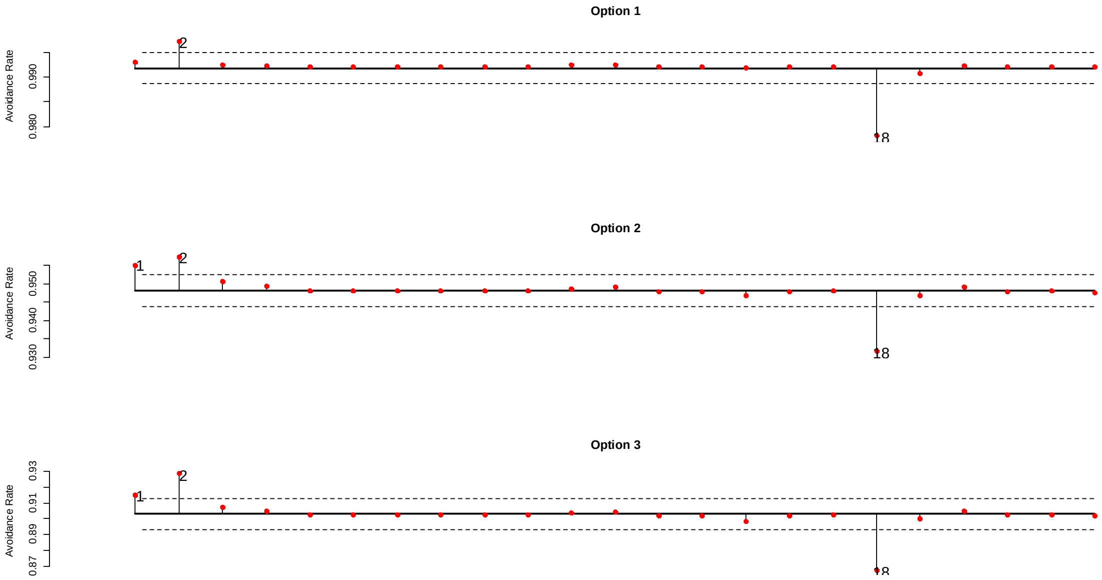


Figure 5.13 Leverage exerted by each site at which within-windfarm avoidance rates were calculated on the overall, mean within-windfarm avoidance rate derived for small gulls. Solid line indicates mean within-windfarm avoidance rate across all sites, broken line indicates mean within-windfarm avoidance rate across all sites ± 1 standard deviation, dots indicate mean within-windfarm avoidance rate with each site excluded from analysis. Sites are considered to have high leverage when their exclusion from the analysis leads to a change of more than 1 standard

deviation in the overall mean within-windfarm avoidance rate. Sites with high leverage are 1 – black-headed gull at Boudwijnkanaal in October 2015, 2 – black-headed gull at Bouin and 18 – common gull at Hellrigg in 2012/13.

For all three model options, the exclusion of data from black-headed gull at Bouin results in an increased within-windfarm avoidance rate (Figure 5.13). This is likely to be due to the presence of the turbines on the edge of a black-headed gull breeding colony. This may have led to a greater number of flights through the rotor-swept area of turbines by adult birds returning to provision chicks and/or newly fledged chicks less experienced at flying. As a result of the relatively high collision rate, including this site in our analysis reduced the overall rate derived using option 1. The leverage of data from Hellrigg in 2012/13 was of a similar magnitude, but in the opposite direction. Despite having the highest level of small gull activity and a high proportion of birds flying at collision risk height, no collisions were recorded at this site, in this year. However, we did not consider there to be a valid reason for excluding these sites from our analysis.

Using options 2 and 3, excluding data for black-headed gull from Boudwijnkanaal in October 2005 also resulted in an increase in the overall within-windfarm avoidance rates. The reason for this differing from the results for option 1 is that the modelled flight height distribution predicts a lower proportion of birds at collision risk height. As a consequence, the predicted collision rate, and therefore the within-windfarm avoidance rate, is reduced.

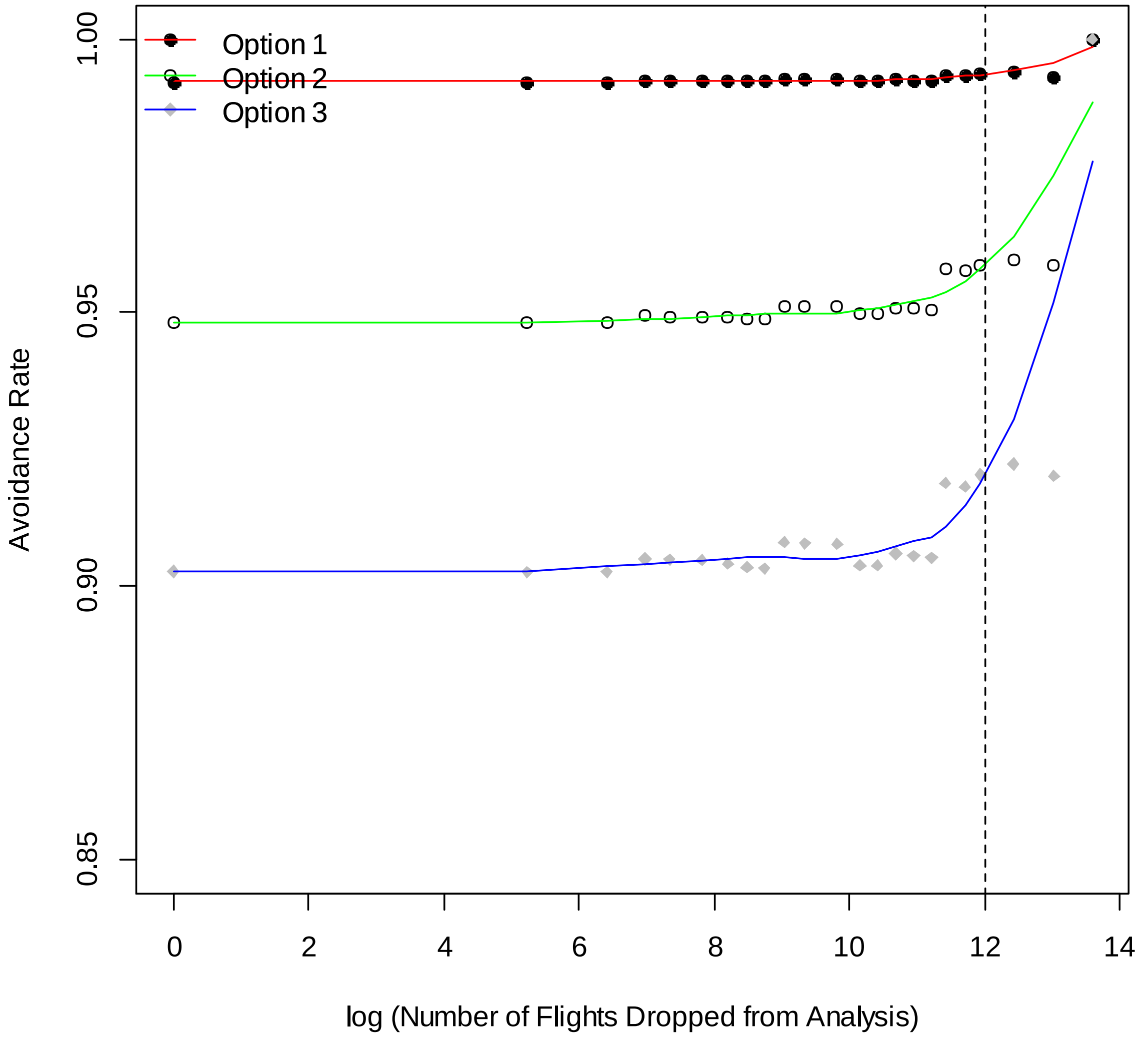


Figure 5.14 Impact of dropping data points (each site-year-species combination) on the within-windfarm avoidance rates derived using ratio estimators for options 1, 2 and 3 of the Band model for small gulls.

Within-windfarm avoidance rates derived using all three model options remain relatively stable as the first 160,000 flights through windfarms were dropped from the analysis (Figure 5.14), before increasing as only the sites with the highest levels of gull activity remain. This reflects the fact that at several of the sites with the highest levels of gull activity, no collisions were recorded, resulting in an overall increase in the within-windfarm avoidance rates as other sites were dropped.

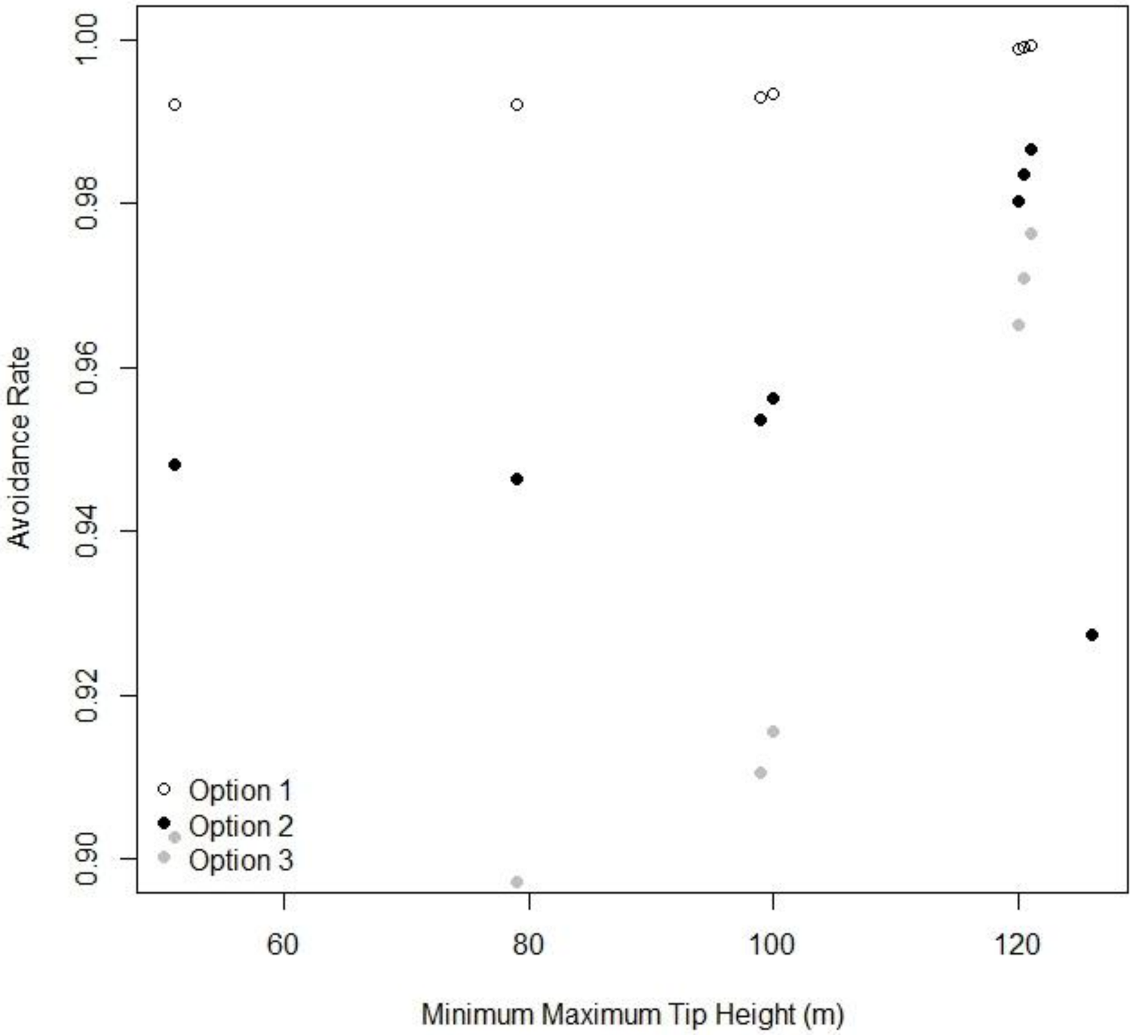


Figure 5.15 Impact of excluding sites with smaller turbines on the within-windfarm avoidance rates derived using ratio estimators for options 1, 2 and 3 of the Band model for small gulls.

Using option 1 of the Band model, there does not appear to be a relationship between turbine size and the within-windfarm avoidance rates derived using ratio estimators (Figure 5.15). However, in the case of options 2 and 3, there is a trend for higher within-windfarm avoidance rates with larger turbines. The reason for this discrepancy is unclear, although it may reflect differences in the proportion of birds at collision risk height between the observed data and modelled distributions.

We consider within-windfarm avoidance rates of 0.9921 (±0.0015 SD) for the basic Band model, and 0.9027 (±0.0068 SD) for the extended Band model to be realistic, precautionary values given the data available. Whilst we identified several sites as having a strong influence over the final values derived, we do not feel there is sufficient reason to exclude these data from our analysis. It should be noted that the influence of these sites occurs in similar magnitudes in both positive and negative directions. We did not identify any strong impact of turbine size on the final within-windfarm avoidance rate derived using option 1 of the Band model.

Large gulls

A total of 639,560 large gulls were expected to have passed through seven sites – Avonmouth (four studies, one species), Boudwijnkanaal (two studies, two species), Bouin (one study, one species), Gniezdzewo (three studies, three species), Hellrigg (three studies, three species), Kessingland (one study, three species) and Zeebrugge (three studies, two species) – over the course of their respective study periods. After adjustments were made to this total to account for the proportion of birds flying at rotor height, the size of the rotor swept area and the probability of birds passing through the rotor-swept area without colliding, this was predicted to result in 3,368 collisions based on option 1, 1,684 collisions based on option 2, and 1,452 based on option 3. However, in total only 42 large gull collisions were recorded across all sites during their respective study periods. This corresponds to within-windfarm avoidance rates of 0.9956 (±0.0004 SD) using option 1 of the Band model, 0.9912 (±0.0007 SD) using option 2 of the Band model and 0.9898 (±0.0009 SD) using option 3 of the Band model.

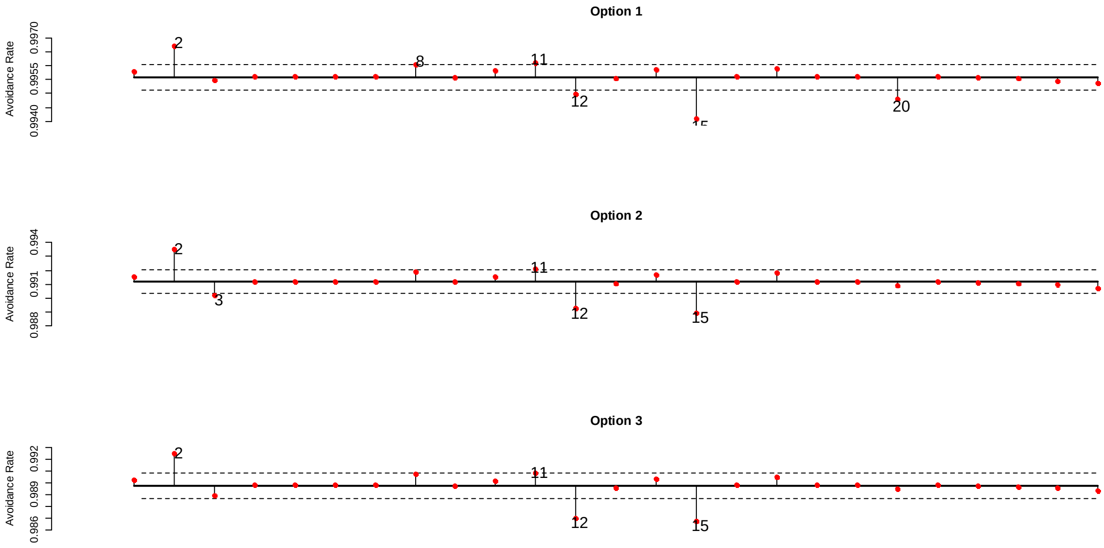


Figure 5.16 Leverage exerted by each site at which within-windfarm avoidance rates were calculated on the overall, mean within-windfarm avoidance rate derived for large gulls. Solid line indicates mean within-windfarm avoidance rate across all sites, broken line indicates mean within-windfarm avoidance rate across all sites ± 1 standard deviation, dots indicate mean within-windfarm avoidance rate with each site excluded from analysis. Sites are considered to have high leverage when their exclusion from the analysis leads to a change of more than 1 standard deviation in the overall mean within-windfarm avoidance rate. Sites with high leverage are 2 – herring/lesser black-backed gull, Boudwijnkanaal (October 2005), 3 – herring gull, Bouin; 8 – herring gull, Kessingland; 11 – herring gull, Zeebrugge (June-July 2001); 12 – herring gull, Zeebrugge (September-October 2001); 15 – lesser black-backed gull, Zeebrugge (September-October 2001); and 20 – herring gull, Hellrigg (2012/13).

There is no obvious pattern to the sites which have high leverage over the final derived within-windfarm avoidance rates (Figure 5.16). Excluding the data for herring/lesser black-backed gull at Boudwijnkanaal in October 2005, herring gull for Kessingland and herring gull for Zeebrugge in June-July 2001 results in an increase in the overall within-windfarm avoidance rate. The size of turbines at these sites varies from small (51 m maximum turbine height at Zeebrugge) to large (126 m maximum turbine height at Kessingland) so the inclusion of different sizes of turbines does not appear to have influenced the within-windfarm avoidance rates derived. In contrast, the inclusion of date for herring gull and lesser black-backed gull at Zeebrugge in September-October 2001 and for herring gull at Hellrigg in 2012/13 results in an increase in the overall within-windfarm avoidance rate derived. In these cases, the increase in the within-windfarm avoidance rates is likely to be linked to the relatively high activity levels at these sites and relatively low collision rates. We do not consider there to be a valid reason for excluding these sites from the analysis.

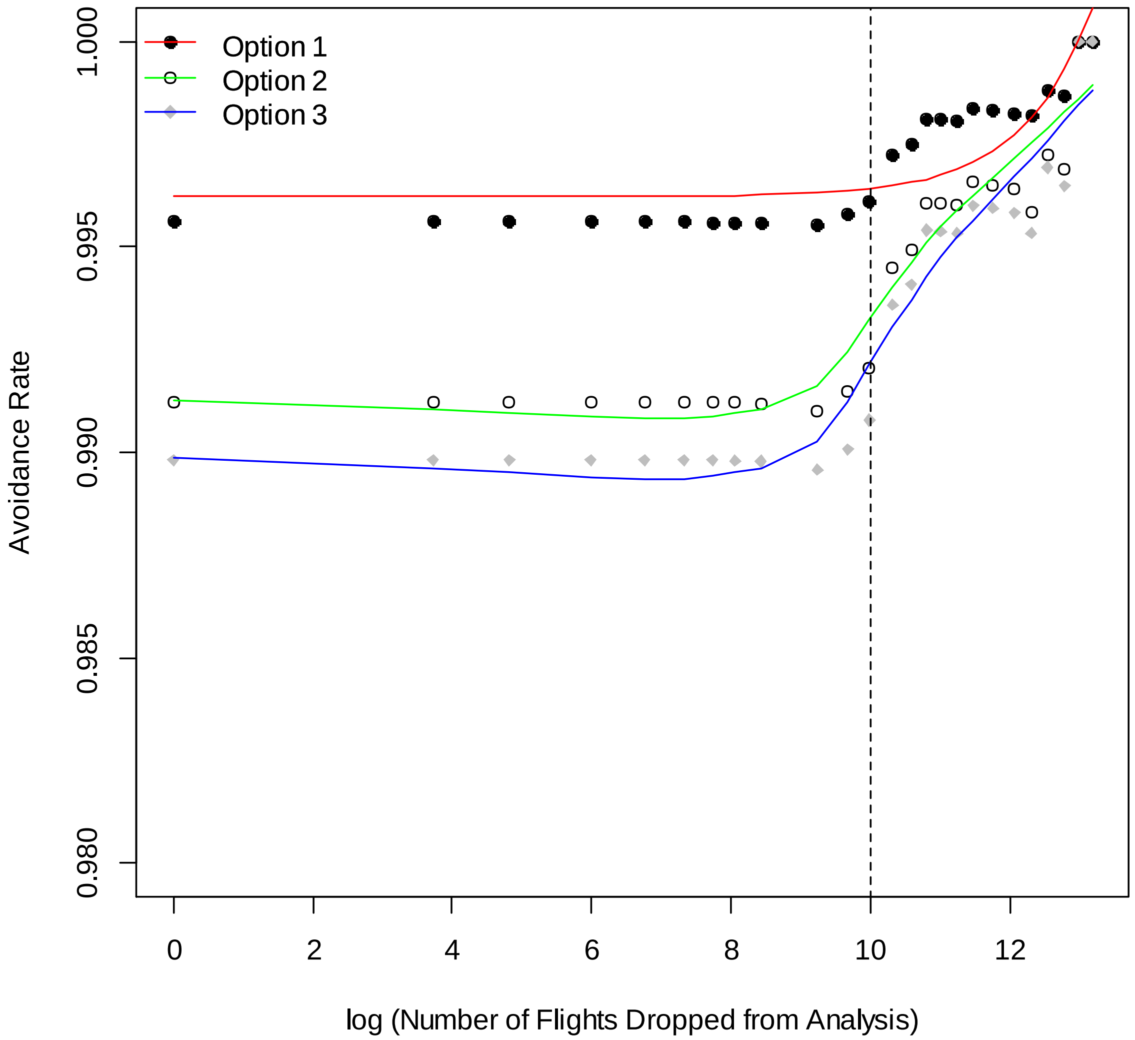


Figure 5.17 Impact of dropping data points (each site-year-species combination) on the within-windfarm avoidance rates derived using ratio estimators for options 1, 2 and 3 of the Band model for large gulls.

Within-windfarm avoidance rates derived using all three model options remain relatively stable as the first 22,000 flights through windfarms are dropped from the analysis (Figure 5.17), before increasing as only the sites with the highest levels of gull activity remain. This reflects the fact that at several of the sites with the highest levels of gull activity, no collisions were recorded, resulting in an overall increase in the within-windfarm avoidance rates as other sites were dropped.

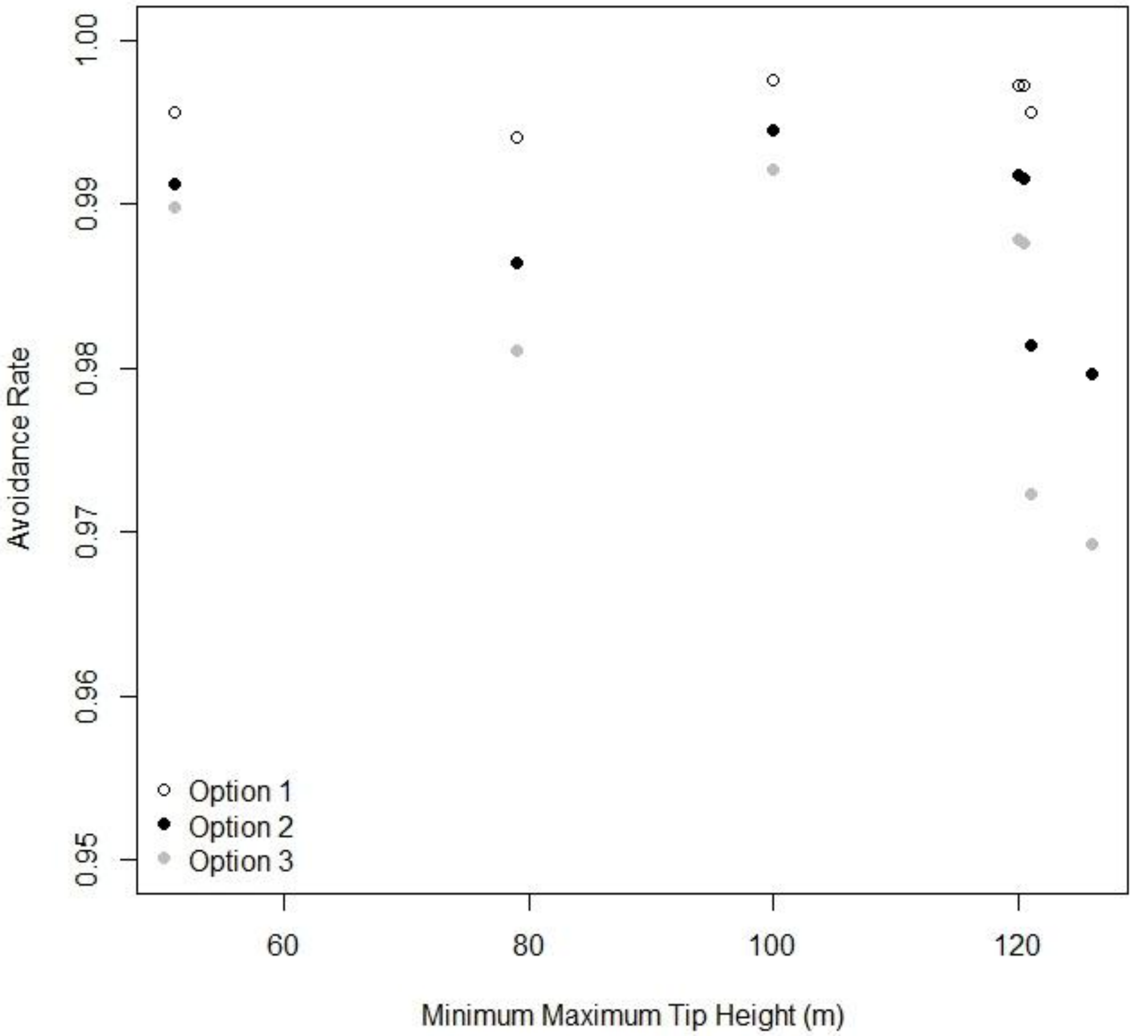


Figure 5.18 Impact of excluding sites with smaller turbines on the within-windfarm avoidance rates derived using ratio estimators for options 1, 2 and 3 of the Band model for large gulls.

Using option 1 of the Band model, there does not appear to be a relationship between turbine size and the within-windfarm avoidance rates derived using ratio estimators (Figure 5.18). However, in the case of options 2 and 3, there is a trend for lower within-windfarm avoidance rates with larger turbines. This apparent discrepancy is likely to reflect differences between the proportion of birds observed flying at collision risk height and the proportion of birds estimated to fly at collision risk height from generic distributions. The generic distributions estimated a lower proportion of birds flying at collision risk height for the larger turbines, meaning the predicted collision rate, and therefore overall within-windfarm avoidance rate, was reduced.

We consider within-windfarm avoidance rates of 0.9956 (±0.0004 SD) for the basic Band model, and 0.9898 (±0.0009 SD) for the extended Band model to be realistic, precautionary values given the data available. Whilst we identified several sites as having a strong influence over the final values derived, we do not feel there is sufficient reason to exclude these data from our analysis. It should be noted that the influence of these sites occurs in similar magnitudes in both positive and negative directions. We did not identify any strong impact of turbine size on the final within-windfarm avoidance rate derived.

All gulls

A total of 2,567,124 gulls were expected to have passed through seven sites over the course of their respective study periods. After adjustments were made to this total to account for the proportion of birds flying at rotor height, the size of the rotor swept area and the probability of birds passing through the rotor-swept area without colliding, this was predicted to result in 10,052 collisions based on option 1, 4,054 collisions based on option 2, and 3,271 based on option 3. However, in total only 107 gull collisions were recorded across all sites during their respective study periods. This corresponds to within-windfarm avoidance rates of 0.9893 (±0.0007 SD) using option 1 of the Band model, 0.9735 (±0.0014 SD) using option 2 of the Band model and 0.9672 (±0.0018 SD) using option 3 of the Band model.

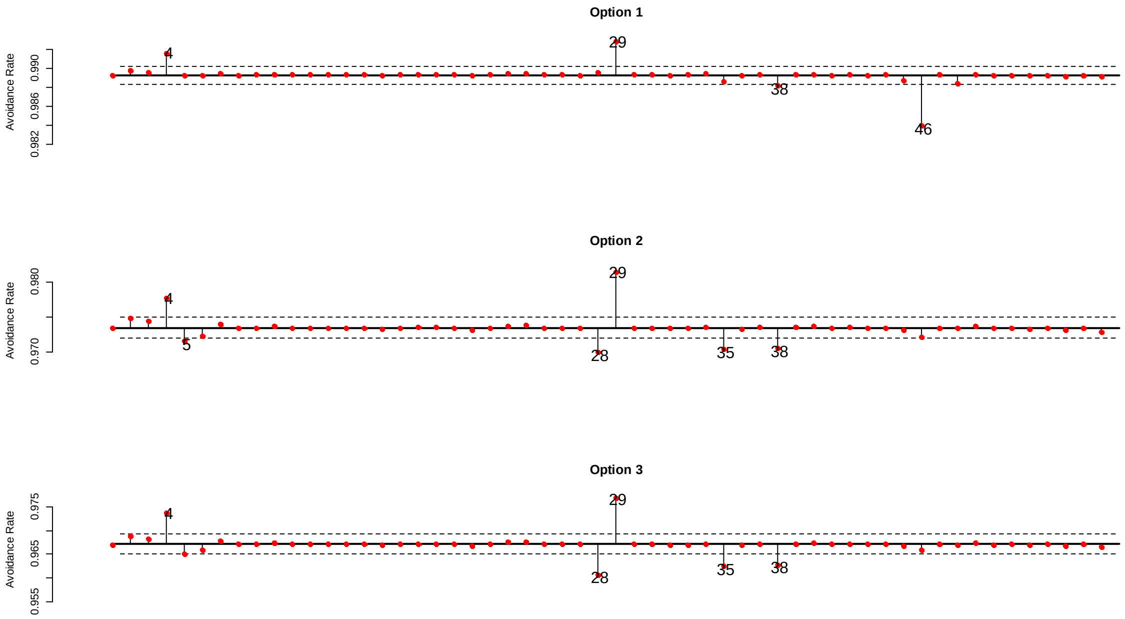


Figure 5.19 Leverage exerted by each site at which within-windfarm avoidance rates were calculated on the overall, mean within-windfarm avoidance rate derived for all gulls. Solid line indicates mean within-windfarm avoidance rate across all sites, broken line indicates mean within-windfarm avoidance rate across all sites ± 1 standard deviation, dots indicate mean within-windfarm avoidance rate with each site excluded from analysis. Sites are considered to have high leverage when their exclusion from the analysis leads to a change of more than 1 standard deviation in the overall mean within-windfarm avoidance rate. Sites with high leverage are 4 – black-headed gulls, Bouin; 5 – gull spp, Bouin; 28 – gull spp, Oosterbierum (autumn 1990); 29 – gull spp, Oosterbierum (spring 1991); 35 – herring gull, Zeebrugge (September-October 2001); 38 – lesser black-backed gull in Zeebrugge (September-October 2001); 46 – common gull, Hellrigg (2012/13).

For all three model options, excluding data for black-headed gulls at Bouin and gull spp at Oosterbierum in autumn 1990, results in an increase in the overall within-windfarm avoidance in the final derived within-windfarm avoidance rates (Figure 5.19). There are no obvious commonalities between these sites. The turbines at Oosterbierum are relatively small with a maximum tip height of 50 m, but those at Bouin are more typical of the sites in our study, with maximum tip heights of 100 m. Using option 1, the exclusion of data from lesser black-backed gull at Zeebrugge in September-October 2001 and common gull at Hellrigg in 2012/13 resulted in decreased within-windfarm avoidance rates. This is likely to reflect relatively high levels of bird activity in combination with very few recorded collisions at these sites, meaning they have a negative bias on the final, derived figures. This pattern was repeated for gull spp at Oosterbierum in spring 1991 and herring gulls and lesser black-backed gulls at Zeebrugge in September-October 2001 using options 2 and 3 and gull spp at Bouin using option 2.

It should be noted that for all three model options, leverage occurred in both directions. We did not feel there was a valid justification for excluding any of these data points from our analysis.

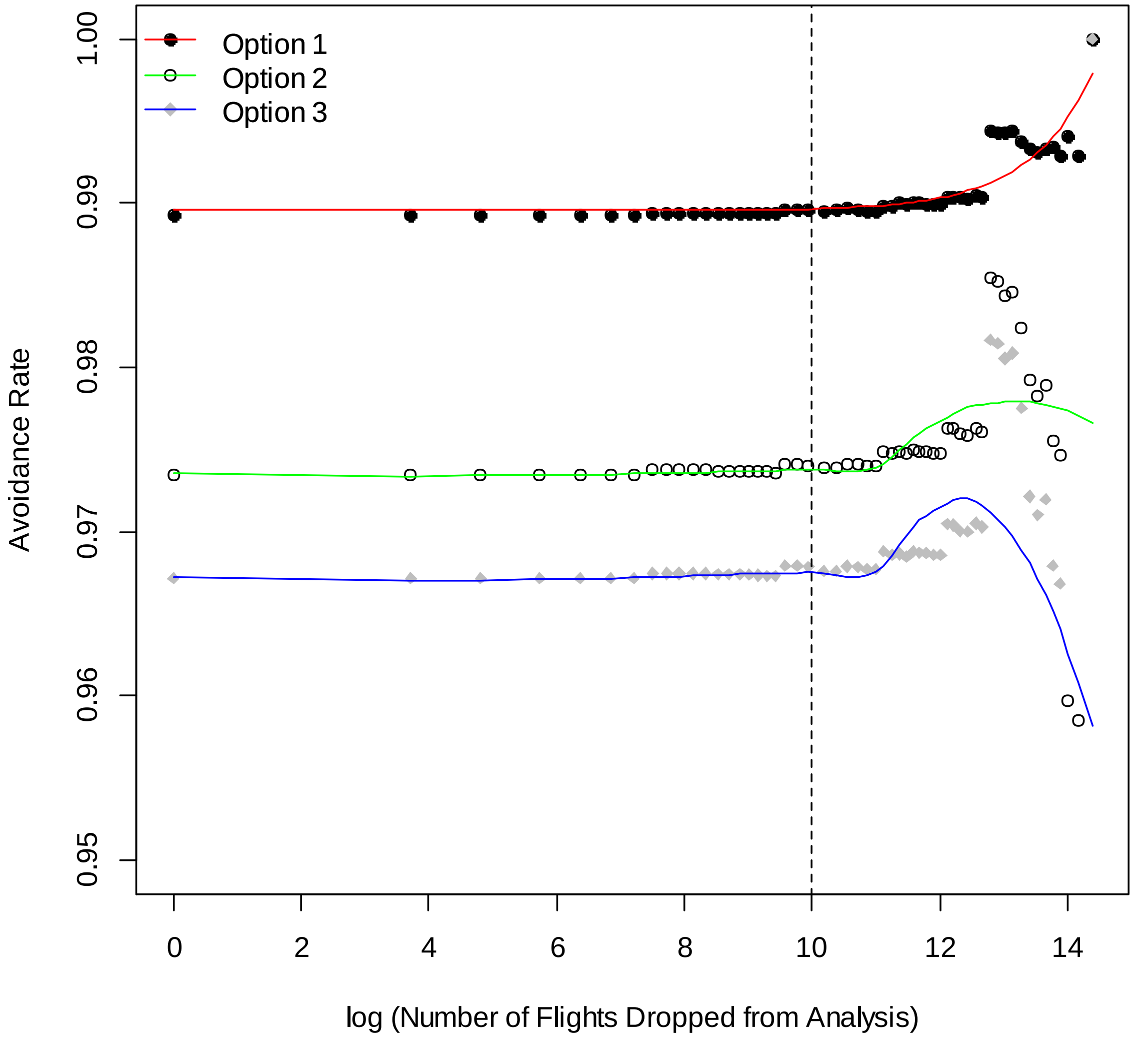


Figure 5.20 Impact of dropping data points (each site-year-species combination) on the within-windfarm avoidance rates derived using ratio estimators for options 1, 2 and 3 of the Band model.

As might be expected, dropping sites from the analysis can influence the final within-windfarm avoidance rates. Only sites at which there is a relatively limited level of flight activity can be dropped from the analysis before the within-windfarm avoidance rates derived become less stable (Figure 5.20). In all three model options, this is noticeable after around 22,000 of the 2,605,681 flights through the windfarms have been removed (Figure 5.20).

Using option 1 of the Band model, dropping sites from the analysis results in an increase in the overall within-windfarm avoidance. This result suggests that, for option 1, a higher flux rate is associated with a higher within-windfarm avoidance rate. Collisions between birds and turbines are relatively rare events, so studies carried out over a month or two may under-estimate mean annual within-windfarm avoidance rates if they are targeted to specific times of year when collisions are more likely. Amongst our datasets, there was a propensity for studies carried out during the breeding season. At Zeebrugge, both herring and lesser black-backed gulls showed a marked increase in their within-windfarm avoidance rates during the autumn than during the breeding season. At present, data are not robust enough to enable detailed analysis of seasonal patterns in within-windfarm avoidance behaviour, but this is an area that would benefit from such analyses as better data become available.

Initially a similar pattern is evident with option 3 of the Band model. However, when only the last few sites are included in the analysis, the final within-windfarm avoidance rates derived using ratio estimators start to fall (Figure 5.20). The decline is driven by breeding season data from Zeebrugge and Bouin, sites where turbines are situated close to the edge of breeding colonies. The reason the pattern is not evident in the within-windfarm avoidance rates derived using option 1 is the variation in the difference between the proportion of birds observed at rotor height in each study and those predicted to occur at rotor height based on the modelled flight height distribution. This is apparent when the differences between within-windfarm avoidance rates derived using options 1 and 2 are considered. Options 1 and 2 differ only in the proportion of birds predicted to fly at collision risk height. The proportion of birds estimated to fly at rotor height tended to be lower than the proportion of birds observed flying at rotor height (Appendix 7). As a result, the predicted collision rate, and therefore mean within-windfarm avoidance rate, was lower using option 2 than option 1. This difference becomes exaggerated under option 3 because, in addition to accounting for a lower proportion of birds flying at rotor height, fewer of these birds are predicted to collide.

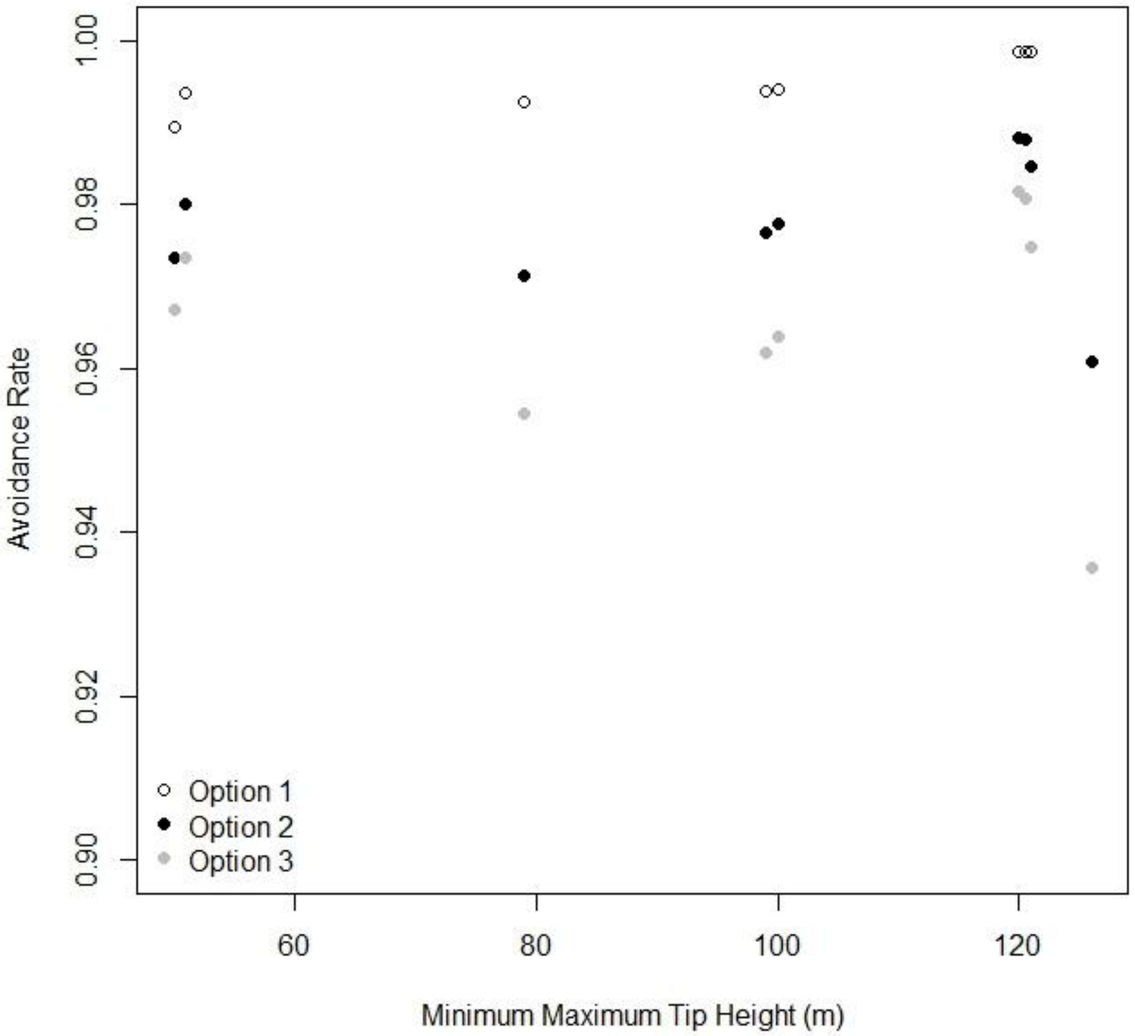


Figure 5.21 Impact of excluding sites with smaller turbines on the within-windfarm avoidance rates derived using ratio estimators for options 1, 2 and 3 of the Band model for large gulls.

Across all three model options there did not appear to be any consistent effect of excluding data collected from sites with smaller turbines on the final within-windfarm avoidance rates derived (Figure 5.21).

We consider that within-windfarm avoidance rates of 0.9893 (±0.0007 SD) for the basic Band model and 0.9672 (±0.0018 SD) for the extended Band model are realistic precautionary within-windfarm avoidance rates given the data available. Whilst we identified several sites as having a strong influence over the final values derived, we do not feel there is sufficient reason to exclude these data from our analysis. It should be noted that the influence of these sites occurs in similar magnitudes in both positive and negative directions. We did not identify any strong impact of turbine size on the final within-windfarm avoidance rate derived.

All terns

A total of 1,286,562 terns were expected to have passed through one site – Zeebrugge – during June 2004 and June 2005. After adjustments were made to this total to account for the proportion of birds flying at rotor height, the size of the rotor swept area and the probability of birds passing through the rotor-swept area without colliding, this was predicted to result in 1,408 collisions based on option 1, 1,299 collisions based on option 2, and 1,011 based on option 3. However, in total only 21 tern collisions were recorded across all sites during their respective study periods. This corresponds to within-windfarm avoidance rates of 0.9851 (±0.0022 SD) using option 1 of the Band model, 0.9838 (±0.0031 SD) using option 2 of the Band model and 0.9792 (±0.0040 SD) using option 3 of the Band model.

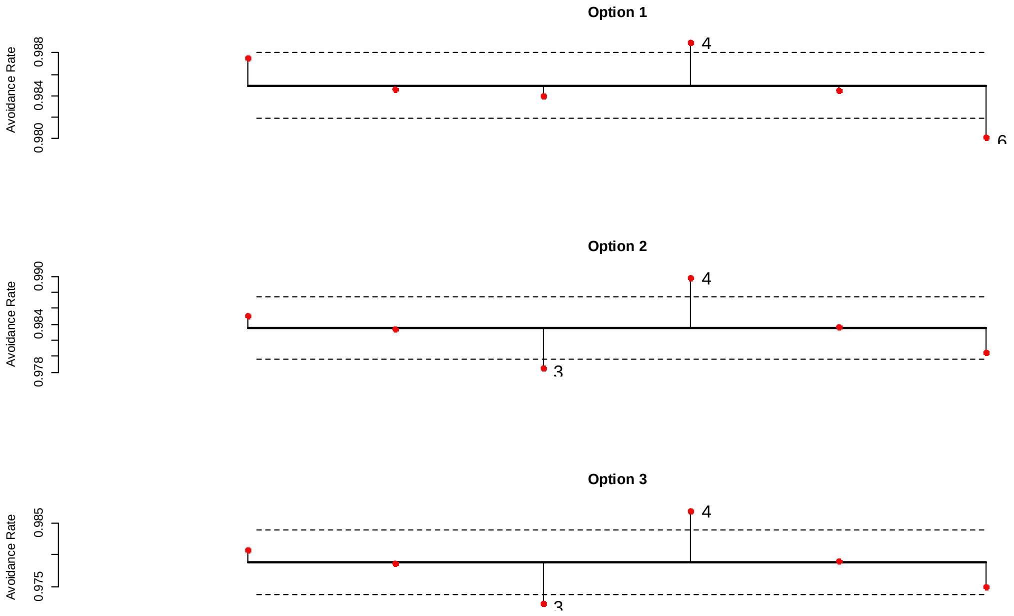


Figure 5.22 Leverage exerted by each site at which within-windfarm avoidance rates were calculated on the overall, mean within-windfarm avoidance rate derived for terns. Solid line indicates mean within-windfarm avoidance rate across all sites, broken line indicates mean within-windfarm avoidance rate across all sites ± 1 standard deviation, dots indicate mean within-windfarm avoidance rate with each site excluded from analysis. Sites are considered to have high leverage when their exclusion from the analysis leads to a change of more than 1 standard deviation in the overall mean within-windfarm avoidance rate. Points with high leverage are 3 – Sandwich tern in June 2004, 4 – common tern in June 2005, 6 – Sandwich tern in June 2005.

There was no obvious pattern in the data points with high leverage. Using all three model options, excluding common tern data from June 2005 was found to result in an increased within-windfarm avoidance rate, reflecting the relatively high collision rate involving this species in this year (Figure 5.22). Using option 1, excluding

Sandwich tern data from June 2005 resulted in a decrease in the within-windfarm avoidance rate derived. Using options 2 and 3 the same was true of Sandwich tern data in June 2004. This is likely to reflect the fact that relatively few collisions were recorded involving this species, despite a high flux rate. Differences between model options are likely to result from differences between the proportion of birds observed at collision risk height during surveys, and that estimated from the modelled distributions. We do not consider there to be a valid reason to exclude any of these data from our analysis when deriving within-windfarm avoidance rates.

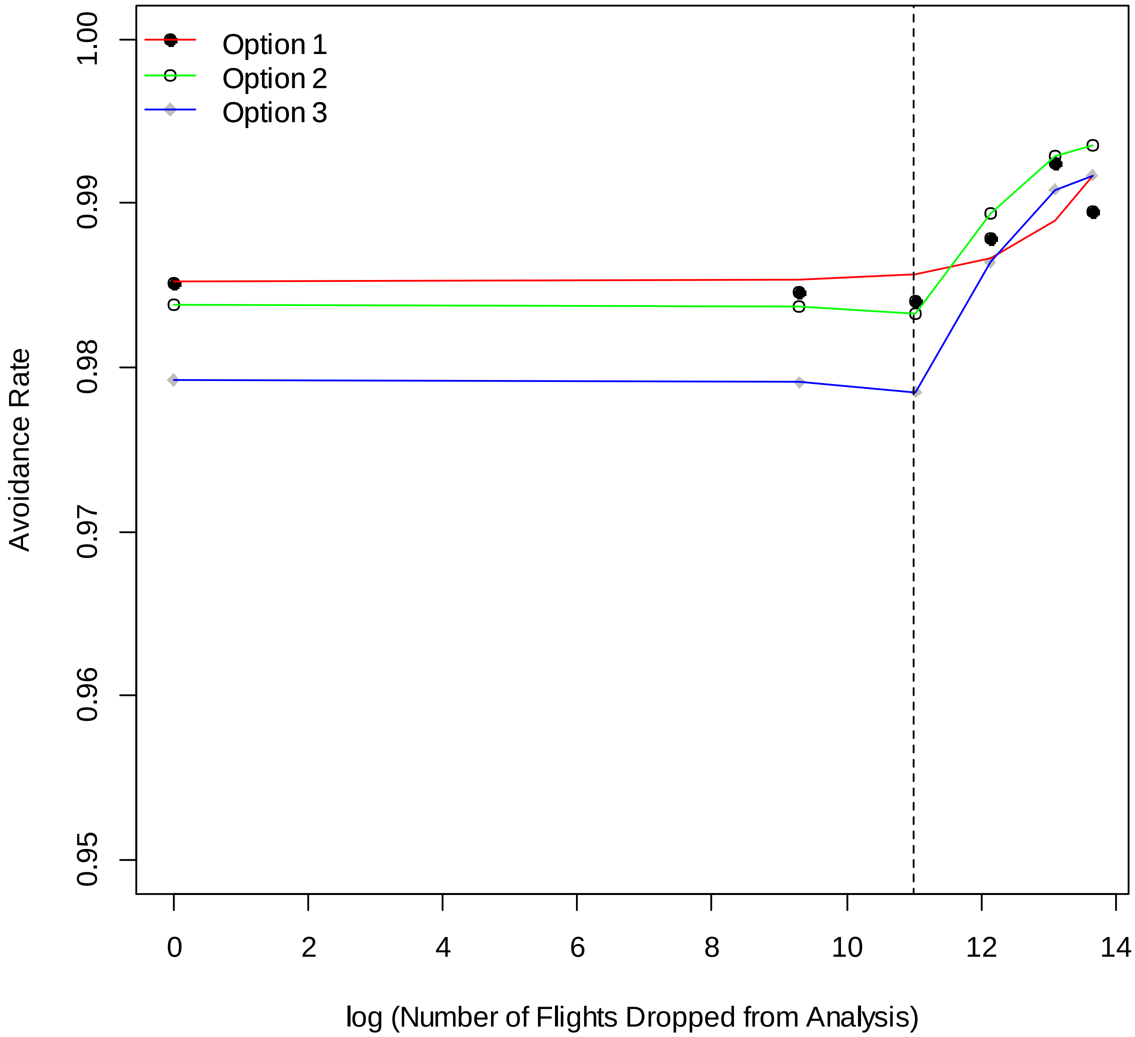


Figure 5.23 Impact of dropping data points (each year-species combination) on the within-windfarm avoidance rates derived using ratio estimators for options 1, 2 and 3 of the Band model for terns.

Within-windfarm avoidance rates derived using all three model options remain relatively stable as the first 660,000 flights through windfarms are dropped from the analysis (Figure 5.23), before increasing as only the species with the highest levels of activity remain. This reflects the fact that Sandwich terns, the species with the highest levels of activity were involved in relatively few collisions, resulting in an overall increase in the within-windfarm avoidance rates as other species were dropped from the analysis.

We consider that within-windfarm avoidance rates of 0.9851 (±0.0022 SD) for the basic Band model and 0.9792 (±0.0040 SD) for the extended Band model are realistic precautionary within-windfarm avoidance rates given the data available. Whilst we determined that some data points had a high level of leverage on the final values derived, we did not feel that there was sufficient justification for excluding them from our analysis. It should be noted that this leverage occurred in both positive and negative directions. However, as data come from only a single site, it is unclear how transferable they are to novel sites.

6. SENSITIVITY ANALYSIS

Within windfarm avoidance rates can be derived from sites at which estimates of collision rates and bird activity are available using the parameters listed in Table 6.1 and following equation 6 (see section 3.1). However, many of these parameters are incorporated into the calculations as mean values, or a range of values, and others must be estimated. Therefore, in order to understand how transferable these values may be between different models and situations, it is important to understand how sensitive the final avoidance rates are to each of the model input parameters. If avoidance rates are found to be highly sensitive to variation in one or more of the input parameters, it may raise questions about whether or not it is appropriate to apply the avoidance rates derived to novel sites.

For this reason, we assess the sensitivity of each of the avoidance rates presented in Appendix 7 to different input parameters. These parameters include corpse correction factors used to correct for the efficacy of corpse searches (observed collision rate in eq. 6), which will be influenced by scavenger behaviour and searcher efficiency, and estimates of the number of birds passing through a windfarm over a given period of time (flux rate in eq. 6). They also include parameters used to calculate collisions in the absence of avoidance behaviour (P_{coll} in eq. 6) including bird behavioural parameters such as flight speed and altitude, and turbine parameters such as rotor speed and pitch.

Avoidance may also be sensitive to a range of additional factors which cannot be easily quantified. These include time of day, weather, proximity to breeding colonies or overlap with migration routes and the size of the turbines concerned. We use a brief literature review to consider how each of these factors may influence the avoidance rates we derive.

6.1 Avoidance rates derived using the basic Band model (options 1 and 2)

The variables used to estimate P_{coll}, the first step to deriving an avoidance rate, are subject to differing levels of uncertainty. Some, such as rotor diameter, blade width and turbine height are fixed and are, therefore, known quantities with very little, if any, uncertainty surrounding them. Others, such as rotor speed and pitch and aspects of bird behaviour, such as flight speed and altitude and the propensity to fly upwind or downwind are subject to a greater degree of uncertainty. As part of the sensitivity analysis, we focus on the parameters which are not fixed and, therefore, subject to varying degrees of uncertainty, in order to determine what influence the inaccurate estimation of each of these parameters has on the final derived avoidance rates. Whilst the focus of much of the interest in collision risk modelling has been on avoidance rates, it is actually 1-avoidance rate, or the non-avoidance rate which is applied in the final step of the Band collision risk model (Band *pers. comm.*, Masden *et al.* in prep). For this reason, we focus our sensitivity analysis on this factor, rather than the avoidance rate.

For each of the sites and species combinations presented in Appendix 7 at which collisions were recorded, we consider the impact that a 10% increase (following Chamberlain *et al.* 2006) in each of rotor speed, rotor pitch, bird flight speed, flux rate and the proportion of flights upwind would have on the avoidance rates derived using option 1. In addition, we also consider the influence of a 10% increase in corpse detection rate.

Table 6.1 Input parameters for the Band (basic and extended model)

Parameter	**Sensitivity assessed**
Species name	No
Bird length	No
Wingspan	No
Flight speed	Yes – 10% increase considered following Chamberlain *et al.* (2006)
Nocturnal activity factor (1-5)	Considered as part of increase in flux rate
Flight type, flapping or gliding	No – Seabirds most likely to engage in flapping flight, which is the higher risk activity
Daytime bird density	Yes – considered as part of increase in flux rate
Proportion at rotor height	Yes – 10% increase in birds flying at risk height considered for basic model, 200 randomly simulated distributions considered for the extended model
Proportion of flights upwind	Yes – 10% increase in the proportion of birds flying upwind considered
Name of windfarm site	No – Fixed parameter
Latitude	No – Fixed parameter
Number of turbines	No – Fixed parameter
Width of windfarm	No – Fixed parameter
Tidal offset	No – Suitable datasets were only available for onshore windfarms
Turbine model	No – Fixed parameter
No. of blades	No – Fixed parameter
Mean rotation speed	Yes – 10% increase considered following Chamberlain *et al.* (2006)
Rotor radius	No – Fixed parameter
Hub height	No – Fixed parameter
Monthly proportion of time operational	Yes – considered as part of increase in flux rate
Max blade width	No – Fixed parameter
Pitch	Yes – 10% increase considered following Chamberlain *et al.* (2006)

6.1.1 Sensitivity to the assumed flux rate at the windfarm

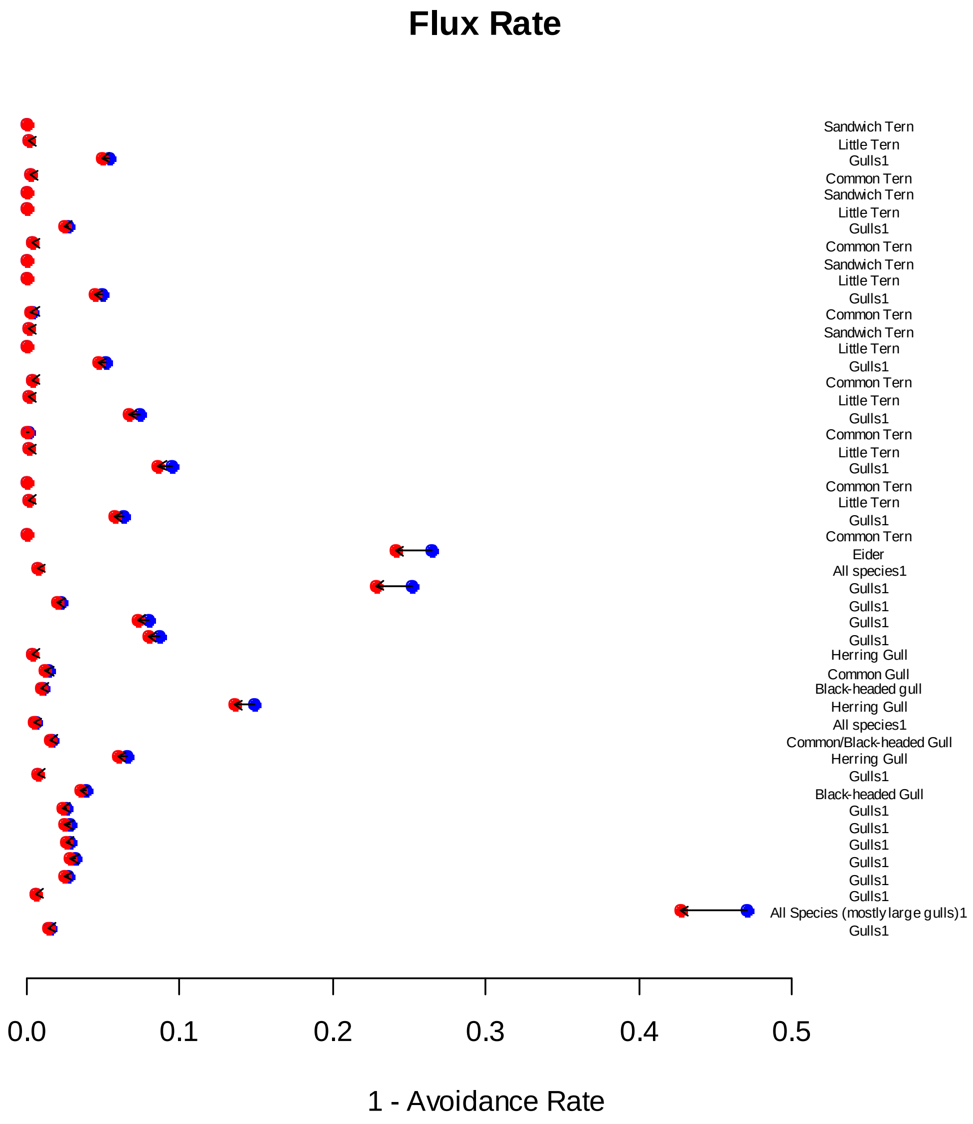


Figure 6.1 Sensitivity of non-avoidance rates for each species and site in Appendix 7 at which a collision was recorded and derived using the basic Band model, to the assumed flux rate at each site. Blue dots indicate the non-avoidance rate derived assuming the flux rate presented in Appendix 7, red dots indicate the non-avoidance rate derived assuming a 10% increase in the flux rate at each site.

Bird flux rate is an estimate of the total number of birds passing through the windfarm when it is operational. As such, it combines estimates of the number of birds recorded within the windfarm, the proportion of birds at collision risk height, corrections for nocturnal activity and an estimate of the monthly proportion of time it is operational. An increase in the flux rate derived at each site results in a decrease in the derived non-avoidance rates (Figure 6.1). This is because, whilst the observed number of collisions remains constant, the number of birds passing through the windfarm increases, meaning that a greater proportion of them are assumed to have avoided collision. These changes are approximately inversely proportional to the increase in the numbers of birds passing through the site. A comparison of the mean non-avoidance rates based on the flux rate presented in Appendix 7, with the mean non-avoidance rates assuming a 10% increase in this flux rate suggests that such an increase may result in a 9.1% decrease in the non-avoidance rate.

6.1.2 Sensitivity to the corpse detection rate at the windfarm

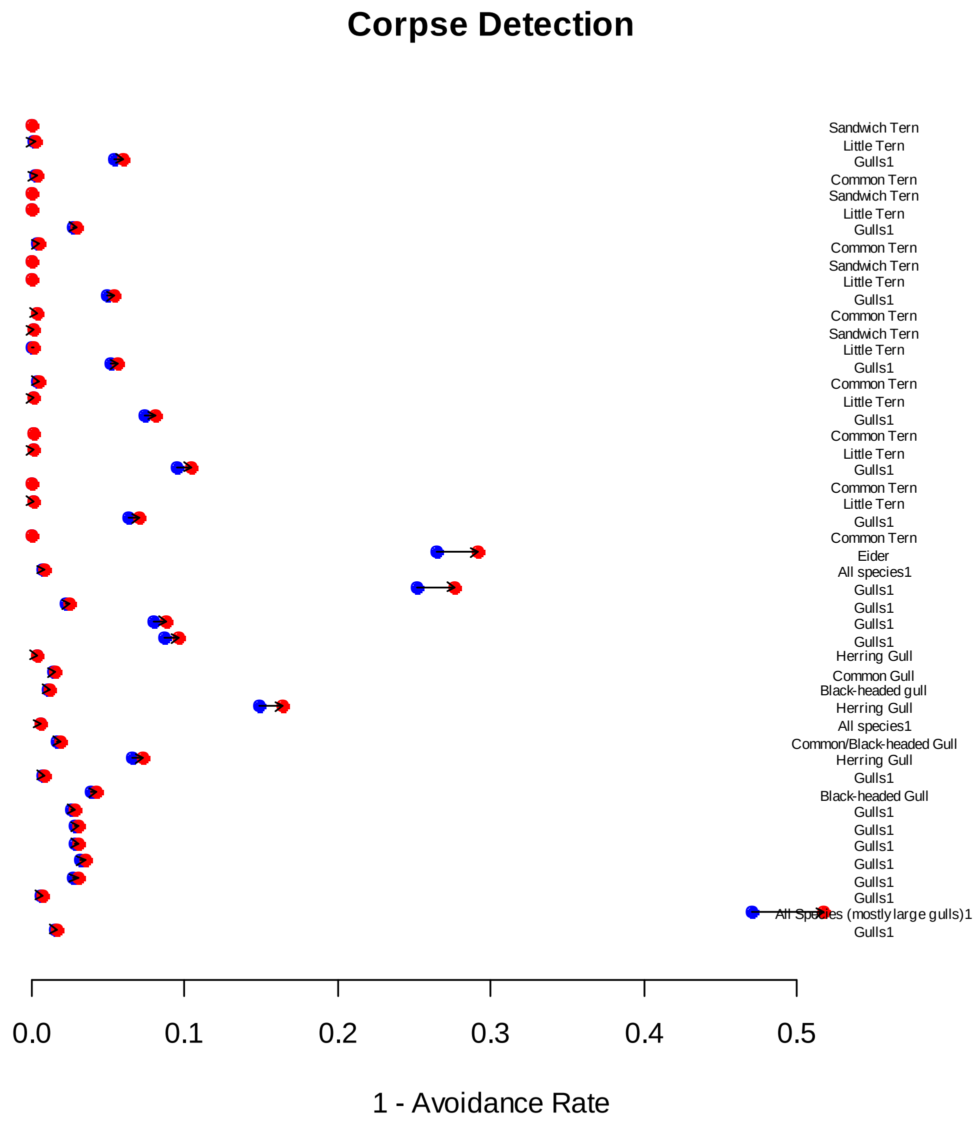


Figure 6.2 Sensitivity of non-avoidance rates for each species and site in Appendix 7 at which a collision was recorded and derived using the basic Band model, to the assumed corpse detection rate at each site. Blue dots indicate the non-avoidance rate derived assuming the

number of collisions presented in Appendix 7, red dots indicate the non-avoidance rate derived assuming a 10% increase in the number of collisions detected at each site.

During the search for collision victims, corpses may be missed either as a result of searcher inefficiency, or through the removal of carcasses by predators (Winkelman 1992). As a result it is often necessary to correct observed collision rates to account for these missing corpses. Assuming an increase in the total number of victims leads to an increase in the derived non-avoidance rate because the total number of birds passing through the windfarm remains constant and it is assumed a higher proportion of them collide with the turbines. These increases in the non-avoidance rate are proportional with the increase in corpse detection (Figure 6.2), with a 10% correction in the number of collisions to account for a failure to detect corpses resulting in 10% increase in the non-avoidance rate.

6.1.3 Sensitivity to the proportion of birds flying upwind

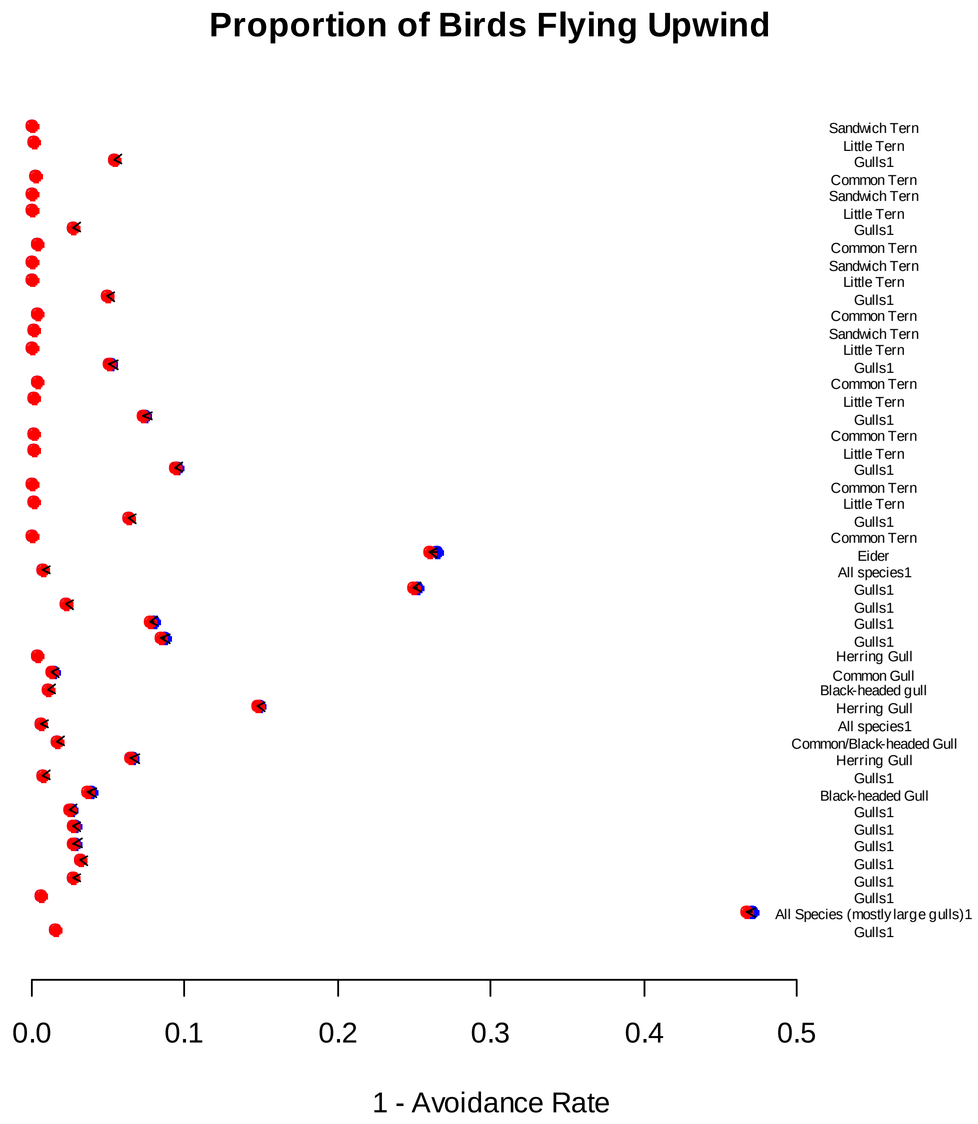

Proportion of Birds Flying Upwind

Figure 6.3 Sensitivity of non-avoidance rates for each species and site in Appendix 7 at which a collision was recorded and derived using the basic Band model, to the proportion of birds flying upwind at each site. Blue dots indicate the non-avoidance rate derived assuming the number of collisions presented in Appendix 7, red dots indicate the non-avoidance rate derived assuming a 10% increase in the proportion of birds flying upwind detected at each site.

A 10% change to the proportion of birds flying upwind resulted in a small decrease in the derived non-avoidance rates (Figure 6.3) of 1.17%. These results suggest that the proportion of birds estimated to fly up or downwind has a relatively small effect on the final, derived non-avoidance rate.

6.1.4 Sensitivity to the mean turbine rotor speed

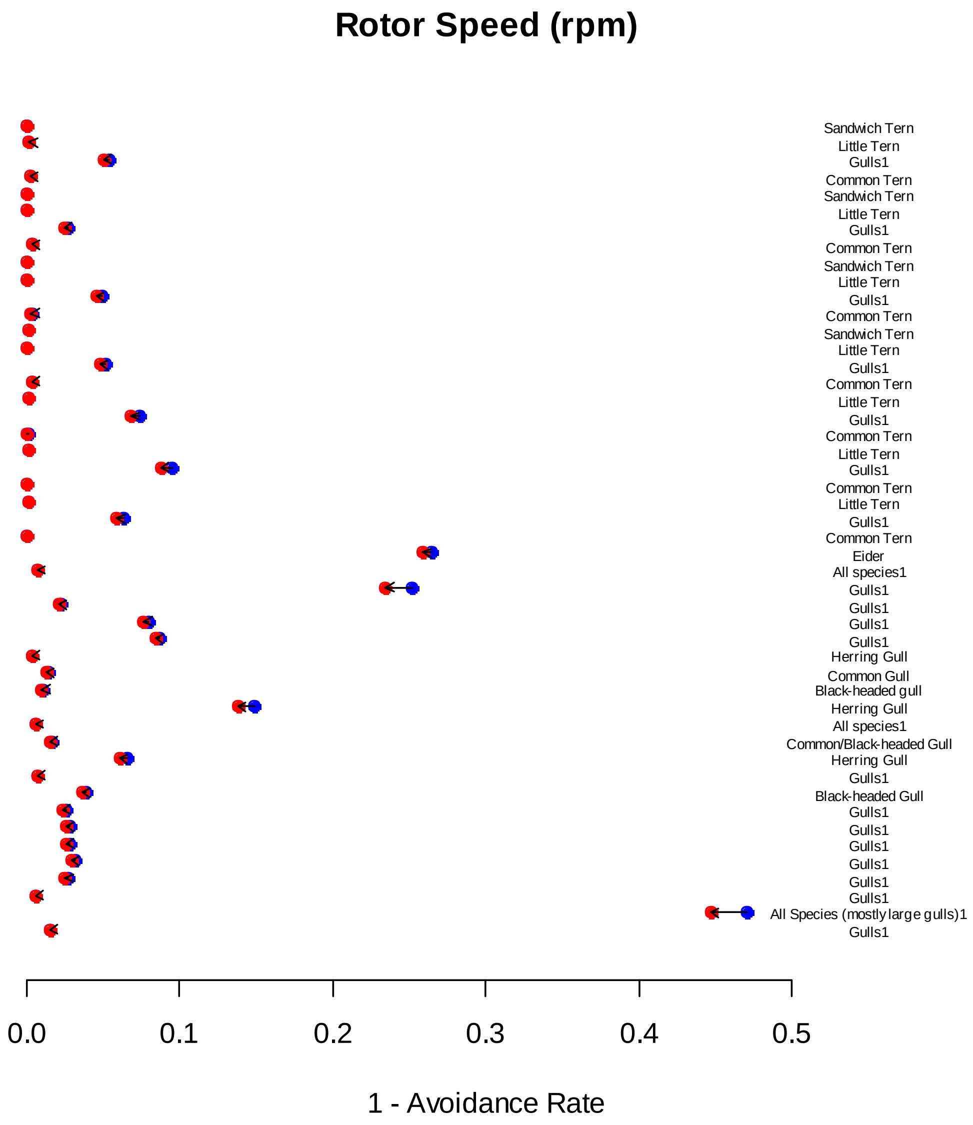


Figure 6.4 Sensitivity of non-avoidance rates for each species and site in Appendix 7 at which a collision was recorded and derived using the

basic Band model, to the turbine rotor speed at each site. Blue dots indicate the non-avoidance rate derived assuming the number of collisions presented in Appendix 7, red dots indicate the non-avoidance rate derived assuming a 10% increase in the turbine rotor speed at each site.

A 10% increase in the mean turbine rotor speed assumed typically resulted in a decrease in the derived non-avoidance rates of approximately 5.5% (Figure 6.4). The reason for this decrease is that as the rotor speed increases, the time available for a bird to pass through unharmed decreases, meaning that the predicted collision rate increases whilst the recorded number of collisions remains constant. Based on the turbines we considered, a 10% increase in mean rotor speed reflects an increase of between 1 and 4 rotations per minute. Published data from turbine manufacturers (http://www.4coffshore.com/) suggests the range of operational speeds for turbines is like to vary by between 5 and 15 rpm. As such, the increase in rotation speed we consider may be somewhat conservative but, without more detailed curves showing the range of operational speeds used by different turbines, assessing this is difficult.

6.1.5 Sensitivity to the turbine pitch

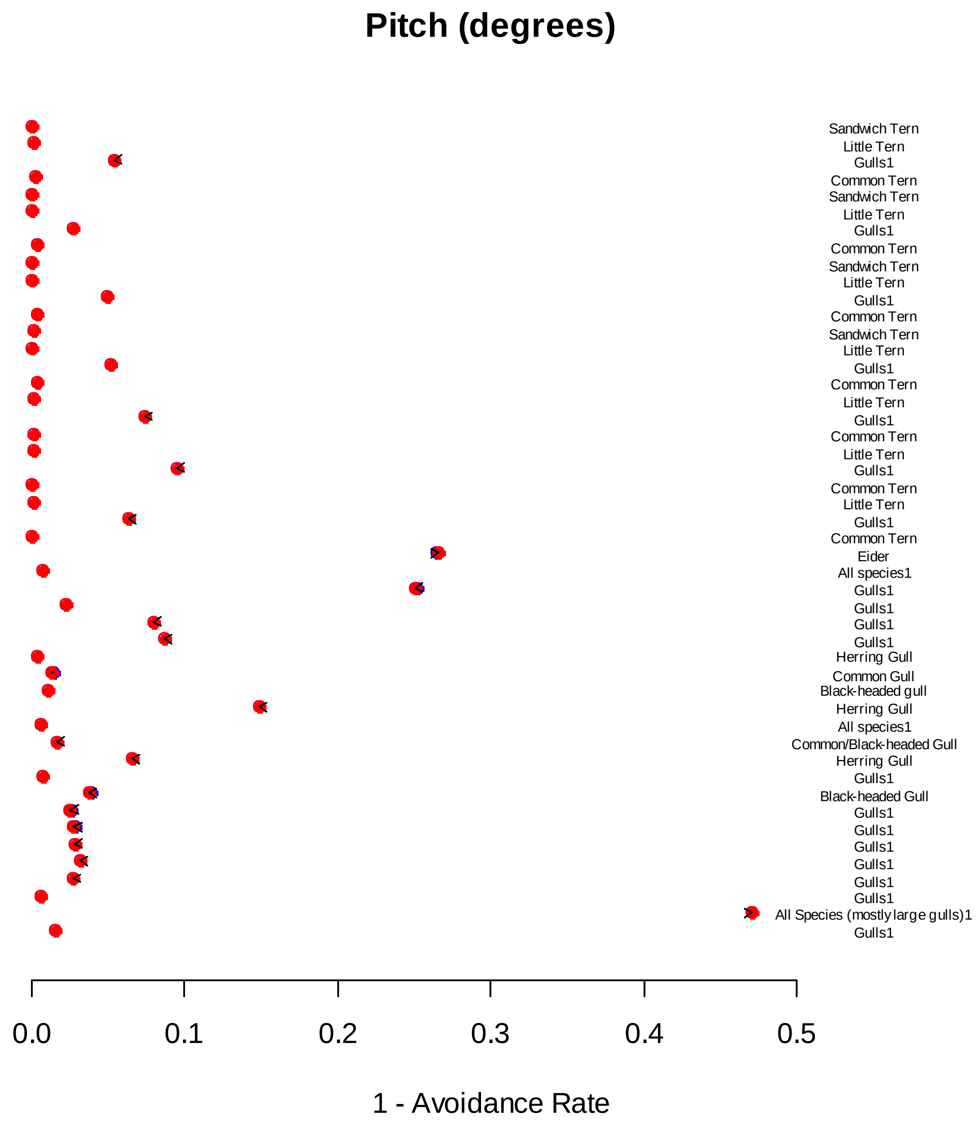


Figure 6.5 Sensitivity of non-avoidance rates for each species and site in Appendix 7 at which a collision was recorded and derived using the basic Band model, to the turbine pitch at each site. Blue dots indicate the non-avoidance rate derived assuming the number of collisions presented in Appendix 7, red dots indicate the non-avoidance rate derived assuming a 10% increase in the turbine pitch at each site.

A 10% change in the assumed turbine pitch resulted in a fairly negligible decrease in the derived non-avoidance rates (Figure 6.5) of 0.2%. Our calculations were based on an assumption of a 10˚ pitch for each turbine, so a 10% increase reflects an 11˚ pitch. Available data describing the pitch of operational turbines are extremely limited. As a consequence, it is not possible to determine how well these values reflect reality at operational turbines.

6.1.6 Sensitivity to the bird flight speed

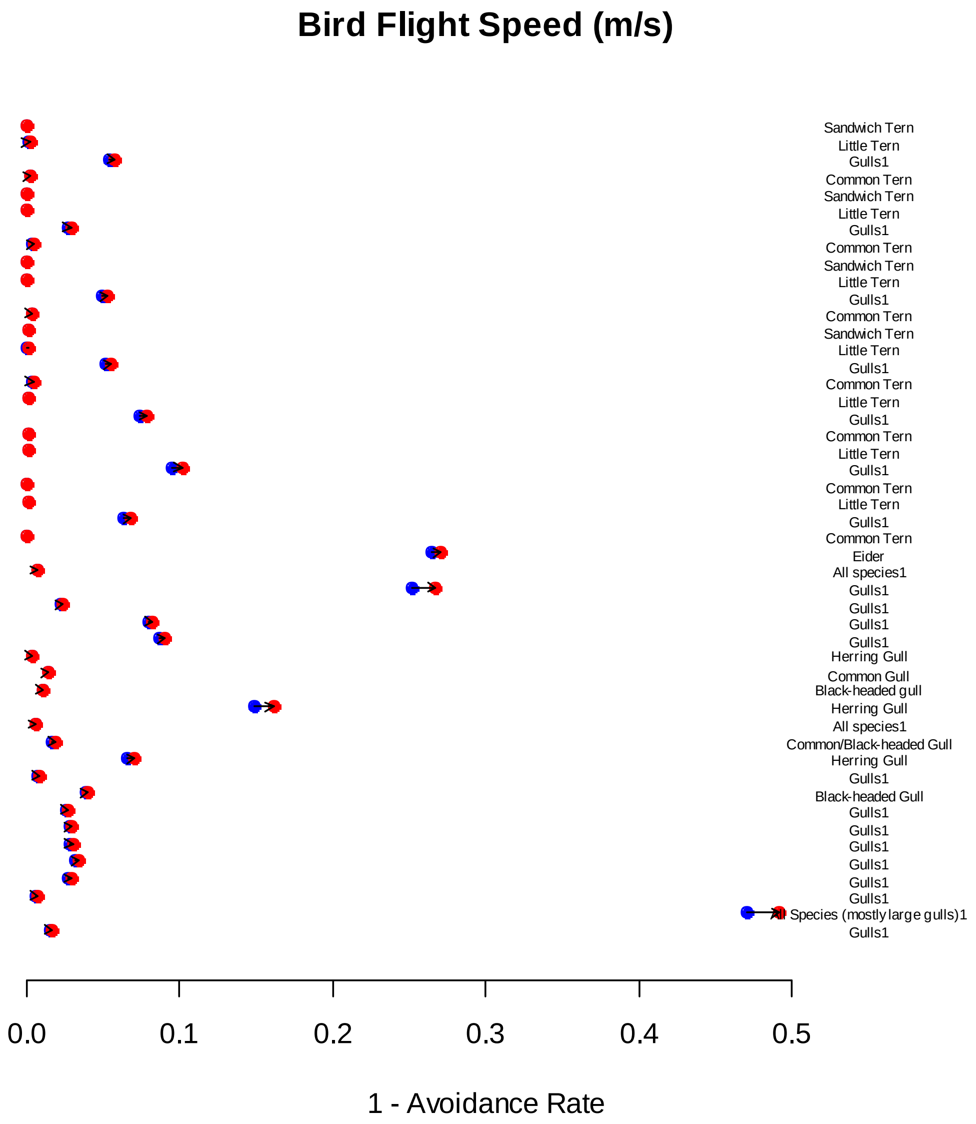

Bird Flight Speed (m/s)

Figure 6.6 Sensitivity of non-avoidance rates for each species and site in Appendix 7 at which a collision was recorded and derived using the basic Band model, to the bird flight speed at each site. Blue dots indicate the non-avoidance rate derived assuming the number of collisions presented in Appendix 7, red dots indicate the avoidance rate derived assuming a 10% increase in the bird flight speed at each site.

A 10% increase in the assumed bird flight speed resulted in an increase in the derived non-avoidance rates (Figure 6.6) of 5.5%. This increase reflects the fact that the faster a bird passes through the rotor swept-area, the less likely it is to be hit. As a result an increase in flight speed results in a decrease in the predicted number of collisions whilst the observed number of collisions remains constant. For our study species a 10% increase in flight speed reflects an increase of 1-1.3 m/s. Alerstam *et al.* (2007) suggest that the standard deviations around the mean flight speeds for our study species are in the region of 1-2 m/s, suggesting that a 10% increase in flight speed may be a realistic, precautionary assumption.

6.1.7 Basic Band model sensitivity conclusions

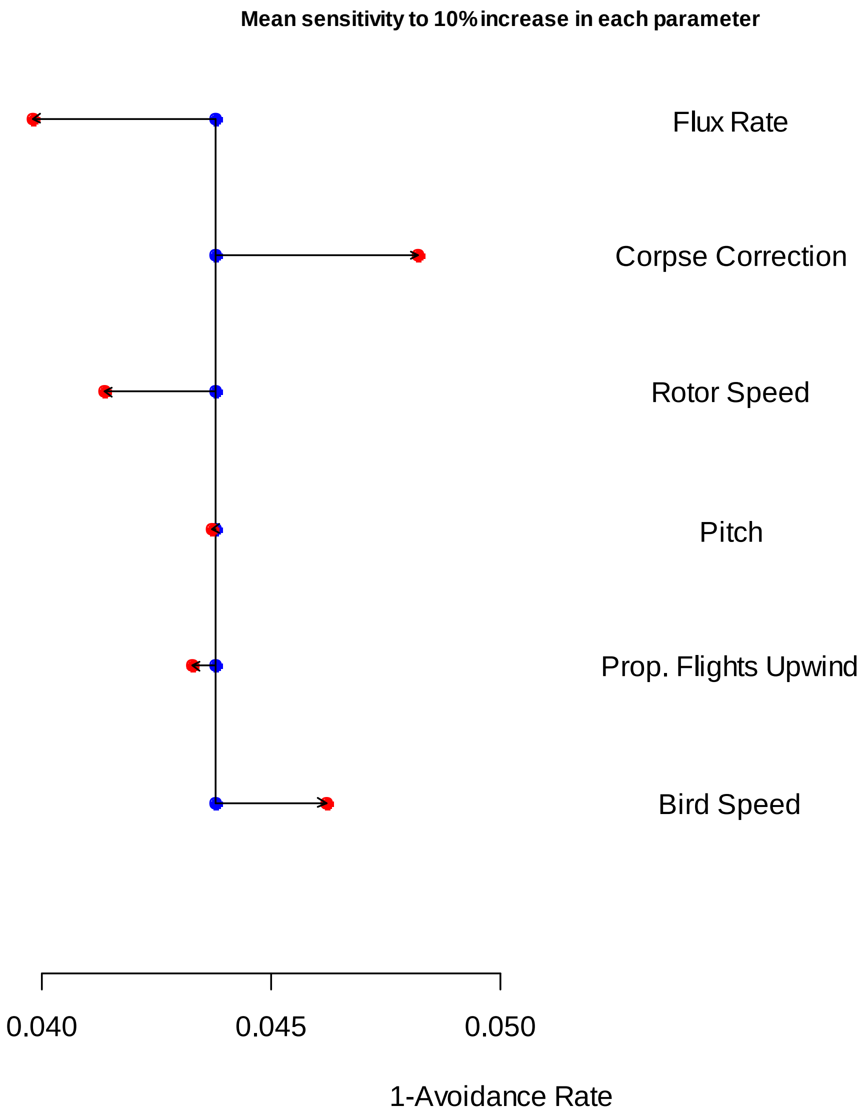


Figure 6.7 Sensitivity of derived non-avoidance rates derived using the basic Band model to a 10% increase in each of the Band model parameters.

Of the parameters considered, the final derived non-avoidance rates were most sensitive to flux rate and the corpse correction (Figure 6.7). An increase in the flux rate meant that the predicted collision rate increased, whilst the observed collision rate remained constant (see eq. 6, section 3.1); as a consequence, the non-avoidance rate decreased in response to an increase in the flux rate. For similar reasons, an increase in the number of corpses detected resulted in an increase in the non-avoidance rate derived. The impacts of assumed rotor speed and bird speed on the derived non-avoidance rates were of a similar magnitude, but in opposite directions. An increased assumed rotor speed results in a decreased non-avoidance rate because faster turbines result in an increased risk of collision. As a consequence, a faster rotor speed would result in an increase in the predicted collision rate, whilst the observed collision rate remains constant. This results in a decrease in the non-avoidance rate. In contrast, an increase in the assumed speed of the birds passing through the rotor swept area of a turbine decreases the risk of collision. As a consequence, the predicted collision rate decreases and, for the reasons stated above, the non-avoidance rate derived increases. Whilst increases in both the assumed pitch and the proportion of flights upwind resulted in decreases in the derived non-avoidance rates, the impact of both parameters was negligible.

6.2 Avoidance rates derived using the extended Band model (option 3)

In addition to the variables described above (section 6.1), non-avoidance rates derived using the extended Band model are also likely to be sensitive to the assumed flight height distributions. Collision risk is not evenly distributed within the rotor swept area of turbines, and is greatest towards the centre of the rotor disk. The extended Band model makes use of flight height distributions, such as those derived by Johnston *et al.* (2014a) to account for this variable risk. However, as these are continuous distributions, it is not appropriate to simply assume, for example, that an additional 10% of birds fly at rotor height as this will have implications for the overall shape of the distribution. Therefore, in addition to the parameters considered for the basic Band model, for each species/site combination we consider, we use 200 random distributions estimated following the methodology of Johnston *et al.* (2014a) to investigate sensitivity to the assumed distribution (Figure 6.8). It is important to note that by comparing between different distributions, the outputs of the sensitivity analysis will not be strictly comparable to the outputs of the sensitivity analyses described above.

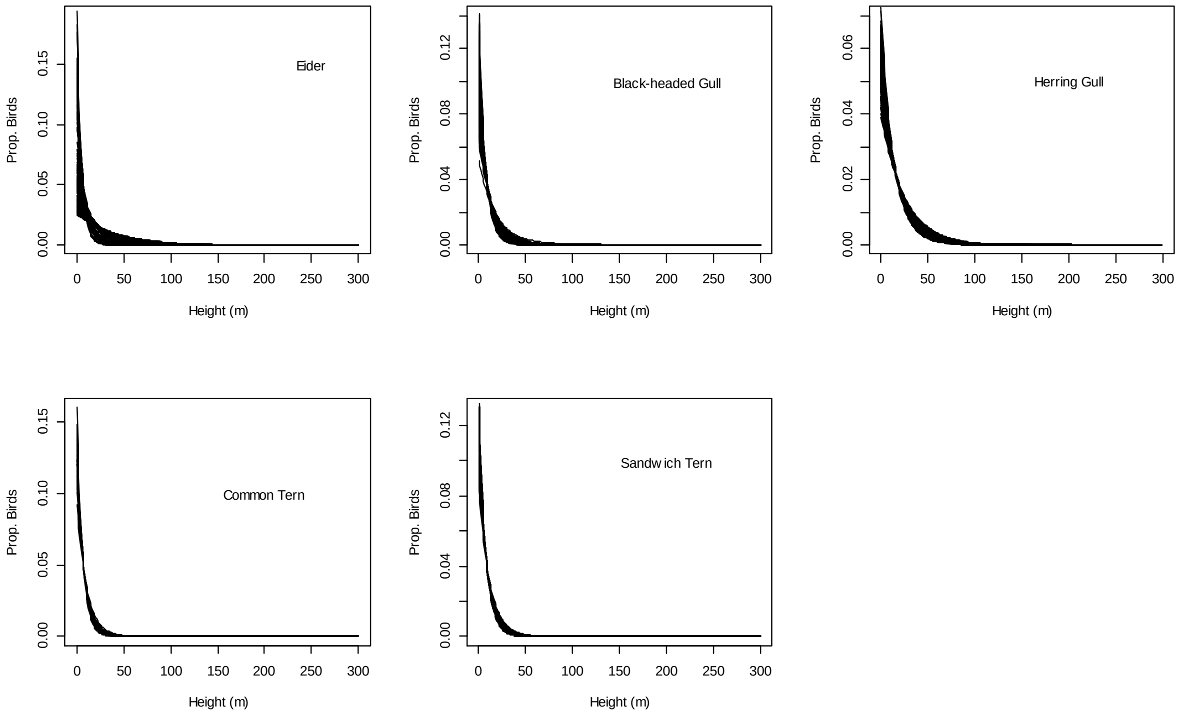


Figure 6.8 200 Random flight height distributions estimated for each of eider, black-headed gull, herring gull, common tern and Sandwich tern, species for which avoidance rates could be derived from a combination of recorded collisions and recorded levels of bird activity, using the methodology set out in Johnston *et al.* (2014a) and used to assess the sensitivity of derived avoidance rates to the assumed flight height distribution.

6.2.1 Sensitivity to assumed flight height distribution

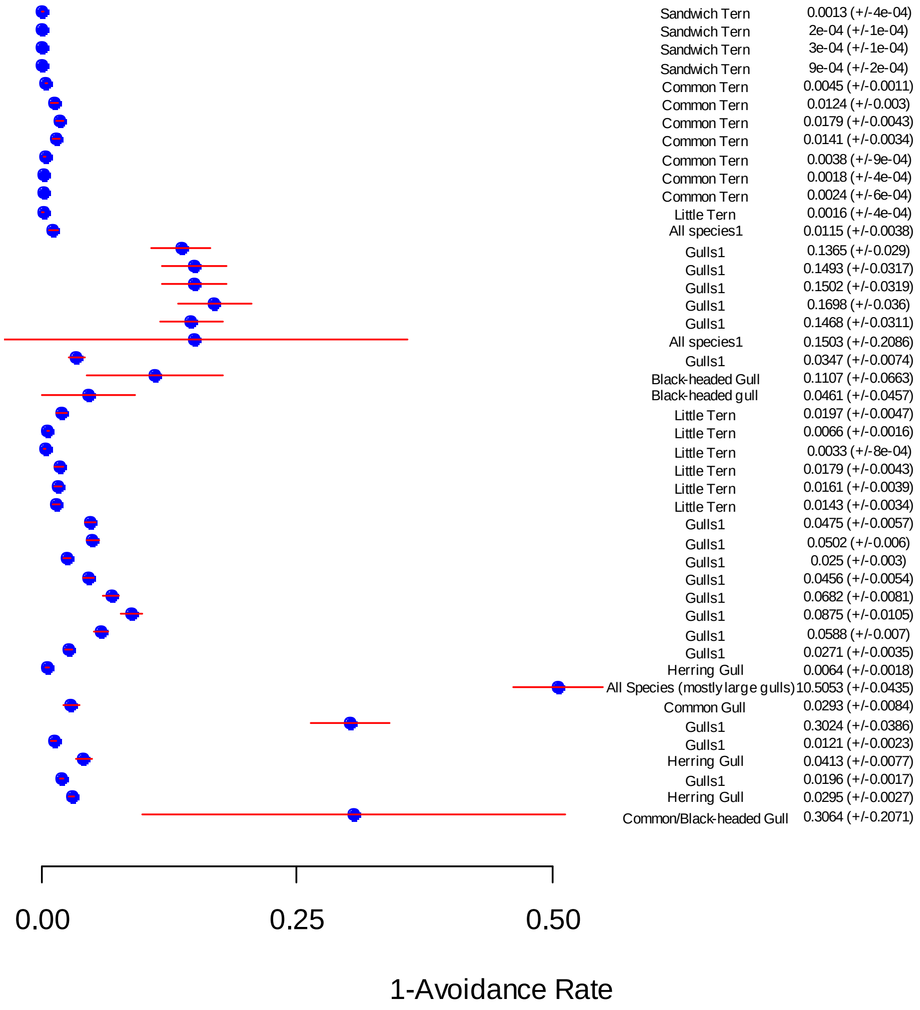


Figure 6.9 Sensitivity of non-avoidance rates derived for each species and site in Appendix 7 at which a collision was recorded and derived using the extended Band model, to the assumed flight height distribution at the site. Blue dots indicate the mean non-avoidance rate values derived from 200 random flight height distributions at each site, red lines indicate the standard deviation around these values, actual values shown alongside plot.

The sensitivity of the derived non-avoidance rates to different flight height distributions appears to be highly variable (Figure 6.9). The greatest sensitivity appears to occur where derived non-avoidance rates are highest. This relationship is likely to reflect the level of activity at any given site. For example, consider two sites, at the first of which 1 flight out of 100 at rotor height results in a collision and at the second of which 1 flight out of 1000 results in a collision. If the estimate of the proportion of birds flying at rotor height increases at each site by 10%, whilst the recorded number of collisions remains constant, this becomes 1 flight out of 110 at the first site and 1 flight out of 1,100 at the second. At the first site the non-avoidance rate decreases from 0.0100 to 0.0091, whilst at the second it decreases from 0.0010 to 0.0009. The overall decrease is therefore greater at the first site, with the lower level of flight activity.

6.2.2 Sensitivity to the assumed flux rate at the windfarm

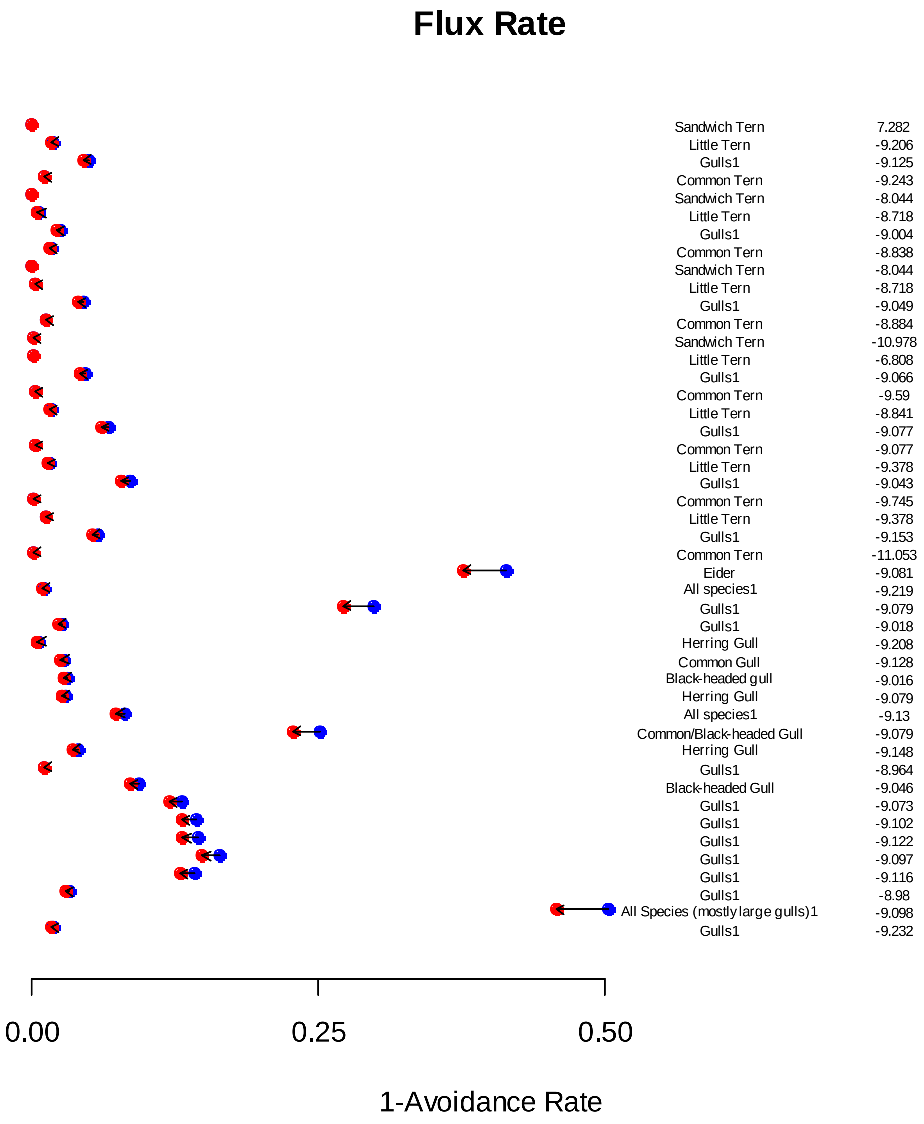


Figure 6.10 Sensitivity of non-avoidance rates derived for each species and site in Appendix 7 at which a collision was recorded and derived using the extended Band model, to the assumed flux rate at each site. Blue dots indicate the non-avoidance rate derived assuming the flux rate presented in Appendix 7, red dots indicate the non-avoidance rate derived assuming a 10% increase in the flux rate at each site. % change in the non-avoidance rates following a 10% increase in flux rate shown alongside graph.

Bird flux rate is an estimate of the total number of bird passing through the windfarm when it is operational. As such, it combines estimates of the number of birds recorded within the windfarm, corrections for nocturnal activity and an estimate of the monthly proportion of time it is operational. An increase in the flux rate derived at each site results in a decrease in the derived non-avoidance rates (Figure 6.10). This is because, whilst the observed number of collisions remains constant, the number of birds passing through the windfarm increases, meaning that a greater proportion of them are assumed to have avoided collision. These decreases are roughly inversely proportional to the increase in flux rate, although in contrast to the case of the basic Band model, this value will vary across sites as a consequence of the different height distributions assumed. A comparison of the mean avoidance rates based on the flux rate presented in Appendix 7, with the mean avoidance rates assuming a 10% increase in this flux rate suggests that such an increase may result in a mean 8.73% decrease in the non-avoidance rate.

6.2.3 Sensitivity to the corpse detection rate at the windfarm

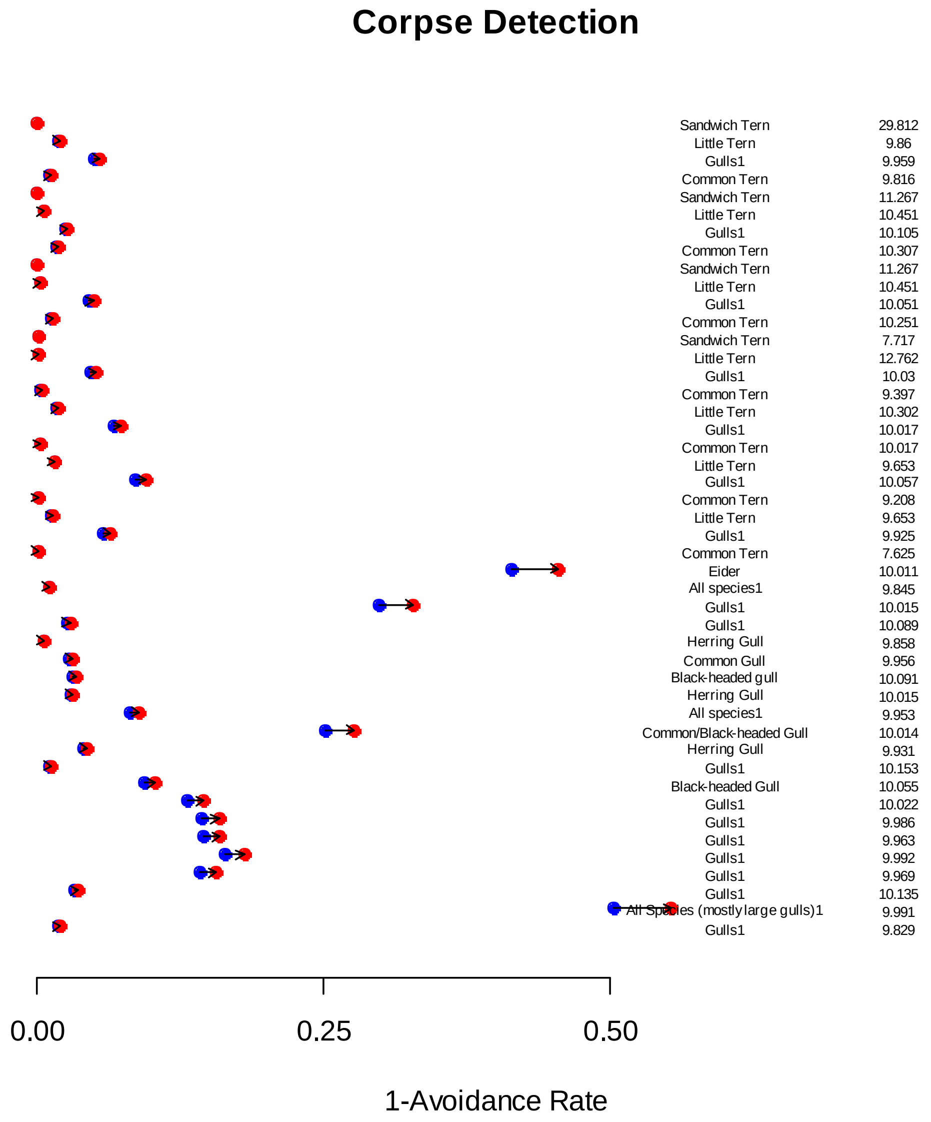


Figure 6.11 Sensitivity of non-avoidance rates for each species and site in Appendix 7 at which a collision was recorded and derived using the extended Band model, to the assumed corpse detection rate at each

site. Blue dots indicate the non-avoidance rate derived assuming the number of collisions presented in Appendix 7, red dots indicate the non-avoidance rate derived assuming a 10% increase in the number of collisions detected at each site. % change in the non-avoidance rates following a 10% increase in the number of collisions detected shown alongside graph.

During the search for collision victims, corpses may be missed either as a result of searcher inefficiency, or through the removal of carcasses by predators (Winkelman 1992). As a result it is often necessary to correct observed collision rates to account for these missing corpses. Assuming an increase in the total number of victims leads to an increase in the derived non-avoidance rate because the total number of birds passing through the windfarm remains constant and it is assumed a higher proportion of them collide with the turbines (Figure 6.11). This increase is broadly proportional with the increase in the flux rate across sites, with a mean 10.43% increase in the non-avoidance rate following a 10% increase in the flux rate.

6.2.4 Sensitivity to the proportion of birds flying upwind

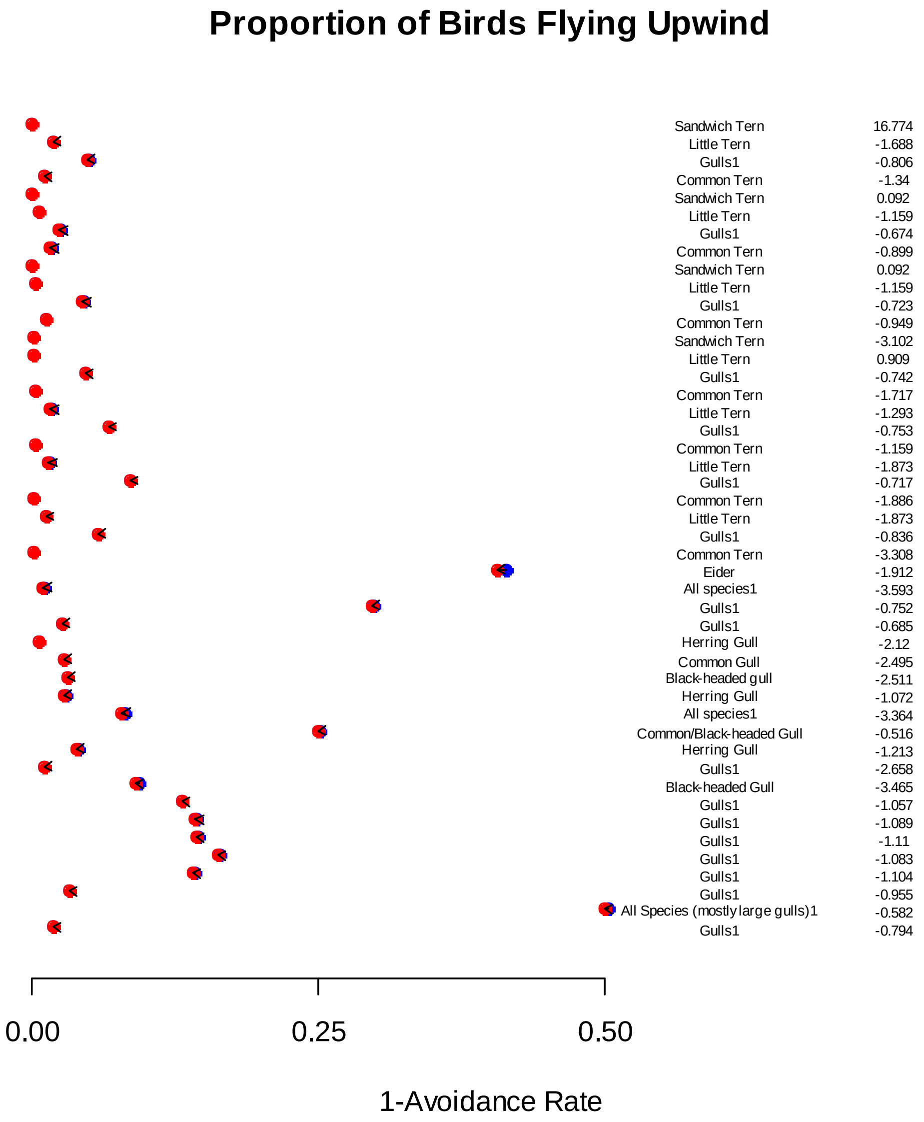


Figure 6.12 Sensitivity of non-avoidance rates derived for each species and site in Appendix 7 at which a collision was recorded and derived using the extended Band model, to the proportion of birds flying upwind at each site. Blue dots indicate the non-avoidance rate derived assuming 50% of birds flying upwind, red dots indicate the avoidance rate derived assuming a 10% increase in the proportion of birds flying upwind at each site. % change in the non-avoidance rates following a 10% increase in the proportion of birds flying upwind shown alongside graph.

A 10% change to the proportion of birds flying upwind resulted in a fairly negligible decrease in the derived avoidance rates (Figure 6.12). The % increases were typically <1%, and across all sites a 10% increase in the proportion of birds flying upwind resulted in a decrease in the non-avoidance rate of approximately 0.97%. These results suggest that the proportion of birds estimated to fly up or downwind has a negligible effect on the final, derived non-avoidance rate.

6.2.5 Sensitivity to the turbine rotor speed

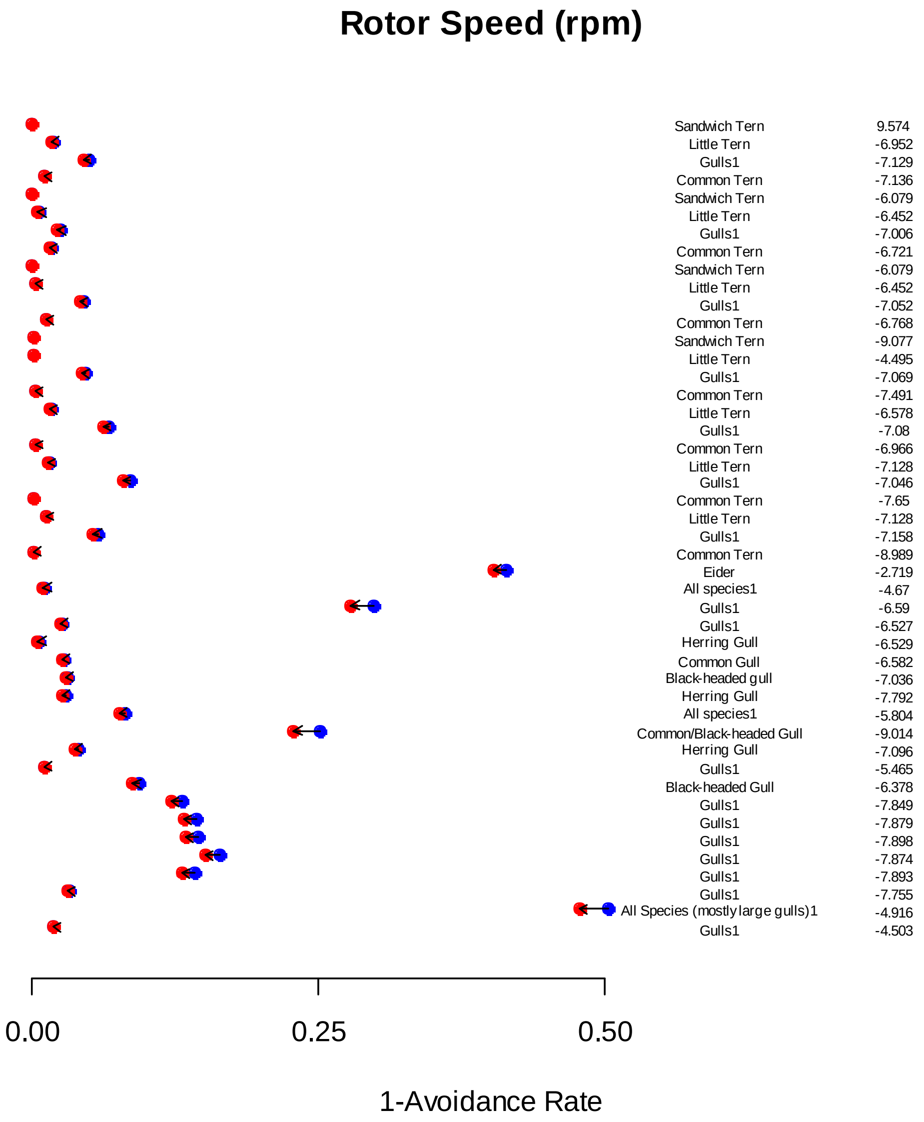


Figure 6.13 Sensitivity of non-avoidance rates derived for each species and site in Appendix 7 at which a collision was recorded and derived using the extended Band model, to the turbine rotor speed. Blue dots indicate the non-avoidance rate derived based on the rotor speed values presented in Appendix 7, red dots indicate the non-avoidance rate derived assuming a 10% increase in these rotor speeds. % change in the non-avoidance rates following a 10% increase in the turbine rotor speed shown alongside graph.

A 10% increase in the assumed turbine rotor speed typically resulted in a decrease in the derived non-avoidance rates (Figure 6.13). Across all sites a 10% increase in the rotor speed resulted in a decrease in the non-avoidance rate of approximately 6.45%.

6.2.6 Sensitivity to the turbine pitch

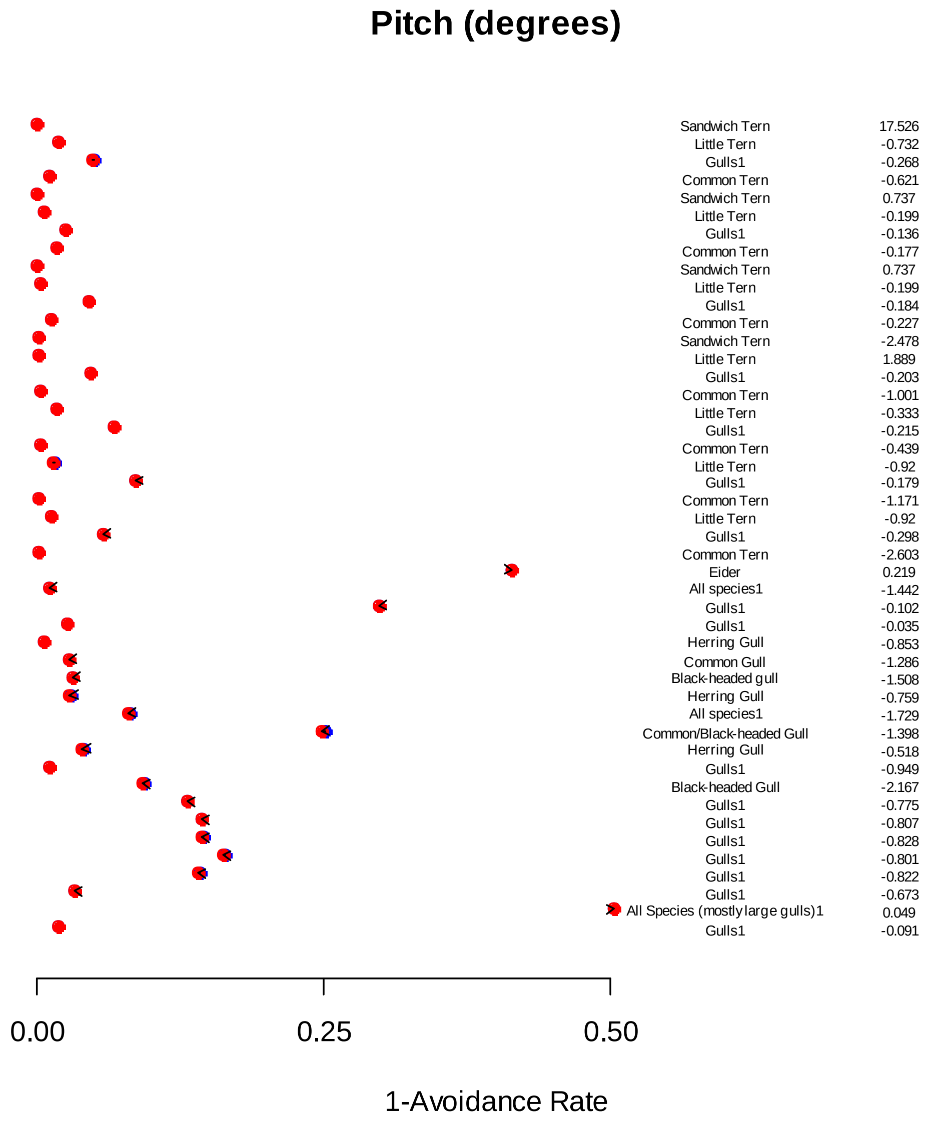


Figure 6.14 Sensitivity of non-avoidance rates derived for each species and site in Appendix 7 at which a collision was recorded and derived using the extended Band model, to the turbine pitch. Blue dots indicate the non-avoidance rate derived based on the rotor speed values presented in Appendix 7, red dots indicate the non-avoidance rate derived assuming a 10% increase in the pitch. % change in the non-avoidance rates following a 10% increase in the turbine rotor speed shown alongside graph.

A 10% change in the assumed turbine pitch resulted in a fairly negligible decrease in the derived non-avoidance rates (Figure 6.14). The % decreases were typically <1%, and across all sites a 10% increase in the turbine pitch resulted in a decrease in the non-avoidance rate of approximately 0.21%.

6.2.7 Sensitivity to the bird flight speed

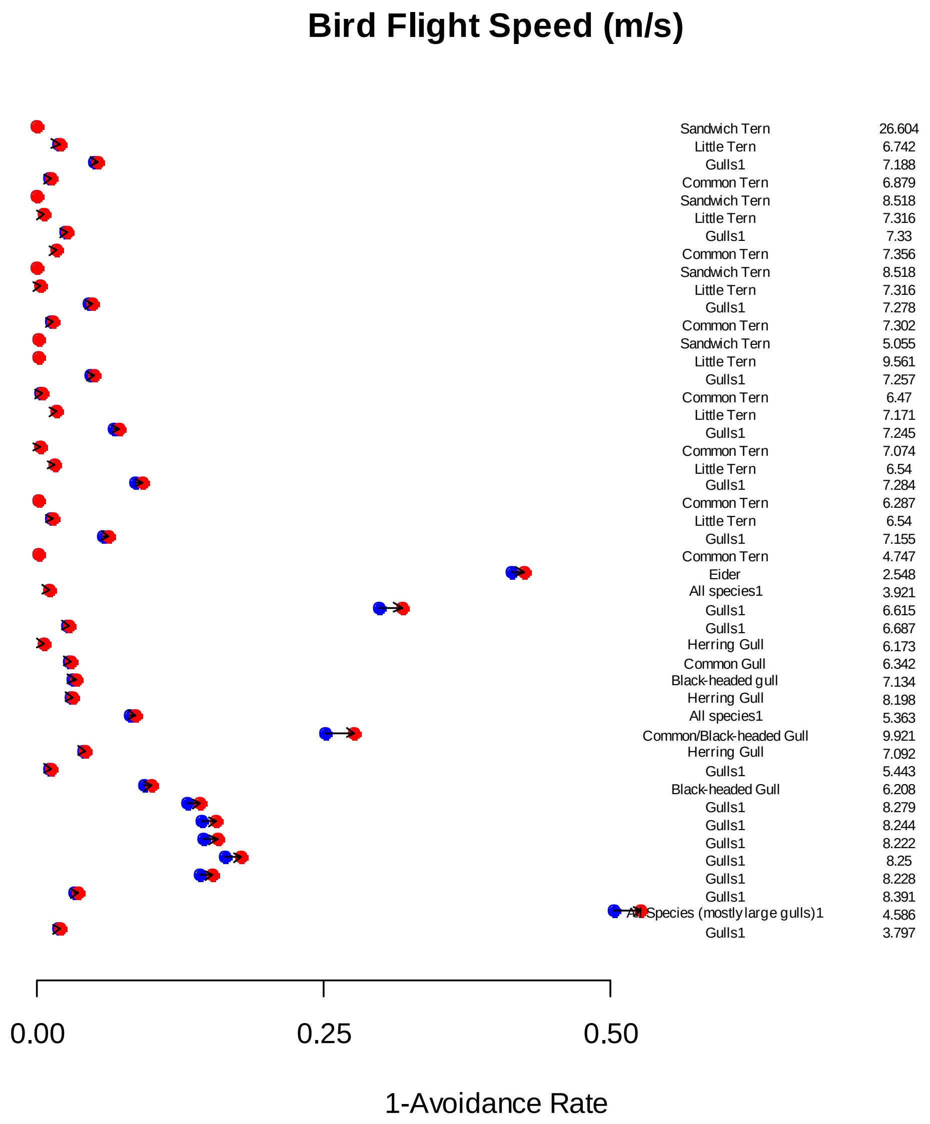


Figure 6.15 Sensitivity of non-avoidance rates derived for each species and site in Appendix 7 at which a collision was recorded and derived using the extended Band model, to the bird flight speed. Blue dots indicate the non-avoidance rate derived based on the bird flight speed values presented in Table 5.4, red dots indicate the non-avoidance rate derived assuming a 10% increase in the bird flight speed. % change in the non-avoidance rates following a 10% increase in the bird flight speed shown alongside graph.

A 10% increase in the assumed bird flight speed typically resulted in an increase in the derived non-avoidance rates (Figure 6.15). Across all sites a 10% increase in the bird flight speed resulted in an increase in the non-avoidance rate of approximately 7.31%.

6.2.8 Extended Band model sensitivity conclusions

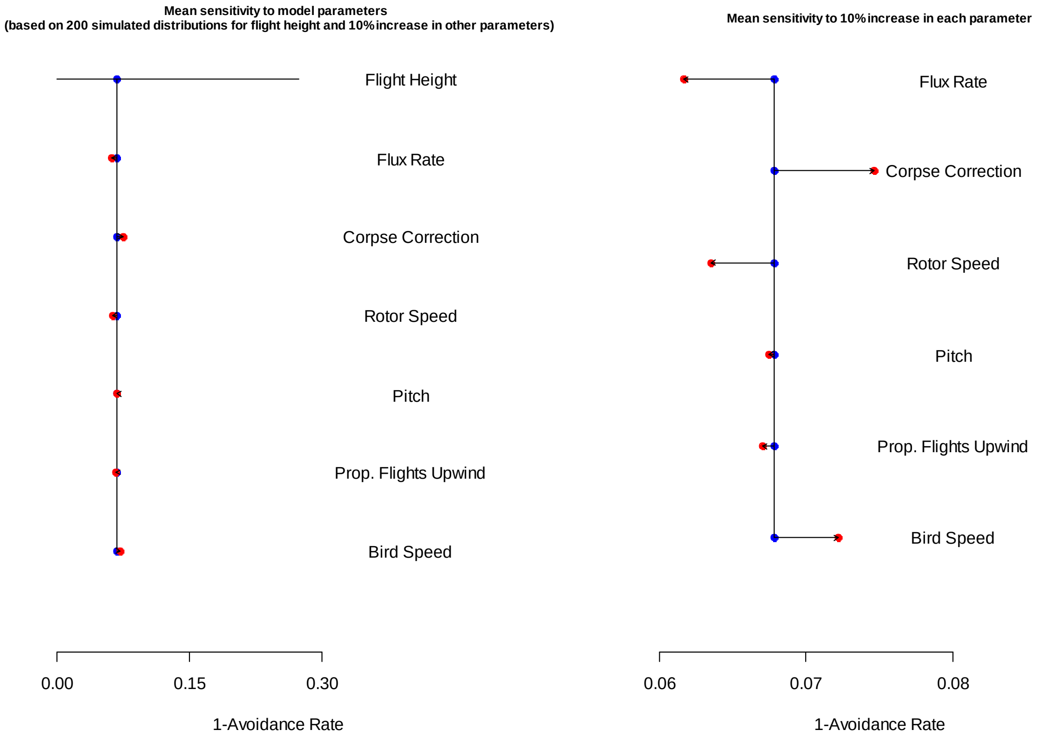


Figure 6.16 Sensitivity of non-avoidance rates derived using the extended Band model to Band model parameters. Sensitivity to flight height distribution is assessed by considering the standard deviation calculated from non-avoidance rates derived using 200 randomly simulated flight height distributions and sensitivity to the remaining parameters is derived from a 10% increase in the values presented in Appendix 7 and Table 5.4.

Of the parameters considered, the derived non-avoidance rates appear to be most sensitive to the assumed flight height distribution (Figure 6.16). However, the assessment of sensitivity for this parameter is not strictly comparable to that for the other parameters as it is not possible to make a simple assumption about a change in a continuous distribution in the same way it is about a change in, for example, rotor speed or bird numbers. Furthermore, the magnitude of the sensitivity in this parameter may be strongly influenced by 11 of the 45 data points, for which there was particularly high variation around the mean values (Figure 6.9). On closer examination, this variation appears to be strongly linked to sites with relatively low levels of bird activity (Figure 6.17).

Of the remaining parameters, the derived non-avoidance rates were most sensitive to changes in the flux rate at the windfarm (the number of birds passing through over the course of the study period) and the accuracy with which corpses were detected.

Both rotor speed and bird speed also appeared to have a moderate influence on the derived non-avoidance rates (Figure 6.18). The sensitivity of the non-avoidance rates to the input parameters appeared to be relatively consistent between option 1 and option 3.

Sensitivity to each parameter also appeared to be strongly linked to the number of birds estimated flying through each monitored windfarm (Figures 6.10 and 6.17). As the number of birds passing through a site increases, the sensitivity of the derived non-avoidance rates to each of the model parameters, including the assumed flight height distribution, drops markedly. This finding is consistent with that of Douglas *et al.* (2012) who found that the sensitivity of predicted collision rates to input parameters dropped as the quantity of observational data increased. In the case of sensitivity to the assumed flight height distributions used, at sites where flight activity is greatest, the derived avoidance rates have a similar level of variability to this and to other parameters. This is because for two sites where similar numbers of collisions are recorded, but at which the levels of bird activity differ, the non-avoidance rate will be higher at the site with the lowest level of bird activity. As a consequence, where an identical change occurs at both sites, the total change in the non-avoidance rate will be greatest at the site with the lowest level of bird activity.

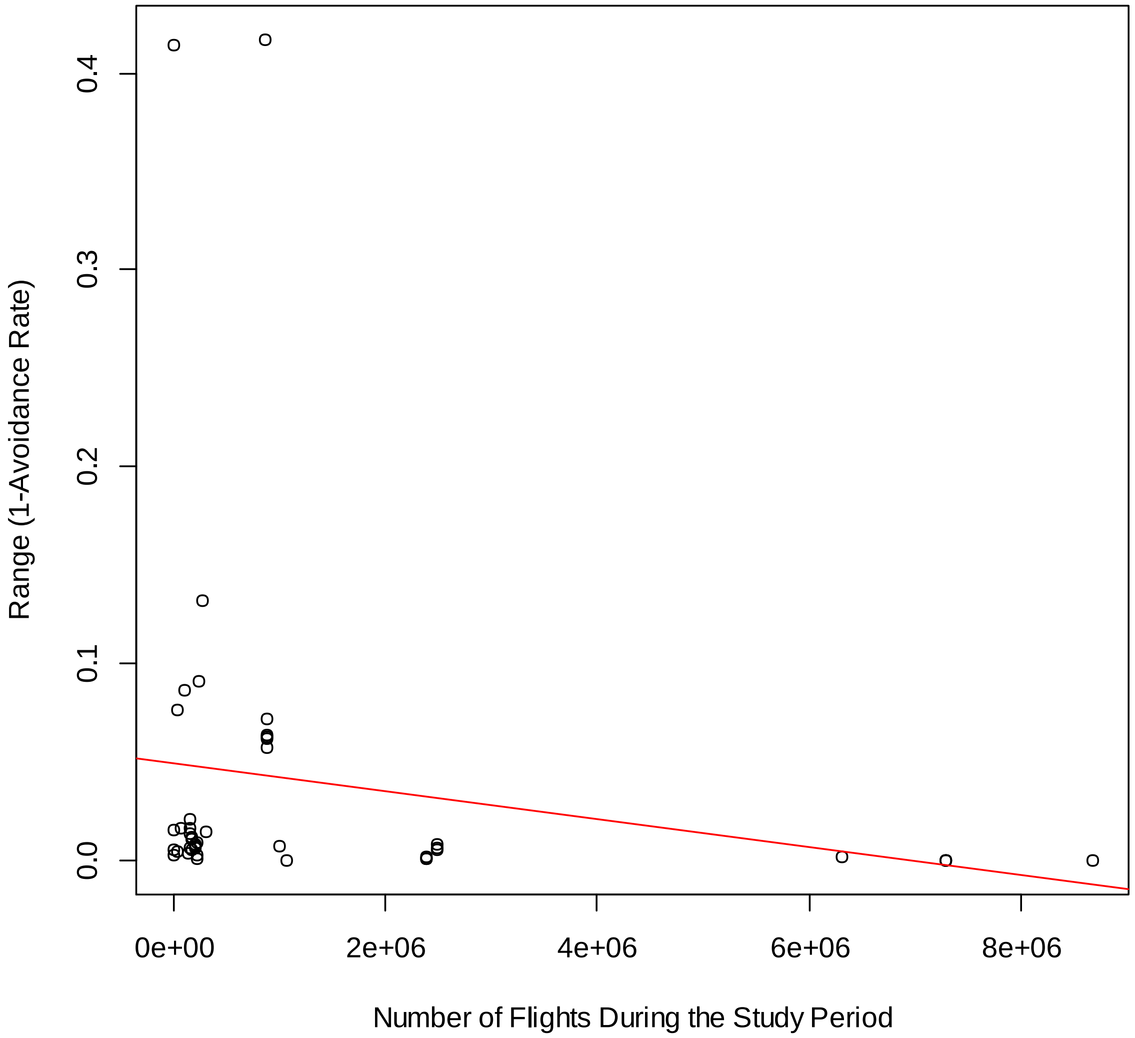


Figure 6.17 Sensitivity of the non-avoidance rate derived using option 3 of the Band model to the assumed flight height distribution.

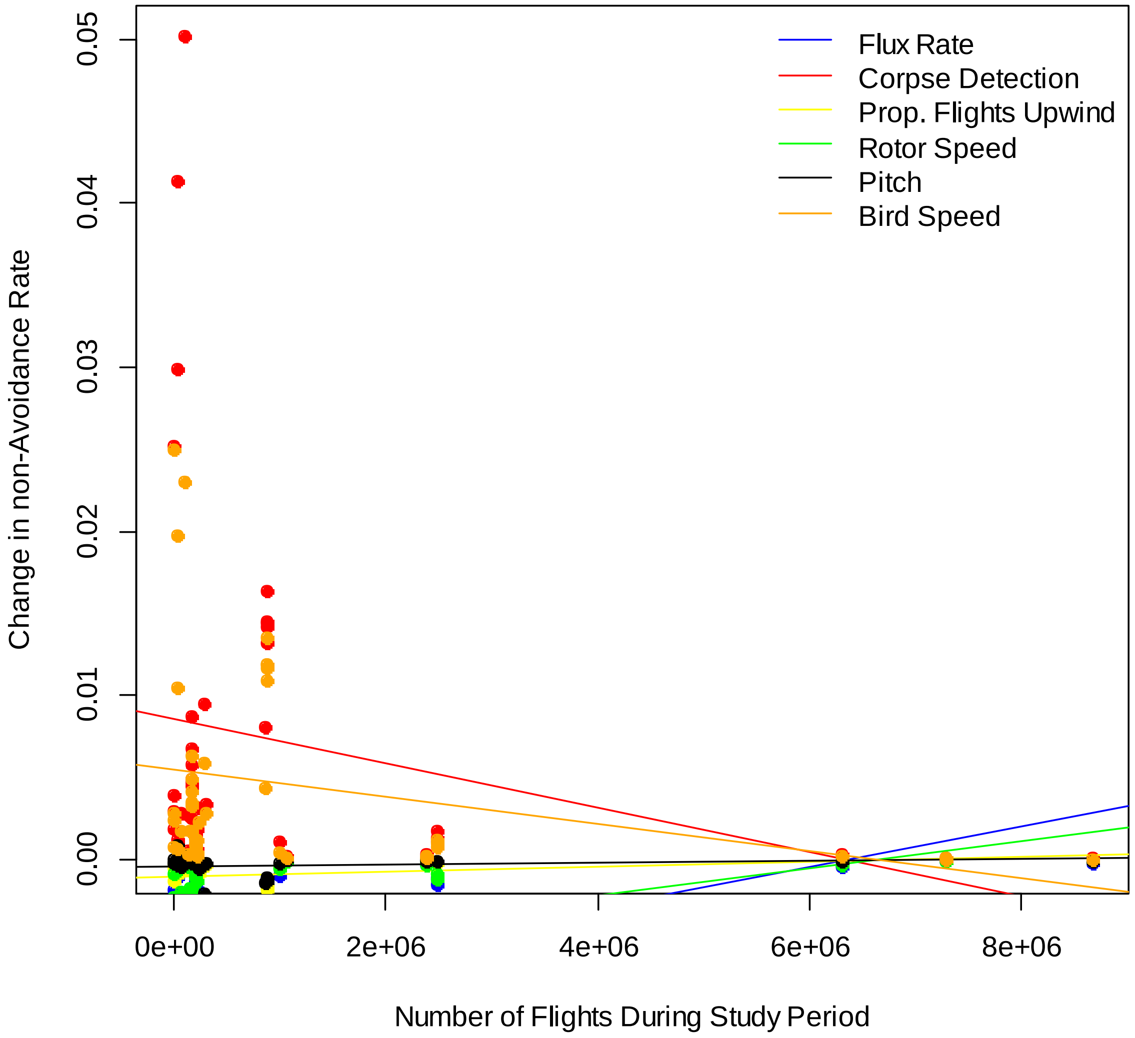


Figure 6.18 Sensitivity of non-avoidance rates derived using option 3 of the Band model to a 10% increase in each of the following parameters – flux rate, corpse detection, proportion of flights upwind, rotor speed, pitch and bird speed.

6.3 Sensitivity analysis conclusions

Avoidance rates derived using both the extended and basic Band models were sensitive to uncertainty surrounding the flux rate, corpse correction factor, rotor speed and bird speed. Whilst we considered a 10% increase in each of these parameter values to test the sensitivity of the models to the underlying assumptions, it would be valuable to consider how this compares to the actual range in each of these parameters experienced at each site. This would enable us to better quantify the uncertainty surrounding the derived avoidance rates. However, such an analysis would be complex, especially given that some parameters may co-vary, or be influenced by factors not included in the model, for example, both rotor speed and bird speed are likely to be influenced by wind speed. Such an analysis would be beyond the scope of this project and has not been considered here.

6.4 Sensitivity to other external factors

6.4.1 Weather

The flight behaviour of birds may be strongly influenced by weather conditions. However, much of the research on this subject has been carried out in relation to migration (e.g. Larkin & Thompson 1980, Gauthreaux 1991, Zehnder *et al.* 2001, Dokter *et al.* 2011). Weather is likely to influence avoidance behaviour in two ways. Firstly, by reducing visibility, making it harder to detect hazards and, therefore, increasing the risk of collision and, secondly, by affecting the manoeuvrability of birds as a result of strong winds or the presence of thermals (Spear & Ainley 1997, Shamoun-Baranes *et al.* 2006, Shamoun-Baranes & van Loon 2006).

Increases in the numbers of recorded collisions between birds and wind turbines, or other man-made objects, have been widely reported following periods of dull, overcast weather (Crawford 1981, Winkelman 1992, Bevanger 1994). This is likely to be because poor visibility reduces the ability of birds to detect turbines, and may lead to them becoming disorientated (Williams *et al.* 1974, Able 1982, Richardson 1990). As a result, the avoidance rates of individual birds are likely to be lower during periods of poor visibility. However, data used for collision risk modelling are based on the abundance of birds in flight within the windfarm, during conditions with good visibility (Camphuysen *et al.* 2004).

In contrast, there is some, limited, evidence that some bird species may be more likely to forage inland, and less likely to fly during periods of poor visibility (Williams *et al.* 1974, Pinder 1989), reducing the number of birds in flight within the windfarm in comparison to baseline survey data used in collision risk modelling. Such a potential reduction in the number of birds in flight needs to be factored into the avoidance rates used in collision risk modelling.

As a result, it is unclear as to the extent to which conditions with poor visibility may affect the avoidance rates necessary for use in offshore windfarms. To understand the potential importance of this, it is necessary to quantify the proportion of birds likely to be in flight, at sea when visibility is poor. Data collected using modern GPS tags has the potential to answer this problem and also inform on nocturnal flight activity.

Wind speed and direction both influence bird flight behaviour (e.g. Spear & Ainley 1997, Safi *et al.* 2013), with potential implications for avoidance rates. At onshore windfarms, birds have been observed to exhibit less risky flight behaviour during periods of increasing wind (Barrios & Rodriguez 2004). During periods of strong winds, Krijgsveld *et al.* (2011) noted a decrease in the number of birds in flight around Egmond aan Zee. However, as these data were collected using radar, they emphasise that these observations may reflect increased clutter from waves, rather than a decrease in the total number of birds.

Studies have demonstrated that birds make use of wind conditions to minimise the energetic cost of flight and optimise the trade-off between the maximum range they can reach and the energy they expend in reaching it (Williams *et al.* 1974, Spear & Ainley 1997, de Lucas *et al.* 2012). They achieve this in two ways. Firstly, birds fly faster into headwinds than tail or crosswinds (Tucker & Schmidt-Koenig 1971, Larkin & Thompson 1980, Wakeling & Hodgson 1992, Spear & Ainley 1997). This would lead to a decrease in the avoidance rates derived above, as the probability of a bird colliding with a turbine would be reduced, reducing the ratio of predicted to observed collisions (see sections 6.1.6 and 6.2.7). Secondly, during stronger winds, birds have a tendency to fly more slowly (Larkin & Thompson 1980, Spear & Ainley 1997). This would lead to an increase in the avoidance rates derived above, as the probability of a bird colliding with a turbine would be increased, increasing the ratio of predicted to observed collisions (see sections 6.1.6 and 6.2.7). As with the influence of visibility, the relative importance of wind direction and speed on avoidance behaviour is hard to quantify. The situation is further complicated as birds may be less likely to fly during periods of heavy wind (Stienen *et al.* 2000). Again, the growth of modern tracking technology has the potential to help address some of these issues.

6.4.2 Habitat use

The avoidance behaviour of birds in relation to an offshore windfarm may relate to how the habitat surrounding the turbines is used – for example, are turbines close to a breeding colony, are turbines situated on a commuting route, or are turbines situated on a key foraging area. Varying responses to the surrounding habitat are likely to manifest themselves in different flight modes, and these different flight modes are likely to have different levels of collision risk associated with them (Martin 2010, 2011). When foraging or searching for roost sites and conspecifics, birds can considerably reduce their detection of obstacles, and therefore increase their risk of collision, by moving their heads vertically (Martin & Shaw 2010). Collision risk at turbines surrounding colonies, as was the case for several of the sites included in our review, may therefore be inflated by birds arriving at the colony searching for their nests. Collision risk at breeding colonies may be further inflated by the display flights undertaken by males at the start of the breeding season (May *et al.* 2013) and by the presence of young birds, whose flight behaviour may place them at greater risk of collision (Henderson *et al.* 1996) at the end of the breeding season.

It is unclear whether foraging may confer a greater collision risk than searching for conspecifics on arrival at breeding colonies. It is, therefore, difficult to say with any certainty whether birds foraging within the area of offshore windfarms may be at lesser or greater risk of collision than those returning to breeding colonies and searching for conspecifics. However, when at sea, species such as northern gannets may restrict their foraging behaviour to relatively discrete areas (Hamer *et al.* 2009, Pettex *et al.* 2010). Therefore, the majority of the area covered at sea is likely to fall within the less risky category of commuting flights. As a consequence, relying on avoidance rates derived from turbines next to breeding colonies, such as those at Bouin and Zeebrugge, for birds at sea is likely to result in an overestimate of the true risk of collision. New technology, for example camera-loggers (e.g. Votier *et al.* 2013), has the potential to help gain a better understanding of collision risk at sea both by revealing more details about activity budgets, and also by allowing quantification of the proportion of flight time spent by birds looking straight ahead, and therefore at less risk of collision, as opposed to looking below.

6.4.3 Turbine Size

Initial analyses suggested that there was no strong relationship between turbine size and the avoidance rates derived for each of the species and groups we considered in our review (see section 5.3.3.2). Plots of avoidance rate against maximum turbine tip height appear to support this conclusion (Figure 6.19).

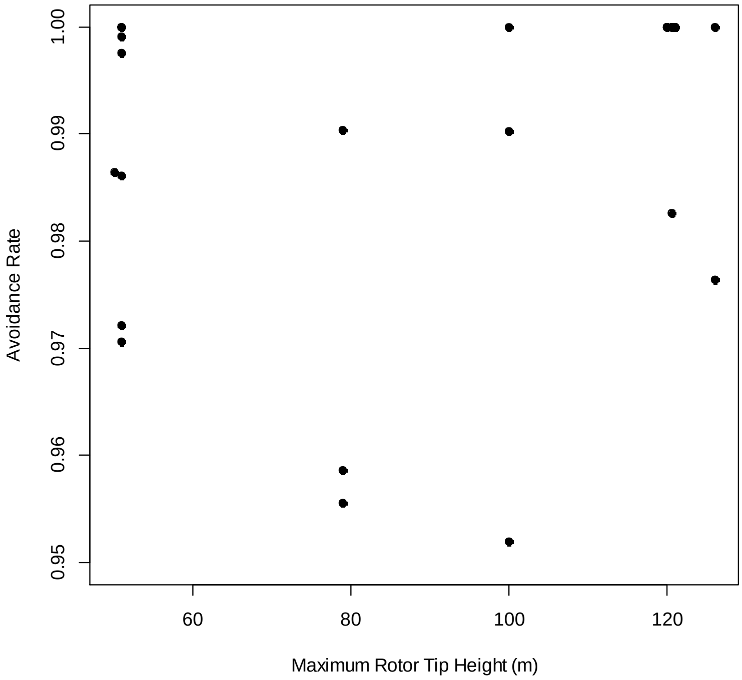


Figure 6.19 Relationship between maximum rotor tip height and the avoidance rate derived using option 1 of the Band model for all gulls.

6.4.4 Seasonality

Our analysis of the data from Zeebrugge present limited evidence that there may be a seasonal aspect to collision risk (see Section 5.1). These data suggest that avoidance rates may be higher in the autumn than in the breeding season. This may be related to two factors. Firstly the presence of younger, inexperienced birds which may have riskier flight behaviour (e.g. Henderson *et al.* 1996). Secondly, given that several of our study sites were located on the edge of breeding colonies, it may be that during the breeding season birds arriving at colonies focus on locating their nests and are therefore less likely to see turbines, increasing the collision risk.

6.4.5 Applicability of avoidance rates between species

Avoidance rates are likely to be linked to a bird’s ability to detect a turbine and perceive it as a potential threat in sufficient time to take action to avoid collision. Whilst we have able to derive a within-windfarm avoidance rate for gulls, we have been unable to come up with a suitable value for northern gannet due to lack of data.

Therefore we consider other supporting evidence to evaluate whether for northern gannet total avoidance rates are likely to be higher or lower than those for gulls.

Total avoidance rates are likely to be a combination of the probability of a bird detecting a turbine and its ability to take last-second action to avoid collision. Ability to take last-second avoidance action is likely to be linked to a species manoeuvrability and a previous review used this as the basis for recommending avoidance rates for different species (Maclean *et al.* 2009). In general, expert opinion suggests that the flight manoeuvrability of northern gannets may be less than that of gulls (Garthe and Hüppop 2004, Furness *et al.* 2013), suggesting that they need more time to react to the presence of a turbine, and may therefore need to detect it earlier. Evidence from our review suggests that a high proportion of northern gannets avoid entering windfarms (Krijgsveld *et al.* 2011, Vanermen *et al.* 2013). In addition, observations undertaken within offshore windfarms suggest that very few northern gannets pass close enough to turbines to be at risk of collision (see section 5.1).

Birds are likely to be better able to detect obstacles, such as turbines, when they are looking straight ahead, as opposed to down, towards the sea-surface (Martin 2010). At sea, it may be reasonable to assume that birds will look downwards when actively foraging, and straight ahead when migrating or commuting between their breeding colonies and foraging areas. Northern gannet typically forage using area-restricted search (ARS) behaviour (based on diving activity) resulting in a relatively small proportion of the total area covered being actively used when at sea (Hamer *et al.* 2009, Votier *et al.* 2013). These ARS zones are found solely on the outbound part of the foraging trip. In contrast, gulls are not likely to limit their foraging area to such restricted zones within foraging trips (Kubetzki and Garthe 2003, Schwemmer and Garthe 2005), and may therefore spend a greater proportion of their time at sea looking towards the sea-surface. The distance over which birds can see is strongly correlated with body size (Brooke *et al.* 1999). As a consequence, northern gannets are likely to be able to detect turbines at a greater distance than gulls. Recent evidence suggests that northern gannets may respond to the presence of fishing vessels over distances of up to 11 km (Bodey *et al.* 2014). These results suggest that, at least theoretically, northern gannets may be capable of responding to the presence of a windfarm over considerable distances.

Whilst insufficient data were available to derive within-windfarm avoidance rates for northern gannets, evidence of strong avoidance of windfarms, in contrast to gulls which appear to show no consistent response, suggests that total avoidance rates for northern gannets are unlikely to be lower than those for gulls.

6.4.6 Comparability of onshore and offshore avoidance rates

The difficulty of recording collisions in the offshore environment has meant that estimates of within-windfarm avoidance rely on data collected from terrestrial windfarms. However, birds may respond differently to onshore and offshore turbines. For example, migrating geese have been found to consistently avoid entering offshore windfarms, demonstrating macro-avoidance, (Plonckzkier & Simms 2012) but may habituate to the presence of onshore turbines (Madsen & Boertmann 2008).

Understanding how avoidance behaviour differs between onshore and offshore environments requires an understanding of how flight behaviour differs between the two. Modern GPS tracking technologies have made such comparisons easier, and it appears that whilst lesser black-backed gulls may spend a similar proportion of their time in flight in both environments (Kolios 2009), there is a tendency to fly lower when offshore (Corman & Garthe 2014, Ross-Smith *et al.* in prep.). As this would result in fewer flights at risk height in the offshore than onshore environment, this would be accompanied by decrease in both the proportion of birds at risk height (and therefore the predicted collision rate) and the actual collision rate of the same proportion. Consequently the avoidance rate would be unchanged between the onshore and offshore environments. However, there remain a number of other possible differences between onshore and offshore flight behaviour. Gulls are capable of adjusting their flight mode in response to airflow patterns which differ between onshore and offshore environments, in order to minimize their energy expenditure (Shamoun-Baranes & van Loon 2006). In the onshore environment they can take advantage of thermals by soaring and wind blowing up slopes or other major topographical features resulting in slope lift soaring. Whereas in the offshore environment a boundary layer can be created as the wind blows over the surface of the sea resulting in differential air wind speeds which some seabirds including gulls can exploit for dynamic soaring (see Alexander 2004). It is unclear how these adjustments between soaring and flapping flight may influence collision risk, though changes in manouverability and flight speed may be important. At present, there are significant gaps in our understanding of how flight behaviour may differ between onshore and offshore environments, though recent technological advances may start to fill these gaps. However, at present, the data describing within-windfarm avoidance rates collected from onshore sites remains our best available evidence.

7 TOTAL AVOIDANCE RATES FOR PRIORITY SPECIES

In this section, we consider total avoidance rates for each of the five priority species – northern gannet, black-legged kittiwake, lesser black-backed gull, herring gull and great black-backed gull.

7.1 Macro-response rates (section 5.1)

For gulls, the present evidence base is equivocal, with some studies suggesting evidence for attraction, others evidence for displacement, and others no significant response. Thus, for these species, the balance of evidence suggests a macro-response of 0 (i.e. no attraction to or avoidance of the windfarm) (Table 7.1).

Northern gannets typically show a strong macro-response to offshore windfarms. However, differences in survey methodologies make it difficult to arrive at realistic macro-response values by combining data from multiple sources. Based on currently available evidence, we believe that 0.64 to be a reasonable value for the macro-response rate (Table 7.1). However, it should be noted that this figure is based on data that are most representative of the non-breeding season.

7.2 Micro-response or meso-response rates (sections 5.2 and 5.3)

The review of existing evidence for avoidance rates in relation to offshore windfarms for the key species considered in this study indicated that insufficient data were available to generate separate micro-avoidance or meso-response rates for any of the species of interest.

7.3 Within-windfarm avoidance rates (section 5.4)

Within-windfarm avoidance rates, representing a combination of meso-responses and micro-avoidance may be derived by comparing observed collisions to those expected in the absence of avoidance (see equation 6 under section 1). Options 1 and 2 of the Band model are mathematically identical (both termed the basic Band model), with the proportion of birds at collision risk height estimated from modelled flight height distributions for option 2 and based on site-specific observational data using option 1. Therefore, it is necessary to use the same avoidance rates for both model options. As the rates derived using option 1 utilise site-specific data, rather than data derived from a generic curve (produced following the methodology of Johnston *et al.* 2013), we feel that these values are the most appropriate to recommend for use with the basic Band model. With respect to the extended Band model, the rate derived should be acknowledged as, potentially, being precautionary as, at several key sites, it is based on an underestimate of the proportion of birds flying at collision risk height (see Appendix 7). As a consequence, when calculating the avoidance rate by comparing the predicted and observed number of collisions, the resulting value is lower than would otherwise be expected. Therefore, where there is a significant difference between the observed proportion of birds at collision risk height and the proportion predicted to be at collision risk height from modelled distributions, the avoidance rates derived for use with the extended model are not considered appropriate as they will be based on an inaccurate assessment of the number of birds at risk of collision.

An alternative methodology with which to derive a within-windfarm avoidance rate for use with the extended Band model is described by in Annex 1 to this report. Following this methodology, the ratio between the number of collisions expected in the absence of avoidance derived using options 2 and 3 of the Band model is used to modify the avoidance rate derived using option 1 of the Band model. However, this requires knowledge of the flight height distribution (e.g. to 1m resolution) at the windfarm concerned – as opposed to the proportions of birds assigned to different flight height categories – in order to separate geometric avoidance (i.e. birds passing the rotor at lower altitudes where the probability of collision is lower) from behavioural avoidance. Whilst it is possible to use this methodology without knowledge of the flight height distribution at the windfarm in question, the result would be that the predicted collision rate using option 3 would be identical to that obtained using option 2. However, this methodology is likely to be of value in the future as data collection techniques improve and detailed flight height distributions are derived on a site-specific basis.

We were able to derive within-windfarm avoidance rates for herring gull and lesser black-backed gull (Table 7.1). Based on a sample of 526,048 predicted flights through windfarms, we derived an avoidance rate of 0.9959 (± 0.0006 SD) for herring gull based on the basic Band model and 0.9908 (± 0.0012 SD) using the extended Band model. For lesser black-backed gull, the derived avoidance rates were 0.9982 (± 0.0005 SD) and 0.9957 (± 0.0011 SD) respectively, based on a sample of 101,746 predicted flights through windfarms. However, the larger sample size and the fact that data originate from a greater number of sites (see Appendix 7) means that the avoidance rates derived for herring gull are more robust than those derived for lesser black-backed gull. We also derived within-windfarm avoidance rates for large gulls as a group. This group includes all birds positively identified as herring gull (this species accounting for 526,048 of the total of 639,560 flights through windfarms), lesser black-backed gull or great black-backed gull, but also those with uncertain species identification (10,638 predicted flights through windfarms), for example those identified as herring/lesser-black backed gull. For the large gulls group, we derived avoidance rates of 0.9956 (± 0.0004 SD) using the basic Band model and 0.9898 (± 0.0009 SD) using the extended Band model. A comparison of the observed and predicted proportions of birds at collision risk height (Appendix 7) shows that whilst there are some notable differences in these values, across most sites they are broadly consistent. For this reason, we feel that the avoidance rates derived using both the basic and extended Band models are appropriate to use.

We also derived within windfarm avoidance rates for small gulls (1,589,953 predicted flights through windfarms) based largely on data collected from common gull (746,668 predicted flights through windfarms) and black-headed gull (841,008 predicted flights through windfarms). For species within the small gulls group, we derived within-windfarm avoidance rates of 0.9921 (± 0.0015 SD) for use with the basic Band model and 0.9027 (± 0.0068 SD) for use with the extended Band model (Table 7.1). However, given significant differences between the proportion of birds observed and predicted to be at collision risk height at a number of key sites, we do not feel that it is appropriate to use the avoidance rate derived for use with the extended Band model for the small gulls grouping. These differences are likely to arise from the fact that the data considered here originate from the terrestrial environment, often close to breeding colonies, whilst the modelled data were collected from the offshore environment.

Finally, we calculated a within-windfarm avoidance rate for all gulls as a group (2,567,124 predicted flights through windfarms). As with the large gull and small gull groups, this incorporated data for individuals with uncertain identification (350,338 predicted flights through windfarms), for example ‘gull spp’. For all gulls, we derived an avoidance rate of 0.9893 (± 0.0007 SD) for use with the basic Band model and 0.9672 (± 0.0040 SD) for use with the extended Band model (Table 7.1). However, as with the small gulls group this includes data for which there were significant differences – due partly to the inclusion of unidentified gulls – between the observed and predicted proportions of birds at collision risk height. For this reason we do not feel that it is appropriate to use the avoidance rate derived for use with the extended Band model for the all gulls groupings.

Insufficient data were available to identify a reliable within-windfarm avoidance rate for northern gannet (Table 7.1).

It is important to note that where we report the standard deviation around the derived within windfarm avoidance rates, this relates variability between sites and not to uncertainty in the model input parameters. Estimating the contribution of the model input parameters to the uncertainty associated with the derived avoidance rates requires a more detailed understanding of the real range of values associated with each parameter than is available currently.

7.4 Total avoidance rates

Total avoidance rates are also provided in Table 7.1. Ideally, total avoidance rates should be calculated using equation 8 (section 3.1). For gulls, the balance of evidence suggests a macro-response of 0 (i.e. no consistent attraction to or avoidance of the windfarm). Consequently, the total avoidance rates for these species are equal to the within-windfarm avoidance rates.

As data describing macro-responses to the windfarm are limited, we are unable to estimate the variability around the macro-response rate. For this reason, whilst we are able to provide an estimate of variability around the within windfarm avoidance rates, we are unable to provide an estimate of variability of uncertainty around the total windfarm rates.

Table 7.1 Derived avoidance rates for priority species and current knowledge gaps based on the review of available data. Empty cells indicate a lack of robust and/or consistent data on which to base conclusions. Colour coding indicates confidence in presented values (based on sample size, representativity of data): green = highest, orange = intermediate, red = lowest (i.e. not suitable for use in CRM). Confidence in total avoidance rates reflects the lower of the confidence ratings given for macro-responses and within-windfarm avoidance rates.

Species/species groupings and sample size for within-windfarm avoidance rate given in parentheses*	**Macro-response[1]**	**Meso-response[2]**	**Micro-avoidance[3]**	**Within-windfarm avoidance basic Band model[4]**	**Within-windfarm avoidance extended Band model[4]**	**Total avoidance basic Band model (1-total avoidance)**	**Total avoidance extended Band model (1-total avoidance)**	**Caveats**
Black-legged kittiwake (0)	None							Whilst data were available for macro-response, no clear patterns were evident across studies. No data available for within-windfarm avoidance.
Lesser black-backed gull (101,746)	None			0.9982 (± 0.0005)	0.9957 (± 0.0011)	0.9982 (0.0018)	0.9957 (0.0043)	Whilst data were available for macro-response, no clear patterns were evident across studies. Within-windfarm rate based on data from only two sites.

Herring gull (526,048)	None			0.9959 (± 0.0006)	0.9908 (± 0.0012)	0.9959 (0.0041)	0.9908 (0.0092)	Whilst data were available for macro-response, no clear patterns were evident across studies. Within-windfarm rate based on a large sample size from seven different sites.
Great black-backed gull (1,128)	None							Whilst data were available for macro-response, no clear patterns were evident across studies. No within-windfarm avoidance rates estimated due to extremely small sample size.

Small gull spp (1,589,953) Comprising: black-headed gull (746,668), common gull (841,008), common/black-headed gull (2,090), little gull (188)	None			0.9921 (± 0.0015)	0.9027 (± 0.0068)	0.9921 (0.0079)	0.9027 (0.0973)	Whilst data were available for macro-response, no clear patterns were evident across studies. Within-windfarm avoidance rates based on large sample size from eight different sites. However, differences between observed and predicted proportions of birds at collision risk height mean it is not appropriate to use value derived for extended model.
Large gull spp (639,560) Comprising: lesser black-backed gull 101,746, herring gull 526,048, herring/Caspian gull 1,417, herring/lesser black-backed gull 8,345, herring/yellow-legged gull 876, great black-backed gull 1,128	None			0.9956 (± 0.0004)	0.9898 (± 0.0009)	0.9956 (0.0044)	0.9898 (0.0102)	Whilst data were available for macro-response, no clear patterns were evident across studies. Within-windfarm avoidance rates based on large sample size from seven different sites.

Gull spp (2,567,124) Comprising: black-headed gull 746,668, common gull 841,008, common/black-headed gull 2,090, little gull 188, lesser black-backed gull 101,746, herring gull 526,048, herring/Caspian gull 1,417, herring/lesser black-backed gull 8,345, herring/yellow-legged gull 876, great black-backed gull 1,128, gull spp. 337,610	None			0.9893 (± 0.0008)	0.9672 (± 0.0018)	0.9893 (0.0107)	0.9672 (0.0328)	Whilst data were available for macro-response, no clear patterns were evident across studies. Within-windfarm avoidance rates based on large sample size from nine different sites. However, differences between observed and predicted proportions of birds at collision risk height mean it is not appropriate to use value derived for extended model.

Northern gannet (0)	0.64							Macro-response rates for northern gannet indicated strong avoidance of windfarms. As data were available from a limited number of sites, the lowest reported value was felt to be most appropriate as a precautionary figure. Note the majority of data comes from the non-breeding season and it is unclear how applicable these findings may be to the breeding season. No data available for within-windfarm avoidance.

[1] See section 5.4; [2] See section 5.1; [3] See section 5.2; [4] see section 5.3.

7.5 Recommended avoidance rates

Please note that these recommendations apply to the five priority species only – northern gannet, black-legged kittiwake, lesser black-backed gull, herring gull and great black-backed gull – they are not intended to be applied to seabirds more generally.

Whilst we have estimated within-windfarm avoidance rates to four decimal places, current guidance from SNH is that expressing avoidance rates to more than three decimal places is unwarranted (SNH 2013). Given the inherent uncertainty in the data we feel that this is a sensible approach to apply to total avoidance rates. For this reason, we round within-windfarm avoidance rates down to three decimal places when deriving recommended total avoidance rates.

- A **macro-response rate of 0.64** is recommended for **northern gannet** (section 5.4). However, no data were available to derive a within-windfarm avoidance rate for this species (section 5.3). Given that there is consistent evidence for high macro-avoidance, and considering the at-sea ecology of northern gannet and gulls (section 6.3.5), we feel that there is no reason to suppose that the total avoidance rates for **northern gannet** should be less than those for all gulls (as opposed to large gulls). A **total avoidance rate** of **0.989** is thus recommended for use with the basic Band (2012) collision risk model. This would reflect a within windfarm avoidance rate of 0.9703. We acknowledge that this is precautionary, but in the absence of more species-specific data, we feel it is appropriate. However, given the evidence available at present, we are unable to recommend an avoidance rate for use with the extended Band (2012) collision risk model.

- No consistent evidence of macro-avoidance was found for **black-legged kittiwake** (section 5.4). It was not possible to derive species-specific within-windfarm avoidance rates for **black-legged kittiwake** (section 5.3**).** However, as black-legged kittiwake have similar wing morphologies (wingspan, wing:body aspect ratio, wing area: Robinson 2005, Alerstam *et al.* 2007), flight speeds (Alerstam *et al*. 2007) and flight altitudes (Cook *et al.* 2012, Johnston *et al.* 2014b) to black-headed and common gulls, which contribute the majority of records for the small gulls group, the within-windfarm avoidance rates derived for the small gulls group were considered appropriate for this species. A **total avoidance rate** of **0.992** is thus recommended for the basic Band model. However, given the evidence available at present, we are unable to recommend an avoidance rate for use with the extended Band (2012) collision risk model (section 5.3).

- No consistent evidence of macro-avoidance was found for **lesser black-backed gull** (section 5.4). Whilst it was possible to derive species-specific within-windfarm avoidance rates for lesser black-backed gull, these were based on limited data and thus the within-windfarm avoidance rates for large gulls were considered more appropriate for use for this species (section 5.3). A **total avoidance rate** of **0.995** is thus recommended for use with the basic Band model and a **total avoidance rate** of **0.989** for use with the extended Band model.

- No consistent evidence of macro-avoidance was found for **herring gull** (section 5.4) and thus total avoidance rates reflect species-specific within-windfarm avoidance rates. A species-specific **total avoidance rate** of **0.995** is thus recommended for use with the basic Band model and a **total avoidance rate** of **0.990** for use with the extended Band model (section 5.3).

- No consistent evidence of macro-avoidance for **great black-backed gull** (section 5.4). It was not possible to derive species-specific within-windfarm avoidance rates for **great black-backed gull.** Given the taxonomic similarity between species within the large gulls group, the avoidance rates derived for use with this group were considered to be appropriate for great black-backed gull (section 5.3). A **total avoidance rate** of **0.995** is thus recommended for the basic Band model and a **total avoidance rate** of **0.989** for use with the extended Band model.

At present, the evidence available does not enable us to recommend a robust avoidance rate for northern gannet or black-legged kittiwake for use with Band model option 3. This does not imply that option 3 is not suitable for these species, and given the programmes of work currently underway in the offshore environment, it is envisaged that an appropriate rate will be derived in the near future. Note, while it is not possible to recommend a robust avoidance rate for use for these species at this time, this does not preclude presenting a no-avoidance collision estimate using option 3 alongside collision estimates derived using option 1 and/or option 2 (with or without using the avoidance rates recommended here) to inform on likely collision risk.

Table 7.2 Recommended total avoidance rates for use in the basic and extended Band models with each of the five priority species.

Species (rate used)	**Basic Band model avoidance rate**	**Extended Band model avoidance rate**
Northern gannet (all gull avoidance rate)	0.989	Not available
Black-legged kittiwake (small gull avoidance rate)	0.992	Not available
Lesser black-backed gull (large gull avoidance rate)	0.995	0.989
Herring gull (species-specific avoidance rate)	0.995	0.990
Great black-backed gull (large gull avoidance rate)	0.995	0.989

8 TRANSFERABILITY OF AVOIDANCE RATES BETWEEN MODELS

There are various collision risk models currently available within the scientific literature to estimate likely collision and mortality of birds due to windfarms (Band 2012; Desholm 2006; Eichorn *et al.* 2012; McAdam 2005; Smales *et al.* 2013; Tucker 1996; Holstrom 2011). The models vary in numerous ways including whether static components such as the tower are included in calculations, if oblique angles of attack are considered and whether single or multiple turbines are assessed, as well as how avoidance behaviour is incorporated. Although the Band model (Band 2012) is the most widely used collision risk model in the UK, it is not the only one available and therefore any developments in our understanding of avoidance behaviour should consider, where possible, these alternative models.

Although described in the literature, avian collision risk models are often not presented in enough detail to reproduce. The majority of models consider avoidance behaviour as an add-on to the process of estimating the probability of collision, separate from the calculation of collision probability for a single rotor transit. From the information available, however, it would seem that the definitions and avoidance rates presented in our report would generally be suitable for use within a range of collision risk models, not only Band (2012). Here we provide examples of how the definitions and rates may align with some of these alternative models.

Desholm (2006) developed a stochastic model analysis of avian collision which included variability in the input parameters and outputs of the model. Although it was a very specific example from an offshore windfarm in the Baltic Sea, the method could be used elsewhere. The definitions used in our project seem suitable for the model. The method considered the different stages at which birds may avoid a windfarm and uses values for the proportion of birds entering the windfarm (1 - macro-avoidance), the proportion within the horizontal/vertical reach of rotor blades (1 - meso-avoidance) and also the proportion trying to cross the area swept by the rotor blades without showing avoidance (1 - micro-avoidance).

Eichorn *et al.* (2012) developed an agent-based, spatially-explicit model of red kite foraging behaviour to assess collision risk related to wind turbines. The model is largely stochastic and combines a spatial model with a collision risk model. Although the study was specific to red kite, the methods could be used more widely. The model uses the method from Band (2007) for calculating probability of collision from a single rotor transit therefore it is likely that any definitions for avoidance behaviour provided by our study will be suitable. The model specifically includes the probability of a bird recognising the threat and actively avoiding, and this avoidance rate is taken from the literature. The value ranges from 0.98 – 0.995 and is therefore likely to be a value for overall avoidance, however the definitions within this study for meso- and micro-avoidance would seem to fit more appropriately because it is a single bird avoiding a single turbine within a 100 m x 100 m grid cell.

Smales *et al.* (2013) describe a collision risk model developed by Biosis Propriety Limited which has been widely used to assess wind energy developments in Australia since 2002. The model uses a deterministic approach and provides a predicted number of collisions between turbines and a local or migrating population of birds. The model uses flight activity data from the windfarm site and applies avoidance rate to the typical number of turbines encountered per flight. Therefore the definitions and rates for within windfarm behaviour should be suitable in this context.

A note of caution when considering avoidance rates and their application within different collision risk models is that although not the intended purpose, avoidance rate may have become a sink for multiple sources of error and uncertainty within a model. Collision risk models rarely state the associated error along with collision estimates. In the process of apportioning overall avoidance into the different components of macro-, meso-, and micro-avoidance, this previous inclusion of model error may need to be considered, and may be model-specific.

9 CONCLUSIONS

We have derived within-windfarm avoidance rates for a variety of species for specific sites. In some cases, these differ from those presented elsewhere using, apparently, the same data (see Natural England/JNCC note). For this reason, we include an appendix (Appendix 7) detailing how each of our values has been derived. Note that the values in Appendix 7 are supplied for illustrative purposes only and that we would recommend the use of the total avoidance rates presented in Table 7.2. Given the variability in the values that have been presented for some datasets, we believe that this level of transparency is crucial to enable readers to come to an informed opinion as to what represents a robust avoidance rate. The derivation of the flux rate through the windfarm is particularly important, as it can have quite a strong influence on the predicted number of collisions, and therefore, the final avoidance rate.

Very little data were available describing separate meso-responses or micro-avoidance. There were limitations in the data from each of the studies we identified. However, observations of flight behaviour around individual turbines indicate that birds very rarely pass close to the rotor blades, suggesting that a significant proportion of avoidance behaviour is likely to occur at a meso-scale. We identified evidence from several sites to suggest that avoidance behaviour may be influenced by both the layout of the windfarm (e.g. the inter-turbine spacing) and the operational status of turbines. There is some limited evidence to suggest that overall avoidance rates may be lower during the breeding season than the non-breeding season, although significantly more data are required to confirm this hypothesis (see section 5.3.3.1).

The availability of suitable data has been a key problem throughout this review. In part, this relates to the difficulty in collecting collision data at sea, leading to gaps in data for key species such as northern gannet and black-legged kittiwake. It is to be hoped that the ongoing ORJIP work will help to address this issue. However, it also relates to the way in which data have been collected as part of post-construction monitoring at offshore windfarms. We identified extremely limited evidence for macro-response rates for our priority species. In many instances, this may be because when impacts which may contribute to macro-avoidance, such as displacement or barrier effects, are considered, the focal species are usually auks, divers and sea-ducks. As a consequence, the impacts on other species, such as northern gannet are less well understood.

Our review highlights that there are still significant data gaps in relation to avoidance rates for marine birds and offshore windfarms, particularly in relation to micro- and meso-responses, as opposed to the correction factors often used as avoidance rates at present. Despite this, we feel that our review represents a significant step forward. We are able to recommend for the first time within-windfarm avoidance rates for gulls using both the basic Band (2012) model (options 1 and 2) and extended Band (2012) model (option 3) based on significantly more data than has been used to make recommendations for geese and raptors in the past (e.g. Pendlebury 2006, Whitfield 2009). Significant data gaps still remain for within-windfarm avoidance behaviour in the northern gannet.

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APPENDIX 1 Evidence review macro-response – barrier effect studies

A1.1 Egmond aan Zee

Location / habitat

Marine, 10-18km offshore

Turbine / array specification

Turbine array consists of 36 Vestas V90 3 MW turbines covering an area of 27 km^2. Distances between turbines are 650 m within rows and 1000 m between rows. Turbine specifications given as hub height 70 m; rotor diameter 90 m; rotor altitude min 25 m (above mean sea level) and max rotor altitude 115 m (above mean sea level).

Case study number 1

Krijgsveld, K.L., Fijn, R.C., Japink, M., van Horssen, P.W., Heunks, C., Collier, M., Poot, M.J.M., Beuker, D. and Dirksen, S. 2011. *Effect studies offshore windfarm Egmond aan Zee: Final report on fluxes, flight altitudes and behaviour of flying birds*. Bureau Waardenburg Report No. 10 - 219.

Lindeboom, H.J., Kouwenhoven, H.J., Bergman, M.J.N., Bouma, S., Brasseur, S., Daan, R., Fijn, R.C., de Haan, D., Dirksen, S., van Hal, R., Hille Ris Lambers, R., ter Hofstede, R., Krijgsveld, K.L., Leopold, M. & Scheidat, M. 2011. Short-term ecological effects of an offshore windfarm in the Dutch coastal zone; a compilation. *Environmental Research Letters* **6,** doi:10.1088/1748-9326/6/3/035101.

Methods

Krijgsveld *et al.* (2011) focussed on the disturbance of flight paths otherwise referred to as barrier effects. Whereas what was termed as the disturbance of locally resting and/or feeding birds were covered by another project (Leopold *et al.* 2011) as birds recorded on the water. Lindeboom *et al.* (2011) reported the impacts of the windfarm on a range of taxonomic groups but with respect to birds focussed on barrier effects, displacement effects and attraction. As the results presented in Lindeboom *et al.* (2011) were based on the preliminary results of Krijgsveld *et al.* (2011), cited as Krijgsveld *et al.* (2010), this paper is not considered further here.

Data collection was carried out during the post-construction period only.

Radar: Horizontal radar was used to record flight paths, with the radar located on a meteorological mast 500 m from the nearest turbine at the south western side of the windfarm). The radar was set to scan up to distances of 5.6 km from the meteorological mast (although it was calculated that gulls could be detected up to shorter distances of 4.5 km). There was no coverage from the angles of 155° to 220° relative to the mast however).

The radar signal was processed and recorded by Merlin (DeTect Inc). Flight paths of birds or groups of birds were visualised in QuantumGIS and grid cells (750 m x 750 m) were set up in order to analyse both the numbers of tracks and flight directions. In order to mitigate for reduced detection of tracks, due to the presence of turbines and decreasing detection rates with increasing distance from the radar, correction factors were applied to the numbers of tracks recorded inside the windfarm.

Visual and auditory observations: Panorama scans from the meteorological mast consisting of hourly 360° scans to record all birds flying within sight of the observation platform. This information was then used to calibrate the radar counts and provided information on species composition, density, flight altitude and flight direction. Additional information was collected at night and included moon watches, call registration by ear, and call registration by an automated bird call recording system. In addition, the opportunistic recording of flight paths of individual birds or bird groups (picked up either visually using a binoculars or a telescope) or on the radar) was carried out.

Study period

Radar: Continuous recording through the period of April/May 2007 to 31 May 2010. Flight path data was obtained for 817 days (out of a possible 918 due to factors such as high winds).

Visual observations: A total of 405 panoramic scans were carried out over 53 days (dawn to dusk) spread throughout the period of Feb 2006 to Dec 2009 and six nights (dusk to dawn) during spring and autumn migration (October 2008 to April 2009). Opportunistic observations of flight paths were carried between and during panoramic scans (n = 666 flight paths of 85 species were recorded with great cormorant (n = 82) and northern gannet (81) being the most commonly observed).

Species

Local seabirds (gull spp, northern gannet, scoter spp, and auks spp); migrating seabirds (diver spp and scoter spp) and migrating non-marine birds (thrushes and geese spp).

Conditions data collected under

Radar: all conditions.

Visual observations: recording carried out in generally dry, relatively calm conditions (all but day had one Beaufort scale of less than 5) and with a range of visibility conditions (0 - 50 km).

Results

Macro-responses (which were regarded by this study as being due to barrier effects), referred to in the report as macro-avoidance rates, were quantified by two methods[2]:

i. Panoramic scans were used to derive the proportion of birds within, at the edge and outside the windfarm. Using the combined values of the first two groupings, it was possible calculate the % of birds that passed through the windfarm[3]. The resulting values were corrected for relative surface area for within and outside the windfarm and then used to derive macro-avoidance rates[4] for northern gannet = 0.64 (n = 282 birds[5]), sea ducks/scoters spp = 0.71 (n = 123 birds), diver spp = 0.68[6] and alcid spp = 0.68[3]. Sample sizes were too small for other species/species groups for values to be derived and, hence, values have to be derived by other means;

ii. Flight path data collected by radar showed that the number of all birds that flew within the windfarm was on average 72% of the numbers outside the windfarm. This was proposed to equate to an average macro-avoidance rate of 0.28 of birds in relation to the windfarm, and when broken down according to time of year, the values ranged from an average of 0.18 (in winter) and 0.34 (in autumn)[7]. For gull spp and great cormorant, the average avoidance rate in winter of 0.18 was used, as the species composition was heavily dominated by those birds at that time of year (as shown by the visual observations). The overall average avoidance value of 0.28 was assumed for grebe spp, tubenoses spp, skua spp, and tern spp (in the absence of other available data or rationale). It was also shown using radar that the percentage of birds flying in the windfarm was significantly higher during the day compared to night (when data from spring was excluded) and these differences were greatest during summer and winter. Hence avoidance was argued to be higher at night.

Results of the opportunistic recording of flight paths indicated deflection rates of 89% for northern gannet and 40% for gulls spp based on sample sizes of 38 and 78 birds respectively[8]. These values were not considered by the authors to provide evidence for macro avoidance (Karen Krijgsveld *pers. comm.*) however.

There was inherent variation in flight direction as recorded by radar with higher variability recorded winter and summer (probably due to the inclusion of locally foraging birds which are less likely to have a consistent flight trajectory than birds migrating through the area) and during the day. Nevertheless, adjustment of flight paths occurred at 750 - 1,500 m from the windfarm when there was a pronounced

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[2] Table 15.1- Krijgsveld *et al.* 2011.

[3] Table 9.3 - Krijgsveld *et al.* 2011.

[4] Macro-avoidance = 100-((x/50)*100). Where x = % of birds that passed through the windfarm and 50 is the correction factor for surface area. Karen Krijgsveld *pers. comm.* Values of x for northern gannet and common scoter were 18 and 14 respectively (sum of the relative abundance inside and at the edge of the windfarm – see Table 9.3).

[5] Taken from Figure 9.25.

[6] Based on the average of northern gannet (0.64) and scoter spp (0.71) which was justified on the grounds of their avoidance behaviour being similar (based on their flight paths).

[7] Figure 9.15 - Krijgsveld *et al.* 2011.

[8] Table 9.6 - Krijgsveld *et al.* 2011

change in flight direction. This was largely based on plots of the mean ± standard errors of flight direction in relation to distance according to season and time of day[9]. The reported changes at 750-1500 m appear to occur before and after midnight in the spring and at dusk during autumn. There were also changes in flight direction at distances further away from the windfarm but these are not highlighted – notably in spring, for most times of day, at distances between 4,500 and 5,250 m.

Numbers of birds were also shown to be highest at 750 - 1500 m, which was taken as evidence of flying birds building up as they were deflected away from the windfarm (also confirmed by visual observations of birds). Moreover, the number of tracks for all seasons in the grid cells circa 750 m from the windfarm was also shown to be significantly higher than the number of tracks for the grids cells containing the adjacent single row of turbines[10].

Assessment of methodology

The values of macro-avoidance derived from the panoramic scans were species specific and were collected in a systematic manner. As for all visual observations, data collection was mostly restricted to days of reasonable visibility and calm conditions.

Macro-avoidance rates (barrier effects) derived using radar were based on mean values across all species and should be interpreted very carefully since there is likely to be variability in response rates between species. Hence this should be borne in mind when citing values derived for gull spp, grebe spp, tubenoses spp, skua spp, and tern spp. It is also unclear whether the actual numbers reported will consist of solely individual birds or whether flocks of birds may have been inadvertently included. Hence as for most radar studies, the avoidance rates cannot be necessarily assumed to correspond to those of individual birds. It is also worth bearing in mind, that the way these data have been collected (comparison of number of flight paths inside and outside the windfarm) could also be potentially considered to be evidence of displacement.

It is also problematic to overlay the arbitrarily selected boundary of 500 m buffer surrounding the outermost turbines used to delineate inside (micro and meso) and outside (macro) the windfarm avoidance (section 3.5) with the grid cell system of 750 km^2 used to analyse the number of tracks.

The grid cell system also does not correspond exactly to the boundaries of the windfarm and hence some cells will overlay areas inside and outside the windfarm which could be an issue for the values cited for % of tracks inside and outside the tracks.

A1.2 Horns Rev

Location / habitat

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[9] Figure 9.28- Krijgsveld *et al.* 2011.

[10] Generalised Linear Model ($t_{2228} = 3.4$, $p < 0.001$) - Krijgsveld *et al.* 2011.

Horns Rev 1: Marine, 14 km offshore.
Horns Rev 2: Marine, 30 km offshore.

Turbine /array specification

Horns Rev 1: Turbine array consists of 80 2.0 MW Vesta turbines. Distance between turbines – north to south (560 m) and east to west (560 m). Turbine specifications given as: hub height 70 m; rotor blade length 40 m (diameter 80 m); and total height 110 m. Height of the lowest tip of rotor blade.

Horns Rev 2: Turbine array consists of 91 turbines. Distance between turbines – north to south (560 m) and east to west (560 m). Turbine specifications given as: hub height 68 m; rotor diameter 93 m; and total height 114.5 m. Height of the lowest tip of rotor blade 21.5 m.

Case study number 1

Petersen, I.K., Christensen, T.K., Kahlert, J., Desholm, M. & Fox, A.D. 2006. *Final results of bird studies at the offshore windfarms at Nysted and Horns Rev, Denmark*. Report commissioned by DONG Energy and Vattenfall A/S. National Environmental Research Institute.

Methods

This report focussed on barrier effects, displacement effects, physical changes to the habitat and collision risk. Work was carried out at the Horns Rev 1 and Nysted offshore windfarms but there were differences in methodology and timing of data collection in relation to the development phase – data collection was carried out during the post-construction period only at Horns Rev 1.

Radar observations*:* Recordings by radar occurred in a circular area of radius ca. 11 km (no coverage in the north east quadrant with the exception of late November 2005). The radar was located on a transformer station located less than 0.6 km from the windfarm. Migration mapped by tracing course of flocks onto a transparency and subsequently digitised. As fewer tracks were recorded both within and beyond the windfarm, due to presence of the turbines and the increasing distance from the radar, densities of tracks were not used to quantitatively to look at barrier effects.

All tracks (n = 468 north of the windfarm and n = 342 east of the windfarm) which were deemed to have a theoretical chance of entering the windfarm were selected using the criteria that they were orientated towards the windfarm at distances between 1.5 and 2 km from the windfarm and had lengths of tracks greater than 2 km.

In order to look at the lateral (horizontal) change in migration route in response (where avoidance occurs) to the windfarm, two sets of transects lines were set up. The first were located east of the windfarm running parallel to the direction of the rows of turbines (from north to south) and were set up at intervals of 50, 100, 150, 200, 250, 300, 400, 500, 1000, 2000, 2500, 3000, 3500 and 4000 m (max. range set by limits of the radar). The second were set up north of the windfarm at 50, 100, 200, 300, 400, 500, 1000, 1500, 2000, 2,500, 3000 m and then at intervals of 1000 m until 7000 m. The orientation of all bird tracks that intersected two adjacent transects were calculated for all of the transects running east and north of the windfarm.

Visual observations: four transects from the transformer station set up, one of which passed diagonally through the windfarm.

Study period

Radar observations: A total of 17 survey periods (shortest = 5 h 30 min, longest = 39 h 30 min) were carried out covering the periods of August to November 2003; March to May 2004; August to September 2004; March to May 2005; and August to November 2005. Total of 243 h 45 min of observations.

Visual observations: 19 surveys (shortest = 7 h 0 min, longest = 29 h 30 min) were carried out covering the periods of April to May 2003; August- November 2003; March to May2004; August to September 2004; March to May 2005; and August to November 2005. Total of 403 h 18 min of observations.

Species

Staging and migrating birds. Based on visual observations of birds during transect counts, likely to consist primarily of diving ducks (by an order of magnitude higher than any other group and consisting almost exclusively of common scoter), gulls (herring gull, little gull, greater back-backed gull and black-legged kittiwake and terns (Sandwich tern and common/Arctic tern)[11].

Conditions data collected under

During day and night, weather conditions not presented.

Results

The annual percentage of bird tracks (based on the years 2003, 2004 and 2005) entering the windfarms from either the northern or the eastern side of the windfarm ranged from 13.6 % (2005, north of windfarm) and 29.3% (2004, east of the windfarm[12]). The number of tracks that these percentages are based upon are relatively small however (ranging from 12 to 39 tracks). These values appear to provide the origins of cited macro-avoidance rates of 0.71 and 0.86. Spring and autumn periods were not differentiated between as it was argued that bird behaviour would be similar regardless of the time of season.

The mean orientation of tracks of migrating birds, as calculated for all intervals between transects, was used as the response variable to look at the lateral deflection of south bound tracks for birds north (n = 2108) and east of the windfarm

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[11] Table 48 - Peterson *et al.* 2006.

[12] Table 55 - Peterson *et al.* 2006.

(n = 1168). For birds north of the windfarm during southbound bird migration, analyses[13] showed that distance to windfarm, wind direction (crosswinds), time of day and the interaction between distance and time of day were significant. Plots of the mean flight orientation with distance to windfarm in relation to time of day wind direction showed that deflections were most pronounced at distances of less than 400 m from the windfarm and that changes at larger distances (<2 km) were more obvious during the daytime compared to the night time period[14]. For birds east of the windfarm analyses[15] found that distance had a significant effect on the orientation of the birds (wind direction, time of day and the interaction between distance to windfarm and wind direction were also significant. Plots of the mean flight orientation with distance to windfarm in relation to time of day wind direction showed that deflections were most pronounced at distances of less than 500 m from the windfarm. Changes in orientation occurred up to 4 km from the windfarm during south bound migrations notably during the day in westerly winds[16].

Assessment of methodology

The derived macro-avoidance rates (based on barrier effects) are a mean value for all birds which occurred during the study and according to visual observations consisted mainly of common scoter. Therefore, these reported avoidance rates may have limited applicability to the less commonly recorded gulls spp and tern spp. In addition these avoidance rates are based on relatively small sample of tracks. Moreover, tracks do not differentiate between individuals or flocks, therefore the reported macro-avoidance rates do not respond to the level of individual birds.

Case study number 2

Blew, J., Hoffman, M., Nehls, G. & Hennig, V. 2008. *Investigations of the bird collision risk and the responses of harbour porpoises in the offshore windfarms Horns Rev, North Sea, and Nysted, Baltic Sea, in Denmark. Part 1: Birds*. Report from the University of Hamburg and BioConsult SH, 145pp.

Methods

The report focussed on the collision risk to migrating birds at Horns Rev 1 and Nysted offshore windfarms and the same methodology was used at both sites.

Blew *et al.* (2008) proposed that avoidance occurred at the three broad scales of: (1) large scale avoidance >2000 m; (2) medium to small scale avoidance 1000 m to 150 m and either horizontally or vertically as measured directly (reactions) or indirectly (comparison of numbers or flight altitudes); (3) last second avoidance. Thus, the

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[13] ANOVA analyses: distance F_{14}=18.93, $p < 0.0001$; wind direction F_1=57.49, $p < 0.0001$; time of day F_1=95.33, $p < 0.0001$; and distance*time of day F_{14} = 3.27 , $p < 0.0001$)- Peterson *et al.* 2006.

[14] Figure 170 - Peterson *et al.* 2006.

[15] ANOVA analyses: distance F_{14}=25.38, $p < 0.0001$; wind direction F_1=13.37, $p = 0.0003$; time of day F_1=132.67, $p < 0.0001$; and distance*wind direction F_{14} = 2.79 , $p = 0.0004$) - Peterson *et al.* 2006.

[16] Figure 172 - Peterson *et al.* 2006.

second category, which was the focus of this report, overlaps with the definitions in section 3 of this report of both macro- and meso-avoidance.

Data collection was carried out during the post-construction period only.

Radar observations*:* Horizontal radar (Bridgemaster E-series and Pathfinder) was deployed from ships with a range of anchoring sites (three, four and four at the eastern, southern and western edges of the windfarm respectively) at distances of 150 to 300 m to the windfarm. Screenshots were captured using a digital camera for the horizontal radar and the angle of tracks and their length were also registered. The range of the radar was set to 1.5 nautical miles. No manual tracking of signals on the horizontal radar was carried out which meant that changes in flight trajectories for individual tracks could not be looked at.

Radar tracks were categorised according to their direction in relation to the first row of the windfarm; flying towards (± 45° either side of perpendicular to the windfarm; flying away; and flying parallel (more or less).

In order to look at lateral avoidance, four intervals ranging from 0-500 m, 500-1,000 m, 1,000-1,500 m and 1,500-2,000 m in relation to the ship and the relative orientation of tracks were recorded in the range of ± 90° with 0° being perpendicular to the windfarm. Due to sample size issues (insufficient number of tracks), it was not possible to report results for Horns Rev, however.

Visual observations: Visual observations were carried out along a 2 km transect which ran perpendicular to the outer edge of the windfarm, with the ship located halfway along it length. On the windfarm side of the transect, the gap between the edge of windfarm as defined by the row of the outer turbines (approximately 300 m from the ship) to 700 m inside the windfarm (or 1,000 m from the ship) was regarded as being inside the windfarm. On the corresponding non-windfarm side, the transect which was between 300–1,000 m from the ship was regarded as being outside the windfarm (in relation to the windfarm this represents a distance of between 600 and 1,300 m). Collectively these were termed as Class A, whereas the transect up to 300 m either side of the ship was Class B (excluding birds within 30 m either side of the ship which were disregarded). Visual observations of flying birds (optics only used for identification purposes) were carried out every half hour for observation periods of 15 minutes from sunrise to sunset. Distance, flight direction and altitude were recorded (classes were largely defined by the upper and lower limits of the rotor blade: 0-5 m: 5-30 m; 30-100 m; >110 m). The results of this work are not considered further here.

Visual observations were carried out for 219.5 and 238.5 h in 2005 and 2006 respectively.

Study period

March to May to coincide with spring migration (27.5 observation days in 2005 and 2006) and September to November to cover autumn migration (39 observation days in 2005 and 2006).

Radar appeared to have been run continuously.

Species

Seaducks, geese, gulls and terns and wide range of songbird species.
Transect counts showed that gulls (many of which were unidentified to the species level) were the most common group recorded in both spring and autumn (with little gull notably more common in the former time of year). Common scoter were also common but more so in spring.

Conditions data collected under

Horizontal radar observations were limited to calm sea state conditions (wind speed < 2 ms^{-1}) and generally dry weather.

Visual observations were stopped when visibility <1 km but visual and acoustic observations were possible for all observation days

Results

During the day, the overall number of tracks flying parallel to the windfarm was higher (n = 1,045) compared to flying away from (n = 486) or towards (n = 386) the windfarm. This pattern was less pronounced at night with the number of birds parallel to the windfarm (n = 253) being only marginally higher compared to flying away from the windfarm (n = 206) but were higher than towards the windfarm (n = 101).

Although the visual observations were designed primarily to look at the differences in flight height distribution, they were able to provide supporting evidence for macro avoidance occurring. For northern gannet, out of 66 gannets recorded only 2 flew within the windfarm. For both little gull and all gull spp (excluding little gull), significantly less birds were present inside the windfarm.

Assessment of methodology

Results from the observations from horizontal radar were limited as only 5% (9% for Nysted) of the observation time yielded screenshots which could be used and these were biased to daytime periods. There was also the additional problem that detection within the windfarm was considerably lower compared to outside due to the presence of the wind turbines (tracks were observed to disappear and reappear when entering and leaving the windfarm).

There were several limitations with working on a ship compared to from land or a fixed platform, including rough sea conditions, which would likely hamper data collection. There were also issues associated with the tidal cycles (particularly at Horns Rev, less so at Nysted) and strong winds which could result in the ship turning and this affected the radar data collected. Another potentially confounding factor is that the ship could also act as an attractant to some species of seabirds (e.g. gull spp) or potentially act as a disturbance to others (e.g. diver spp and duck spp).

In terms of demonstrating macro-avoidance, horizontal radar was unable to provide quantitative evidence. Avoidance appeared to be implied by the percentage of birds flying parallel being higher than those values reported for birds flying towards and away from the windfarm and this pattern was more pronounced during the day when the windfarm was more visible. The significance of birds tracks running parallel to as opposed to being orientated towards or away from the windfarm was not explained, however, and there was a lack of pre-construction information to make comparisons with. There was also insufficient data to look at potential changes in the orientation of tracks (but enough data was available for Nysted – see section 5.4.4). Similarly the visual observations did not provide quantitative evidence of macro-avoidance rates.

Case study number 3

Skov H., Leonhard, S.B., Heinänen, S., Zydelis, R., Jensen, N.E., Durinck, J., Johansen, T.W., Jensen, B.P., Hansen, B.L., Piper, W. & Grøn, P.N. 2012. *Horns Rev 2 Monitoring 2010-2012. Migrating Birds*. Orbicon, DHI, Marine Observers and Biola. Report commissioned by DONG Energy.

Methods

This report focussed on migrating birds in relation to Horns Rev 1 and 2.

Radar observations: Horizontal radar was used from observation stations located to the north east of Horns Rev 1 (assumed to be the same as used in previous studies at Horns Rev 1, 560 m distance to the windfarm) and to the east of Horns Rev 2 (no distance provided but estimated to be less than 2 km away). Radar range was set at 6.0 km and covered a circular area. Additional information on species identification was possible by use of “a real-time tracking” procedure whereby tracks of individual birds or tracks could be followed on background images to produce videos. Videos were produced using a frame grabber connected to the radar and tailor made software provided the video as a back ground image on the PC screen. Whilst one observer followed the trace on the screen, a second attempted to locate the target in the field using a binocular or telescope to provide names, number of birds and altitude. Identification on tracks was not always possible during busy periods. Track densities were estimated for a 100 m^2 grid system within the radius of the radar.

Laser range finders: Laser range finders (Vectronix 21 Aero) were also used from the observation stations used to collect species-specific data up to distances of 2-3 km for large bird species (depending on the field of view and flight mode of the bird). Positions and altitudes of birds were logged automatically via GPS recorded at intervals of 10-15 sec. Data from the laser range finders were used to supplement data collected by the radar. Calibrations in order to correct the readings provided by the GPS were necessary due to interference by the observation tower.

Track data for range finders and radar were also integrated with weather data including wind direction, wind speed, air pressure, clearness, humidity, total precipitation and air temperature. In addition, the relative flight direction of the bird in relation to wind direction was also calculated.

Generalised Additive Models (GAMs) with a Tweedie distribution were used to look at track densities derived by radar for all bird tracks and common scoter tracks in relation to distance to the radar and distance to the windfarm. Generalized Additive Mixed Models (GAMMs) with a correlation structure (to deal with spatial and temporal autocorrelation) were used to look at the flight altitude in relation to weather variables and distance to the nearest wind turbine. However, this information could not be used to quantify an avoidance rate.

Study period

Data collection carried out during spring and autumn from September 2010 to May 2012. No further details given.

Radar observations: 15 min per h during daylight.

Laser range finders: operated permanently with observation periods of a minimum of 15 min per h.

Species

All spring and autumn migrants (seabirds, water birds, ducks and passerines).

Conditions data collected under

Not specified.

Results

Tracks recorded by both horizontal radar and the laser range finders were mapped for a range of species/groups in order to visualise movement patterns. It was proposed that diver spp (small sample size), northern gannet and common scoter tended to migrate along corridors along the periphery of the windfarms, although looking at the maps provided it is clear that northern gannet[17] and common scoter[18] did occur within the windfarms, notably Horns Rev 2. This was thought to be a result of the bathymetry as common scoters seemed to associate with waters less than 10 m in depth.

At Horns Rev 2 both distance to radar and distance to the windfarm were significant predictors of the densities for all birds tracks combined [19] and common scoter tracks. Response curves[20] produced by the models were similar for both analyses, which was unsurprising given the relative proportion of all tracks that were from common scoter. A peak in the density of birds occurred at around 1,500-2,500 m from the windfarm and was argued to provide evidence for a barrier effect due to birds

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[17] Figure 5-14- Skov *et al*. 2012.

[18] Figure 5-15- Skov *et al*. 2012.

[19] GAM; Distance to radar F=321.5, $p < 0.01$ and distance to windfarm F=286.4 , $p < 0.01$. Overall deviance explained 18.6% - Skov *et al*. 2012.

[20] Figure 5-23 - Skov *et al*. 2012.

altering their flight path. Similarly at Horns Rev 1, both distance to radar and distance to the windfarm were significant predictors for all bird tracks and common scoter tracks. In terms of the response curves, distance to windfarm the peak for all birds was between 2,000-3,000 m, whereas for common scoter it was around 1,000-2,000 m[21].

Assessment of methodology

From the results provided it is not possible to quantify an overall macro-avoidance rate although this study did provide information on the distances to which barrier effects were observed.

A1.3 Nysted offshore Windfarm

Location / habitat

Marine, offshore 10 km.

Turbine /array specification

Turbine array consists of 72 2.3 MW Bonus turbines covering 24 km^2. Distance between turbines – north to south (480 m) and east to west (850 m). Turbine specifications given as: hub height 69 m; rotor blade length 41 m; total height 110 m. Clearance above water is 28 m.

Case study number 1

Petersen, I.K., Christensen, T.K., Kahlert, J., Desholm, M. & Fox, A.D. 2006. *Final results of bird studies at the offshore windfarms at Nysted and Horns Rev, Denmark*. Commissioned by DONG Energy and Vattenfall A/S. National Environmental Research Institute.

Desholm, M. & Kahlert, J. 2005. Avian collision risk at an offshore windfarm. *Biology Letters* 1: 296-298[22].

Methods

Peterson *et al.* (2006) focussed on barrier effects, displacement effects, physical changes to the habitat and collision risk. Work was carried out at Horns Rev and Nysted offshore windfarm but there were differences in methodology and timing of data collection. Study at Nysted covered the three phases of: baseline (1999-2002); during construction (2002-2003) and post-construction (2003-2005). Desholm and Kahlert (2005) reported the results from the barrier effects and collision risk work only.

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[21] Figure 5-26 -- Skov *et al*. 2012.

[22] Assumed to be derived from the same data as Peterson *et al.* 2006.

Radar observations: Recordings by radar (Furuno FR125) were carried out from an observation tower, 5 km north-east of the windfarm area. The range was approximately 11 km and covered a circular area of 388 km^2. Migration was mapped by tracing the course of flocks onto a transparency and subsequently digitised. Only tracks longer than 5 km were included in the analyses.

The lateral response to the windfarms was investigated by setting a number of transects: the eastern gate (located along the full length the most eastern edge of the windfarm); the northern gate (located along the full length the most northern edge of the windfarm) and the buoy transect (running from north to south from the observation tower to a buoy, 6.9 km in length). During autumn migration, tracks of flocks of birds travelling in a westerly direction which crossed the buoy transect were selected to see if they crossed the eastern gate (in order to derive the percentage of birds which did so). In contrast, during spring migration the flight behaviour of birds was studied after they passed the windfarm and so is not considered further here. The total numbers of flocks of birds crossing the eastern and northern gate were also counted. In addition, migration intensities were compared for an area within the windfarm with an adjacent area outwith the windfarm (both less than 11 km^2 in area). Each area was subdivided into squares of 0.1 km^2 and within each cell, the lengths of radar tracks (bird flocks) were expressed as the total sum of track meters (the track density). In order to derive the change due to the windfarm, proportional differences in the bird densities within and inside the windfarm from the baseline data (pre-construction) were used to correct the data collected post-construction to derive avoidance rates.

In order to determine the response distance (where avoidance occurs) to the windfarm, transect lines to the east of the windfarm were set up which ran parallel to the direction of the rows of turbines (from north to south). These were spaced at intervals of 100, 200, 300, 400, 500 m and then at intervals of 500 m to 4,000 m and after which there were a further two transects at 5,000 and 6,000 m. The mean ± s.d. migration course of tracks were calculated for each transect (based on the gap between the transect itself and the 100 m interval to the west).

Visual observations*:* Abundance, phenology, diurnal pattern and flock sizes of species were recorded along the buoy transect. Count data was then converted into number of birds per 15 mins for all westerly bound birds in autumn and easterly bound birds in spring (although again the latter represents the number of birds after passing through the windfarm).

Study period

Radar observations: spring (easterly-orientated migration) and autumn (westerly-orientated migration) periods covered. Total number of hours or breakdown by season not reported.

Visual observations*:* During the main survey periods of 14 March to 19 April and 30 August to 12 November from 1999 - 2005, observations were carried out two days per week covering day and night time periods. A total of 259 h and 579 h observations gathered for the spring and autumn periods.

Species

Staging and migrating birds but common eider and geese spp most commonly recorded.

Conditions data collected under

Not specifically described but very little data of conditions under poor visibility (<1 km).

Results

The probability of birds crossing the windfarm was analysed using a logistic regression model and included the following explanatory terms and first order interactions (phase of development; distance to the observation tower when crossing the buoy transect), time of day, direction of winds (all of which were found to have significant effects). It was shown through comparison of data from the baseline and operation phases that 0.78 of all birds[23], which consisted mostly of common eider, avoided entering the windfarm post-construction during autumn migration. This was based on 40% of flocks entering the eastern edge of the windfarm during the baseline period compared to 9% during operation[24]. This was suggested to equate to 8 out of 10 flocks crossing the eastern gate during the baseline study then avoiding the windfarm during the post-construction phase. It was also shown that during the post-construction phase, the numbers of flocks crossing the eastern gate were higher at night than during the day (Desholm and Kahlert 2005 cited values of 13.5 % and 4.5 % respectively).

More specifically there was notable inter-annual variation in macro-avoidance rates for autumn migrating birds, again mostly common eider, ranging from 0.63 and 0.83[25] in the use of the windfarm post-construction compared to the baseline. These rates were derived from figures of 0.08-0.09 of flocks passing the eastern side of the windfarm compared to 0.24-0.48 passing the eastern gate of the windfarm during the pre-construction period[26].

There was a difference in migration intensity during the baseline period as the track densities in the eastern windfarm were 60% of the reference area which suggested a problem with detection rate. Nevertheless a significant reduction in track densities was reported for the post-construction period but there was acknowledgement that a reduction could be partially explained by problems of what is termed a shadow effect to do with individual turbines.

The standard deviation of the orientation was used to determine the lateral deflection as means of quantifying response distance to the windfarm (citing Kahlert *et al.*

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[23] Figure 121. Calculated as 1-(0.09/0.40) - Peterson *et al.* 2006.

[24] Desholm and Kahlert (2005) reported the proportion of flocks entering the windfarm decreased from 40.4% during pre-construction to 8.9% during initial operation. Data collection methods were not extensively described - Peterson *et al.* 2006.

[25] Calculated as 1-0.08/0.48 and 1-0.09/0.24 - Peterson *et al.* 2006.

[26] Figure 122 - Peterson *et al.* 2006.

(2005) as justification for this approach). Analyses of data collected during the autumn migration, showed a significant interaction between the phase of development and distance to the windfarm (other terms were also significant but not discussed here due to lack of information presented which can be evaluated with respect to providing evidence for the response distance)[27]. Plots of the means of annual standard deviation values showed that there was little change in orientation for distances between 100 m and 5 km from the windfarm during the baseline period[28]. However, during the operation period, the orientation of tracks steadily changed over the distances 5 to 1 km away from the windfarm (orientation of birds at 3 km from the windfarm were significantly different to the baseline period) and the greatest deflection occurred between 500 m and 100 m (note that the way the transects were set up, there was a gap between 500 m and 1 km). A tendency was also reported for the first deflection to be recorded at greater distances during the day compared to the night time period (based on the multiple use of pair-wise t tests across each distance interval)[29].

Assessment of methodology

As there was a before and after comparison carried out at Nysted this was argued to provide greater confidence (compared to Horns Rev) that any changes were as a direct result of the windfarm presence.

The response distance was only possible for birds entering the windfarm during autumn (the area used during spring migration was beyond the edge of the radar range and hence the derived figures are based on autumn migration only. Moreover, tracks do not differentiate between individuals or flocks, therefore the reported macro-avoidance values do not respond to the level of individual birds.

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[27] Table 41- Peterson *et al.* (2006).

[28] Figure 119 - Peterson *et al.* (2006).

[29] Table 42 - Peterson *et al.* (2006).

Case study number 2

Blew, J., Hoffman, M., Nehls, G. & Hennig, V. 2008. *Investigations of the bird collision risk and the responses of harbour porpoises in the offshore windfarms Horns Rev, North Sea, and Nysted, Baltic Sea, in Denmark. Part 1: Birds*. Report from the University of Hamburg and BioConsult SH, 145pp.

Methods

Methods used were exactly the same as used for Horns Rev (Appendix 1, section A1.2)

Study period

March to May to coincide with spring migration (44 ship days in 2005 and 2006) and September to November to cover autumn migration (51.5 ship days in 2005 and 2006).

Radar appeared to have been run continuously.

Species

Wide range of non-pelagic waterbirds with high numbers of common eider as well as raptors and songbirds. Transect counts showed that in spring, the common eider was by far the most common bird recorded and in autumn it was the great cormorant.

Conditions data collected under

Horizontal radar observations were limited to calm sea state conditions (wind speed < 2 ms^{-1}) and generally dry weather. Weather and sea state conditions tended to be better than those experienced at Horns Rev where fewer observation days were possible.

Visual observations were stopped when visibility <1 km.

Results

Radar tracks were categorised according to their direction in relation to the first row of the windfarm: flying towards (± 45° either side of perpendicular to the windfarm; flying away; and flying parallel (more or less). Initially tracks were presented regardless of their location (and therefore distance) in relation to the windfarm (but included tracks within the boundary of the outer row of the windfarm). During the day the overall number of tracks flying parallel to the windfarm was higher (n = 2,274) compared to towards (n = 1,725) or away (n = 563) from the windfarm. This pattern was not evident at night when the numbers flying towards (n = 968) and parallel (n = 804) were more similar but still much higher than flying away (n = 216).

In terms of determining whether horizontal avoidance occurred, the mean (and standard deviations) of angles of the approaching tracks were presented for the four

500 m width distance bands, for all anchor points east and west of Nysted offshore windfarm. It was reported that the angles did not increase (as would be predicted if horizontal avoidance occurred) or differ with decreasing distance to the windfarm (no statistical analyses were carried out).

Although the visual observations were designed primarily to look at the differences in flight height distribution, they were able to provide supporting evidence for macro avoidance occurring. For all gull spp significantly less birds were present inside the windfarm. No results for northern gannet were provided.

Assessment of methodology

See Appendix 1, section A1.2 for a discussion regarding the work carried out on radar and visual observations at Horns Rev where the same approach was used. With respect to looking for evidence of horizontal avoidance this study was unable to show evidence for a change in flight orientation. It was unclear though whether this was due to relatively wide bands being used (500 m in width) as other studies have used smaller intervals of 100 m at distances less than 1,000 m from the windfarm.

APPENDIX 2 Evidence review macro-response – displacement and attraction studies

A2.1 Egmond aan Zee

Leopold, M.F., Dijkman, E.M. & Teal, L. 2011. *Local Birds in and around the Offshore Windfarm Egmond aan Zee (OWEZ) (T-0 & T-1, 2002-2010)*. Texel, The Netherlands: Wageningen IMARES.

Leopold M.F., Camphuysen C.J., van Lieshout S.M.J., ter Braak C.J.F. & Dijkman E.M. 2004. *Baseline studies North Sea windfarms: Lot 5 marine birds in and around the future site Nearshore Windfarm (NSW)*. Alterra-rapport 1047.

Lindeboom, H.J., Kouwenhoven, H.J., Bergman, M.J.N., Bouma, S., Brasseur, S., Daan, R., Fijn, R.C., de Haan, D., Dirksen, S., van Hal, R., Hille Ris Lambers, R., ter Hofstede, R., Krijgsveld, K.L., Leopold, M. & Scheidat, M. 2011. Short-term ecological effects of an offshore windfarm in the Dutch coastal zone; a compilation. *Environmental Research Letters* **6**. doi:10.1088/1748-9326/6/3/035101.

Location/habitat

Marine 10-18 km offshore.

Turbine /array specification

Hub height 70 m and a rotor diameter 90 m (rotor altitude min 25 m, max rotor altitude 115 above mean sea level). Turbine array consists of 36 Vestas V90 3 MW turbines covering an area 27 km^2. Distance within turbines is 650 m within rows and 1000 m between rows.

Methods

The focus of Leopold *et al.* (2004) and (2011) was to look at avoidance and attraction by birds to the windfarm at Egmond aan Zee for what were termed local birds (although the survey work did cover the Princess Amalia windfarm site, results specific to this windfarm site were not presented). Survey periods covered the pre-construction and post-construction phases of the development. Lindeboom *et al.* (2011) reported the impacts of the windfarm on a range of taxonomic groups but with respect to birds presented less detail than the above reports and therefore is not considered further here.

The study area was approximately 725 km^2 (22 x 33 km). It was selected on the basis that it would include an adjacent offshore windfarm, Princess Amalia, and an anchorage area, where ships wait before entering the nearby major port. Ten transect lines were selected running east to west at distances of 2.47 km apart (with eight additional transect lines added in 2008 running north east to south west). The aim was to cover each transect twice (this was possible until the additional transect lines were added) and the transect lines were sailed in the same order each survey period. Successfully completed surveys ranged between 4-8 days in duration.

Ship based strip census surveys based on the methods adopted in the baseline studies in 2002-2004 (described in Leopold *et al.* 2004) which were originally derived from Tasker *et al.* (1984); Komdeur *et al.* (1992) and Camphuysen and Garthe (2004). All swimming birds were assigned to distance bands: AB (0-100 m); C (100-200 m) and; D (200-300 m) and all observations were assigned to five minute intervals. Flying birds were recorded using the snap shot methodology at intervals of 1 min.

Although BACI design was originally set to look at bird responses to the windfarm, there was considerable annual variation in seabird presence which hampered the ability to look for any differences between pre-construction and post-construction. Therefore the results focussed on comparisons within surveys (e.g. species-specific monthly counts). Presence/absence data were used as the response in Generalised Additive Mixed Models, which took into account temporal auto-correlation, for all individual species/month combinations there were sufficient data for. Otherwise a more simple General Additive Model was used or, in some cases, statistical models could not be run (birds were counted less than 10 occasions). Therefore, the number of surveys that were available for further analyses varied according to species and were a reflection of the relative abundance of birds each month. Presence /absence data were argued to be more appropriate as they were less affected by the large numbers of zero counts or the few counts with very large numbers of birds recorded. These models took into account the distance to coast, the northing value and the presence of impact area as factors (Egmond aan Zee, Princess Amalia and the anchorage area were considered individually within these models). The model output was then used to predict and subsequently map the probability of birds occurring across the survey area.

Within surveys, there was the possibility of four outcomes: attraction (probability of finding birds inside the windfarm was significantly higher than expected on the basis of the general distribution pattern); avoidance (probability of finding birds inside the windfarm was significantly lower than expected); indifference (probability of finding birds within the perimeter was not impacted by the windfarm and insufficient data.

Study period

Baseline/pre-construction surveys: T-0 = September and October 2002; April, May, June, August and November 2003; February 2004 (described in Leopold *et al.* 2004).

Post-construction surveys: T-1a = April, June, August, September, November (incomplete) 2007 and January 2008 (May was not repeated); T-1b = April, June, August, September (incomplete), November 2008; January, 2010; T-1c = April, June, August, October (September not possible) November 2009 and; January and February 2010.

Species

Local seabirds as defined as those which reside for some time in the study area. Species accounts were presented for: diver spp, great crested grebe, northern fulmar, northern gannet, great cormorant, common scoter, little gull, black-headed gull, common gull, lesser black-backed gull, herring gull, great black-backed gull, black-legged kittiwake, Sandwich tern, common/arctic tern, common guillemot and razorbill).

Conditions data collected under

Generally aimed to survey in conditions with a Beaufort scale of less than 6 Bft but there were a number of transects that were carried out in higher winds of 6-7 Bft (when light conditions permitted).

Results

Northern gannet: Northern gannet tended to occur on all sides around Egmond aan Zee windfarm but rarely within the perimeter of the windfarm[30]. Observations recorded that those few birds that did enter only went one turbine deep. Where presence/absence analyses were possible for the post-construction period (n = 10 surveys), it was shown that the presence of the species was significantly negatively related to the Egmond aan Zee windfarm for only two surveys. Anecdotally it was reported that gannets never entered Princess Amalia Windfarm (which has a higher turbine density[31]). Also highlighted was the lack of searching feeding, resting in the windfarms during the surveys.

Lesser black-backed gull: It was evident that lesser black-backed gulls were often seen within perimeters of windfarm[32]. These birds tended to be either resting on the water or foundation structures or feeding at the tidal wakes around the monopiles. Presence/absence analyses for the post-construction period (n = 12 surveys), found that the presence of the species was negatively related to the Egmond aan Zee windfarm for only one survey (the rest were also negative but insignificant). This was counter to what would have been predicted as large fishing vessels only operated outside the windfarm which should have in effect reduced the numbers of birds inside the windfarm (resulting in an apparent avoidance). Most observations of lesser black-backed birds were anecdotally reported to be associated with, looking out for or resting in the wake of active fishing vessels.

Herring gull: Birds did occur in the windfarm area but overall fewer birds were recorded in the offshore environment compared to other gulls (notably in August where herring gulls remain mostly near shore). Like lesser black-backed gulls they were often associated with fishing vessels. Presence/absence analyses for the post-construction period (n = 14 surveys), found that the presence of the species was negatively related to Egmond aan Zee windfarm for eight surveys although this effect was only significant in three cases. Herring gull distribution patterns were thought to be likely to be attributable to overall latitudinal variation, as evidenced by the strong effect of distance to coast in the models (significant p values for six surveys).

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[30] Figure 31 - Leopold *et al.* 2011.

[31] 60 2 MW turbines which are evenly spaced (550 m apart) in area of 14 km^2 - Leopold *et al.* 2011.

[32] Figure 34 - Leopold *et al.* 2011.

Great black backed gull: Birds were reported as occurring in the windfarm area[33]. Presence/absence analyses for the post-construction period (n = 18 surveys), found that the presence of the species was positively related to the Egmond aan Zee windfarm in five cases, four significantly, although this effect was only apparent at low densities. There were also two surveys in which significant effects were reported. As reported for lesser black-backed gull, birds did tend to feed around fishing vessels but not in the same high numbers.

Black-legged kittiwake: birds were recorded within the windfarm and in general numbers declined with decreasing distance to shore (apart from in November and one January). Presence/absence analyses for the post-construction period (n = 5 surveys), found that the presence of the species was positively related to the Egmond aan Zee windfarm in three cases, one significantly.

Assessment of methodology

Overall, there was lack of consistent evidence for either displacement or attraction for any of the species. This could have been partly due to the importance of factors operating at the larger scale of study area. For the larger gull species, there was a strong association with fishing vessels in the study area. Since fishing was no longer permitted in the windfarm areas, this could have confounded any results reported to do with possible attraction or avoidance of windfarms. There was also evidence that distance to coast was an important factor in determining the overall distribution patterns of herring gulls.

There were potential issues relating to the choice of statistical approach. As comparisons of pre-construction and post-construction data was deemed not to be possible, multiple tests for individual surveys were carried out which may have led to the possibility of a Type 2 error (increased chances of reporting a false significant result). Also the numbers of observations were low for northern gannet and gull spp and consequently the modelling power was very low (Lindeboom *et al.* 2011). Moreover, the model outputs were in the form of p values and model co-efficients which could not be converted into avoidance rates without further details being presented (even if consistent effects had been observed). Therefore, from the results provided, it is not possible to derive displacement/attraction rates or thus macro-response rates for the study species.

A2.2 Robin Rigg

Natural Power. 2014. Analysis of Marine Ecology Monitoring Plan Data from the Robin Rigg Offshore Windfarm, Scotland (Post-construction Year 3). Draft Technical Report. E.ON Climate & Renewables.

Location/habitat

Marine, offshore < 11 km

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[33] Figure 38 - Leopold *et al.* 2011.

Turbine /array specification

Turbine array consists of 60 3.0 MW Vestas turbines which are positioned approximately 500 m apart. Turbine specifications are given as turbine towers 80 m high and a rotor blade length of 44 m.

Methods

The purpose of this report was to look at: displacement of key species; changes in patterns of abundance and distribution; compare observed patterns with predicted impacts/sensitivities from the EIA process.

Data collection was carried out during the pre-, during and post-construction periods.

Boat based surveys based on standard European Seabirds-At-Sea (ESAS) survey methods were carried our (e.g. prior to the publication of Camphuysen *et al.* 2004) as used in the baseline period. In order to ensure comparability between the different phases of the development, methods were kept the same throughout. Additional survey work has been carried out from year 3 of the post-construction period which corresponds to current best practice. The main difference between the two approaches is that for flying birds the former records flying birds using transect methodology whereas the latter uses the snap shot methodology currently regarded as best practice. A total of 10 parallel transects running in a south west to north east direction of 18 km in length and spaced 2 km apart.

For the purpose of analyses, each survey was divided into individual blocks of 600 m^2 (corresponding to the 300 m either side of the transect line as both sides of the boat are surveyed). In terms of the data, there was a cleaning process applied. Uneven sampling effort across the different phases of the development (some months were surveyed twice) was identified as an issue and therefore a single survey at random was selected. The study area was also cropped to remove an area in the northeast where shallow waters sometimes prevented access and two transects in the southeast were removed due to under surveying during the pre- and during construction phases. There was also a gap during the construction period where there was no building activity (January and July 2008) and these were also excluded from the analysis.

Birds on the water and birds in flight were analysed separately. Datasets that had fewer than 300 non-zero observations were not considered. Raw observations were mapped and summary statistics for the three development phases were calculated in order to provide an initial indication of any change. These included: mean number of sightings (groups of animals), mean number of individuals per segment and mean number of individuals per segment per month. These are not discussed here however and the results of models output are focussed upon.

Distance Sampling techniques were not used to correct the survey counts and a correction factor derived using the detection function was applied instead. Generalised Additive mixed effects mixture modelling carried out within a Bayesian framework were applied in order to deal with zero inflation (high number of zeros). Transect and survey were incorporated as random effects in order to deal with spatial and temporal autocorrelation. Covariates used in the models were latitude, longitude month (or season) and time of day.

Outputs of the models were used to produce density surface maps of the predicted distribution during the three different phases of the development. Abundance and density estimates for each species within the windfarm and the study area were produced for each phase. In order to look at avoidance, model outputs were used to predict the number of animals within the windfarm and for buffers 0.5, 1, 1.5 and 2 km of the three different windfarm phases. Model outputs were presented only for the comparisons of pre-construction to construction and pre-construction to post-construction (but it was not clear which of the spatial scales they related to).

Study period

Baseline surveys: monthly basis between May 2001 and April 2002. Further pre-construction surveys April and May 2003 and then on a monthly basis between January 2004 and September 2004 (excluding April and June) with further work in July 2007. Construction surveys: monthly basis between January 2008 and February 2010 (excluding November 2009). Post-construction: monthly surveys from March 2010 to February 2013 – scheduled to continue until 2015.

Species

Data were collected for a wide range of species (e.g. seabirds, seaducks, waders, passerines). Species accounts were only presented for the following key species: scaup, common scoter, red-throated diver, Manx shearwater, northern gannet, great cormorant, black-legged kittiwake, herring gull, great black-backed gull, common guillemot, and razorbill.

Conditions data collected under

Not specified but ESAS provide guidance regarding suitability of conditions.

Results

Northern gannet: Modelling of the numbers of northern gannet on the water was not possible as there were too few sightings. The predicted numbers of northern gannet in flight across the three different phases of the development were found not to be significantly different. There appeared to have also been relatively little change in the predicted densities for the windfarm site, windfarm plus buffers (at any of the scales) or even at the level of the study area[34]. Although northern gannet was recorded throughout the study area, densities of the gannets were reported as being generally low[35].

Black-legged kittiwake: The predicted numbers of black-legged kittiwake on the water across the three different phases of the development were found not to be

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[34] Figure 3.55 – Natural Power 2014.

[35] Figures 3.56-3.61- Natural Power 2014.

significantly different. There appeared to have also been relatively little change in the predicted densities for the windfarm site, windfarm plus buffers (at any of the scales) or even at the level of the study area[36]. A similar result was found for black-legged kittiwakes in flight[37].

Herring gull: Modelling of the numbers of herring gull on the water was not possible as there were too few sightings. The predicted number of herring gull in flight across the three different phases of the development were found to be significantly different with the numbers within the windfarm decreasing over the development (pre-construction to construction p = 0.0021, parameter estimate -0.750 and pre-construction to post-construction p = 0.0013, parameter estimate - 0.841).

Great black-backed gull: Modelling of the numbers of herring gull on the water was not possible as there were too few sightings. The predicted number of herring gull in flight were found to significantly differ from pre-construction to construction (p = 0.0166, parameter estimate -1.133) but not from pre-construction to post construction (p = 0.7854).

Assessment of methodology

There were insufficient data to allow modelling of the observations of birds on the water for northern gannet, herring gull, and great black-backed gull. For birds in flight, there was evidence for a significant decrease for herring gull both during the construction and post-construction periods whereas this decrease was only noted during construction for great black-backed gull. Northern gannet and black-legged kittiwake did not appear to respond to the presence of the windfarm. From the results provided, it was not possible to derive macro-response rates since it was not clear what models have been fitted and it was not apparent whether the changes were due to the presence of the windfarm or as result of changes at the scale of the overall study site. It is acknowledged though that despite this being year 3 of the post construction, it is not the final report and any reported results should be considered as preliminary findings.

A2.3 Blighbank

Vanermen, N., Stienen, E.W.M., Courtens, W., Onkelinx, T., Van de walle, M. & Verstraete, H. 2013. *Bird monitoring at offshore windfarms in the Belgian part of the North Sea - Assessing seabird displacement effects*. Rapporten van het Instituut voor Natuur- en Bosonderzoek 2013 (INBO.R.2013.755887). Instituut voor Natuur- en Bosonderzoek, Brussel.

Location/habitat

Marine, 42 km offshore

Turbine /array specification

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[36] Figure 3.82 - Natural Power 2014.
[37] Figure 3.83 - Natural Power 2014.

55 turbines. Additional information was not presented.

Methods

This report looked at Blighbank and Thorntonbank windfarms (but also referred to the more recent development of Lodewijckbak) in what is termed the windfarm concession zone located in the north eastern edge of the Belgian Part of the North Sea (BPNS). Surveys at both windfarms are still on going.

Data collection was carried out during the pre-construction, during and post-construction periods.

A BACI approach was adopted in order to monitor sea bird displacement. A control area of comparable size was selected on the basis of having similar attributes in terms of number of birds, environmental conditions and having sufficient historic data. A buffer zone of 3 km was applied to the boundary of the windfarm (and the control area), in order to reflect the distance to which the effects of the windfarm could be an issue for birds.

Boat transects were carried out on a monthly basis (citing Tasker *et al.* 1984) from 2008. The time interval used in this survey for recording was 10 minutes (a number of other windfarm surveys use 1 min). Although only transect routes used post 2012 were shown[38], despite some apparent minor shifts in the location the overall configuration was considered to be the same over the whole monitoring period (Nicolas Vanermen *pers. comm.*). An overview was provided of all the ESAS counts carried out by INBO during the period of 1992-2012 based on location of counts, this could not be used to look at survey effort which varied over the study period[39]. Count effort for Blighbank[40] (as shown by the number of surveys) indicated overall higher effort in the pre-construction period (but this included data possibly dating back to 1992). There was also marked monthly variation in effort in the preconstruction phase with peaks in February/August for the pre-construction period and in March/December for the post construction period.

Although distance sampling was used to correct count data to estimate the total numbers of birds within the BPNS (based on Buckland *et al*. 2001), it was not applied for modelling of the windfarm data (this was on the grounds that the correction factor used for both control and the windfarm area was likely to be the same Nicolas Vanermen *pers. comm.*). In order to analyse the count data, generalised linear models were used, with a negative binomial distribution assumed in order to cope with over dispersion. Modelling was carried out using area (the reference area or the impact area) and month (as a as a continuous variable in order to model seasonality) included as explanatory terms in what was termed the reference model (based on data collected prior to April 2008). The best model was then selected using a backward approach using a Wald test and looking at the resulting AIC values. The impact model was a simple extension of the count

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[38] Figure 27 - Vanermen *et al*. 2013.

[39] Figure 2 - Vanermen *et al.* 2013.

[40] Figure 29 - Vanermen *et al*. 2013.

component of the reference model with before and after being added as factor variables to the model. Although not carried out in this report, the natural exponent of the model coefficients can be used to derive the factorial change (and hence the overall percentage change in numbers from pre to post construction – see Table A6.1).

Species' preference for the windfarm area was calculated using Jacob's Selectivity Index (calculated using the proportion of birds that occur inside the entire windfarm concession zone compared to the total numbers within the BPNS and the proportion of the surface area of the concession zone to the total area of the BPNS) whereby values of -1 represent total avoidance and + 1 is total preference (attraction). However this data was only carried out for the baseline data and hence are not considered further here.

The impact of the windfarm was considered separately for the post-construction phase at the scale of the windfarm, the windfarm and buffer, and the buffer without the windfarm[41]. Displacement-related coefficients and their respective p values were reported.

Study period

The baseline period (reference period) referred to data pre-September 2009. The construction period ran from September 2009 to August 2010, and the post-construction period was from September 2010 onwards. Data collected during the initial construction period were not used in subsequent assessment due to access issues over this period. Results are presented for up until December 2012.

Species

Northern fulmar, northern gannet, great skua, little gull, common gull, lesser black-backed gull, herring gull, great black-backed gull, black-legged kittiwake, common guillemot, and razorbill.

Conditions data collected under

Not specified in the report. Conditions were, however, mostly favourable - boat surveys are cancelled when wave heights > 1.8 m, and in poor visibility (Nicolas Vanermen *pers. comm.*).

Results

Northern gannet: Model coefficients were significant for the scale of the windfarm and buffer and buffer without the windfarm (see Table A6.1). Therefore there were highly significant decreases in numbers of northern gannet in the windfarm and the buffer of 3 km at all three spatial scales considered.

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[41] Table 18 - Vanermen *et al.* 2013.

Lesser black-backed gull: Model coefficients were significant for the windfarm and buffer, and buffer without the windfarm, and were only just not significant for just the windfarm. Therefore there was a significant increase in numbers of lesser black-backed gull in the windfarm and the buffer of 3 km relative to the pre-construction period.

Herring gull: The model coefficient was only significant at the scale of the windfarm, indicating an increase in numbers in the windfarm area relative to the pre-construction period.

Great black-backed gull: The model coefficients were not significant, indicating no changes in numbers of the species relative to the pre-construction period.

Black-legged kittiwake: The model coefficients were not significant, indicating no changes in numbers of the species relative to the pre-construction period.

Table A2.1 Model outputs of Negative binomial modelling converted into factorial changes

Species	Scale	Model coefficient	P value	Factorial Change*	Overall change as a proportion
Northern gannet	Windfarm	-1.83	0.000	0.16	0.84
	Windfarm plus buffer	-1.52	0.000	0.22	0.78
	Buffer	-1.32	0.003	0.27	0.73
Lesser black-backed gull	Windfarm	1.57	0.059	4.81	-3.81
	Windfarm plus buffer	2.39	0.004	10.91	-9.91
	Buffer	2.37	0.006	10.70	-9.70
Herring gull	Windfarm	3.97	0.000	52.98	-51.98
	Windfarm plus buffer	1.26	0.111	-	-
	Buffer	0.83	0.269	-	-
Great black-backed gull	Windfarm	1.08	0.127	-	-
	Windfarm plus buffer	0.47	0.447	-	-
	Buffer	0.54	0.428	-	-
Black-legged kittiwake	Windfarm	0.25	0.605	-	-
	Windfarm plus buffer	0.50	0.264	-	-
	Buffer	0.77	0.092	-	-

*natural exponent of the model co-efficient.

Assessment of methodology

The results of this report should be considered as being preliminary since further data was collected for 2013. Nevertheless, northern gannet was shown to decrease in response to the presence of windfarm by a value of 0.84. This value could be taken as being indicative of macro-avoidance. Whereas both lesser black-backed gull and herring gull shown quite marked attraction to the windfarm. Great black-backed gull and black-legged kittiwake showed no overall response to the windfarm. From the results provided it was not possible to look at seasonal variation in displacement or attraction.

Sampling effort was biased towards the pre-construction phase and was characterised by variable effort on a monthly basis. Spatial coverage over the whole study period is likely to have been fairly consistent however. The data presented in this report is based on a BACI approach and potentially has limited value in looking at changes in the wider area but long term monitoring in the BPNS has continued throughout the study period and hence there is scope to include this at a later stage if required.

A2.4 Thorntonbank

Vanermen, N., Stienen, E.W.M., Courtens, W., Onkelinx, T., Van de walle, M. & Verstraete, H. 2013. *Bird monitoring at offshore windfarms in the Belgian part of the North Sea - Assessing seabird displacement effects*. Rapporten van het Instituut voor Natuur - en Bosonderzoek 2013 (INBO.R.2013.755887). Instituut voor Natuur- en Bosonderzoek, Brussel.

Location/habitat

Marine, 27 km offshore.

Turbine /array specification

Initially six turbines, final array to consist of 54 turbines.

Methods

See Appendix 2, section A2.3 for overall approach.

The impact of the windfarm was considered separately for the two different operation phases: phase 1 (turbine array consisting of six turbines) and; phase 2 (second construction period). Models were run at the scale of the windfarm and buffer only[42].

Power analyses were also carried out for the reference data collected in the Thorntonbank study area in order to determine the power required to detect change in numbers of birds (25, 50 and 75% decrease) and the length of the monitoring period required.

Study period

Monthly surveys were started in 2005 (although additional data were available from 1993 based on surveys that have been carried out of the whole region of the BPNS but coverage was uneven spatially and temporally). The baseline period (reference period) referred to data pre-April 2008. The construction period ran from April 2008 to May 2009, and the post-construction period (called here the impact period) was from June 2009 to April 2011. Thereafter there was another period of construction from May 2011 that was ongoing at the time of the report.

Species

Northern fulmar, northern gannet, great skua, little gull, common gull, lesser black-backed gull, herring gull, great black-backed gull, black-legged kittiwake, Sandwich tern, common tern, common guillemot, razorbill.

Conditions data collected under

Not specified but ESAS provide guidance regarding suitability of conditions.

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[42] Table 15 - Vanermen *et al.* 2013.

Results

Northern gannet: For both phase 1 and phase 2, the model coefficients were not significant, indicating no changes in numbers of the species relative to the pre-construction period.

Lesser black-backed gull: For phase 1, the model co-efficient was not significant. For phase 2, a significant model co-efficient of 2.13 was reported ($p = 0.052$) for the scale of the windfarm, indicating a decrease inside the windfarm (but this effect was not found for the other models at the scales of the windfarm plus buffer, and buffer without the windfarm).

Herring gull: For both phase 1 and phase 2, the model coefficients were not significant, indicating no changes in numbers of the species relative to the pre-construction period.

Great black-backed gull: For phase 1, the model co-efficient was reported as 1.5 and was found to be significant ($p = 0.024$) for the windfarm plus buffer indicating an attraction to the windfarm. Whereas for phase 2, the model coefficients were not significant, indicating no change in numbers of the species relative to the pre-construction period.

Black-legged kittiwake: For both phase 1 and phase 2, the model coefficients were not significant, indicating no changes in numbers of the species relative to the pre-construction period.

Assessment of methodology

The results of this study were derived from when the windfarm only consisted of 6 turbines (phase 1) or during the next phase of construction of a further 48 turbines (phase 2). Hence the years covered by this report do not include the post-construction phase of a fully post-construction windfarm. Hence the results are not considered further here as part of this review.

A2.5 Nysted

Petersen, I.K., Christensen, T.K., Kahlert, J., Desholm, M. & Fox, A.D. 2006. *Final results of bird studies at the offshore windfarms at Nysted and Horns Rev, Denmark*. Commissioned by DONG Energy and Vattenfall A/S. National Environmental Research Institute.

Location/habitat

See under section 5.4.1.3 under barrier effects.

Turbine /array specification

See under section 5.4.1.3 under barrier effects.

Methods

Aerial transect surveys were carried out using methodology described in Kahlert *et al.* 2004 (which prior to the publication of Camphuysen *et al.* 2004 was commonly cited by other studies as the standard methodology). A total of 26 parallel transects running north to south separated by distances of 2 km were carried out covering an area of 1,700 km^2. The area was extended by four additional transect lines in 2002 to increase the area to 1,846 km^2.

Jacob's selectivity indexes (D) were used in order to look at displacement and attraction. This approach essentially determines bird preferences for the windfarm area and a buffer zone (2 and 4 km) where birds could still be impacted, in relation to their preference to the whole study area. Values fell between -1 (displacement) and +1 (attraction). Bird encounters (for both individuals and groups here termed as clusters) rather than estimates of bird densities were used. Bird preferences were then compared by looking at the pre- and post-construction D values, based on a simple comparison of number rather by formal statistical analyses, in order to describe the change in bird utilisation of the windfarm.

Bird encounter rate (number of birds reported per km of survey route per observer) was used as a proxy of density in order to calculate mean densities in the windfarm area and in the buffer zone. Comparisons of the mean densities pre- and post-construction were carried out using Student's t-test with corrections for unequal variance. Sufficient data (with respect to the five priority species) was available for comparisons for herring gull at Nysted in January and Horns Rev in March.

Study period

Pre-construction period = August 1999 to August 2002 (n = 21 surveys); construction period = January 2003 to August 2003 (n = 3); post-construction period = January 2003 (sic) to November 2005 (n= 8). The timing of the actual surveys (e.g. by month were not reported). Only the pre-construction and post-construction surveys were used. There was a lack of autumn surveys for the post-construction phase and therefore only winter and spring surveys were available.

Species

Diver spp, great cormorant, long-tailed duck, common eider, common scoter, red-breasted merganser, herring gull and great black-backed gull.

Conditions data collected under

Not specified.

Results

Herring gull: Comparisons of pre- and post-construction selectivity indices for numbers clusters of birds showed no change[43]. Whereas selectivity indices for

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[43] Table 27 and 29 - Peterson *et al*. 2006.

numbers individuals showed a tendency towards decreased selectivity (e.g. less birds were using the area) for the windfarm as well as both buffer zones[44]. There was no significant difference between bird encounter rate between the pre- and post-construction phases in the windfarm area or the 4 km zone but a significant difference was found for the 2 km buffer. The report concluded there was no evidence for either attraction or avoidance.

Great black-backed gull: outputs of the models were all found to be insignificant apart for the selectivity indices for individual birds post-construction and hence are not reported further here as they have no meaningful comparison for pre-construction.

Assessment of methodology

Overall there was little evidence that herring gull showed any response to the presence of the windfarm.

There are a number of potential limitations of the approach used. There may be issues to do temporal coverage – from the information provided, it was difficult to be able to evaluate how sampling effort varied over the different phases of the development. Also whilst the Jacob's selectivity indices may provide an indication of the likely direction of response, these cannot be directly translated into displacement rates. Also the comparison of pre- and post-construction bird encounter rate had limited value since they provided no indication of changes in distribution that may have occurred at a wider scale (and therefore nothing to do with the presence of the windfarm).

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[44] Table 28 and 30 - Peterson *et al*. 2006.

A2.6 Horns Rev

Location/habitat

See under Appendix 1, section A1.2.

Turbine /array specification

See under section Appendix 1, section A1.2.

Methods

Aerial surveys: Aerial transect surveys were carried out using methodology described in Kahlert *et al.* (2004) which prior to the publication of Camphuysen *et al.* (2004) was commonly cited by other studies as the standard methodology. A total of 26 parallel transects separated by distances of 2 km were carried out covering an area of 1,350 km^2.

Study period

Pre-construction period = August 1999 to January 2002 (n = 16 surveys); construction period = March 2002 to August 2002 (n = 3); post-construction period January 2003 to November 2005 (n= 15). The timing of the actual surveys (e.g. by month were not reported). Only the pre-construction and post-construction surveys were used.

Species

Diver spp, northern gannet, common eider, common scoter, little gull, Arctic/common tern and guillemot.

Conditions data collected under

Not specified.

Results

Northern gannet: There were no observations of northern gannet inside the windfarm pre- or post-construction. Comparisons of pre- and post-construction selectivity indices for the buffer zones indicated increased avoidance at the 2 and 4 km zone. Insufficient numbers of birds were recorded in order to be able look at encounter rates and limited further interpretation of what the likely overall response of northern gannet to the windfarm.

Herring gull: Comparisons of pre- and post-construction selectivity indices for clusters and individuals of birds indicated a reduced avoidance of the windfarm area. The bird encounter rate revealed no significant difference between the pre- and post-construction period. It was concluded that despite an increased preference being found during construction (citing Christensen *et al.* 2003), attraction was not observed post-construction.

Black-legged kittiwake: Model outputs were not significant for numbers of clusters of birds post-construction and for both pre- and post-construction for numbers of individual birds. Hence the results are not reported here.

Assessment of methodology

See Appendix 2, section A2.5.

A2.7 Alpha ventus demonstration site

Bundesamt fur Seeschiffart und Hydrographie, BSH 2011. Okologische Begleitforshung bei alha ventus erste Ergebnisse (Environmental research at alpha ventus – first results). Contributions from the Event of 10 May 2010, Katholische Akademie Hamburg.

Mendel, B., Kotzerka, J., Sommerfeld, J., Schwemmer, H., Sonntag, N. & Garthe, S. 2014. Effects of the Alpha Ventus offshore test site on distribution patterns, behaviour and flight heights of seabirds. *In* Ecological Research at the Offshore Windfarm Alpha Ventus: Challenges, Results and Perspectives. Editors Federal Maritime and Hydrographic Agency, Federal Ministry for the Environment, Nature Conservation and Nuclear Safety. Springer Spektrum.

All the post-consent monitoring reports from this OWF demonstration site are written in German (Stefan Garthe *pers. comm.*). The first reference reviewed is a report (BSH 2014) which has a full English translation. The second reference (Mendel *et al.* 2014) is a book chapter and is written in English. Neither reference can be considered to be fully comprehensive in the level of detail provided but given the importance of this OWF site this information should be included. The information which is cited below is largely taken from Mendel *et al.* (2014).

Location/habitat

45 km offshore

Turbine /array specification

Twelve turbines. Two designs (jacket foundation and tripod steel foundations) – no further information provided.

Methods

Two study areas were selected: the key study area, the size of which was in excess of 30 times the size of the windfarm itself and; a reference site which appeared to be nearly twice the size of the study area. Boat based surveys were carried out according to standard European Seabirds-At-Sea (ESAS) survey methods. Aerial-based methods were based on methods described in Pihl and Frikke (2002), Noer *et al.* (2000) and Diederichs *et al.* (2002) (full citations are given in Mendel *et al.* 2014). As well as data from the EIA studies, additional data from eight multiple-day ship-based surveys and 21 aerial surveys carried out in both study areas were available. No further information was provided, however (e.g. on the timing of the surveys in relation to season).

In order to carry out analyses of the changes in distribution patterns for pre-and post-construction data, data were collated into grid cells of 1 km^2 and only data from the key study area were used. A total of six species or species groups were looked at (divers, northern gannet, lesser-black backed gull, little gull, black-legged kittiwake and common guillemot) and only the most important period/s for each of these were focussed upon. Data were also collated over large time periods (usually seasons).

Changes in abundance were looked at using the pre- and post-construction data and only two species were considered (lesser-black backed gull and common guillemot). Generalised Linear Mixed Models of the abundances of birds at different distances in relation to the windfarm (0-2 km, 2-6 km and 6-10 km) were tested in three different models using a Poisson error distribution.

The percentage of birds recorded in each behavioural category was calculated for the key study areas and the reference area for lesser black-backed gull only.

Study period

Data from 2000-2008 were regarded as pre-construction (construction started in September 2008) and data from 2010-2012 represented the post-construction period.

Species

Northern gannet, northern fulmar, black scoter, skua spp, gull spp, and auks spp. Key species: Red-throated diver, black-throated diver, lesser black-backed gull, black-legged kittiwake, little gull, common guillemot and razorbill.

Conditions data collected under

Data collected according to ESAS methods (sea state < 5Bft).

Results

Changes in distribution

The statistical significance of the following results was not provided and interpretation of results was largely based on maps representing densities of birds for the 1 km^2 grid cell system of the key study area. Overall lower abundances were reported post-construction for six of the species/groups but only the relevant species are reported further here.

Northern gannet: the impact of the windfarm was hard to qualify due to the very low numbers recorded within the key study area. This species was reported to have occurred on seven occasions (nine individuals) within the windfarm area during the pre-construction period and none were observed post-construction. Data were taken from March to September and hence represented the breeding season.

Lesser-black-backed gull: a ‘clear decrease’ was reported to have occurred from the pre- to the post-construction period. Although low to medium densities were reported post construction within the windfarm area, the highest densities were found a few kilometres away from the windfarm site (previously some of the highest were found within the perimeter of the windfarm area during pre-construction). Data were taken from May to July and hence represented the breeding season.

Black-legged kittiwake: a ‘remarkable decline’ occurred post-construction not only within the perimeter of the windfarm but at the scale of the whole key study sites. Numbers recorded overall were very low however (e.g. highest number of birds recorded per km^2 was 5). Data were taken from November to April and hence represented the non-breeding season.

Changes in abundance

Lesser-black backed gull: Statistically significantly lower abundances were reported for the 0-2 km, 2-6 km and 6-10 km distance class and the models suggested that the disturbance effect was strongest within 2 km of the windfarm,

Assessment of methodology

Based on the information provided, it is not possible to carry out a proper assessment of the methodology used. The overall abundance of northern gannet was very low and therefore this study cannot be cited as evidence of the windfarm having an impact on their distribution. There is some evidence to suggest that displacement may be occurring for lesser black-backed gull and black-legged kittiwake based on the maps of the distribution of bird densities for pre- and post-construction, but there was a lack of statistical analyses. However a statistically significant reduction in the abundance of lesser black-backed gulls was reported for all the three distances classes from the windfarm.

APPENDIX 3 EVIDENCE REVIEW HORIZONTAL MESO-RESPONSE

A3.1 De Put, Nieuwkapelle

Everaert, J. 2008. *Effecten van windturbines op de fauna in Vlaanderen Onderzoeks resultaten, discussie en aanbevelingen*. INBO, Brussels

Methods

Baseline data describing bird movements within the area, prior to turbine construction, were collected on six days between December 2004 and February 2005 at periods of dawn and dusk. Following turbine construction, additional data were collected on six days between December 2005 and March 2006, again at dawn and dusk. Changes in the number of birds flying within 100 m and 300 m of each turbine pre- and post-construction were then modelled using a factorial ANOVA.

Seasons / time of day

Data were collected over the winter at dawn and dusk.

Species

Black-headed and common gulls.

Conditions data collected under

Not specified.

Location / habitat

Terrestrial site in Belgium.

Turbine / array specification

A two turbine array. Each turbine has a mast height of 75 m and a rotor diameter of 48 m.

Results

No significant differences were recorded in the number of black-headed or common gulls passing within 300 m or 100 m of the turbines between the pre- and post-construction periods.

Assessment of methodology

A key flaw in this study is the lack of a control site with which to compare differences in movement pre- and post-construction. A consequence of this is that it is not possible to determine whether the lack of significant changes reflects the local population remaining relatively stable or whether the overall proportion, but not numbers, of a variable local population passing the turbines has changed.

A3.2 Egmond aan Zee

Krijgsveld, K.L., Fijn, R.C., Japink, M., van Horssen, P.W., Heunks, C., Collier, M.P., Poot, M.J.M., Beuker, D. & Dirksen, S. 2011. *Effect studies Offshore Wind farm Egmond aan Zee: Final report on fluxes, flight altitudes and behaviour of flying birds*. Bureau Waardenburg

Methods

Radar Observations

Between July 2009 and March 2010, the flight paths of birds within the windfarm were recorded using a horizontal radar with range of 0.75 nautical miles. The study area included six turbines and it was possible to collect data on 235 out of the 239 days during the study period, although it was necessary to filter out data on an additional 59 days due to the incidence of ‘clutter’. Data were then analysed using a t-test to assess whether birds were distributed evenly within the windfarm by comparing the number of birds passing within 50 m of a turbine to the number of birds elsewhere.

Seasons / time of day

Data were collected during daylight on eight occasions between July and December.

Species

Not stated

Conditions data collected under

All conditions.

Location / habitat

Marine 10 km offshore.

Turbine / array specification

Egmond aan Zee Offshore Windfarm covers an area of 27 km^2 and contains 36 turbines. Each turbine has a hub height of 70 m and rotor diameters of 90 m. Turbines are arranged in four rows, with 650 m between turbines in each row and 1 km between rows. The study of horizontal meso-responses covered six turbines at the edge of the windfarm.

Results

There was a statistically significant difference in the numbers of birds flying within 50 m of the turbines in comparison to the proportion of birds elsewhere in the study area. Over the course of the study period, this reflected a horizontal meso-response rate of 0.34 (i.e. the number of birds within 50 m of a turbine was 66% of that elsewhere within the windfarm).

Assessment of methodology

Data used in this study have been collected using radar, meaning near-continuous data collection was possible. In order to detect finer scale movements of birds in relation to the windfarm, the resolution of the radar was reduced to cover a distance of 0.75 nautical miles. As a consequence, it was possible to detect movements of birds that were as close as 1 m to turbines. However, a key limitation of the data is that it is not possible to relate echoes to individual species, or to determine whether a single echo reflects an individual birds, or a flock. An additional limitation is that birds at low altitudes may have been obscured by high waves, which they exploit in order to minimise energy expenditure.

A3.3 Horns Rev I and II

Skov, H., Leonhard, S.B., Heinanen, S., Zydelis, R., Jensen, N.E., Durinck, J., Johansen, T.W., Jensen, B.P., Hansen, B.L., Piper, W., Grøn, P.N. 2012. *Horns Rev 2 Monitoring 2010-2012. Migrating Birds.* Orbicon, DHI, Marine Observers and Biola. Report commissioned by DONG Energy

Methods

Radar Monitoring

Between September 2010 and May 2012 Bird movements were recorded using horizontal radar at stations within the Horns Rev I and Horns Rev II offshore windfarms. All movements within 6 km of the radar were recorded. Two observers were used during the data collection. The first observer followed the tracks and recorded information within a database. The second observer attempted to locate each of the tracked objects in the field using binoculars or a telescope and relayed information on the species identification, number and altitude to the first observer.

Seasons / time of day

Data were collected during the spring and autumn migration periods during the hours of daylight.

Species

Northern gannet (442 birds), common scoter (2,374 birds), large gulls (408 birds), terns (617 birds).

Conditions data collected under

Data were generally collected during relatively calm conditions (little wind or rain and good visibility).

Location / habitat

Horns Rev I is located 17.9 km from the Danish coast and Horns Rev II is located 31.7 km from the Danish coast.

Turbine / array specification

Horns Rev I is an array of 80 turbines, each with a hub height of 70 m and a rotor diameter of 80 m. Horns Rev II is an array of 91 turbines, each with a hub height of 68 m and a rotor diameter of 93 m.

Results

The study estimated the mean, minimum and maximum distances from turbines recorded by each species. On average, northern gannets were recorded passing within 1,119 m of turbines (range 0-2,840 m), common scoter were recorded passing within 921 m of turbines (range 0-4,302 m), large gulls were recorded passing within 783 m of turbines (range 50-2,252 m) and terns were recorded passing within 840 m of turbines (range 0-2,355 m). In practice, without knowing the shapes of these distributions, it is hard to use this information to estimate the magnitude or direction of horizontal meso-responses to the turbines. In practice, the mean distance to turbines is likely to be strongly influenced by the body size of the species concerned, or by their tendency towards flocking behaviour, both of which are likely to increase their detection at greater distances. However, of the 408 large gulls tracked, none passed within 50 m of the turbines, suggesting a strong, negative meso-response to the turbines occurring at a distance of at least 50 m.

Assessment of methodology

The way data are presented make it difficult to disentangle meso-responses to the turbines. In particular, biases may exist relating to the detectability of different species, which may make the estimates of mean distance to turbines unreliable. Of the information presented, the minimum distance to turbines for large gulls is of value in estimating a meso-response rate.

A3.4 Hungary

Janoska, F. 2012. Investigations of Bird Collisions in 2 Wind farms. *International Scientific Conference on Sustainable Development & Ecological Footprint*, Sopron, Hungary, March 26-27 2012

Methods

Between November 2010 and November 2011, two Hungarian windfarms were visited every two weeks. During visits, the altitude and flight direction of birds were noted.

Seasons / time of day

Data were collected throughout the year.

Species

Yellow-legged gull

Conditions data collected under

No Details given.

Location / habitat

Two terrestrial sites in Hungary.

Turbine / array specification

No details given.

Results

Of the yellow-legged gulls recorded, only 2.5% (23/917) were recorded flying within 75 m of turbines, reflecting a meso-response of 0.975, and only 0.6% (6/917) were recorded flying within 25 m of turbines, reflecting a meso-response of 0.994.

Assessment of methodology

Very little detail is given describing the methodology used. As a consequence, these data must be interpreted with extreme caution. In particular, it is unclear to what extent data reflect avoidance, and to what extent they more generally reflect the flight paths taken by birds passing through the area.

APPENDIX 4 EVIDENCE REVIEW VERTICAL MESO-RESPONSE

A4.1 Barrow Offshore Windfarm

Barrow Offshore Wind Limited. *Post Construction Ornithological Monitoring – Third Year Report and Overall Conclusions*.

Methods

Boat-based estimation of flight heights.

Following the construction of Barrow Offshore windfarm, boat-based surveys were carried out during the breeding season and autumn migration in 2006, 2008, 2009 and 2010. In total 12 surveys, each lasting a single day were carried out, of which 8 were during the breeding season (May to August) and 4 during autumn migration (September to November). Boat survey data were collected within the windfarm according to standard protocols (Camphuysen *et al.* 2004) and flying birds were assigned to height bands of <5 m, 5- 15 m, 15-100 m and >100 m. Birds at risk of collision were assumed to be all those flying >15 m. The proportion of birds observed flying at heights presenting a risk of collision were then summarised across all surveys. Pre-construction proportions at collision risk height within the windfarm were compared to post-construction proportions at collision risk height, although no detailed analyses were undertaken.

Seasons / time of day

Data were collected during the breeding season and autumn migration periods.

Species

Auk spp. (238 recorded in 2010), common guillemot (2,002 recorded in 2010), razorbill (691 recorded in 2010), great cormorant (5 recorded in 2010), red-throated diver (2 recorded in 2010), black-headed gull (6 recorded in 2010), common gull (5 recorded in 2010), great black-backed gull (23 recorded in 2010), herring gull (142 recorded in 2010), black-legged kittiwake (132 recorded in 2010), lesser black-backed gull (425 recorded in 2010), gull spp. (51 recorded in 2010), Arctic skua (2 recorded in 2010), northern gannet (53 recorded in 2010), Manx shearwater (12 recorded in 2010), Sandwich tern (30 recorded in 2010), common scoter (10 recorded in 2010),

Conditions data collected under

No details given.

Location / habitat

Marine 7 km Offshore

Turbine / array specification

An array of 30 turbines covering an area of 10 km^2 and arranged in four rows of seven or eight turbines each. The rows are separated by a distance of 750 m and within the rows, each turbine is separated by a distance of 500 m. Each turbine has a hub height of 75 m above sea-level and a rotor diameter of 90 m.

Results

Several species were not present in sufficient numbers to allow a reliable estimate of the changing proportion of birds flying at a height placing them at risk of collision. Of those that were, common guillemot, great black-backed gull, herring gull, lesser black-backed gull and Sandwich tern all showed a decline in the proportion of birds flying at risk height, with meso-responses of 1, 0.29, 0.65, 0.28 and 0.55 respectively. However, other species (or groups) showed an increase in the proportion of birds flying at risk height including black-legged kittiwake, unidentified gulls and northern gannet, with meso-responses of -0.41, -0.85 and -0.59 respectively, reflecting an apparent attraction to the rotor-swept area of the turbines.

Assessment of methodology

Boat-based data collection was robust, following standard methodologies (Camphuysen *et al.* 2004). However, in assessing the vertical response to turbines there is a key flaw in the available data. In order to compare flight height data to that collected pre-construction, the same flight height bands were used in both study periods, and it was assumed that all birds flying at a height of more than 15 m above sea-level were potentially at risk of collision. However, as the rotor-swept area covers an area from 30 m to 120 m above sea-level, this may lead to a significant over-estimate of the actual number of birds flying at collision risk height. As a result, the meso-response rates of birds within the windfarm may be underestimated. An additional, arguably less serious, flaw in the data collection is that estimates of the birds at collision risk height refer to flocks, rather than individuals. Flock size is likely to show significant variation, making it difficult to infer what the proportional changes mean in relation to actual numbers of birds.

A4.2 Blyth Offshore Windfarm

Rothery, P., Newton, I., Little B. (2009) Observations of seabirds at offshore wind turbines near Blyth in northeast England. *Bird Study* **56,** 1-14

Methods

Shore based observations were undertaken between 18 April 1998 and 30 August 2003 covering the pre-construction, construction and post-construction periods of Blyth Offshore Windfarm. Observations were carried out at pre-determined times, at least twice a month. All passing birds were recorded, and it was stated that all birds were visible at a range of 1 km, although the turbines are only likely to comprise a small part of the total observation area. All birds were assigned to one of four height categories – 0-9.1 m, 9.1-26.4 m, 26.4-92.4 m and >92.4 m. A total of 70.3 hours of monitoring were available for the pre-construction period and 351.6 hours for the post-construction period, although no analyses were undertaken to assess the significance of any changes in flight height.

Seasons / time of day

Data were collected throughout the year and during daylight hours.

Species

Northern gannet (432 birds post-construction), great cormorant (352 birds post-construction), common scoter (341 birds post-construction), common eider (1,034 birds post-construction), black-headed gull (978 birds post-construction), herring gull (1,408 birds post-construction), great black-backed gull (564 birds post-construction), black-legged kittiwake (1,350 birds post-construction), Sandwich tern (2,135 birds post-construction).

Conditions data collected under

Data collected under all conditions in which visibility was at least 1 km.

Location / habitat

A shallow spit, approximately 1 km from shore.

Turbine / array specification

Two turbines spaced 200 m apart with a hub height of 59.4 m above mean sea-level and a rotor diameter of 66 m.

Results

For each species, the change in the proportion of birds flying at altitudes greater than 9.1 m above mean sea-level pre and post-construction are available. For most species, a greater proportion of birds fly above 9.1 m post-construction than pre-construction. The increase in the proportion of gulls flying above 9.1 m varied from 114-238% during the summer and 267-2,900% in the winter. Similarly during the summer, the proportion of gannets flying above 9.1 m increased by 2,800%.

Assessment of methodology

Despite the authors’ assurances, it is unlikely that all birds were detected over the full range of the observation area. In particular, birds at lower altitudes may be obscured by waves, or be less visible against the sea surface. As a result, the proportion of birds at lower altitudes may have been under-estimated. In addition, the change in observation platform between pre- and post-construction periods is likely to have afforded an improved view of the observation area. These factors mean that pre- and post-construction comparisons of the estimates of birds at different altitudes may not be reliable. In addition, the presence of the turbines offering a fixed structure with which to assess birds' flight heights against, is likely to have improved the accuracy of estimates of flight heights made post-construction. Finally, by limiting the comparison to birds above 9.1 m, well below the rotor sweep of the turbines, the proportion of birds at risk is likely to be vastly over-estimated.

A4.3 Egmond aan Zee

Krijgsveld, K.L., Fijn, R.C., Japink, M., van Horssen, P.W., Heunks, C., Collier, M.P., Poot, M.J.M., Beuker, D., Dirksen, S. 2011. *Effect studies offshore wind farm Egmond aan Zee. Final report on fluxes, flight altitudes and behaviour of flying birds.* Bureau Waardenburg

Methods

Visual observations

Between spring 2007 and December 2009, 405 panorama scans were carried out from a met mast on the edge of the Egmond aan Zee Windfarm. Scans were undertaken once an hour during daylight covering a 360° angle around the windfarm with a pair of 10 x 42 binoculars fixed on a tripod. During each observation period, two scans were undertaken, the first to capture birds close to the sea surface and the second to capture birds at greater altitudes. The height of birds was estimated using trigonometry to combine the distance and angle between the bird and observer. Birds could be viewed to a distance of up to 3 km, although imperfect detection is likely to be an issue at these distances. The area covered by each panorama scan is approximately 50% within the windfarm and 50% outside, allowing for simple comparisons to be made of birds inside and outside of the windfarm, although differences were not assessed statistically.

Seasons / time of day

Data were collected during daylight, throughout the year. There was increased effort during the spring and autumn migration periods.

Species

Northern gannet, great cormorant, black-legged kittiwake, black-headed gull, common gull, lesser black-backed gull, herring gull, great black-backed gull, Sandwich tern, small gull spp., large gull spp., gull spp.

Conditions data collected under

Data collected under all conditions.

Location / habitat

Marine 10 km offshore.

Turbine / array specification

Egmond aan Zee Offshore Windfarm covers an area of 27 km^2 and contains 36 turbines. Each turbine has a hub height of 70 m and rotor diameters of 90 m. Turbines are arranged in four rows, with 650 m between turbines in each row and 1 km between rows.

Results

Species varied in their vertical responses to wind turbines. Of the 13 species or groups considered, the proportion flying at rotor height was lower inside the windfarm than outside for kittiwake, black-headed gull, northern gannet, great black-backed gull, Sandwich tern and unidentified gull species (no numbers were presented). Large gulls appeared to show little, or no vertical response to the turbines, with roughly the same proportion flying at rotor height inside as outside. In contrast, the proportions of great cormorants, common gulls, little gulls and other small gulls flying at rotor height showed a noticeable increase inside the windfarm.

Assessment of methodology

Data are presented as the proportions of birds at rotor height both within and outside the windfarm. Without any details on the number of birds involved, it is difficult to determine the strength of these data, and the subsequent findings. Of particular concern is the way in which data for unidentified gulls have been presented and the apparent inconsistency in the results for each category which show roughly the same proportion of unidentified large gulls at rotor height inside as outside the windfarm, more small gulls at rotor height inside than outside the windfarm, but unidentified gulls assigned to neither category significantly less likely to be at rotor height within the windfarm. Without more details of the species likely to be covered by each category, and their abundance within the study area, it is difficult to assign levels of confidence to the results presented.

A4.4 Gunfleet Sands I and II

RPS. 2008. *Gunfleet Sands Monitoring Report*, RPS, London

NIRAS. 2011. *Gunfleet Sands 2 Offshore Wind farms. Year 1 Post-construction Ornithological monitoring*. NIRAS, Cambridge

GoBe Consultants Ltd. 2012. *Gunfleet Sands Offshore Wind farm I & II – Post Construction Year 2 – Marine Licence Environmental Monitoring Report.* Prepared for DONG Energy.

Methods

Boat surveys

Pre- and post-construction monitoring data were collected as part of boat surveys following standardised methodologies (Camphuysen *et al.* 2004). Flying birds were assigned to one of the following flight height bands <5 m, 5- 15 m, 15 -150 m. Pre-construction surveys were carried out between October 2007 and March 2008. Post-construction surveys were carried out between October 2010 and March 2011 and between October 2011 and March 2012. However, differences were not assessed statistically.

Seasons / time of day

Data were collected over winter, during periods of daylight.

Species

Red-throated diver, black-headed gull, common gull, great black-backed gull, gull spp., herring gull, black-legged kittiwake, lesser black-backed gull

Conditions data collected under

No details given.

Location / habitat

Gunfleet Sands I & II offshore windfarms, approximately 7 km from the coast.

Turbine / array specification

Gunfleet Sands I and II contain 48 turbines between them, each with a hub height of 75 m and a rotor diameter of 107 m. The projects cover a total area of 16 km^2.

Results

The proportion of red-throated divers flying at collision risk height declined following the construction of the windfarm, by 39% in winter 2010/11 and by 96% in winter 2011/12. In contrast, the proportion of great black-backed gulls at rotor height showed an increase following construction, by 75% in winter 2010/11 and 53% in winter 2011/12. The proportion of herring gulls at rotor height showed little change between pre-construction years and either post-construction survey. Results for other species were less consistent. For example common gulls showed an increase in the proportion at rotor height in 2010/11 compared to pre-construction data, but a decrease in 2011/12.

Assessment of methodology

Data were collected following a relatively robust methodology and the height bands used were a reasonable match for the dimensions of the rotor swept area of each turbine meaning the proportions of birds at risk height are less likely to be significantly over-estimated. However, the limited duration of pre- and post-construction surveys, reflected in the quantity of data available, means that there may only be limited power to detect significant changes in species flight heights.

A4.5 Nysted/Horns Rev

Blew, J., Hoffman, M., Nehls, G., Hennig, V. 2009. *Investigations of the bird collision risk and the responses of harbour porpoises in the offshore windfarms Horns Rev, North Sea, and Nysted, Baltic Sea, in Denmark*.

Methods

X-Band Radar

The spring and autumn migration periods were monitored at Horns Rev and Nysted in 2005 and 2006 using x-band radar mounted on vessels anchored in each windfarm. In total, across both windfarms 71.5 days of monitoring were carried out during the spring and 93.5 days during the autumn. Data were captured up to a height of 1,500 m and movements were examined in two height bands <200 m and 200-500 m. All birds tracked for > 100 m and showing a change in movement of >20 m were considered to have changed altitude.

Seasons / time of day

Data were collected throughout spring and autumn in 2005 and 2006.

Species

Having used radar, it was not possible to determine the species captured by the radar.

Conditions data collected under

It was not possible to collect data during periods of strong wind or heavy rain. However, all other conditions were covered.

Location / habitat

Horns Rev 17.9 km from the Danish North Sea Coast.
Nysted 10.8 km from the Danish Baltic Sea Coast.

Turbine / array specification

Horns Rev is an array of 80 turbines covering an area of 21 km^2. Each turbine has a hub height of 70 m and a rotor diameter of 80 m.

Nysted is an array of 72 turbines covering an area of 26 km^2. Each turbine has a hub height of 69 m and a rotor diameter of 82 m.

Results

Across both windfarms, and within the 0-200 m observation band, 4.8% of birds flying towards the windfarm were shown descending by more than 20 m and 13.4% were shown ascending by more than 20 m during the day time. At night time, the values were 2.9% and 13.6% respectively. However, these proportions did not differ significantly from the observations within the 200-500 m band, suggesting that the change in flight heights did not differ from what may be expected to occur by chance and are therefore unlikely to reflect avoidance behaviour.

Assessment of methodology

The rotor-swept area of each turbine covers altitudes from 20-110 m. Consequently, as data were relatively coarse and restricted to all flights within a band of 0-200 m, it may not have been possible to detect responses to turbines. In addition, having used radar, any responses to turbines that had been recorded could not have been identified to species level.

A4.6 Robin Rigg

Natural Power Consultants. 2013. *Analysis of Marine Environmental Monitoring Plan Data from the Robin Rigg Offshore Wind farm, Scotland (Operational Year 3).* Natural Power, Castle Douglas.

Methods

Boat-based surveys

Pre- and post-construction boat surveys were carried out within the windfarm following standard methodologies (Camphuysen *et al.* 2004). Birds in flight were assigned to bands of 0-5 m, 6-25 m, 26-34 m, 35-125 m, 126-200 m and >200 m. Surveys were carried out on a bi-monthly basis during pre-construction monitoring (2001-2007), and on a monthly basis during post-construction monitoring (2010-2011). Where sufficient data were available, differences in the proportions of birds flying at rotor height were assessed using a chi-squared test.

Seasons / time of day

Surveys were carried out throughout the year, during daylight.

Species

Common scoter, red-throated diver, diver spp., Manx shearwater, northern gannet, great cormorant, black-legged kittiwake, herring gull, great black-backed gull, gull spp, common guillemot, razorbill, auk spp.

Conditions data collected under

No details given.

Location / habitat

Robin Rigg Offshore Windfarm, 11 km from shore.

Turbine / array specification

Robin Rigg is an array of 60 turbines, each with a hub height of 80 m and a rotor diameter of 88 m. The turbines are spaced at intervals of approximately 500 m.

Results

There were no significant differences in the proportions of birds flying at rotor height during pre- and post-construction surveys for common scoter and red-throated diver. However, the proportion of northern gannet, great cormorant, black-legged kittiwake and large gull species flying at rotor height within the windfarm all increased between pre- and post-construction. However, the low power of the data was noted raising concerns over the validity of the results.

Assessment of methodology

Flight height data were not collected following the standard ESAS methodology and concerns are raised that this is likely to lead to a double counting of individuals, meaning estimates of changes in the proportion of birds at collision risk height may not be reliable.

APPENDIX 5 EVIDENCE REVIEW MICRO-AVOIDANCE

A5.1 Egmond aan Zee

Krijgsveld, K.L., Fijn, R.C., Japink, M., van Horssen, P.W., Heunks, C., Collier, M.P., Poot, M.J.M., Beuker, D., Dirksen, S. 2011. *Effect studies offshore wind farm Egmond aan Zee. Final report on fluxes, flight altitudes and behaviour of flying birds.* Bureau Waardenburg

Methods

Between July and December 2009, the flight paths of birds around six turbines were observed visually. These flight paths were then related to short range radar tracks in order to estimate the altitude and distance to nearest turbine. As a result, a dataset containing high resolution observations of bird behaviour around turbines was created. Birds were assigned to 5 m horizontal distance bands beginning at the rotor hub. All birds flying between 20 and 120 m above sea-level (reflecting the rotor-swept area of each turbine) were considered to be at risk of collision and the number of birds within each 5 m band was compared to the number of birds that would have been expected if they had been distributed evenly. To assess the level of last-second avoidance action taken, the number of birds within the 45-50 m band (just outside the rotor-sweep) was compared to the number of birds recorded between 0 and 45 m from the rotor hub.

Seasons / time of day

Data were collected during daylight on eight occasions between July and December.

Species

Seabirds, waterbirds and other migrants.

Conditions data collected under

All conditions.

Location / habitat

Marine, 10 km offshore.

Turbine / array specification

Egmond aan Zee Offshore Windfarm covers an area of 27 km^2 and contains 36 turbines. Each turbine has a hub height of 70 m and rotor diameters of 90 m. Turbines are arranged in four rows, with 650 m between turbines in each row and 1 km between rows. The study of micro-avoidance covered six turbines at the edge of the windfarm.

Results

Whilst 1,610 birds in 409 groups were recorded over the course of the study, only 115 in 52 groups were recorded passing within 50 m of the turbines. Of these, only 36 birds were recorded between 20 and 120 m, at heights placing them at risk of collision. Of the 36 birds passing within 50 m of the turbine and at rotor height, it is reported that 0.926 did not fly within the rotor swept window of the turbine (i.e. 2-3 birds). This would reflect a micro-avoidance rate of 0.926.

Assessment of methodology

The described methodology of combining visual and radar observations to record the tracks of birds approaching turbines is robust. This makes it possible to relate tracks to individual species and to determine how close each individual, or flock, gets to a turbine. Focussing on the area 50 m either side of the rotor hub and comparing the proportion in the 45-50 m band to the proportion in the 0-45 m band data is likely to capture the type of last-minute action covered by micro-avoidance.

However, only limited weight can be given to the data presented here. Observations were recorded on only four days, during which only 36 birds were recorded passing within 50 m of the turbine, the distance presented to represent micro-avoidance. This figure may be substantially inflated as it includes a single observation of a flock of 28 skylark.

A5.2 Greater Gabbard

RPS. 2011. *Galloper Wind farm Project Environmental Statement – Technical Appendices 2: Appendix 4: Greater Gabbard post-construction vantage point surveys*, RPS, Glasgow

Methods

Visual Observations

Two surveyors collected data from 180˚ arcs to the port and starboard sides of a stationary vessel within Greater Gabbard Offshore Windfarm. Each arc had a radius of 2 km and all birds entering each arc were recorded during snapshot counts taken every 15 seconds. The location of the boat and the viewing area, which covered a total of 15.9 km^2, included seven operational turbines and a total of 36 hours of data were collected during the survey. The flight paths of each bird within the viewing area were noted, as was the proportion of time each bird spent at different heights.

Seasons / time of day

Data were collected between 1st June 2011 and 28th July 2011, with each survey lasting four hours.

Species

Northern gannet (0.14 birds/hr), Arctic skua (0.03 birds/hr), lesser black-backed gull (3.69 birds/hr), herring gull (0.11 birds/hr), black-legged kittiwake (1.28 birds/hr).

Conditions data collected under

Conditions were limited to sea-states one and two, to ensure the vessel remained as a stable observation platform.

Location / habitat

Greater Gabbard, UK (offshore).

Turbine / array specification

The survey monitored seven operational turbines, each with a hub height of 77.5 m and a rotor diameter of 107 m.

Results

Over the course of the study period, 190 flights through the area were recorded. Of these, the vast majority did not pass close to the turbines. Given the proportion of the total study area occupied by turbines, this is unsurprising. As a consequence, only a single evasive manoeuvre, involving a kittiwake, was recorded.

Assessment of methodology

The length of the observation periods carried out during this study were extremely limited, so it is difficult to make an accurate assessment of how widespread different avoidance actions are. In addition, records of avoidance action have been made in a subjective fashion, both in relation to assessing the number of birds on a collision course for the turbines, and in assessing the actions recorded. For these reasons, it is not possible to quantify the micro-avoidance behaviour reported in this study.

A5.3 Kessingland Windfarm

Wild Frontier Ecology. 2013. *Kessingland Windfarm Annual Post-construction Monitoring Report Year 2.* Wild Frontier Ecology, Norfolk

Methods

Bird activity was monitored within the windfarm through nine two-hour vantage point surveys at each turbine carried out between November 2012 and March 2013. In total 36 hours of survey effort was completed throughout the study period. The response of birds whose flight paths were likely to overlap with turbines was noted.

Seasons / time of day

Late morning – early afternoon during winter.

Species

Black-headed gull (97 birds/hr), common gull (31.4 birds/hr), lesser black-backed gull (11 birds/hr), herring gull (56.72 birds/hr), great black-backed gull (0.28 birds/hr).

Conditions data collected under

No details given.

Location / habitat

Kessingland, Suffolk, UK (terrestrial).

Turbine / array specification

Two turbines with hub heights of 80 m and rotor diameters of 92 m. Distance between turbines within each row is not described.

Results

All birds recorded as being on a collision course with the turbines were observed to take evasive action to avoid collision. Typically this action occurred at a distance of 0-50 m from the turbine. Over the course of the study period, five black-headed gulls, two lesser black-backed gulls and a herring gull were recorded taking evasive action. In three instances this involved a change in altitude to fly below the rotor blades, whilst in other instances it involved a change to flight direction. In the case of the two lesser black-backed gulls, both were observed to take last minute evasive action at just five metres from the blades.

Assessment of methodology

The length of the observation periods carried out during this study were extremely limited, so it is difficult to make an accurate assessment of how widespread different avoidance actions are. In addition, records of avoidance action have been made in a subjective fashion, both in relation to assessing the number of birds on a collision course for the turbines, and in assessing the actions recorded and the distances at which they occur. For these reasons, it is not possible to quantify the micro-avoidance behaviour reported in this study.

A5.4 Nysted

Desholm, M. 2005. *TADS investigations of avian collision risk at Nysted offshore wind farm, autumn 2004*. NERI, Denmark

Petersen, I.K., Christensen, T.K., Kahlert, Desholm, M., Fox, A.D. 2006 *Final results of bird studies at the offshore wind farms at Nysted and Horns Rev, Denmark*, NERI, Denmark

Methods

Using a Thermal Animal Detection System (TADS) all bird movements past a single turbine during spring and autumn 2004 and spring and autumn 2005 were recorded. Birds were detected at distances of up to 120 m.

Seasons / time of day

Data were collected throughout both day and night in the spring and autumn.

Species

Mostly migrant passerines and waterbirds.

Conditions data collected under

All conditions.

Location / habitat

Located approximately 11 km offshore in the Danish part of the Baltic Sea.

Turbine / array specification

An array of 72 turbines arranged in eight rows of nine turbines each. Turbines have a hub height of 69 m and a rotor diameter of 92 m.

Results

In over 123 days of continuous monitoring, cameras captured 5,507 video sequences of which only 14 were found to include birds. Of these, none revealed birds passing close to the turbine.

Assessment of methodology

The methodology is robust with sufficient capability to record all birds passing the turbine over the study period. However, the low frequency with which birds were recorded passing close to the turbine suggests that the data are unlikely to have sufficient power to detect avoidance activity.

APPENDIX 6 EVIDENCE REVIEW WITHIN-WINDFARM AVOIDANCE

A6.1 Avonmouth Docks

The Landmark Practice. 2013. *Birds and Wind Turbines At Avonmouth Docks. Year 5 Monitoring Report for Ecotricity.* The Landmark Practice, Bristol

Methods

Monitoring was undertaken at the Avonmouth Docks windfarm between October and March in the winters of 2007/08, 2008/09, 2009/10, 2011/12. Three vantage point surveys, each lasting three hours, were carried out in each month to record bird activity at the site. Flight altitude was estimated in five bands 0-20 m, 20-40 m, 40-80 m, 80-160 m and >160 m.

During the visits for each vantage point survey, a search with a radius of 60 m around each turbine was carried out for corpses. Additional surveys were carried out following periods of severe weather. In total 343 checks were carried out around the base of each turbine in the post-construction period.

Seasons / time of day

Vantage point surveys were carried out between October and March, and timed so that periods of rising, falling and high tide were covered each month.

Species

Black-headed gull (4.4 birds/hr 2007/08, 7.1 birds/hr 2008/09, 2.9 birds/hr 2009/10, 12.8 birds/hr 2011/12), herring gull (6.8 birds/hr 2007/08, 13 birds/hr 2008/09, 18.8 birds/hr 2009/10, 38.2 birds/hr 2011/12)

Conditions data collected under

Not stated.

Location / habitat

Avonmouth Docks, coastal.

Turbine / array specification

A line of 3, 2 MW Enercon E82 turbines, with a hub height of 79m and a rotor diameter of 83 m.

Results

A single black-headed gull was identified as a probable collision victim in the winter of 2007/08. An average of 4.4 black-headed gulls were recorded passing through the site over the study period, suggesting a total flux rate of 10,530 birds, of which 57 were predicted to collide based on option 1 of the Band model, 2 were predicted to collide based on option 2 of the Band model and 1 was predicted to collide based on option 3 of the Band model. This reflects avoidance rates of 0.9826 using option 1 of the Band model, 0.5152 using option 2 of the Band model and -0.0005 using option 3 of the Band model.

Assessment of methodology

The corpse search methodology is likely to provide an accurate estimate of collision numbers as previous studies have shown that the majority of corpses are recovered within 40 m of a turbine base (Orloff & Flannery 1992, Munster *et al.* 1996, Howell 1997). Furthermore, corpses were examined to confirm collision as cause of death. No corrections were carried out to account for searcher efficiency or predator activity. However, given the habitat surrounding the turbines and the frequency of searches through the study period, it is unlikely corpses would have been missed. Bird activity surveys were carried out throughout the study period and are therefore likely to give a realistic impression of bird activity in the area.

As the bird activity surveys were carried out concurrently with the corpse searches and covered the same area, these data were combined with data from other sites to estimate representative avoidance rates.

A6.2 Altamont Pass

Thelander, C.G., Smallwood, K.S. & Rugge, L. 2003. *Bird risk behaviours and fatalities at the Altamont Pass Wind Resource Area*. National Renewable Energy Laboratory, Colorado.

Methods

Visual observations and fatality searches.

Circular areas with a 50 m radius around the base of 685 wind turbines were searched for corpses every five to six weeks between 1998 and 2000. These searches were combined with 1,958 30 minute point counts carried out in 20 study plots on 303 different days between 1998 and 2000.

Seasons / time of day

Counts carried out throughout the year and between 0700 h and dusk.

Species

Gulls (0.48 birds/hour).

Conditions data collected under

All conditions unless wind or rain resulted in visibility dropping to <60 m.

Location / habitat

Altamont Pass, California, U.S.A. (terrestrial).

Turbine / array specification

685 turbines arranged in 109 rows across an area of 50 km^2. Turbine hub heights ranged from 14 m-30 m, with rotor diameters of 17-23 m. Distance between turbines within each row is not described.

Results

At this site, a total of five gulls, of unknown species, were recovered following collision with turbines. Across the study plots as a whole, the average rate at which gulls passed through the windfarm was 0.48 birds per hour, reflecting a total of 7,428 gull movements within the area over the two year study period. Site specific flight height data were not available, so it was not possible to calculate an avoidance rate based on option 1 of the Band model. Assuming no avoidance behaviour, and a bird with the characteristics of a herring gull, the total number of collisions expected would have been 296 per annum under option 2 of the Band model and 295 under option 3 of the Band model. The collision rate of five birds over the study therefore indicates a within-windfarm avoidance rate of 0.9831 using option 2 and 0.9831 using option 3. The similarity between these values reflects the relatively small size of the turbines installed at the site, in particular the rotor-swept area, diameters of 17-23 m are significantly smaller than many of the turbines installed at offshore sites.

Assessment of methodology

The corpse search methodology is likely to provide an accurate estimate of collision numbers as previous studies have shown that the majority of corpses are recovered within 40 m of a turbine base (Orloff & Flannery 1992, Munster *et al.* 1996, Howell 1997). Furthermore, corpses were examined to confirm collision as cause of death. Correction factors were applied to account for carcass removal by scavengers, but not to correct for searcher efficiency. However, the limited size of the search area and terrain made it unlikely that any corpses would have been undetected.

To minimise the effects of observer bias in point counts, paired observations were carried out during the early part of the study period so that different observers calibrated their perceptions of altitude, distance and behaviour with one another. However, no correction was applied for the detection distance of different species. This is a concern given that study plots were up to 4 km^2, meaning that the total number of birds present within the study areas may have been an underestimate and that, therefore, the final, derived avoidance rate would also have been an underestimate.

However, as it has been necessary to extrapolate bird activity data across the site, this has not been combined with data from other sites to identify representative avoidance rates.

A6.3 Blyth Harbour

Visual observations and fatality searches.

Lawrence, E.S., Painter, S. & Little, B. 2007. Responses of birds to the wind farm at Blyth Harbour, Northumberland, UK. In de Lucas, M., Janss, G.F.E., Ferrer, M. 2007. *Birds and Wind farms Risk Assessment and Mitigation*, Quercus, Madrid.

Newton, I. & Little, B. 2009. Assessment of wind farm and other bird casualties from carcasses found on a Northumbrian beach over an 11-year period. *Bird Study*, **56,** 158-167.

Methods

Once a week over an 11 year period, a 4.7 km stretch of beach near Blyth in Northumberland was searched for corpses. Depending on the condition of the birds, an attempt was made to assign a cause of death to each carcass, and those with symptoms thought to be typical of collision with a wind turbine – head or one or both wings missing, broken bones blood in body cavity and a ruptured liver – were identified. The total number of carcasses found was then corrected to account for those lost to scavengers, those not washed up on the beach and those not found during searches.

Between October 1996 and August 1998, 31 three hour-long periods of observation were made of flight activity perpendicular to the turbine row and in the vicinity of five of the nine turbines. Observations were made from a point on the shore opposite the turbines, at a distance of approximately 80 m. In total 93 hours of observational data were collected.

Seasons / time of day

Fatality data were collected throughout the year. Bird activity data were also collected throughout the year, between the hours of 0800 and 1500 h, with observation periods split equally between the morning and afternoon.

Species

Around 80% of the flight activity within the windfarm involved herring gull and great black-backed gull, and other gull species made up a significant proportion of the remaining species. However, as species-specific data were not available regarding the corpses collected and it was stated that the majority of those collected belonged to gulls, to calculate an avoidance rate, it was necessary to consider gulls collectively.

Conditions data collected under

No details given.

Location / habitat

Blyth Harbour breakwater, Northumberland, UK (coastal).

Turbine / array specification

Nine turbines arranged in a row along a harbour breakwater. The turbines are spaced at 200 m intervals and have a hub height of 25 m with a 25 m rotor diameter.

Results

Results were presented as average collision rates and passage rates over the study period as a whole. Based on the data presented an average of 417,954 birds, most of which were large gulls, would have been expected to pass through the windfarm over the study period. Of these, approximately 3,047, assuming birds with the characteristics of a herring gull, would have been expected to collide with turbines in the absence of avoidance behaviour using option 1 of the Band Model and 3,083 using option 2 and 3,007 using option 3. Having corrected for the imperfect detection of corpses, between 148.5 and 193.5 collisions with wind turbines were expected in an average year. This suggests a within-windfarm avoidance rate of 0.3966-0.5369 using option 1, 0.4037-0.5423 using option 2 and 0.3886-0.5308 using option 3.

Assessment of methodology

The fatality searches were intensive throughout the study period and followed a robust methodology to account for corpses that went undetected. In particular, the potential for corpses to wash up within the study area was tested experimentally.

The observational data were limited to a two year period in the middle of the study. The data may have underestimated gull movements within the surrounding area for two key reasons. Firstly, no corrections were applied to account for imperfect detection of birds. Secondly, by limiting observations to the period between 0800 and 1500 h, key movements of gulls to and from roost sites may have been missed during the summer and autumn. Underestimating bird activity within the area would lead to an underestimate of the number of collisions expected in the absence of avoidance behaviour, and consequently, the final derived avoidance rates would also be underestimated.

Activity data were only collected between 206 and 2008 and only between turbines 5 and 9. As the mean annual collision rates relate to the whole of the study period, and to all 9 turbines, it is necessary to extrapolate activity data both temporally and spatially to derive a flux rate. Therefore, these data have not been included when deriving representative avoidance rates.

A4.4 Blyth Offshore Windfarm

Rothery, P., Newton, I. & Little, B. 2009. Observations of seabirds at offshore wind turbines near Blyth in northeast England. *Bird Study*, 56, 1-14

Methods

Visual observations

Following the installation of the offshore turbines, observations of birds in the vicinity of the turbines were made on 177 occasions between 12 January and 30 August

2003, totalling almost 352 hours of observation. Observations were made from the shore and distances and heights of flying birds were calibrated against objects of known size and fixed locations.

Seasons / time of day

Observations were made between January and August. Data collection was focussed on the period between 1130 and 1600, consequently, during the summer movements to and from breeding colonies may have been missed.

Species

Northern gannet (1.23 birds/hr), great cormorant (1 bird/hr), common scoter (0.96 birds/hr), common eider (2.77 birds/hr), black-headed gull (2.78 birds/hr), herring gull (4 birds/hr), great black-backed gull (1.6 birds/hr), black-legged kittiwake (3.83 birds/hr), Sandwich tern (6.07 birds/hr).

Conditions data collected under

No details given.

Location / habitat

Blyth, Northumberland, UK (offshore).

Turbine / array specification

Two turbines separated by 200 metres. Each turbine had a hub height of 59.4 m above mean sea-level and a rotor diameter of 66 m.

Results

Throughout the study period, no collisions were recorded involving any of the species observed in the vicinity of the windfarm, reflecting a within-windfarm avoidance rate of 1.0000 for each species considered (Northern gannet, great cormorant, common scoter, common eider black-headed gull herring gull, great black-backed gull, black-legged kittiwake and Sandwich tern).

Assessment of methodology

No corrections were applied to account for the imperfect detection of birds during the survey. Consequently, the true level of bird activity within the study area was likely to have been underestimated. Additionally, it was not possible to search for carcasses, meaning that inferences about avoidance behaviour can only be drawn from the failure of observers to detect a collision from a total of 352 hours of monitoring. Given the low probability of a collision occurring, and the levels of flight activity recorded, this outcome is unsurprising. It is also important to note that the size of the OWF was very small (two turbines) and therefore caution must be applied when considering how applicable these avoidance rates are for much bigger arrays.

As insufficient observational data have been collected to record a collision, these data have not been included in those used to derive representative avoidance rates.

A6.5 Boudwijnkanaal

Everaert, J. 2008. *Effecten van windturbines op de fauna in Vlaanderen: Onderzoeksresultaten, discussie en aanbevelingen*. INBO, Brussels

Everaert, J. & Kuikjen, E. 2007. *Wind turbines and birds in Flanders (Belgium): Preliminary summary of the mortality research results.* INBO, Brussels

Everaert, J., Devos, K. & Kuijken, E. 2002. *Windturbines en vogels in Vlaanderen: Voorlopige onderzoeksresultaten en buitenlandse beviningen*. INBO, Brussels

Methods

Visual observations and fatality searches.

Systematic fatality searches were carried out once every 14 days between 2001 and 2006. Searches were carried out within a circular area, with a radius of 100 m, centred on each turbine. Corrections were applied to the data to account for imperfect detection and searcher efficiency.

Observational data describing the number of birds passing the turbine hub were collected between September and December 2005 between turbines 8 and 14. The resultant data were used to extrapolate the total number of birds likely to have passed the turbines over this period. Observational data are presented as a mean daily total collected during the period from two hours before dawn to four hours after dusk in October, reflecting a total of 17 hours of observations.

Seasons / time of day

Fatality data were collected throughout the year, behavioural data were collected between September and December.

Species

Gulls (1,075 birds/day).

Conditions data collected under

No details given.

Location / habitat

Boudwijnkanaal, Brugge, Belgium (terrestrial).

Turbine / array specification

A row of 14 turbines, each with a hub height of 55 m and a rotor diameter of 48 m. Distance between turbines within each row is not described.

Results

Collisions involving gulls were recorded in each year of the study, with a minimum of 21.2 collisions occurring in 2001 when only five of the 14 turbines were operational and a maximum of 264.6 collisions occurring in 2003, when all 13 turbines were operational. Behavioural data were only collected between September and December 2005 from between turbines 8 and 14. Extrapolating from these data to estimate the total number of collisions expected in each year in the absence of any avoidance action gives predictions of 550 collisions in 2001 using option 1 of the Band model, 252 using option 2 and 227 using option 3, and 3,262 collisions in each year between 2002 and 2006 using option 1, 1,497 using option 2 and 1,348 using option 3. Based on these analyses, within-windfarm avoidance rates would have been 0.9615 in 2001, 0.9299 in 2002, 0.9189 in 2003, 0.9284 in 2004, 0.9287 in 2005 and 0.9338 in 2006 using option 1. Using option 2, meso-micro avoidance rates would have been 0.9160, 0.8472, 0.8232, 0.8440, 0.8446 and 0.6990. Using option 3, meso-micro-avoidance rates would have been 0.9067, 0.8302, 0.8037, 0.8268, 0.8273 and 0.6656 respectively.

However, bird activity was only recorded around turbines 8 and 14 in October 2001 and October 2005. If we consider collisions recorded around these turbines in each of these time periods, the predicted number of collisions is 103 herring gulls in October 2001 and 145 black-headed gulls, 90 herring gulls and 260 birds in total during October 2005. The actual number of collisions recorded was 1, 6, 4 and 11 respectively, reflecting avoidance rates of 0.9903, 0.9586, 0.9556 and 0.9577 using option 1 of the Band model, 0.9789, 0.3658, 0.7865 and 0.8077 using option 2 of the Band model and 0.9765, 0.1886, 0.7629 and 0.7865 using option 3 of the Band model.

Assessment of methodology

Fatality data have been collected on a regular basis and following a robust methodology. Corrections have been applied to these data to account for the imperfect detection of corpses due to scavenger behaviour and searcher efficiency.

The observational data that have been collected are extremely limited. Data collection has been restricted to the September to December period in a single year. It is unclear how accurately this reflects bird movements within the windfarm over the rest of the study period. This may have a significant, but unquantifiable impact on the final, derived within-windfarm avoidance rates. In addition, it is unclear whether corrections have been applied to the observational data to account for the imperfect detection of birds.

Using the overall data, it is necessary to make both spatial and temporal extrapolations to estimate the avoidance rates. For this reason, we only use the data collected around turbines 8-14 in October 2001 and 2005 to derive representative avoidance rates.

A6.6 Bouin

Dulac, P. 2008. *Evaluation de l'impact du parc eolian de Bouin (Vendee) sur l'avifaune et les chauves-souris. Bilan de 5 anness de suivi*. Ligue pour la Protection des Oiseaux, Nantes.

Methods

Visual observations and fatality searches.

Weekly searches were carried out for corpses at the foot of turbines between 2002 and 2006. Searches were restricted to a 100 m^2 box centred on each turbine. To aid searching, each box was divided into a grid with squares of 25 m^2.

Observational data were collected from four points, covering 1 km each. Each month a two hour count was made from each point, with a total of 474 hours of observational data collected from the site as a whole between 2002 and 2006.

Seasons / time of day

Data were collected throughout the year and protocols were designed so that full day was covered.

Species

Black-headed gull (16.23 birds/hr), herring gull (2.26 birds/hr), other gulls (2.09 birds/hr).

Conditions data collected under

The observational protocol was designed to collect data throughout the tidal cycle and in all weather conditions.

Location / habitat

Bouin, Baie de Bourgneuf, France (Coastal)

Turbine / array specification

A single row of eight turbines, each with a hub height of 60 m and a diameter of 80 m. Distance between turbines within each row is not described.

Results

At this site, 30 gulls were recovered from turbine bases over the course of a four year study period. Of these, 28 were black-headed gulls, one was a yellow-legged gull and one was a Mediterranean gull. Using option 1 of the Band model, 584 black-headed gulls and 206 'other' gulls were predicted to collide with the turbines, reflecting avoidance rates of 0.9520 and 0.9903 respectively. For option 2, 483 and 354 birds were predicted to collide respectively, reflecting avoidance rates of 0.9421 and 0.9943. For option 3, the corresponding figures were 237 and 251 birds predicted to collide reflecting avoidance rates of 0.8820 and 0.9920. No collisions were recorded for herring gulls, despite a predicted collision rate of 216 per annum, reflecting a within-windfarm avoidance rate of 1 for options 1, 2 and 3 of the Band model.

Assessment of methodology

Fatality data were collected following a robust protocol, with corrections applied to account for birds lost to scavengers and search efficiency. The intensive nature of these searches, weekly over a four year period, is likely to mean that fatality rates were estimated with a high degree of accuracy.

Observational data were collected over a four year period. However, no corrections were applied to account for imperfect detection. Consequently, bird activity in the area and the derived within-windfarm avoidance rates were likely to have been underestimated.

As activity data were a spatial and temporal match for the period over which collision data were collected, these data were included when estimating representative avoidance rates.

A6.7 Buffalo Ridge

Johnson, G.D., Erickson, W.P., Strickland, M.D., Shepherd, M.F., Shepherd, D.A. 2000. *Avian Monitoring Studies at The Buffalo Ridge, Minnesota Wind Resource Area: Results of a 4-year study.* Western EcoSystems Technology Inc., Wyoming.

Methods

Visual observations and fatality searches.

Fatality searches were carried out within 126 m x 126 m plots, centred on 61 turbines. Searches were carried out every two weeks and observers covered the area by walking parallel transects separated by a distance of 6 m. This was combined with a series of large bird counts carried out every two weeks for a 0.8 km radius surrounding each of six observation stations. During each survey, two 30 minute observations were made, one in the morning and one in the afternoon. In total 70 hours of survey data were collected over the course of the study period.

Seasons / time of day

Fatality searches were carried out throughout the year. Large bird counts were carried out between 0800 and 1600 h and restricted to the period from 15 March to 15 November.

Species

Herring gull (0.1 birds/hour).

Conditions data collected under

No details given.

Location / habitat

Buffalo Ridge, Minnesota, U.S.A. (terrestrial).

Turbine / array specification

143 turbines arranged in 26 rows with between 100 m and 200 m between each turbine. Each 750 kW turbine had a hub height of 50 m and a diameter of 48 m.

Results

At this site, one herring gull was recovered following collision with turbine. Across the study plots as a whole, the average rate at which herring gulls passed through the windfarm was 0.03 birds per hour, reflecting a total of 625 gull movements within the area over the two year study period. Assuming no avoidance behaviour, the total number of collisions expected would have been 3 using option 1 of the Band model, 5 under option 2 of the Band model and 5 under option 3 of the Band model. The collision rate of 1 bird over the study therefore indicates a within-windfarm avoidance rate of 0.6503 using option 1, 0.8149 using option 2 and 0.7923 using option 3.

Assessment of methodology

The methodology was generally sound with a well-structured search likely to detect all corpses within the study area. Corrections were made for both corpses removed by scavengers and also searcher efficiency. The large bird survey also followed a sound methodology, with corrections applied to account for imperfect detection. However, as observations were limited to 0800 to 1600 h and November to March, it is possible that they failed to detect daily or seasonally important gull movements. This may reflect the fact that raptors were the primary concern at this site.

As it was necessary to extrapolate bird activity data spatially to estimate an avoidance rate, these data have not be included when deriving representative avoidance rates.

A6.8 De Put

Everaert, J. 2008. *Effecten van windturbines op de fauna in Vlaanderen: Onderzoeksresultaten, discussie en aanbevelingen*. INBO, Brussels

Everaert, J. & Kuikjen, E. 2007. *Wind turbines and birds in Flanders (Belgium): Preliminary summary of the mortality research results.*

Methods

Visual observations and fatality searches.

Systematic fatality searches were carried out once every 14 days between April 2005 and March 2006. Searches were carried within a circular area, with a radius of 100 m, centred on each turbine. No correction factors were used to account for scavengers or imperfect searcher efficiency.

Observational data describing the number of birds passing within 100 m of the turbine hub were collected between January and February 2006, the period in which the corpses were recovered. The resultant data were used to estimate the total number of birds likely to have passed the turbines over this period. In total 18 hours of survey data were collected.

Seasons / time of day

Observational data were collected throughout the day during January and February 2006.

Species

Black-headed gull and common gull (3,186 during the study period).

Conditions data collected under

No details given.

Location / habitat

De Put, Nieuwkapelle, Belgium (terrestrial).

Turbine / array specification

A row of two turbines, each with a hub height of 75 m and a rotor diameter of 100 m.

Results

In January and February 2006, the corpses of two gulls, one common gull and one black-headed gull, were recovered. Based on the number of birds estimated to have passed through the windfarm during the study period, the combined number of collisions predicted in these two species would be 19 using option 1 and none using options 2 and 3. The two recorded collisions therefore reflect a micro-meso avoidance rate of 0.8928 for common and black-headed gulls using option 1, -9.1051 using option 2 and -11.8383 using option 3.

Assessment of methodology

Whilst fatality searches appear to have been relatively robust and intensive throughout the study period, no corrections were applied to account for the imperfect detection of corpses, either through searcher inefficiency or through loss to scavengers. This may have led to an underestimate of the total number of collision victims.

Details of the methodology used to collect observational data of bird behaviour within the windfarm were sparse. In particular, no details were given of the length of observations used to collect data during the study. There also appears to have been no attempt to account for the imperfect detection of birds, meaning the total number passing through the study area may have been an underestimate. This, in turn would also mean that the final within-windfarm avoidance rate had been underestimated.

As bird activity and collision data have been collected concurrently, these data have been included when deriving representative avoidance rates.

A6.9 Gneizdzewo

Zielinski, P., Bela, G. & Marchlewski, A. 2008. *Report on monitoring of the wind farm impact on birds in the vicinity of Gniezdzewo (gmina Puck, pomorskie voivodeship)*

Zielinski, P., Bela, G. & Marchlewski, A. 2010. *Report on monitoring of the wind farm impact on birds in the vicinity of Gniezdzewo (gmina Puck, pomorskie voivodeship)*

Zielinski, P., Bela, G. & Marchlewski, A. 2011. *Report on monitoring of the wind farm near Gniezdzewo impact on birds (gmina Puck, pomorskie voivodeship)*

Zielinski, P., Bela, G. & Marchlewski, A. 2012. *Report on monitoring of the wind farm near Gniezdzewo impact on birds (gmina Puck, pomorskie voivodeship)*

Methods

Visual observations and fatality searches.

Collision surveys were carried out in the autumns of 2008, 2010, 2011 and 2012 (September-November). Corpse searches were carried out within 70 m radius of each turbine, on average every 2-3 days.

Over the same periods each year (mid-September – mid-November), activity surveys were carried out within the windfarm. Between 60 and 70 hours of observational data were collected each year, with observation sessions lasting up to 6 hours.

Seasons / time of day

Data were collected throughout the day during the autumn migration period in each year.

Species

Great cormorant (0.17-1.44 birds/hr), gulls (3.88-44.14 birds/hr), little gull (0.23 birds/hr), common gull (0.57 -1.73 birds/hr), black-headed gull (0.51-4.94 birds/hr), herring gull (1.06-5.39 birds/hr).

Conditions data collected under

All conditions

Location / habitat

Gniezdzewo, Poland (terrestrial).

Turbine / array specification

An array of 19 turbines arranged in four rows. Each turbine had a rotor diameter of 80 m and a hub height of 80 m.

Results

In the four autumns over which data have been collected, only a single collision involving a gull was recorded, a black-headed gull during the 2010 field season. No site specific flight height data were available, so it was necessary to use the distributions presented in Johnston *et al.* (2014a) and option 2 of the Band Model to estimate avoidance rates. In the 2010 field season, 460 black-headed gulls were predicted to have passed through the windfarm, with a predicted collision rate of 0.2 birds. The avoidance rate for black-headed gulls during autumn 2010 would, therefore, have been –3.9524, suggesting that a significant number of birds were attracted to the rotor swept area of the turbine. Using option 3 of the Band model, the collision rate was predicted to be 0.1 birds, reflecting a within-windfarm avoidance rate of -8.9238. However, it should be noted that this collision rate is based on a relatively low number of birds passing through the windfarm and as a result may be unreliable. The unusual nature of this result is confirmed as in three additional years of monitoring, no black-headed gull collisions were recorded, despite often higher levels of flight activity. The avoidance rate for cormorants and all other gull species in all years would have been 100%.

Assessment of methodology

The search for collision victims has been robust, with specially trained dogs used to increase detection. However, no corrections have been applied to account for birds lost to scavengers, potentially meaning the collision rates have been under-estimated.

No correction has been applied to the activity surveys to account for the imperfect detection of birds. As a consequence, the total number of birds passing through the area, and therefore potentially the final avoidance rates, may be under-estimated.

As collision and activity data were collected concurrently, from the windfarm as a whole, throughout the study period, they have been included when deriving representative avoidance rates.

A6.10 Greater Gabbard

RPS. 2011. *Galloper Wind farm Project Environmental Statement – Technical Appendices 2: Appendix 4: Greater Gabbard post-construction vantage point surveys*, RPS, Glasgow

Methods

Visual observations

Two surveyors collected data from 180˚ arcs to the port and starboard sides of a stationary vessel within Greater Gabbard Offshore Windfarm. Each arc had a radius of 2 km and all birds entering each arc were recorded during snapshot counts taken every 15 seconds. The location of the boat and the viewing area, which covered a total of 15.9 km^2, included seven operational turbines and a total of 36 hours of data were collected during the survey. The flight paths of each bird within the viewing area were noted, as was the proportion of time each bird spent at different heights.

Seasons / time of day

Data were collected between 1st June 2011 and 28th July 2011, with each survey lasting four hours.

Species

Northern gannet (0.14 birds/hr), Arctic skua (0.03 birds/hr), lesser black-backed gull (3.69 birds/hr), herring gull (0.11 birds/hr), black-legged kittiwake (1.28 birds/hr).

Conditions data collected under

Conditions were limited to sea-states one and two, to ensure the vessel remained as a stable observation platform.

Location / habitat

Greater Gabbard, UK (offshore).

Turbine / array specification

The survey monitored seven operational turbines, each with a hub height of 77.5 m and a rotor diameter of 107 m.

Results

The predicted number of collisions, in the absence of avoidance behaviour, within the 36 hour study period would have been less than 1 bird from each species. However, no collisions were recorded reflecting an avoidance rate of 1.000 for all species over the course of the study period.

Assessment of methodology

No corrections were applied to account for the imperfect detection of birds during the survey. Consequently, the true level of bird activity within the study area is likely to have been underestimated. Additionally, it was not possible to search for carcasses, meaning inferences about avoidance behaviour can only be drawn from the failure of observers to detect a collision with 36 hours of monitoring. Given the low probability of a collision occurring, and the levels of flight activity recorded, this outcome is unsurprising.

Given the limited data collection during the study period, these data have not been included when deriving representative avoidance rates.

A6.11 Groettocht

Krijgsveld, K.L., Akershoek, K., Schenk, F., Dijk, F. & Dirksen, S. 2009. Collision risk of birds with modern large wind turbines. *Ardea* **97:** 357-366.

Methods

Radar observations and fatality searches.

Fatality searches were carried out within a 100 m radius around each turbine every 2-3 days. Searches were carried out by walking parallel transects, each separated by 4-6 m. Searches were carried out between October and December 2004.

Flight movements were quantified using a 12 kW x-band marine surveillance radar overnight between 1800 and 0700 h on 20 October 2004, 22 November 2004 and 22 December 2004, and the number of radar echoes up to 140 m (the maximum turbine height) were estimated as a measure of flux through the windfarm area.

Seasons / time of day

Resultant data reflect overnight collision rates of birds between October and December 2004.

Species

Key movements recorded included gulls travelling between Lake Ijsselmeer and a nearby roost site around dusk and dawn. However, amongst the five corpses encountered, there were only two gull carcasses, a common gull and a herring gull.

Conditions data collected under

Not specified.

Location / habitat

Agricultural area in the Netherlands.

Turbine / array specification

The array consists of a single line of seven turbines, each separated by 285 m. Turbines have a hub height of 78 m and a rotor diameter of 66 m. However, only the areas under five turbines were searched for carcasses.

Results

The average flux of birds through the area was 370 birds/km/hr, reflecting a movement of 873,534 birds through the study period as a whole. Site specific flight height data were not available for the site, so it was necessary to use the distributions presented in Johnston *et al.* (2014a) to estimate the proportion of birds at collision risk height, and option 2 of the Band model to estimate predicted collision numbers. In total, the remains of five birds (one herring gull, one common gull, one redwing, two unidentified species) were retrieved. Given that it is not possible to relate the radar tracks to individual species, we calculated the probability of collision based on a bird with the characteristics of first a herring gull, giving a predicted collision rate of 2131 birds over the study period, and an overall avoidance rate of 0.9991 based on option 2 and a collision rate of 1648 birds over the course of the study, with an avoidance rate of 0.9988 using option 3.

Assessment of methodology

The total collision rate may be an underestimate as the initial searching rate of once every three days was lowered to once every two days following the outcome of depredation tests. However, all corpses present were likely to be discovered as only turbines where the surrounding vegetation was low were searched for remains. With the exception of concerns over the depredation rate, the fatality searches were robust.

Flux rates were estimated using x-band radar, with the considerable disadvantage that it cannot be used to estimate the flux rates of different species. As a consequence, using individual species collision rates to estimate an avoidance rate may have led to an inaccurate estimate of the true value. In addition, as a single radar echo may represent multiple birds, there was a considerable risk that the true movement of birds through the area was underestimated and that, therefore, the overall avoidance rate was also underestimated.

As it was necessary to extrapolate activity data both spatially and temporally to estimate the avoidance rates, these data have not been used to derive representative avoidance rates.

A6.12 Haverigg

RPS. 2011. *Galloper Wind farm Project Environmental Statement – Technical Appendices 2: Appendix 3: Information on gull flight behaviour at operational wind farms and the estimation of avoidance rates for use in the Band Collision Risk Model*, RPS, Glasgow.

Methods

Visual observations

In July and August 42 hours of vantage point surveys were carried out at Haverigg Windfarm following the standard SNH vantage point methodology (SNH 2010).

Seasons / time of day

Surveys were carried out in July and August.

Species

Gulls (19.90 birds/hr).

Conditions data collected under

Not stated.

Location / habitat

Haverigg Windfarm, Cumbria, UK (terrestrial).

Turbine / array specification

Haverigg Windfarm consists of two groups of four turbines. The first four turbines have a hub height of 45 m and a rotor diameter of 42 m, whilst the remaining four, larger, turbines have a rotor diameter of 52 m.

Results

During 42 hours of vantage point observations, a total of 836 gulls, mostly herring and lesser black-backed gulls were recorded entering the windfarm at a rate of 19.90 birds/hr. However, during the observation periods, no collisions were recorded, reflecting an avoidance rate of 1 over the course of the study period under options 1,2 and 3 of the Band model.

Assessment of methodology

The evidence provided by the survey is limited as no corpse searches were carried out in the area surrounding the windfarm. Whilst 42 hours of survey effort were carried out, no collisions were recorded. However, given the likely rarity of collisions occurring, this is unsurprising. Furthermore, the levels of flight activity within the windfarm are likely to have been underestimated as no correction was made for the imperfect detection of birds.

As insufficient monitoring data have been collected to observe collisions, these data have not been included when deriving representative avoidance rates.

A6.13 Hellrigg

Percival, S. 2012. *Hellrigg Wind farm: Goose Refuge Monitoring Report Winter 2011-12*, Ecology Consulting, Durham

Percival, S. 2013. *Hellrigg Wind farm: Goose Refuge Monitoring Report Winter 2012-13*, Ecology Consulting, Durham

Methods

An area covered by a 100 m radius around the base of each turbine was searched between December and March in the winters of 2011/12 and 2012/13 on a weekly basis. Searches were carried out slowly and carefully, with particular care taken over areas containing large clumps of vegetation. The locations of each corpse were carefully noted, and each was left in place to provide information about decay rates and detectability.

Bird activity data were collected through vantage point surveys from a single point following standard SNH guidance. The flight lines of each species were noted and flight altitudes estimated. In total 38 hours of flight observations were collected in this way each winter.

Seasons / time of day

Data were collected between December and March each year, with effort made to cover dawn and dusk movements of birds as well as general daytime movements of birds.

Species

Common gull (8.47 birds/hr in 2011/12 and 507.17 birds/hr in 2012/13), lesser black-backed gull (0.3 birds/hr in 2011/12 and 0.41 birds/hr in 2012/13), herring gull (3.71 birds/hr in 2011/12 and 72.49 birds/hr in 2012/13), great black-backed gull (0.05 birds/hr in 2011/12 and 0.49 birds/hr in 2012/13), black-headed gull (4.79 birds/hr in 2011/12 and 131.48 birds/hr in 2012/13)

Conditions data collected under

Not stated.

Location / habitat

Hellrigg windfarm, onshore.

Turbine / array specification

An array of four turbines with a hub height of 80 m and a rotor diameter of 82 m.

Results

A single collision involving a herring gull was recorded in 2011/12. Based on the passage rate of 3.71 birds/hr, 13 collisions would have been expected in the absence of avoidance behaviour based on option 1 of the Band model, 3 collisions based on option 2 of the Band model and 2 collisions based on option 3 of the Band model. This reflects avoidance rates of 0.9209, 0.6635 and 0.5133 respectively.

Assessment of methodology

Analysis of the length of time corpses remained at the site, suggested that the mean time to disappearance was 22 days, well in excess of the 7 day search intervals. In combination with the systematic and methodical searches carried out at the site, this suggests it is unlikely any corpses went undetected.

Bird activity data were collected following standard SNH vantage point methodology. However, as no correction was made for imperfect detection, the levels of flight activity at the site and, therefore, the overall avoidance rates, may have been underestimated.

As mortality and activity data were collected concurrently at the site, following robust methodologies, these data were used when deriving representative avoidance rates.

A6.14 Keewaunee County

Howe, R.W., Evans, W. & Wolf, A.T. 2002. *Effects of Wind Turbines on Birds and Bats in Northeastern Wisconsin*. Report to Wisconsin Public Service Corporation and Madison Gas and Electric Company.

Methods

Visual observations and fatality searches.

Intensive searches were carried out between July 1999 and July 2001. Searches were carried out at least once a week. Surveyors visited a 60 m x 60 m area centred on each of the turbines and covered a series of nine 60 m transects in each. These searches were complemented by a series of 3,214 3 minute short counts carried out on 160 dates between 1998 and 2001, to estimate the number of birds within the area.

Seasons / time of day

Surveys were carried out between June and November, with a bias towards data collection during the morning.

Species

Herring gull (0.012 birds/hour), Franklin’s gull (0.019 birds/hour), ring-billed gull (1.589 birds/hour).

Conditions data collected under

No details given.

Location / habitat

Keewaunee County, Wisconsin, U.S.A. (terrestrial).

Turbine / array specification

31 turbines with a hub height of 65 m and a rotor diameter of 47 m, within three clusters of 8, 9 and 14 turbines. Distance between turbines within each row is not described.

Results

At this site, one herring gull was recovered following collision with turbine. Across the study region as a whole, the average rate at which herring gulls passed through the area was 0.012 birds per hour, reflecting a total of 131 gull movements within the area over the two year study period. No site specific flight height data were available, meaning it was necessary to use the flight height distributions presented in Johnston *et al.* (2014a) and option 2 of the Band model. Assuming no avoidance behaviour, no collisions would have been expected under options 2 or 3 of the Band model. The collision rate of 1 bird over the study therefore indicates a within-windfarm avoidance rate of -12.0935 using option 2 and -13.5238 using option 3.

Assessment of methodology

The methodology was generally sound with a well-structured search likely to detect all corpses within the study area. Corrections were made for both corpses removed by scavengers and also searcher efficiency. However, no corrections were made to account for imperfect detectability during the bird surveys.

As it was necessary to extrapolate bird activity data spatially to estimate avoidance rates, these data have not been used to derive representative avoidance rates.

A6.15 Kessingland Windfarm

Wild Frontier Ecology. 2013. *Kessingland Wind farm Annual Post-construction Monitoring Report Year 2.* Wild Frontier Ecology, Norfolk.

Methods

Visual observations and fatality searches.

Fatality searches were undertaken around the bases of each turbine on nine occasions between November 2012 and March 2013. Surveyors walked a series of transects, separated by 10 m, within 65 m of the turbine base to search for corpses. A corpse correction factor of 1.79 was applied to account for corpses removed by scavengers.

Bird activity was monitored within the windfarm through nine two-hour vantage point surveys at each turbine carried out between November 2012 and March 2013. In total 36 hours of survey effort was completed throughout the study period.

Seasons / time of day

Data collection was carried out over winter 2012/13, between November and March. Surveys were carried out for two hour periods between 0800 and 1500 h.

Species

Black-headed gull (48.5 birds/hr), common gull (15.69 birds/hr), lesser black-backed gull (5.5 birds/hr), herring gull (28.36 birds/hr), great black-backed gull (0.14 birds/hr).

Conditions data collected under

No details given.

Location / habitat

Kessingland, Suffolk, UK (terrestrial).

Turbine / array specification

Two turbines with hub heights of 80 m and rotor diameters of 92 m. Distance between turbines within each row is not described.

Results

Black-headed, common, lesser black-backed, herring and great back-backed gulls were recorded within the study area at varying frequencies. Three gulls were found to have collided with the turbines – one black-headed gull, one common gull and one herring gull. After applying corpse correction factors, these estimates were revised to 1.79 birds of each species. No site specific flight height data were available, so it was necessary to use the modelled flight height distributions presented in Johnston *et al.* (2014a) and option 2 of the Band model. Given the number of birds likely to have passed through the windfarm during the study period, the predicted collision numbers would have been 28, 21 and 76 respectively. Using option 2, the avoidance rate for black-headed gull would therefore be 0.9367, for common gull it would be 0.9147 and for herring gull it would be 0.9764. Using option 3, the expected collision rates were 13, 12 and 51 respectively, reflecting avoidance rates of 0.8664, 0.8505 and 0.9647. No collisions were recorded involving lesser or great black-backed gulls, reflecting avoidance rates of 1.000 for these species.

Assessment of methodology

The fatality searches appear to have been robust, with corpse correction factors applied to account for loss of corpses to scavengers. However, during vantage point surveys, no corrections were applied to account for imperfect detection. As a result, bird activity within the area was likely to be underestimated, and therefore, the final, derived avoidance rates were also likely to be underestimated.

As collision and bird activity data were collected concurrently over the same area, these data were included when deriving representative avoidance rates.

A6.16 Kleine Pathoweg

Everaert, J. 2008. *Effecten van windturbines op de fauna in Vlaanderen: Onderzoeksresultaten, discussie en aanbevelingen*. INBO, Brussels

Everaert, J. & Kuikjen, E. 2007. *Wind turbines and birds in Flanders (Belgium): Preliminary summary of the mortality research results.*

Methods

Throughout 2005 and 2006, an area covered by a 100 m radius around the base of each turbine was searched for collision victims once every 2 weeks. Correction factors were applied to the resultant data to account for searcher efficiency and the removal of corpses by scavengers.

Between September and December 2005, bird activity data were collected between turbines 3 and 7. Data were collected from 2 hours before sunrise to 4 hours after sunset and presented as an average number of birds/day – reflecting an average of 16 hours of survey effort over this period.

Seasons / time of day

Bird activity data were collected between September and December, from 2 hours before sunrise to 4 hours after sunset.

Species

Black-headed gulls (345 birds/day), ‘large’ gulls (327 birds/day).

Conditions data collected under

Not stated.

Location / habitat

Kleine Pathoweg (Belgium), terrestrial.

Turbine / array specification

A line of 7 turbines, each separated by 280 m. Turbines had a hub height of 85 m and a rotor diameter of 70 m.

Results

In 2005, 240.9 gulls were believed to have collided with turbines once corrections had accounted for imperfect corpse detection. In 2006, this figure was 220.3. Based on a passage rate of 42 birds per hour, in 2005 these figures reflect an avoidance rate of 0.8795 using option 1 of the Band model, -0.2529 using option 2 of the Band model and -0.6887 using option 3 of the Band model. In 2006, these figures reflect an avoidance rate of 0.8898 using option 1 of the Band model, -0.1458 using option 2 of the Band model and -0.5443 using option 3 of the Band model.

Assessment of methodology

Fatality data have been collected on a regular basis and following a robust methodology. Corrections have been applied to these data to account for the imperfect detection of corpses due to scavenger behaviour and searcher efficiency.

The observational data that have been collected are extremely limited. Data collection has been restricted to the September to December period in a single year. It is unclear how accurately this reflects bird movements within the windfarm over the rest of the study period. This may have a significant, but unquantifiable impact on the final, derived within-windfarm avoidance rates. In addition, it is unclear whether corrections have been applied to the observational data to account for the imperfect detection of birds.

As it has been necessary to make spatial and temporal extrapolations to estimate avoidance rates, these data have not been used when deriving representative avoidance rates.

A6.17 Oosterbium

Methods

Visual observations and fatality searches.

Searches were carried out within a 50 m radius of the base of each turbine in autumn 1990 and spring 1991. Searches were carried out on 25 days in the spring and 40 days during autumn. All corpses were assessed in order to determine the cause of death and identify those killed by turbines. Corrections were applied to the data to account for searcher efficiency and scavenger activity.

Bird activity within the windfarm and a surrounding 500 m buffer was assessed during spring 1991 and autumn 1990. These activity levels were used to extrapolate the number of bird-days spent within the windfarm for each species or group of species.

Seasons / time of day

Data covered both the nocturnal and diurnal movements of birds in the spring and autumn.

Species

Gulls (158,600 bird days, autumn 1990; 43,800 bird days, spring 1991).

Conditions data collected under

No details given.

Location / habitat

Oosterbierum, Netherlands (terrestrial)

Turbine / array specification

A cluster of 18 turbines with hub heights of 35 m and a rotor diameter of 30 m, situated within 55 hectares of farmland. Distance between turbines within each row is not described.

Results

Gulls were recorded within the area more commonly during the autumn than the spring. However, the number of collisions was greatest during the spring, when 37 corpses were recovered in comparison to 12 in the autumn. No site specific flight height data were available so it was necessary to use the modelled distributions presented in Johnston *et al.* (2014a) and option 2 of the Band model. During the autumn, the predicted number of collisions in the absence of avoidance was 883 birds. Therefore, the 12 collisions recorded during the autumn reflects a meso-micro avoidance rate of 0.9864. Using option 3, the predicted number of collisions was 846, reflecting a meso-micro avoidance rate of 0.9858. During the spring, the predicted number of collisions in the absence of avoidance was 244 using option 2 and 234 using option 3. Therefore, the 37 collisions recorded during the spring reflects a meso-micro avoidance rates of 0.8483 and 0.8417 respectively.

Assessment of methodology

Fatality searches were carried out intensively throughout the spring and autumn seasons. They followed a robust methodology with corrections made for both searcher efficiency and scavenger activity.

Activity data were collected throughout the period covered by the fatality searches. However, it appears no corrections were made to the data to account for imperfect detection, meaning activity levels in the area may have been underestimated. As a consequence, the number of collisions predicted in the absence of avoidance, and therefore the derived avoidance rate would also have been underestimated.

As activity and mortality data were collected concurrently and no spatial extrapolation was necessary, these data were used when deriving representative avoidance rates.

A6.18 Waterkapptocht

Krijgsveld, K.L., Akershoek, K., Schenk, F., Dijk, F. & Dirksen, S. 2009. Collision risk of birds with modern large wind turbines. *Ardea* **97,** 357-366.

Methods

Radar observations and fatality searches.

Fatality searches were carried out within a 100 m radius around each turbine every 2-3 days. Searches were carried out by walking parallel transects, each separated by 4-6 m. Searches were carried out between October and December 2004.

Flight movements were quantified using a 12 kW x-band marine surveillance radar overnight between 1800 and 0700 h on 18 October 2004, 17 November 2004 and 20 December 2004, and the number of radar echoes up to 140 m (the maximum turbine height) were estimated as a measure of flux through the windfarm area.

Seasons / time of day

Resultant data reflect overnight collision rates of birds between October and December 2004.

Species

Key movements recorded included gulls travelling between Lake Ijsselmeer and a nearby roost site around dusk and dawn. However, amongst the seven corpses encountered, there was only a single gull carcass, that of a black-headed gull.

Conditions data collected under

Not specified.

Location / habitat

Agricultural area in the Netherlands.

Turbine / array specification

The array consists of a single line of eight turbines, each separated by 300 m, with a larger 1 km gap between turbines 4 and 5. Turbines have a hub height of 78 m and a rotor diameter of 66 m. However, only the areas under five turbines were searched for carcasses.

Results

The average flux of birds through the area was 251 birds/km/hr, reflecting a movement of 1,195,011 birds through the study period as a whole. In total, the remains of seven birds (one common pheasant, one oystercatcher, one black-headed gull, one skylark and two goldcrests) were retrieved. No site specific flight height data were available, so it was necessary to use the modelled distributions presented in Johnston *et al.* (2014a) and option 2 of the Band model. Given that it was not possible to relate the radar tracks to individual species, we calculated the probability of collision based on a bird with the characteristics of a black-headed gull, giving a predicted collision rate of 1,446 birds over the study period, and an overall avoidance rate of 0.9952. Using option 3, the predicted number of collisions was 1,118 birds, reflecting an overall avoidance rate of 0.9937.

Assessment of methodology

The total collision rate may have been an underestimate as the initial searching rate of once every three days was lowered to once every two days following the outcome of depredation tests. However, all corpses present were likely to be discovered as only turbines where the surrounding vegetation was low were searched for remains. With the exception of concerns over the depredation rate, the fatality searches were robust.

Flux rates were estimated using x-band radar, with the considerable disadvantage that it cannot be used to estimate the flux rates of different species. As a consequence, using individual species collision rates to estimate an avoidance rate may lead to an inaccurate estimate of the true value. In addition, as a single radar echo may represent multiple birds, there was a considerable risk that the true movement of birds through the area was underestimated and that therefore the overall avoidance rate has also been underestimated.

As it was necessary to make temporal and spatial extrapolations with these data, they were not used to derive representative avoidance rates.

A6.19 Yttre Stengrund/Utgrunden Offshore Windfarm

Petterson, J. 2005. *The impact of Offshore Wind farms on Bird Life in Southern Kalmar Sound, Sweden: A final report based on studies 1999-2003*. Lund University.

Methods

Visual observations

Field data were collected from three observation points located within the Southern Kalmar Sound – Eckelsudde in Oland in the east of the observation area, Olsang in the west of the observation area and Utgrunden Lighthouse in the centre of the Sound of Kalmar. The observation points made it possible to cover the whole of the Sound of Kalmar, including both windfarm sites. The sound was divided into four 5 km zones, each of which was further subdivided into 1-2 km wide zones. The observation point at Olsang covered the first of these 5 km zones, the Utgrunden Lighthouse covered the second and third 5 km zones and the Eckelsudde observation point, the fourth. Observers recorded to the exact minute the location of all flocks of migrating waterbirds they encountered, so that data could be combined into a single dataset at a later date.

Seasons / time of day

Data were collected throughout the spring (22 March to 8 April) and autumn (6 to 28 October) migration periods between 2001 and 2003.

Species

Conditions data collected under

All conditions.

Location / habitat

Southern Kalmar Sound, Sweden (offshore).

Turbine / array specification

Five 2 MW turbines with a hub height of 60 m and a rotor diameter of 72 m at Yttre Stengrund.

Seven 1.5 MW turbines with a hub height of 65 m and a rotor diameter of 70 m at Utgrunden. Distance between turbines within each row is not described.

Results

No collisions were recorded amongst any species during the spring migration periods, reflecting an avoidance rate of 1. No site specific flight height data were available at this site, so it was necessary to use the modelled distributions presented in Johnston *et al.* (2004). A single collision event was recorded involving four common eider during autumn 2003, reflecting an avoidance rate of 0.1861 using option 2 of the Band model and -0.1098 using option 3. No other collisions were recorded amongst other species, again indicating an avoidance rate of 1.

Assessment of methodology

Methodology is sound with careful calibration of estimates of distance between observers and co-ordination of counts to minimise double-counting. However, there was no correction applied to account for imperfect detection, meaning the total number of birds may have been under-estimated.

As insufficient data have been collected to detect a collision amongst any of the priority species, these data have not been used to derive representative avoidance rates.

A6.20 Zeebrugge

Everaert, J. 2008. *Effecten van windturbines op de fauna in Vlaanderen: Onderzoeksresultaten, discussie en aanbevelingen*. INBO, Brussels

Everaert, J. & Kuikjen, E. 2007. *Wind turbines and birds in Flanders (Belgium): Preliminary summary of the mortality research results.* INBO, Brussels

Everaert, J. & Stienen, E.W.M. 2007. Impact of wind turbines on birds in Zeebrugge (Belgium) Significant effect on breeding tern colony due to collisions. *Biodiversity and Conservation*, 16, 3345-3359

Everaert, J., Devos, K. & Kuijken, E. 2002. *Windturbines en vogels in Vlaanderen: Voorlopige onderzoeksresultaten en buitenlandse beviningen*. INBO, Brussels

Methods

Visual observations and fatality searches.

Between 2001 and 2007 systematic fatality searches were carried out within a 50 m radius around the base of turbines on a fortnightly basis, increasing to 3-4 times a week during the breeding season. Every turbine was searched, and corrections were made to account for searcher efficiency and scavenger activity.

An initial set of bird activity surveys were carried out at the site in 2000 and 2001. Bird activity within a 400 m section of the breakwater was monitored on four days between June and July in 2000 and 2001, with eight days data collected in total. An additional four days of monitoring were carried out on four days and two nights between September and October 2001.

In June 2004 and 2005, a second set of bird activity were carried out. In each year, two full days of monitoring data were collected covering the period from dawn to dusk. During this period, data were collected between turbines 7 and 12, covering a 720 m section of the breakwater.

Seasons / time of day

Fatality searches were carried out throughout the year. Activity surveys were limited to the breeding season and autumn. Data were collected throughout the day between dawn and dusk, with additional nocturnal surveys carried out during the autumn.

Species

Gulls (234 birds/day), little tern (375-1,860 birds/day), common tern (4,228-10,263 birds/day), Sandwich tern (11-12,334 birds/day).

Conditions data collected under

No details given.

Location / habitat

Zeebrugge, Belgium (Coastal)

Turbine / array specification

25 turbines arranged along Zeebrugge Harbour breakwater. Turbines vary in size from hub heights of 23-55 m and rotor diameters of 22-48 m. Details of collisions at individual turbines are not given, so avoidance rates are estimated assuming turbines with a hub height of 34 m and rotor diameter of 34 m, the most common turbine within the windfarm. Distance between turbines within each row is not described.

Results

Collisions were recorded in every year. For Sandwich terns, collisions varied from seven to 54 birds per year. Using option 1 of the Band model, the estimated number of collisions per year, in the absence of avoidance behaviour, varied from 6,383 birds to 10,299, 8,024 to 10,326 using option 2 and 5,984 to 8,035 using option 3. The meso-micro avoidance rates derived from the values are 1 between 2001 and 2003, 0.9915 in 2004, 0.9972 in 2005, 0.9992 in 2006 and 0.9993 in 2007 using option 1, and 1 between 2001 and 2003, 0.9948 in 2004, 0.9963 in 2005, 0.9989 in 2006 and 0.9991 in 2007 using option 2. Using option 3, the avoidance rates are 1 between 2001 and 2003, 0.9933 in 2004, 0.9952 in 2005, 0.9986 in 2006 and 0.9989 in 2007. Collision data were also obtained relating to June 2004 and June 2005, the periods in which bird activity data were collected and relating to only the turbines around which activity was monitored. In both years, 3 Sandwich terns were observed to have collided between turbines 7-12 in June. Given passage rates of 896 birds/hr in June 2004 and 725 birds/hr in June 2005, this reflects an avoidance rate in 2004 of 0.9895 using option 1 of the Band model, 0.9935 using option 2 of the Band model and 0.9917 using option 3 of the Band model. In 2005, the corresponding values are 0.9940, 0.9920 and 0.9897.

For little terns, collisions varied from two to 12 birds per year. Using option 1 of the Band model, the estimated number of collisions per year, in the absence of avoidance behaviour, varied from 990 birds to 1,087, 165 to 838 using option 2 and 128 to 650 using option 3. The meso-micro avoidance rates derived from the values are 0.9923 in 2001, 0.9914 in 2002, 0.9904 in 2003, 0.9950 in 2004, 0.9982 in 2005, 0.9963 in 2006 and 0.9890 in 2007 using option 1, and 0.9516 in 2001, 0.9455 in 2002, 0.9395 in 2003, 0.9940 in 2004, 0.9884 in 2005, 0.9768 in 2006 and 0.9304 in 2007 using option 2. Using option 3, the avoidance rates were 0.9516 in 2001, 0.9455 in 2002, 0.9395 in 2003, 0.9940 in 2004, 0.9884 in 2005, 0.9768 in 2006 and 0.9304 in 2007. No little tern collisions were recorded in the June 2004 and 2005 data relating to turbines 7-12.

For common terns, collisions varied from 12 to 164 birds per year. Using option 1 of the Band model, the estimated number of collisions per year, in the absence of avoidance behaviour, varied from 4,503 birds to 6,869, 2,475 to 6,530 using option 2 and 1,931 to 5,094 using option 3. The meso-micro avoidance rates derived from the values are 0.9970 in 2001, 0.9977 in 2002, 0.9951 in 2003, 0.9758 in 2004, 0.9812 in 2005, 0.9761 in 2006 and 0.9834 in 2007 using option 1, and 0.9919 in 2001, 0.9939 in 2002, 0.9871 in 2003, 0.9833 in 2004, 0.9501 in 2005, 0.9365 in 2006 and 0.9559 in 2007 using option 2. Using option 3, meso-micro avoidance rates were 0.9896 in 2001, 0.9922 in 2002, 0.9834 in 2003, 0.9786 in 2004, 0.9360 in 2005, 0.9186 in 2006 and 0.9434 in 2007. Collision data were also obtained relating to June 2004 and June 2005, the periods in which bird activity data were collected and relating to only the turbines around which activity was monitored. In 2004 6 common terns were observed to have collided between turbines 7-12 in June, in 2005, this figure was 9. Given passage rates of 603 birds/hr in June 2004 and 248 birds/hr in June 2005, this reflects an avoidance rate in 2004 of 0. 9703 using option 1 of the Band model, 0.9796 using option 2 of the Band model and 0.9738 using option 3 of the Band model. In 2005, the corresponding values are 0.9720, 0.9255 and 0.9045.

For gulls, collisions varied from 110 to 354 birds per year. Using option 1 of the Band model, the estimated number of collisions per year, in the absence of avoidance behaviour, varied from 2,334 birds to 2,537, 2,856 to 3,104 using option 2 and 2,698 to 2,932 using option 3. The meso-micro avoidance rates derived from the values are 0.8979 in 2001, 0.8481 in 2002, 0.8817 in 2003, 0.9105 in 2004, 0.9173 in 2005, 0.9547 in 2006 and 0.9092 in 2007 using option 1, and 0.9166 in 2001, 0.8758 in 2002, 0.9033 in 2003, 0.9268 in 2004, 0.9324 in 2005, 0.9630 in 2006 and 0.9258 in 2007 using option 2. Using option 3 meso-micro avoidance rates were 0.9117 in 2001, 0.8686 in 2002, 0.8976 in 2003, 0.9226 in 2004, 0.9285 in 2005, 0.9608 in 2006 and 0.9214 in 2007.

Data were also obtained relating to black-headed, lesser black-backed and herring gull collisions in June-July 2000, June-July 2001 and September-October 2001, periods corresponding to the times during which gull activity data were collected and restricted to the turbines around which gull data were collected. No collisions were reported involving black-headed gulls. In June-July 2000, a single collision was reported involving a herring gull, reflecting an avoidance rate of 0.9861 using option 1, 0.9829 using option 2 and 0.9819 using option 3. In June-July 2001 and September-October 2001, two collisions were reported involving herring gulls, reflecting avoidance rates of 0.9722 and 0.9976 respectively using option 1, 0.9659 and 0.9959 using option 2 and 0.9639 and 0.9957 using option 3. Single collisions were reported involving lesser black-backed gulls in each of June-July 2001 and September to October 2001, reflecting avoidance rates of 0.9706 and 0.9990 respectively using option 1, 0.9680 and 0.9977 using option 2 and 0.9656 and 0.9975 using option 3.

Assessment of methodology

The study at Zeebrugge offers one of the most comprehensive datasets for collisions involving marine birds. Fatality data have been collected over a seven year period following a robust methodology with corrections made to account for searcher efficiency and scavenger activity. However, a key limiting factor in the dataset is the accompanying bird activity data. In the case of terns, activity data is limited to the period of peak tern activity in June. As a consequence, extrapolating from this to cover the full period when terns are present is likely to vastly over-estimate activity in the area, and therefore the predicted collision numbers. This means that the avoidance rates derived for each year are likely to be significantly over-estimated. This reflects the limitations in the way data are presented within the reports. Ideally, collisions would be broken down on a month by month and turbine by turbine basis, so that avoidance rates could be calculated for the areas in which activity data were collected, rather than extrapolating across the windfarm as a whole.

We used only the collision data collected from gulls during the period in which activity data were collected, and from only those turbines around which activity data were collected, in deriving representative avoidance rates.

ANNEX 1

USING A COLLISION RISK MODEL TO ASSESS BIRD COLLISION RISKS FOR OFFSHORE WINDFARMS
(SOSS Guidance: March 2012)

SUPPLEMENT – AVOIDANCE RATES USING THE BASIC AND EXTENDED MODELS
March 2014 – Bill Band

This is a supplement to guidance prepared for the Crown Estate as part of the Strategic Ornithological Support Services programme, project SOSS-02[45]. That provides guidance for offshore wind developers, and their ecological consultants, on using a collision risk model to assess the bird collision risks presented by offshore windfarms. The March 2012 version of the guidance enabled use to be made of flight height distribution data.

This supplement is an addition to Stage E – Avoidance and Attraction. That section describes how, having used the collision model to calculate the potential collision rate if birds take no avoiding action, one should then apply an avoidance rate A to allow for the fact that many species of birds do in fact take avoiding action, either at long range (macro) or at close range (micro).

Paragraph 80 notes that

'if the extended model taking account of flight height distribution is used, it is important that the calculations on which avoidance rates are based also start with a no-avoidance collision rate derived using the extended model'.

Most of the published literature on avoidance rates is currently based on using the basic model. **This supplement shows how such avoidance rates may be modified to enable their application to the extended model.**

Avoidance in the basic and extended models

The two models – basic and extended – yield different predictions of the rate of collisions before avoidance is taken into account. The extended model is a more refined model which takes into account the effect of flight height distribution. It takes into account the fact that, for a given number of flights at risk height, a flight height distribution skewed towards low altitude leads to a smaller proportion of birds passing through the rotor, and bird passages through parts of the rotor with less risk, than if the distribution were uniform.

The outputs of the two models may be formally compared if the data input to the basic model on the proportion of flights at risk height (Q'_{2R}) is derived from the same

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[45] Project SOSS -02: see http://www.bto.org/science/wetland-and-marine/soss/projects

flight height distribution used in the extended model, as in Option 2 of the spreadsheet accompanying the SOSS guidance. That is, the comparison should be made between the collision rate using the basic model (Option 2) in the spreadsheet, and the extended model (Option 3).

The collision rates (before avoidance) projected by the two models are:

Basic model (Option 2):

$C_{basic} = v(D_A/2R)(T\pi R^2)t \times Q_{2R}' \times p_{average} \times Q_{op}$
(guidance eq.5[46])
i.e. flux factor $\times Q_{2R}' \times p_{average} \times Q_{op}$

Extended model (Option 3):

$C_{extended} = v(D_A/2R)(T\pi R^2)t \times (2/\pi) \iint d(y)\, p(x,y)\, dxdy \times Q_{op}$
(guidance eq. 9)
i.e. flux factor x collision integral x Q_{op}

Where the bird flight height distribution is skewed towards low altitude, the extended model prediction $C_{extended}$ is usually less than C_{basic}, because this equation takes full account of the reduction in risk at lower parts of the rotor. Let g be the ratio $C_{extended} / C_{basic}$, g is thus usually less than 1. The value of g may be obtained by dividing the second of the above equations by the first:

g = $C_{extended} / C_{basic}$ = collision integral / ($Q'_{2R} \times p_{average}$) …. eq. S1

and this is readily calculated from the 'Overall collision risk' spreadsheet
g = cell D35 / (cell D33 x cell D27)

The expected collision rate must then take into account the proportion A of birds avoiding the turbines (e.g. by displacement, or by evasive action), by multiplying the above no-avoidance collision rates by the proportion (1-A) which do not avoid. Values of A are typically in the range 90-100%. It is more helpful to think in terms of the non-avoidance rate A' = 1 – A , such that A' is the small proportion of birds which do not avoid the turbines. The expected collision rate is then

$A'_{basic}\, C_{basic}$ in the basic model, or .. eq. S2a
$A'_{extended}\, C_{extended}$ in the extended model. .. eq. S2b

The two models require the use of different non-avoidance rates. The calculation of $C_{extended}$ takes account of the effect of a skewed flight distribution, such that the

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[46] Strictly, equation (5) of the guidance refers to Q_{2R} derived from site survey, as used in the basic model (Option 1), rather than Q'_{2R}, derived from the assumed flight height distribution, as required here.

factor $A_{extended}$ (= 1 – $A'_{extended}$) refers only to genuine behavioural avoidance. The calculation of C_{basic} in the basic model does not, such that any such effect, in the basic model, must be covered by the avoidance factor A_{basic}.

Establishing avoidance rates from reference windfarms

Values of A'_{basic} and $A'_{extended}$ for use in the two models are obtained by monitoring collisions at one or more reference windfarms, and working back from the two models. For either model we have

Non-avoidance rate A' = Actual no of collisions / Predicted number of collisions C.

		using basic model	*using extended model*
Actual no of collisions	=	$A'_{basic} \times C_{basic}(ref)$	= $A'_{extended} \times C_{extended}(ref)$

thus $A'_{extended} = A'_{basic} \times C_{basic}(ref) \; / \; C_{extended}(ref)$

but $g(ref) = C_{extended}(ref) / C_{basic}(ref)$

so $\mathbf{A'_{extended} = A'_{basic} / g\,(ref)}$ … eq. S3

$A'_{extended}$ is the non-avoidance rate from the reference windfarm, for use with the extended model. Equation (S3) describes how it is related to the value of A'_{basic} derived using the basic model, using the g factor for this reference windfarm.

Where data from several reference windfarms are used to yield an average A'_{basic} , then the value for $A'_{extended}$ should be the average of A'_{basic} / g(ref) as calculated for each of the reference windfarms.

Applying reference avoidance rates to new or projected windfarms

Avoidance rates, derived from collision studies at one or more reference windfarms, may be used to inform the calculation of collision rate at a new or projected windfarm. The assumption in applying such avoidance rates is that the birds' behavioural response to the new windfarm will be similar to their response to the reference windfarm, and hence the proportion of birds avoiding the turbines of the new windfarm, further to the calculation of a no-avoidance collision rate, is likely to be the same as for the turbines of the reference windfarm.

Thus, having established values A'_{basic} and $A'_{extended}$ for non-avoidance, as derived from the reference windfarm, these same values may be assumed to apply to new or projected windfarms for the same bird species. If C_{basic}(new) and $C_{extended}$ (new) are the no-avoidance collision rates calculated for the new windfarm, the predicted collisions after avoidance for the new windfarm are:

basic model: $\mathbf{A'_{basic}\ C_{basic}\ (new)}$ .. eq. S4a

extended model: $\mathbf{A'_{extended}}$ $\mathbf{C_{extended}}$ **(new)** .. eq. S4b

A'_{basic} is the avoidance rate established from the reference windfarm(s) using the basic model, and $A'_{extended}$ that using the extended model; they are related as in equation (S3).

Dealing with lack of information on g(ref)

Published information on avoidance rates for reference windfarms has not so far included information on avoidance using the extended model, or on g(ref), the ratio between the outputs (before avoidance) of the extended and basic models. Calculation of g(ref) requires information on bird size and speed, turbine parameters, and the flight height distribution at the reference site; however it does not need information on bird density, levels of flight activity, or number of transits. If this limited subset of data is available, then it should be possible to calculate g(ref) for the reference windfarm, for the bird species under assessment.

Example:

Monitoring studies have established that for a certain bird species, an overall avoidance rate of 98% may be assumed. This has been derived using theoretical collision rates derived using the basic model, and comparing these with the actual collision mortality observed on an existing windfarm – the ‘reference’ windfarm.

$\mathbf{A_{basic} = 98\%}$ so the non-avoidance rate $A'_{basic} = 2\%$.

Using the extended model, the calculated g factor for this reference windfarm is 0.46. Thus the non-avoidance rate appropriate for use with the extended model is

$A'_{extended} = 2\%/0.46 = \mathbf{4.38\%}$

The corresponding avoidance rate for use with the extended model is

$\mathbf{1 - A' = 95.62\%}$

A developer now undertakes collision risk assessment for a proposed offshore windfarm. The CRM extended model which takes account of flight height distributions may be used, provided that it makes use of the avoidance rate appropriate for the extended model.

For the proposed windfarm, the projected collision rates are 23 (basic model) and 8 (extended model) per year. Applying the above non-avoidance rates of 2% and 4.38% respectively yields

expected collisions(basic) = 2% x 23 = 0.46 birds/annum

expected collisions (extended) = 4.38% x 8 =0.35 birds/annum

The two models yield different results because the second model takes flight height distribution into account, a factor ignored in the basic model.

It is recommended that any future publication of reference avoidance rates, derived from collision monitoring studies, should state both that for use in the basic model and that for use in the extended model. This will require application of both models to the reference windfarm.

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Quantifying avian avoidance of offshore wind turbines: Current evidence and key knowledge gaps

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ABSTRACT

The risk of collision between birds and turbines is seen as one of the key issues in the planning process for offshore wind farms. In some cases, predictions of collision risk have led to projects either being withdrawn from the planning process, or refused planning consent. Despite this, the evidence base on which collision risk is assessed is extremely limited and assessments rely on models which can be highly sensitive to assumptions, notably about bird collision avoidance behaviour. We present a synthesis of the current state of knowledge about collision risk and avoidance behaviour in seabirds. Evidence suggests species-specific responses to turbines and that in order to avoid collision, most birds adjust their flight paths at some distance from the turbines, rather than making last-second adjustments. We highlight the key gaps in knowledge and make recommendations for future data collection.

1. Introduction

Offshore wind energy is likely to play a key role in efforts to combat climate change through the production of renewable energy (Kaldellis and Zafirakis, 2011). At present the industry is well-developed in northern Europe, and is expanding globally (Breton and Moe, 2009; Snyder and Kaiser, 2009). However, there are concerns over the potential for offshore wind farms to negatively affect wildlife, with impacts on seabirds frequently cited as a key concern (Furness et al., 2013; Garthe and Huppop, 2004).

The main effects of offshore wind farms on seabirds are thought to be: i) collision mortality; ii) displacement and attraction effects and; iii) barrier effects (Desholm and Kahlert, 2005; Everaert and Stienen, 2007; Masden et al., 2009; Vanermen et al., 2015). Barrier effects occur when the wind farms physically exclude birds causing extended flight journeys around the development during migration or when commuting between colonies and foraging areas (Masden et al., 2010, 2009). Displacement is regarded as a response that results in a functional loss of the habitat available within a wind farm, as opposed to a change in flight trajectory around the wind farm (Drewitt and Langston, 2006; Furness et al., 2013). Whereas attraction to wind farms is argued to be a consequence of turbines serving as a platform for roosting birds or the base acting as a reef resulting in an increase in food availability (Dierschke et al., 2016). Collision mortality describes birds colliding with turbines and associated infrastructure and has received a significant level of attention by the onshore industry as a result of well documented events (de Lucas et al., 2008; Everaert and Stienen, 2007; Loss, 2016; Thaxter et al., 2017a). However, the feasibility of collecting corpses or observing collision events in the marine environment is challenging and, to date, only two studies have reported birds colliding with offshore turbines (Desholm, 2006; Pettersson, 2005).

In the absence of more detailed information about collision rates, Collision Risk Models (CRM) are routinely used to predict the risk posed by offshore wind farms to seabird populations as part of pre-construction Environmental Impact Assessments (EIAS) in Europe (e.g. Ministry of Economic Affairs, 2015; NIRAS, 2015). CRMs are also being used in a range of countries where the offshore wind industry is in the early stages of development including the USA (Cranmer et al., 2017; Fammler and Kuris, 2010; Jenkins et al., 2018; Stumpf et al., 2011). They have also been used in a post-construction context in order to quantify likely collision rates (Skov et al., 2012) and to help estimate the cumulative impact of collisions at multiple offshore wind farms

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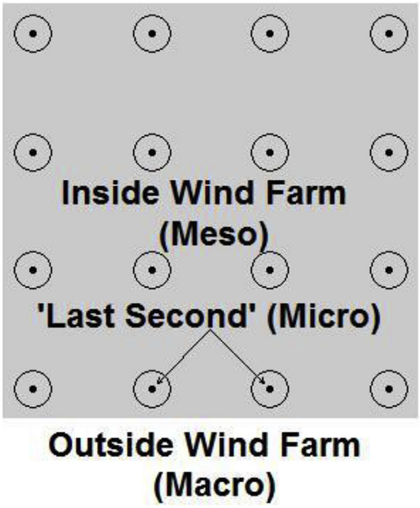

Fig. 1. Different scales of avoidance behaviour in relation to an offshore wind farm, turbines indicated by black dots. Macro-avoidance reflects birds either taking action to avoid entering, or birds being attracted to, a wind farm, meso-avoidance reflects birds taking action to avoid individual turbines and micro-avoidance reflects birds taking last-second action to avoid colliding with rotor blades (i.e. within circles surrounding each turbine).

through extrapolation (Brabant et al., 2015; Busch and Garthe, 2017). A variety of different CRMs are available, but at their core most calculate the probability of a bird colliding based on the likelihood of it occupying the same space as a turbine blade. The collision risk to an individual bird is then scaled up based on the number of birds likely to pass through a wind farm over a given time period. The final stage is the application of an avoidance rate which takes into account the proportion of birds likely to take action to avoid a collision (Masden and Cook, 2016). However, outputs from CRMs are known to be sensitive to assumptions made about the avoidance behaviour of the species concerned, notably flight height and flight speed, which are often based on extremely limited data (Chamberlain et al., 2006; Masden, 2015).

Whilst avoidance behaviour can be seen as a continuum over space and time, there is a need to break this down into different components which correspond to how birds may respond to both the wind farm and to individual turbines. Technological limitations associated with measurement have also influenced the definitions but, currently avoidance behaviour is recognised at three different scales (Fig. 1), termed macro, meso, and micro (Cook et al., 2014). May (2015) developed a framework for understanding avian avoidance based on the underlying behavioural mechanisms and set out how this related to these three classifications. Macro-avoidance (avoidance of the wind farm as a whole) can arise through a functional habitat loss and is observed as displacement. May (2015) went on to argue that attraction could be included under the term displacement, resulting in what are in effect negative avoidance rates. However macro-avoidance can also include barrier effects, a type of evasive behaviour which can be classified as being impulsive or anticipatory, the latter of which requires early detection or a prior experience or knowledge. Meso-avoidance is the anticipatory or impulsive evasion of rows of turbines within a wind farm. Micro-avoidance reflects the last-second action taken to avoid collision with the turbine blades and may be thought of as an escape response (May, 2015).

Collisions with turbines may not only have significant conservation implications (Everaert and Stienen, 2007) but important economic consequences as well. In the UK, the Docking Shoal Offshore Wind Farm was refused planning consent over the estimated numbers of Sandwich terns *Thalasseus sandvicensis* predicted to be killed (DECC, 2012), a decision with major implications for both the developer and regulators. Considering the respective economic and conservation concerns, it is vital that decisions about offshore wind farms are made based on the best available evidence. Despite this, there has been no clear agreement about how data describing avoidance behaviour should be collected (Cook et al., 2014; May, 2015). There is a risk that this situation may lead to "decision paralysis" whereby decision-making is constantly postponed whilst additional data are collected (Milner-Gulland and Shea, 2017).

Northern gannet *Morus* bassanus, lesser black-backed gull *Larus fuscus*, herring gull *Larus argentatus*, great black-backed gull *Larus marinus* and black-legged kittiwake *Rissa tridactyla* are viewed as being at a high risk of collision with offshore wind farms due to their flight altitude (Furness et al., 2013; Johnston et al., 2014; Ross-Smith et al., 2016). In northern Europe, the foraging ranges of these species also often overlap with the currently planned offshore wind farm developments (Bradbury et al., 2014; Johnston et al., 2015; Soanes et al., 2013; Thaxter et al., 2015). Here we consider what evidence currently exists to quantify avoidance behaviour for these species. We then describe how these data can be best combined to calculate an overall avoidance rate suitable for use in CRMs for the five key species. In so doing we present an approach which can be adapted for other species and also allows for sufficient flexibility for the inclusion of future data for our example species. Finally, we highlight any gaps in knowledge that we have identified as part of our review.

2. Methods

We focussed our literature search on operational wind farms in northern Europe at which the five key bird species were likely to occur. An online database (www.4coffshore.com) was used to identify offshore wind farm sites, relevant developers and their environmental consultants in order to obtain available reports and data. Web of Science and Google Scholar were used to search for relevant peer-reviewed papers, reports, conference proceedings and book chapters relating to the impacts of wind farms on the five priority species, following literature trails where appropriate. We also referred to previous reviews on the topic (Marine Management Organisation, 2014; Smartwind/Forewind, 2013) to ensure that all sources of primary literature had been identified. Where appropriate, we also considered data relating to the five key species collected from coastal sites, as currently these may reflect the best or only available data on which to base decisions (potential biases are highlighted in section 4.2-Limitations).

2.1. Macro-avoidance

Methodologies which have been used to look at macro-avoidance may not actually distinguish between birds displaced from a wind farm and those exhibiting barrier effects since both can be manifested as a decrease in the numbers of birds in flight within the wind farm area. For the purposes of this review however, we considered studies according to the effect they were designed to investigate. This was not considered an issue as barrier, displacement and attraction effects collectively describe the overall macro-avoidance rate. The key studies included based boat or aerial surveys or from counts from panoramic scans but supporting information was included from GPS tracking studies or radar studies where species identification had been possible. Rates of macro-avoidance were taken directly from the studies cited or calculated using model coefficients (see Cook et al., 2014 for more details).

2.2. Meso-avoidance

To assess evidence for meso-avoidance, we considered studies in which the distribution or movement patterns of birds within a wind farm were assessed. Studies selected for inclusion in the review were those which compared the distribution of bird densities or bird movements in the area surrounding individual turbines to the density elsewhere within the wind farm. Surveys were carried out using either visual observations or with radar in combination with visual observations to identify target birds to species level.

2.3. Horizontal vs vertical macro- and meso-avoidance

We considered both macro- and meso-avoidance to have two components, a vertical component and a horizontal component. For the horizontal component, we considered studies in which the distribution of birds (densities) or flightpaths outside the wind farm were compared to the distribution within the wind farm (macro) or with respect to turbines or turbine rows within the wind farm itself (meso).

A significant proportion of birds are likely to fly below rotor-swept height where no turbines are present (Johnston et al., 2014). Consequently, in order to estimate vertical avoidance, a comparison must be made of the proportion of birds at rotor-swept height pre- and post-construction or, inside and outside the wind farm. We searched for studies which met these criteria. These studies were mainly derived from fairly limited visual observations collected from boats or other observation platforms. Data collected using radar were considered, but were found to be at too coarse a scale to be of use.

2.4. Micro-avoidance

Studies of micro-avoidance require detailed behavioural observations of the interaction between birds and turbines due to the fact that they involve last-second escape responses. Therefore, to assess micro-avoidance we considered studies in which interactions between birds and turbines were recorded visually by observers or remotely using radar or turbine-mounted cameras. There was a lack of information which was identified for this scale however.

2.5. Within-wind farm avoidance

Ideally, micro and meso avoidance could be quantified separately for each species in order to generate robust estimates of avoidance behaviour. However, such data may not be available and, given pressures in the decision making process, decisions often rely on the best available data (Milner-Gulland and Shea, 2017). In these circumstances, it is possible to derive avoidance rates by comparing recorded collision rates with estimates of bird flight activity within a wind farm (Band, 2012). As this approach considers all bird movements within a wind farm, it is effectively a combination of both meso-avoidance and micro-avoidance although, it should be noted that it also incorporates elements of bias introduced by estimates of flight activity from the model itself (Band, 2012). We therefore refer to avoidance rates calculated in this way as within-wind farm avoidance. At present, technologies to record collision rates in the offshore environment are still under development (e.g. Collier et al., 2011). Consequently, avoidance rates derived using this approach rely on data from the onshore environment. However, analysis of GPS data indicates that there are likely to be strong differences between flight behaviour on the coast and in marine areas in comparison to the terrestrial environment (Ross-Smith et al., 2016). Consequently, we limited data considered for this analysis to those collected from coastal locations, where flight behaviour may be more similar to that observed offshore, although there may still be some differences (Ross-Smith et al., 2016).

Records of collisions between birds and turbines are frequently reported as a collision rate per year, or a collision rate per turbine per year (Musters et al., 1996). However, in order to estimate a within-wind farm avoidance rate, these data must be combined with estimates of the number of birds passing through the site i.e. the flux rate. We therefore restricted our analyses to sites where estimates of flight activity were also made. Due to likely seasonal and spatial patterns in flight activity, analyses were restricted to sites in which collision and flight activity data were collected during the same months in order to ensure that collision rates reflected seasonal patterns in flight activity data. For these reasons, reported avoidance rates may not match those presented in the original studies. However, we feel it is important that data across sites should be assessed in a consistent way. In order to ensure transparency, Table S3 includes the data and calculations used to estimate the flux rates and within-wind farm avoidance rates at each site. Within-wind farm avoidance rates at each site were estimated using equation (1) (Scottish Natural Heritage, 2010).

$$Within - wind\ farm\ avoidance\ rate = 1 - \left(\frac{Observed\ Collisions}{Probability\ of\ collision\ \times Flux\ Rate}\right) \quad (1)$$

Probability of collision is the likelihood of the blade and bird occupying the same location in space and time based on simplified geometry and is derived using the Band model (Band, 2012), assuming turbine characteristics presented in Table S1 and bird behaviour and morphology presented in Table S2. The probability of collision for each species at each site is given in Table S3. The flux rate is estimated by calculating the number of birds expected to have passed through the wind farm per m^2 per hour scaled up to cover the total turbine frontal area and the total time period during which corpses were collected, and corrected for the proportion of birds at collision risk height and the level of nocturnal activity. The assumptions made during calculations can have a significant impact on the final estimates, and we therefore include Table S3 in supplementary information which shows the step by step process by which we estimated each within wind farm avoidance rate. We then used ratio estimators (Cochran, 1977) to combine avoidance rates across multiple sites and the delta method (Powell, 2007) to estimate the standard deviation associated with the derived avoidance rates.

2.6. Derivation of recommended total avoidance rates

Collision risk estimates are typically based on pre-construction estimates of the total number of birds within a wind farm (Cook et al., 2014). Consequently, the avoidance rates used in collision risk models must account for changes in the total number of birds within the wind farm between the pre- and post-construction periods as well as any redistribution arising from behavioural responses to turbines within the wind farm. The total avoidance rate can then be estimated by combining the macro-, meso- and micro-avoidance rates as shown in equation (2) (Cook et al., 2014; Krijgsveld et al., 2011), or the macro- and within wind farm-avoidance rates as shown in equation (3) (adapted from Equation (2)).

$$\begin{aligned}(1-\text{Total Avoidance Rate}) = {} & (1 - \text{Macro}-\text{Avoidance}) \\ & \times (1 - \text{Meso}-\text{Avoidance}) \\ & \times (1 - \text{Micro}-\text{Avoidance}) \quad (2)\end{aligned}$$

$$\begin{aligned}(1-\text{Total Avoidance Rate}) = {} & (1 - \text{Macro}-\text{Avoidance}) \\ & \times (1 - \text{Within}-\text{wind farm avoidance}) \quad (3)\end{aligned}$$

Equations (2) and (3) can accommodate situations where birds are attracted at a macro- or meso-scale. Within these formulae, a value of 1 relates to total avoidance, a value of 0 relates to neither avoidance nor attraction and values less than 0 relate to attraction (i.e. -0.1 would relate to a 10% increase), meaning the avoidance rate is reduced when birds are attracted to the wind farm or individual turbines.

3. Results

3.1. Macro-avoidance – barrier effects

Overall there was limited evidence of macro-avoidance as an apparent consequence of barrier effects for the five priority species

Table 1
Summary of key studies of barrier effects, displacement and attraction for the five priority species (B = barrier effects; D = displacement; A = attraction and NR = no response). Blank cells indicate species which were not covered by that particular study. Where given, estimated rates are either those reported in the study concerned or, derived from published effect sizes.

	Horns Rev (Petersen et al., 2006)	Horns Rev (Petersen et al., 2006)	Nysted (Petersen et al., 2006)	Egmond aan Zee (Krijgsveld et al., 2011)	Horns Rev 2 (Skov et al., 2012)	Egmond aan Zee (Leopold et al., 2013)	Princess Amalia (Leopold et al., 2013)	Alpha Ventus (Mendel et al., 2014)	Robin Rigg (Natural Power, 2014)	Greater Gabbard (APEM Ltd, 2014)	Blighbank (Vanermen et al., 2015)	Alpha Ventus (Welcker and Nehls, 2016)
Northern gannet	D	B (1.00)		B (0.64)	B (0.86)	D	D	NR	NR	D (0.95)	D (0.85)	D (0.92)
Lesser black-backed gull						D	NR	D			A (− 4.25)	
Herring gull	A		NR			NR	NR		NR		A (− 8.4)	A (− 1.79)
Great black-backed gull						NR	NR		NR		NR	A (− 2.00)
Black-legged kittiwake					B (0.69)	NR	D	D	NR		NR	NR
Gulls (*Larus* spp.)				B (0.18)	B (0.56)							

(Table 1). Systematic panoramic scans of densities of birds in flight within and around the Egmond aan Zee wind farm in the Netherlands revealed a macro-avoidance rate of 0.64 for northern gannet (n = 81) and 0.18 for gull spp combined (Krijgsveld et al., 2011). Using a combination of radar and laser range finders at Horns Rev, a macro-avoidance rate of 0.84 was calculated based on the numbers of tracks of migrating gannets (n = 74) which did not enter the wind farm (Skov et al., 2012). The same study also reported an avoidance rate of 0.56 for large gulls (n = 84) and 0.69 for kittiwakes (n = 11). An earlier study at the same wind farm reported that out of 126 tracks representing 268 individual migrating gannets, none of these entered the wind farm. For migrating gulls (herring, great black-backed, little and kittiwake, 442 tracks out of a total of 461 did not enter the wind farm — although as neither species or size of flock were reported, the avoidance rate cannot be estimated (Petersen et al., 2006). However, in these studies data were collected during the post-construction period only and caution should therefore be applied when interpreting their significance in the absence of pre-development data. Furthermore, data collection also tended to be focused on outside the breeding season and the extent to which this information is relevant to birds when they are tied to their colonies is unclear.

3.2. *Macro-avoidance – displacement*

From studies of displacement, macro-avoidance was estimated for northern gannet using ship based surveys at the Blighbank wind farm in Belgium (Vanermen et al., 2015) – a rate of 0.85 (Table 1) – and at the Alpha Ventus wind farm in Germany – a rate of 0.92 –although this study was based in an area where gannets densities were low (Welcker and Nehls, 2016). As before, whether these data are representative of behaviour during the breeding season is uncertain as the majority of the data were from the non-breeding season (Vanermen et al., 2013). Of the remaining studies, one reported displacement at two wind farms (Leopold et al., 2013) and another three reported no response of northern gannet, possibly as a result of low densities of birds being present pre- and post-construction (Mendel et al., 2014; Natural Power, 2014; Petersen et al., 2006). An aerial based survey at Greater Gabbard in the UK estimated an avoidance rate of 0.95 (APEM, 2014) during the autumn passage period and based on the post construction period only. An additional study of three GPS-tagged northern gannets also indicated that they avoided entering wind farms (Garthe et al., 2017).

One study reported great black-backed gulls as being attracted to offshore wind farms (Welcker and Nehls, 2016), whilst the others reported no response (i.e. no attraction to or displacement from). For lesser black-backed gulls the evidence for macro-avoidance was equivocal with studies reporting attraction, displacement and no response to the wind farms. A recent study of GPS-tagged lesser black-backed gulls suggests that while individuals may differ in their response to offshore wind farms, overall the species did not consistently exhibit displacement or attraction (Thaxter et al., 2017b). Herring gull largely showed no response to wind farms with the notable exception at Blighbank and Alpha Ventus where attraction effects were reported (Vanermen et al., 2015; Welcker and Nehls, 2016) possibly linked to increased roosting opportunities provided by the wind farm (the same effect was observed for lesser black-backed gulls at the same site). Black-legged kittiwake showed both displacement effects and no response to wind farms.

3.3. *Macro-avoidance – combining all effects*

For the species considered in this review, there was evidence that northern gannet exhibit macro avoidance. At this stage, we believe the lower of the available values, 0.64 (Krijgsveld et al., 2011), is an appropriate macro-avoidance rate for northern gannet. This is based on a precautionary approach given that estimates were often based on small sample sizes leading to limited power to detect change combined with most data being collected outside the breeding season. In contrast, based on the studies we identified, none of the gull species appear to show a consistent response to wind farms. In the absence of consistent evidence, we are unable to recommend a suitable macro-avoidance rate for gulls.

3.4. Horizontal meso-avoidance

Meso-avoidance is likely to reflect the anticipatory or impulsive evasion of individual turbines. We identified four studies in which the distribution of birds or flight paths within a wind farm were quantified. Using radar, Krijgsveld et al. (2011) and Skov et al. (2012) found strong evidence of horizontal meso-avoidance of individual turbines. Krijgsveld et al. (2011) reported that the density of birds within 50 m of a turbine was 66% of the density elsewhere in the wind farm. Assuming that, in the absence of turbines, birds would be expected to be evenly distributed across the area of the wind farm, this reflects a meso-avoidance rate of 0.34. It is likely that this figure reflects an under-estimate of total meso-avoidance as it is based on data collected using horizontal radar and will, therefore, include birds flying above or below the turbines thus not at risk of collision. Skov et al. (2012) found a stronger response, with none of the 408 large gulls they recorded passing within 50 m of a turbine. However, the primary purpose of this analysis was to collect information describing species flight heights rather than their proximity to turbines. Tracks from radar suggested some birds may approach the turbines more closely. Using visual observations, Janoska (2012) recorded only 23 out of 917 gulls passing within 75 m of a turbine, reflecting a meso-avoidance rate of 0.975. By contrast, also using visual observations, Everaert (2008) reported no significant difference in the number of gulls passing within 100 m of a turbine (or its proposed site) between pre- and post-construction periods, possibly reflecting the location of the site on a flight line between a roost and a foraging/loafing area. These studies suggest that gulls may have a strong horizontal meso-avoidance of turbines, but that this may be site or context specific. Whilst the data in the studies described above are informative, it should be noted that they are not sufficiently robust to allow firm conclusions to be drawn about the likely magnitude of any meso-avoidance.

In addition to the studies described above, several studies reported anecdotal evidence describing how the relative location of the turbines may influence the distribution of birds within a wind farm. Petersen et al. (2006) provided evidence to suggest that birds may be more likely to respond to turbines as the number of turbine rows they passed increased, suggesting stronger avoidance towards the middle of the wind farm than at the edge. Similarly, Winkelman (1992) noted that there were fewer collision victims towards the centre of a wind farm. These data suggest that the strength of any horizontal meso-avoidance may vary with distance from the wind farm centre. There was also evidence from three sites – Horns Rev, Alpha Ventus and Egmond aan Zee – to suggest that birds respond to the operational status of turbines, with higher densities recorded when turbines were not operational, although this effect may be more noticeable at night (Krijgsveld et al., 2011; Mendel et al., 2014; Petersen et al., 2006; Schulz et al., 2014).

3.5. Vertical meso-avoidance

We identified three sites at which the proportions of birds of different species at rotor-swept height could be compared pre- and post-construction – Barrow (Barrow Offshore Wind Limited, n.d.), Gunfleet Sands (GoBe Consultants Ltd., 2012; NIRAS Consulting, 2011) and Robin Rigg (Natural Power, 2013) – and a fourth – Egmond aan Zee (Krijgsveld et al., 2011) – where flight heights were compared inside and outside a wind farm (Table 2). Across these sites, there was no consistent pattern indicating an increase or decrease in the proportion of birds at rotor-swept height in response to the presence of a turbine. However, given the extremely limited evidence, no firm conclusions can be drawn about the extent or direction of any vertical meso-response in any species of marine birds. Furthermore, where flight heights are estimated by observers by eye, it should be noted that any comparison may be confounded by the fact that heights are easier to estimate once turbines have been installed as they offer fixed reference points of known height.

3.6. Micro-avoidance

Micro-avoidance reflects a last second action to avoid collision and, may be thought of as an escape response (May, 2015). We identified 17 sites at which visual observations of the interactions between birds and turbines had been reported (Table 3). Over the course of these studies, only two collision events were directly observed, the first involving a flock of four common eider *Somateria mollisima* at a single turbine at the Utgrunden Offshore Wind Farm in Sweden and the second, a passerine or bat at Nysted Offshore Wind Farm in Denmark (Desholm, 2006; Pettersson, 2005). Whilst it should be noted that collisions may have occurred between observation periods, the number of birds observed interacting with turbines without colliding suggests that collisions are likely to be rare events (Table 3).

Of the studies we identified, only six provided sufficiently detailed descriptions of birds' interactions with wind turbines to characterise micro-avoidance (Table 3; Desholm, 2005; Krijgsveld et al., 2011; RPS, 2011; Schulz et al., 2014; Thaxter et al., 2017b; Wild Frontier Ecology, 2013), although these reflected significant effort across multiple sites. Despite this effort, there were very few records of birds flying close enough to turbines to require micro-avoidance. Indeed, Desholm (2005) did not record any birds passing within 20 m of a turbine. Similarly, a detailed analysis of two GPS-tagged Lesser Black-backed Gulls indicated that these birds significantly avoided entering the turbine rotor-swept area (Thaxter et al., 2017b). Across the remaining studies only 59 birds were recorded as passing close to the turbine rotor-swept area, of which 54 were recorded as taking action to avoid the rotor-swept area (Table 3). The data for micro-avoidance would appear to be consistent with those reported above for meso-avoidance, suggesting that a last second escape reflex may be required relatively rarely as, within a wind farm, avoidance behaviour is driven by a high level of anticipatory or impulsive evasion. However, given the differences in the species recorded interacting with turbines and the relatively low number of birds recorded, it is difficult to draw firm conclusions about the extent of micro-avoidance behaviour.

3.7. Within-wind farm avoidance

We identified nine coastal sites (Table 3) from which data describing the recorded number of collisions were available from the same time periods as estimates of the total number of birds passing through turbine rotor-swept areas. Based on the data presented in the studies highlighted in Table 3, it was possible to calculate species-specific within-wind farm avoidance rates for herring gull and lesser black-backed gull, as well as rates for small gulls (e.g. black-legged kittiwake), large gulls (e.g. great black-backed gull) and all gulls (Table 4) but not for northern gannet.

These analyses confirmed that within-wind farm avoidance rates were likely to be very high (> 0.99) (Table 4). Avoidance rates were similar between species with rates of 0.9959 for herring gull and 0.9982 for lesser black-backed gull. We also estimated group-specific avoidance rates of 0.9921 for small gulls (birds identified as little, common or black-headed gulls), 0.9956 for large gulls (those identified as lesser black-backed, herring or great black-backed gull or described as large gull spp.) and 0.9893 for all gulls (those identified to species level or described as large gull, small gull or gull spp.).

Whilst the level of precision expressed by these values may seem excessive, it should be noted that it is the non-avoidance rate (1-avoidance) which is incorporated in the collision risk model. When

Table 2
Vertical meso-avoidance rates obtained for the five priority species and for birds classified as unidentified gulls from comparisons of the number of birds at rotor height pre- and post-construction, or the number of birds at rotor height inside and outside a wind farm. Values of 0 reflect no increase or decrease in the proportion of birds at rotor height, values > 0 reflect a decrease in the proportion of birds at rotor height (avoidance) and values < 0 reflect an increase in the proportion of birds at rotor height (attraction).

	Barrow (Barrow Offshore Wind Limited, n.d.)	Egmond aan Zee (Krijgsveld et al., 2011)	Gunfleet Sands 2010/11 (GoBe Consultants Ltd., 2012; NIRAS Consulting, 2011)	Gunfleet Sands 2011/12 (GoBe Consultants Ltd., 2012; NIRAS Consulting, 2011)	Robin Rigg (Natural Power, 2013)
Northern gannet	−0.59	0.49			
Black-legged kittiwake	−0.41	0.20	−0.47	0.05	−1.00
Lesser black-backed gull	0.72	attraction	−0.44	0.00	
Herring Gull	0.35	No change	−0.02	0.11	−8.00
Great black-backed gull	0.28	avoidance	−0.75	−0.53	−0.67
“small” gulls		−0.26			
“large” gulls		no change			
Gull spp.	−0.85	avoidance	−1.98	−1.13	

presented to four decimal places, the non-avoidance rates typically had 2 significant figures (Table 4), which we feel reflects a reasonable level of precision. The difference between an avoidance rate of 0.995 and 0.9959 would result in an 18% increase in the collision rate predicted from the CRM. We would also argue that this level of precision is justified given the estimated level of uncertainty surrounding each value (Table 4).

3.8. Recommended total avoidance rates

Evidence suggests that the response of gulls to turbines at a macro scale may be highly variable. Consequently, at present, we consider all gull species (including kittiwake) to have an average macro-avoidance rate of zero. Few studies were available with which to draw conclusions about meso- and micro-avoidance in gulls. Consequently, the total avoidance rates for gulls can be considered to be equal to the within-wind farm avoidance rates. However, the evidence base for macro-avoidance in gulls was limited meaning it was not possible to produce robust estimates of uncertainty surrounding macro-avoidance rates. Therefore, when combining macro- and within-wind farm avoidance rates, we are not able to give an estimate of uncertainty surrounding the total avoidance rate. Additionally, given the limited evidence base for macro-avoidance, we present the total avoidance rate to three, rather than four, significant figures and round down in order to be precautionary. We recommend total avoidance rates of 0.998 for lesser black-backed gull and 0.995 for herring gull. Based on flight behaviour and morphology, we believe it is reasonable to include the great black-backed gull in the large gull spp. grouping, and the black-legged kittiwake in the small gull spp. grouping. We therefore recommend total avoidance rates of 0.995 and 0.992 respectively for these species.

Fewer data were available to support a total avoidance rate for northern gannet. However, given the evidence of strong macro-avoidance of wind farms, it was felt that the total avoidance rate was unlikely to be below that obtained for all gulls. Consequently, a rate of 0.989 is recommended for northern gannet.

4. Discussion

May (2015) suggests that alertness is likely to increase with decreasing distance to turbines, meaning birds are more likely to take action as they get closer to a turbine. We believe our review supports this hypothesis as, despite significant survey effort, we uncovered very little evidence of birds approaching turbines close enough to be at risk of collision. Of those that did, a high proportion were recorded taking last-second action to avoid collision, termed an escape response by May (2015). In a behavioural context, this suggests that most avoidance action is likely to be caused by functional habitat loss or anticipatory or impulsive evasion, rather than a last second escape reflex. There was also evidence to suggest that the avoidance rate may vary in relation to both the position of a turbine in an array and whether or not turbines are operational (Krijgsveld et al., 2011; Mendel et al., 2014; Petersen et al., 2006; Schulz et al., 2014; Winkelman, 1992), a conclusion consistent with the predictions made by May (2015). Such responses highlight the ability of some species, particularly gulls, to adapt to the presence of wind turbines.

4.1. Use of avoidance rates in collision risk models

Previous guidance of the use of avoidance rates in CRMs was that 0.98 should be considered the default value for seabirds (Scottish Natural Heritage, 2010). Whilst significant gaps in knowledge remain, this review highlights that, for the species most likely to be affected by collision, avoidance rates are estimated to exceed 0.99. Whilst this may seem a trivial difference, it will result in the predicted collision rate being more than halved. These avoidance rates are applicable to models such as Band (2012), as well as others including the models of Tucker (1996) and Eichhorn et al. (2012). However, care must be taken when using these avoidance rates in models which account for the vertical distribution of birds when estimating the probability of collision (e.g. the Extended Band Model Band, 2012). Accounting for the vertical distribution of birds will reduce the number of collisions predicted in the absence of avoidance as the number of birds within the central, and more risky part of the rotor-swept area will be reduced (Johnston et al., 2014). Consequently, within-wind farm avoidance rates suitable for use with models such as the extended Band model (Band, 2012), which do account for vertical distribution, are likely to be lower than those suitable for use with simpler models. At present, insufficient data are available with which to estimate robust avoidance rates for use in the extended Band model for most species. However, with ongoing data collection in the offshore environment, for example through the UK Offshore Renewables Joint Industry Project (Davies et al., 2013), it is to be hoped that this review will help inform for the collection of appropriate data in future.

4.2. Limitations

At present, our recommended avoidance rates only consider horizontal avoidance. We identified some evidence suggesting birds may alter their flight altitudes when within a wind farm in order to reduce collision risk (Table 2). However, this evidence was inconclusive and further studies are required in order to fully understand vertical avoidance behaviour. Technological advancements, for example, the

Table 3

Data sources used to estimate micro-avoidance and within-wind farm avoidance rates for marine species. Rows in bold indicate sites from which data were used to derive within wind farm avoidance rates.

Wind Farm (citation)	Survey Method	N Hours observations	N Turbines Covered	N Birds recorded during point counts	Reported Fatalities (N collisions directly observed)	Behavioural interactions with turbines
Alpha Ventus (Schulz et al., 2014)	Remote Camera	8741	1	241	< 1 (0)	Of 14 objects reliably identified as birds, at least 12 had successfully passed through the rotor swept area of the turbine. Whilst collisions were assumed, none were directly recorded by the cameras
Avonmouth (The Landmark Practice, 2013)	**Visual**	**108**	**3**	**5616**	**1 (0)**	
Blyth (Rothery et al., 2009)	Visual	352	2	8534	0 (0)	
Blyth Harbour (Newton and Little, 2009)	Visual	93	9	791	1410–1,838[a] (0)	
Boudwijnkanaal (Everaert, 2014)	**Visual**	**34**	**5–7**[b]	**1847**	**12 (0)**	
Bouin (Dulac, 2008)	**Visual**	**370**	**8**	**8243**	**30 (0)**	
De Put (Everaert, 2014)	**Visual**	**18**	**2**	**54**	**2 (0)**	
Egmond aan Zee (Krijgsveld et al., 2011)3	Visual		6	1610	0 (0)	Of 36 birds (2 lesser black-backed gulls, 4 great black-backed gulls, 2 starlings, 28 skylarks) recorded within 50 m of a turbine, 33 were recorded as being beyond the reach of the turbine blades
Gneizdzewo (Zielinski et al., 2012, 2011, 2010, 2008)	**Visual**	**620**	**19**	**4443**	**1 (0)**	
Greater Gabbard (RPS, 2011)	Visual	36	7	189	0 (0)	1 kittiwake noted making an evasive manoeuvre to avoid collision, no other birds reported close enough to turbines to require evasive manoeuvres
Groettocht (Krijgsveld et al., 2011)	Radar	39	7	6825	5 (0)	
Haverigg (RPS, 2011)	Visual	42	8	836	0 (0)	
Hellrigg (Percival, 2012, 2013)	**Visual**	**74.5**	**4**	**26,638**	**1 (0)**	
Kessingland (Wild Frontier Ecology, 2013)	**Visual**	**36**	**2**	**3535**	**3 (0)**	**5 black headed gulls, 2 lesser black-backed gulls and 1 herring gull reported taking evasive action within 50m of turbines. No birds observed colliding**
Kleine Pathoweg (Everaert, 2014)	Visual	16	7	672	0 (0)	
Nysted (Desholm, 2005)	Remote Camera	476	1	55	0 (0)	Despite potential to record birds interacting with turbines, no birds were recorded within 20 m of a turbine
Oosterbierum (Winkelman, 1992)3	**Radar**		**18**	**202,400**	**49 (0)**	
Walney I, Walney II, West of Duddon Sands, Ormonde & Barrow Offshore Wind Farms (Thaxter et al., 2017b)	GPS Tag	2112	270	2	0 (0)	2 lesser black-backed gulls spent 1.2% and 2.7% of their time within a 3-dimensional rotor swept area around turbines, neither collided with the blades
Waterkaaptocht (Krijgsveld et al., 2011)	Radar	39	8	14,430	6 (0)	
Yttre Stengrund (Pettersson, 2005)	Visual	219.5	5	404,146	4 (4)	
Zeebrugge (Everaert, 2014)	**Visual**	**43.7**	**4**	**2491**	**7 (0)**	

[a] Extrapolated from mean annual collision rates corrected for corpses lost at sea or undetected by observers.

[b] Five turbines covered in 2001, seven turbines in 2005.3Total time not stated.

Table 4
Within-wind farm avoidance rates for seabirds.

Species	N birds observed (N collisions recorded)	Non-avoidance rate	Within-wind farm avoidance rate (± SD)
Lesser black-backed gull	101,746 (2)	0.0018	0.9982 (± 0.0005)
Herring gull	546,047 (9)	0.0041	0.9959 (± 0.0006)
Small gull spp.	1,598,953 (42)	0.0079	0.9921 (± 0.0015)
Large gull spp.	639,560 (14)	0.0044	0.9956 (± 0.0004)
Gull spp.	2,567,124 (139)	0.0107	0.9893 (± 0.0008)

availability of radar (Kunz et al., 2007; Skov et al., 2016; Ward et al., 2016) and GPS tags (Corman and Garthe, 2014; Garthe et al., 2017; Thaxter et al., 2017b) which can collect detailed information about the movement patterns of individual birds may mean these data could be collected in the near future. Combining horizontal and vertical avoidance rates in order to derive a three-dimensional avoidance rate is unlikely to be straightforward as birds may employ both strategies at the same time, meaning simple formulae like equation (2) are unlikely to be appropriate. However, approaches such as that used with GPS tracking data by Thaxter et al. (2017a,b) may prove valuable.

Within wind farms (i.e. at meso- and micro-scales) a lack of data from the offshore environment is an issue, particularly in relation to northern gannet. Whilst data from terrestrial sites are informative about how birds may interact with individual turbines, evidence suggests that flight behaviour may differ between onshore and offshore environments (Corman and Garthe, 2014; Ross-Smith et al., 2016) potentially affecting how birds respond to turbines and therefore avoidance rates. Whilst we have attempted to minimise the impact of this by focussing on data collected from coastal locations, some differences may remain, notably in relation to flight height and speed (Corman and Garthe, 2014; Ross-Smith et al., 2016; Spear and Ainley, 2008). Birds tend to fly higher over land than offshore and, there is also a greater tendency for them to fly at altitudes within the rotor-swept area of turbines in the terrestrial environment (Corman and Garthe, 2014; Ross-Smith et al., 2016). This greater potential exposure to turbine blades means that collision rates in the terrestrial environment may be greater than is the case offshore. Seabird flight speed can be strongly influenced by weather conditions, particularly wind strength and direction (Shamoun-Baranes and van Loon, 2006; Spear and Ainley, 2008), potentially affecting their manoeuvrability and ability to respond to obstacles. As wind conditions can differ markedly between the onshore and offshore environments, this is likely to have implications for collision risk. However, at present insufficient data are available to enable us to understand in which direction this may influence collision risk.

The within-wind farm avoidance rates presented here are higher than those derived by Everaert (2014). This may partly reflect the broader range of sites covered by our review, especially as some of the sites covered in Everaert (2014) appear to have particularly high collision rates. In particular Everaert (2014) highlights the proximity of one wind farm to a breeding colony as a key reason for a high collision rate. This highlights the importance of considering site-specific variation in avoidance behaviour, which we have attempted to capture by estimating confidence intervals around our within-wind farm avoidance rates. However, even when we estimate within-wind farm avoidance rates from similar data, the values derived as part of our review differ from those presented by Everaert (2014) (Table S3). A key reason for this is that, in order to ensure data were consistent with those collected elsewhere, we have only considered data where no spatial or temporal extrapolation was required in order to combine collision and flight activity data. We recognise that there are a number of ways in which avoidance rates can be derived, and that small differences in the way some parameters are derived (i.e. passage rate), and biases due to survey technique, can strongly influence the final estimated avoidance rates. This is undesirable as it can increase uncertainty in the consenting process, increasing costs for those involved. For this reason, we strongly suggest that authors provide detailed calculations showing how the rates presented have been estimated in order to enable readers to come to an informed decision about the results (see Table S3).

To date, there has been little consideration of factors which are likely to influence avoidance behaviour and to what extent there is seasonal- or site-specific variation in the offshore context. Avoidance rates for non-seabird species at onshore wind farms have been reported to vary by site and even within wind farms (Garvin et al., 2011), as well as by season, whether birds are resident or migrants and the relative distance to the wind farm from roost sites or nest locations (Campedelli et al., 2014). It is apparent, therefore, that the magnitude of any avoidance behaviour is likely to be linked to the ecological importance of a site to a species at a given point in time, and how it is being used. Seabirds act as central place foragers during breeding (Stephens and Krebs, 1986; Thaxter et al., 2012). This may manifest itself in spatial differences in behaviour, dependent upon whether the area covered by an offshore wind farm is used for active foraging or for commuting between foraging grounds and the breeding colony. Such behavioural differences may be associated with varying levels of collision risk and avoidance behaviour. There may also be a temporal element to avoidance behaviour. Stage-dependent changes in foraging behaviour between the incubation and early chick-rearing period have explained the change in spatial overlap with offshore wind farms (Thaxter et al., 2015). The presence of newly fledged birds in the population towards the end of the breeding season may also affect avoidance rates as these naive individuals may unintentionally engage in riskier flight behaviour (Henderson et al., 1996). Individual seabirds may also show consistency in their preferred foraging areas (Irons, 1998; Soanes et al., 2013) or have limited alternative habitats available. Where wind farms overlap with these preferred foraging areas, displacement may be less likely and macro-avoidance rates therefore lower for these individuals. These studies suggest that there are likely to be both spatial and temporal elements to avoidance behaviour for seabirds, neither of which have yet been properly quantified. There is also some evidence to suggest that group size and social interactions can influence the likelihood of collision and hence by association, the avoidance behaviour of birds (Croft et al., 2013). Other factors which influence collision risk have also been reviewed extensively (Marques et al., 2014; May et al., 2015; Thaxter et al., 2017a; Wang et al., 2015) and include aspects of: species characteristics (morphology, flight behaviour, sensory perception, phenology); site features (landscape, food availability, weather); and wind farm features (type of turbines and design of array).

4.3. Future data collection — displacement and functional habitat loss

Whilst this study has advanced our understanding of avoidance behaviour of seabirds in relation to offshore wind farms, a number of significant gaps in knowledge remain. Collecting the data necessary to quantify avoidance behaviour in relation to offshore wind farms can be extremely costly and therefore requires well designed studies involving both industry and regulators (e.g. Davies et al., 2013). The cost and challenging nature of these studies means that it is important to utilise robust analytical approaches that make the most of any data collected.

Studies of displacement/attraction have typically used Before-After-Control-Impact (BACI) survey design (Stewart-Oaten et al., 1986) but have been hampered by inadequate survey design notably gaps in spatial or temporal coverage and inappropriate choice of control sites

(Marine Management Organisation, 2014). Recently developed approaches, such as Before-After-Gradient (BAG) analyses are increasingly used to assess the impacts of wind farms with the focus on collecting data over much more extensive areas around the wind farm site starting in the pre-construction period (Jackson and Whitfield, 2011; Mackenzie et al., 2013; Marine Management Organisation, 2014; May, 2015). By incorporating environmental covariates (e.g. sea surface temperature, tidal cycles) to help describe spatial and temporal variation in seabird distributions and abundance at sea, further changes associated with the construction and operation of wind farms can be more accurately attributed (Mackenzie et al., 2013), and therefore better inform macro-avoidance rates. Species assumed to be at risk of displacement (Furness et al., 2013), tend to have estimates of avoidance based largely on data collected at the macro-scale. In the case of species for which displacement is not perceived to be a significant issue, for example gulls, there is often less focus on data collection at this scale, meaning the macro level response to wind farms is often less well understood. By focussing data collection on the scale perceived to be most relevant for the species concerned, there is a risk that avoidance behaviour at other spatial scales is overlooked. Whilst this is primarily an issue for data collected using observational surveys, it may also be an issue for data collected using radar depending on the range over which the system operates.

To help to provide a better evidence base for macro-avoidance, future analyses should distinguish between birds in flight and those on the water, as only those in flight are at risk of collision. Ideally, such studies should also incorporate measurements of flight altitude so that birds flying above, or below, the collision risk window can be excluded from subsequent analyses. However, in collecting these data a key consideration needs to be whether the survey has sufficient power to detect change between the pre- and post-construction periods. The power to detect change is related to a variety of factors including the frequency of, and area covered by, the surveys as well as inherent spatial and temporal variability in seabird distribution and relative abundance (Maclean et al., 2013; Pérez Lapeña et al., 2010). This is a particular issue where the pre-construction population of a species is small, and is always likely to be an issue where baseline sampling has not taken account of statistical power for detection of change. This exacerbates the risks of a change in the number of birds using a site either giving the false impression of a significant effect (false positive response) or where no change is found, the results are wrongly interpreted as a lack of response to the presence of the wind farm by the particular species (false negative response). A recent review (Marine Management Organisation, 2014) of post-consent monitoring of offshore wind farms concludes that the power to detect such changes by existing studies is likely to be low and the responses of seabirds to wind farms may have been incorrectly quantified. Careful consideration must also be given to biases associated with survey methodology. In particular, data collected from different platforms (e.g. visual aerial surveys vs digital aerial surveys) can give very different estimates of abundance (Buckland et al., 2012). Consequently, when estimating macro-avoidance based on displacement as functional habitat loss, it is important to ensure that the data used to do so are directly comparable.

Ideally, the effect size and associated confidence intervals should always be reported as standard in the results of ecological studies (Masden et al., 2015; Nakagawa and Cuthill, 2007). However, of the studies we considered, only Vanermen et al. (2015) and Natural Power (2014) did so in respect to the studies of displacement and attraction. If these practices were adopted as standard when measuring avoidance behaviour, not only would it make it more straightforward to quantify avoidance rates and compare across studies, it would also give us an understanding of the uncertainty and variability surrounding these rates.

4.4. Future data collection — anticipatory or impulsive evasion

Radar can be deployed in order to investigate anticipatory or impulsive evasion of wind farms or turbines. However, deriving species-specific avoidance rates from data collected in this way can be challenging given the difficulty of identifying species from radar tracks. Where species-specific macro-avoidance rates have been derived using radar, this has been possible because the majority of tracks could be assigned to a single species (e.g. during mass migration events when only a few species are represented; Desholm and Kahlert, 2005; Petersen et al., 2006). However, recent studies have demonstrated effective use of radar monitoring in combination with visual observations in order to be able to identify more complex suites of species moving in and around wind farms (Skov et al., 2012). Ideally these studies should also aim to collect data on the vertical distribution of birds and in-flight changes in behaviour (e.g. flight speed and turning angles).

4.5. Future data collection — escape response

In order to collect data describing micro-avoidance, carefully designed experiments and analyses are required. Approaches such as the use of turbine mounted cameras (Desholm, 2005) may be suitable, but must be capable of detecting abrupt changes in flight direction and/or altitude. Given that micro-avoidance behaviour is likely to be an extremely rare event, careful consideration must be given to ensure that any methods used have the necessary statistical power to estimate robust avoidance rates.

5. Conclusions

Our study assesses the evidence for avoidance behaviour in five key seabird species, perceived to be at particular risk of collision, at three different spatial scales. We have demonstrated how the different types of data which have been collected fit within the framework for describing avoidance behaviour developed by May (2015). Whilst we have done this in the context of offshore wind farms, this approach is also likely to be applicable to other situations where collision risk is likely to be an issue, for example in relation to tidal turbines.

Lack of data on avoidance behaviour has been acknowledged as an issue for some time (Chamberlain et al., 2006). As the wind industry has developed both onshore and offshore, the evidence base has developed. This review summarises the evidence that has been collected to date and represents a significant step forward by presenting estimates of avoidance behaviour for five seabird species. It is important to acknowledge that these values are largely based on data from coastal, rather than offshore locations. However, in our opinion, this remains the best available evidence with which to quantify avoidance behaviour in seabirds. Significant knowledge gaps remain and key areas to be addressed include distinguishing between vertical and horizontal avoidance and gaining a better understanding of how seasonal and spatial processes may influence avoidance behaviour. This is particularly important given the rapid growth of the offshore wind sector and the potential for the cumulative impacts of collisions from multiple wind farms on species and populations of concern (Brabant et al., 2015; Busch and Garthe, 2017).

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Appendix A. Supplementary data

Supplementary data related to this article can be found at http://dx.doi.org/10.1016/j.marenvres.2018.06.017.

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BTO Research Report No. 665

The Scientific Validity of Criticisms made by the RSPB of Metrics used to Assess Population Level Impacts of Offshore Wind Farms on Seabirds

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1. INTRODUCTION

The purpose of this report is to consider the validity of criticisms that have been made by the RSPB on the use of risk-based metrics to assess the response of seabird populations to impacts from offshore wind farms. Following the project start-up meeting on 3rd December 2014, it was agreed that this consideration should also include approaches that have been used to set thresholds of additional mortality including Potential Biological Removal (PBR), Acceptable Biological Change (ABC) and reduced uncertainty Acceptable Biological Change (ruABC).

A variety of metrics and methodologies have been used to assess the population level impacts of effects associated with proposed offshore wind farms (Table 1). With respect to the recent assessment of potential impacts of offshore wind farm projects on populations of seabirds in Scotland, the RSPB has made three key criticisms of the approaches that have been used (Green 2014):

1. ***Procedures for calculating effects of wind farms on seabird per capita mortality rates and breeding success as a result of collision, displacement and barrier effects do not have a firm foundation in empirical data, so scientifically robust, and therefore defensible, collision rate values and confidence intervals for collision mortalities and displacement and barrier effects cannot be calculated.***
2. ***As a consequence of (1), no probabilistic methods for assessing risk of population impacts of a given type and magnitude are scientifically robust and therefore defensible. This applies to both the Acceptable Biological Change (ABC) method, reduced uncertainty Acceptable Biological Change (ruABC) as well as other proposed methods to estimate the difference in the probability of a specified population outcome (e.g. a population decline) between impacted and unimpacted scenarios.***
3. ***Attempts to identify thresholds either side of which estimates of population-level impacts of developments are considered negligible (or non-negligible) on biological grounds are mistaken and should be abandoned.***

In addition to these general criticisms, Green (2014) makes more specific criticisms of some of the approaches used, notably PBR and ABC. In their written[1] and oral[2] responses to the Hornsea Project One Offshore Wind Farm, the RSPB indicate further criticisms of the approaches that have been used to date. The RSPB suggest that the most appropriate approach is to use PBR in an initial screening process to determine which projects will clearly have an unacceptable impact on populations. For the remaining projects, they suggest that a density-independent Leslie matrix model should be used to estimate the population size at the end of the lifetime of the project with and without the demographic impacts of the wind farm using matched simulations (Green 2014).

The British Trust for Ornithology (BTO) is an independent research organisation that provides data and analysis to inform decisions impacting on biodiversity and the environment. Our approach is strictly impartial and based on rigorous science. We have contributed to the development of guidance for the offshore wind industry through projects under the Collaborative Offshore Wind Research into the Environment (COWRIE) programme, operated the Strategic Ornithological Support

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[1] http://infrastructure.planningportal.gov.uk/wp-content/ipc/uploads/projects/EN010033/2.%20Post-Submission/Representations/Comments/Other%20Comments/Royal%20Society%20for%20the%20Protection%20of%20Birds.pdf

[2] http://infrastructure.planningportal.gov.uk/wp-content/ipc/uploads/projects/EN010033/2.%20Post-Submission/Hearings/Issue%20Specific%20Hearing%20-%2029-04-2014%20-%200930%20-%20Humber%20Royal,%20Grimsby/Royal%20Society%20for%20the%20Protection%20of%20Birds.pdf

Services (SOSS) Secretariat on behalf of The Crown Estate for the period 2010-12, and provided substantial input to published reviews on seabird flight heights (Cook *et al.* 2012, Johnston *et al.* 2014), avoidance rates (Cook *et al.* 2014), and post-consent monitoring (MMO 2014). Building on this expertise and impartial approach, the BTO was invited to comment on the points raised in the RSPB report, "*Misleading use of Science in the assessment of probable effects of offshore wind projects on populations of seabirds in Scotland*" (Green 2014), and the RSPB written[1] and oral[2] representations in relation to the proposed Hornsea Project One offshore wind farm. We first identify the range of different approaches that have been taken in the assessment of offshore wind farms on seabird populations before, second, considering both the general and specific criticisms made by the RSPB of the methodologies used to assess the population level impacts of offshore wind farms on seabirds. The purpose of this report is, specifically, to consider the criticism of metrics that are used to assess population level impacts of offshore wind farms on seabirds, put forward in Green (2012) and in relation to the Hornsea Project One offshore wind farm[1,2], and how these metrics may be used in the future. We do not critique the use of these approaches in past assessments, nor analyse the casework undertaken in relation to any particular offshore wind farm applications.

As part of BTO's Quality Assurance procedures, this report has been internally reviewed by the BTO Director of Science, Dr James Pearce-Higgins, and by the Head of the Wetland and Marine Research Team, Dr Niall Burton. The report has also been commented on by the project steering group consisting of representatives from the Joint Nature Conservation Committee, Natural England, Scottish Natural Heritage, Marine Scotland Science, Natural Resources Wales and Department of Environment, Northern Ireland. In order to maintain the impartiality of this work, the BTO were under no obligation to address any of the comments raised by the project steering group, however, we incorporated those comments that improved the clarity of writing or corrected factual inaccuracies. Throughout this report, references to "we" solely reflect the views of the BTO, and not the positions of the project steering group. We consider each of the general, then specific, criticisms raised in turn, and, to aid the reader, we provide a brief summary of the key points for each in *italics*.

2. METRICS USED TO ASSESS POPULATION LEVEL IMPACTS OF OFFSHORE WIND FARMS ON SEABIRDS

There is widespread concern that offshore wind farm developments may impact on seabird populations (*e.g.* Drewitt & Langston 2006; Everaert & Stienen 2007). There is, therefore, much effort being invested in understanding the potential effects (i.e. collision, displacement and barrier effects) of such developments and their impacts on seabirds at a population level, with a range of predictive approaches being employed in an attempt to quantify these future impacts.

Having considered projects currently listed on the infrastructure planning portal website[3], the Marine Scotland Licensing Portal[4] and other recent, high profile examples, we identified 27 proposed sites at which the population level impacts of offshore wind farms on seabirds had been considered during assessment: Aberdeen Offshore Wind Farm, Beatrice, Burbo Bank Extension, Docking Shoal, Dogger Bank Creyke Beck A, Dogger Bank Creyke Beck B, Dogger Bank Teesside A, Dogger Bank Teesside B, Dudgeon, East Anglia One, Fife Wind Energy Park, Galloper, Hornsea Project One, Inch Cape, London Array Phase II, MacColl, Navitus Bay, Neart na Gaoithe, Race Bank, Rampion, Seagreen Alpha, Seagreen Bravo, Stevenson, Telford, Triton Knoll 3, Walney I & Walney Extension. Across these 27 sites, we identified 12 metrics, many of which are necessarily inter-related, that have been used to estimate population level effects of offshore wind farms (Table 1). **For the purposes of this report, we define a metric as any value or rule upon which a decision about whether or not a population level effect associated with the impacts of an offshore wind farm is deemed acceptable.**

Table 1 Description of metrics used to estimate population level impacts of proposed offshore developments.

Potential Biological Removal (PBR)
PBR was initially developed for use in marine mammal populations (Wade 1998). PBR provides an estimate of the maximum level of mortality, in addition to that expected to occur naturally, which a population can experience and still remain viable. A key advantage of PBR is that it requires very little input data, only the minimum current population size, mean age at first breeding and mean adult survival (Niel & Lebreton 2005). In addition, it is relatively simple to calculate following the methodology set out by Dillingham & Fletcher (2008). This simplicity makes PBR an extremely attractive approach with which to assess the potential population level impacts of offshore wind farms on seabirds.
Population growth rate
The population growth rate measures the extent to which the size of the breeding population changes on an annual basis. By considering the growth rate of the population in the presence of an offshore wind farm, it should be possible to consider whether the population will remain stable (growth rate=1), increase (growth rate>1) or decrease (growth rate<1) through the life time of the project.
Probability that growth rate <1
As part of the SOSS programme, guidance was produced for using Population Viability Analysis (PVA) to assess the potential impacts of collision-related mortality associated with offshore wind farms (WWT Consulting 2012). Under a PVA approach, stochastic models are used to simulate the impact of additional mortality on populations of species of interest and the proportion of

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[3] http://infrastructure.planningportal.gov.uk/

[4] http://www.scotland.gov.uk/Topics/marine/Licensing/marine/scoping

simulations where the population declines (i.e. growth rate <1) calculated.

Probability that population decreases below initial size

The impact of a development is typically assessed in relation to a baseline population size, which may be either the pre-construction population size, the population size of a protected site at designation, the population size from Seabird 2000 (Mitchell *et al.* 2004), or some other appropriate value. Using stochastic models, the proportion of simulations in which the population drops below this baseline, either at any point in the lifetime of the project or by the end of the project, could be assessed. Alternative baseline populations, for example, the size of the population at designation in the case of a breeding colony at a protected site, could be used. Mathematically, this metric is nearly identical to the previous metric.

Probability of a population being a given magnitude below the median size predicted in the absence of an impact

With the simulations from stochastic models, rather than looking at the probability or magnitude of a decline, it may be more meaningful to estimate the median population size estimated across all simulations. This could be done either for a single fixed point in time, or at given intervals. A metric to assess the population level impact of a development could be derived by estimating a median size for a population in the absence of it and then calculating the proportion of simulations for a population in the presence of the development that are (a given magnitude) below this median population size.

Ratio of median impacted to unimpacted growth rate

Considering the growth rate of a population only in the presence of an offshore development enables an assessment of whether the population will remain stable, increase or decrease over time, but it does not make it possible to quantify the impact of the development on that growth rate. By comparing the growth rate of the population in the presence of a development to that expected in its absence it is principle possible to quantify what annual impact the development is having on a population.

Ratio of impacted to unimpacted population size

Population models can be used to estimate the size of a population through time both with and without the impact of an offshore development. Comparing the ratio of the size of these two populations offers a relatively easy to interpret statistic with which to assess the population level impact. The ratio could be derived either from a simple deterministic model, or taken from the mean or median values simulated using a more complex stochastic model, with or without density-dependence. The ratio of population sizes could be estimated either at a fixed point in time, e.g. the end of a project, or at a series of intervals throughout the life time of a project.

Change in probability that growth rate <1

Where simulations show that a population may already be at risk of declining in the absence of a development, for example if more than 50% of simulations have a growth rate <1, simply quantifying the probability of a population decline in the presence of an offshore development may not be meaningful. To assess the population level impact it would be necessary to determine how much greater the probability of a decline is in the presence of the development than in its absence. This could be done either at a single fixed point in time, or at intervals throughout the life time of the project.

Change in probability of a population decreasing by a given magnitude

At many colonies throughout the UK, seabird populations are already declining (JNCC 2014). As a consequence, the presence of a development is unlikely to increase the probability of the growth rate at these colonies being <1, especially if all the simulations from the baseline scenario already have a growth rate <1. However, the presence of the development may cause a further reduction in the magnitude of growth rate. It may, therefore, be more meaningful to consider the change in probability of a population decreasing by a given (though almost certainly artificial) threshold, e.g. a 10% increase in the probability of a 5% decline.

Probability of growth rate being x% less than unimpacted growth rate

With growth rates simulated from stochastic models, it may be desirable to estimate a mean or median value for the unimpacted population and calculate the proportion of simulations in which the growth rate of the impacted population is lower, or a given percentage lower, than this value. This approach has the advantage of allowing a probabilistic forecast of the impact of the offshore development on a population, e.g. there is a 50% chance that the development will reduce the population growth rate by 10%.

Acceptable Biological Change (ABC)

ABC was set out as a method for assessing the population impact of an offshore wind farm by Bennet (2013). Using terminology from the Intergovernmental Panel on Climate Change (IPCC), ABC attempts to set out an acceptable risk to a population. ABC allows for a change of up to one-third in the probability of a defined management target being achieved as a result of the impact of an offshore wind farm. For example, a management objective could be that the population size at the end of the life time of the project should be that which is more likely than not (i.e. in IPCC terminology has a probability of 0.667 or more) to occur in the absence of the project. If the impacted population size is greater than that which is expected to occur with a probability of 0.667 in its absence, the impact could be deemed acceptable. The approach allows also alternative targets, e.g. in reference to the site's conservation status, to be set.

Reduced Uncertainty Acceptable Biological Change (ruABC)

An acknowledged weakness of ABC is that when there is larger uncertainty surrounding the input (demographic) parameters, this can result in wider confidence limits surrounding the estimated population sizes, and therefore greater declines being deemed acceptable (JNCC & SNH 2014). As a result, JNCC and SNH refined the methodology to account for this discrepancy, referred to as Reduced Uncertainty Acceptable Biological Change. The approach considers the model prediction uncertainty, taken to be the difference between the population predicted with a probability of 0.5 and the population predicted with a probability of 0.667. The incorporation of additional data from the regional population can help reduce the model prediction uncertainty. The model prediction uncertainty from the regional population can then be applied to the median population size for the colony of interest to estimate an acceptable level of biological change.

3. GENERAL CRITICISMS BY THE RSPB OF APPROACHES USED TO DATE

3.1. Procedures for calculating effects of wind farms on seabird per capita mortality rates and breeding success as a result of collision, displacement and barrier effects do not have a firm foundation in empirical data, so scientifically robust and therefore defensible collision rate values and confidence intervals for collision mortalities and displacement and barrier effects cannot be calculated – p3 of Green (2014)

Whilst this criticism is not directly related to the metrics used to assess population level effects of offshore wind farms on seabirds, it does relate to the way in which predicted impacts are incorporated into models used to assess population level effects. There is significant uncertainty associated with the assessment of offshore wind farms on seabird populations (Stewart *et al.* 2007, Masden *et al.* 2014). This uncertainty has been the subject of much discussion for some time (Fox *et al.* 2006, Drewitt & Langston 2006, Chamberlain *et al.* 2006) and has led to significant work programmes, including the COWRIE and SOSS programmes, and other associated research-based advice. This uncertainty has led to considerable debate about what constitutes a precautionary approach when estimating the impact of an offshore wind farm.

As more offshore wind farms become operational, increasing amounts of data are also now being collected in order to address these uncertainties. Whilst inconsistencies in data collection methodologies can make interpreting these data challenging (MMO 2014), they do reflect a growing evidence base from which to draw conclusions about the potential impacts of the effects associated with offshore wind farms on seabird populations.

Key to understanding the likely population level impacts of offshore wind farms on seabirds is an understanding of how many birds a development may affect. This requires a detailed understanding of how many birds are present in an area prior to construction, and therefore at risk from collision, barrier effects or displacement. As a consequence, there is a requirement to carry out detailed surveys in order to estimate the population size of birds within the proposed development area (Camphuysen *et al.* 2004). These data may be collected using boat and/or aerial surveys and sophisticated modelling approaches are being developed in order to generate robust population estimates from these data (e.g. Johnston *et al.* 2015). However, there remains uncertainty still over the breeding populations from which these birds originate, making an assessment of population-level impacts more challenging.

Using radar to study the movement of birds in and around wind farms has enabled some quantification of barrier effects (e.g. Desholm & Kahlert 2005, Masden *et al.* 2009, Plonczkier & Simms 2012). However, these studies have so far focussed on migrant waterbirds (geese and ducks) rather than seabirds foraging in and around the areas developed as offshore wind farms.

Several recent studies have attempted to quantify displacement rates of seabirds in relation to offshore wind farms. These studies have revealed that displacement may be highly species-specific with a range of responses recorded covering total avoidance, attraction and no response (Petersen *et al.* 2006, Leopold *et al.* 2011, Natural Power 2014, Vanermen *et al.* 2013, 2014). Whilst estimates of both displacement rates and barrier effects are available, interpreting this evidence has been hampered by inconsistencies and methodological issues in post-consent monitoring programmes (MMO 2014). Despite this, the growing evidence base for displacement means that rates can be estimated for a number of species.

As seabirds are typically long-lived and able to delay breeding attempts in unfavourable conditions, it is believed that the key impacts of displacement would be on productivity and over-winter survival, rather than adult survival in the breeding season (Furness 2013, Searle *et al.* 2014). Recent work by Searle *et al.* (2014) has modelled the impact of displacement and barrier effects on seabird populations. The study considered the population level consequences of 66 scenarios linked to different levels of displacement and barrier effects, prey availability and distribution on five species of seabird from four different protected sites. These analyses quantified the potential population level consequences of a range of displacement and barrier effects scenarios on seabird populations and showed the potential for decreases in the survival and productivity rates for all species linked to the amount of time spent foraging within the wind farm zones.

The potential impacts on seabird populations of the mortality associated with collision risk have been a key focus for concern (Fox *et al.* 2006, Drewitt & Langston 2006). Whilst collision risk models have been found to be mathematically sound (Chamberlain *et al.* 2005), their outputs are highly sensitive to input parameters including the avoidance behaviour, flight heights and speeds of birds within wind farms (Chamberlain *et al.* 2006, Fox *et al.* 2006, Masden 2015). Consequently, concern has been raised about a lack of knowledge of precise values for these parameters and, more recently about the uncertainty associated with them (Chamberlain *et al.* 2006, Masden *et al.* 2014, Masden 2015). However, recent work has sought to derive more robust estimates of both flight height and avoidance behaviour of birds within wind farms (Cook *et al.* 2012, 2014, Johnston *et al.* 2014), including generating estimates of uncertainty around these values. In addition, ongoing work (Masden 2015) is seeking to combine these uncertainty estimates within the Band collision risk modelling framework in order to derive robust estimates of the uncertainty surrounding collision risk estimates.

It should be acknowledged that there remains significant uncertainty surrounding some of the key parameters used to estimate these impacts and that the models themselves lack empirical validation. However, we believe that the estimates of the impacts of collision and displacement on seabird populations have been made with reference to the best available evidence and utilising mathematically sound models, and are therefore defensible, ***given the data available****.*

3.2. As a consequence of (1), no probabilistic methods for assessing risk of population impacts of a given type and magnitude are scientifically robust and therefore defensible. This applies to both the Acceptable Biological Change method, reduced uncertainty Acceptable Biological Change and proposed methods to estimate the difference in the probability of a specified population – p3 of Green (2014)

As stated above (3.1), we believe that while the robustness of the methods used to assess the population level impact of offshore wind farms needs empirical validation, given the data available at present, estimates of the magnitude of these impacts are defensible. That said, as highlighted by Masden *et al.* (2014), it is important to highlight uncertainty surrounding the estimated impacts, something which, given analytical limitations, is rarely done. However, given the data collection currently underway in and around existing offshore wind farms, reviews of the data that have been collected (Cook *et al.* 2012, 2014, Furness *et al.* 2013, Johnston *et al.* 2014) enable us to estimate variability around some of the key parameters used to estimate these impacts. These data can then be used within simulation modelling frameworks currently under development (e.g. Searle *et al.* 2014, Masden 2015) in order to estimate the uncertainty surrounding predicted impacts. Where estimates of uncertainty are obtained around predicted impacts, these can be used to make a probabilistic assessment of the population level effect of these impacts.

We believe that, whilst they have not been presented to date, it is possible to generate defensible estimates of uncertainty around the impacts associated with offshore wind farms.

3.3. Attempts to identify thresholds either side of which estimates of population-level impacts of developments are considered negligible (or non-negligible) on biological grounds are mistaken and should be abandoned – p3 of Green (2014)

In assessing the impacts of an offshore wind farm (or any other development) on seabirds, impacts must be considered at three different levels: (i) Is individual fitness (i.e. survival, productivity) impacted by predicted effects, e.g. collision, displacement or barrier effects? (ii) Do these individual impacts alter the population trajectory of the species concerned? (iii) Is the population level impact acceptable when considered in the context of the economic or societal benefits of the development? These levels can be considered as a continuum with each reflecting a more significant biological impact. Article 2 of the Directive of the European Parliament and Council on the Conservation of Wild Birds (2009/147/EC, the 'Birds Directive') states that:

"*Member States shall take the requisite measures to maintain the population of the species referred to in Article 1 at a level which corresponds in particular to ecological, scientific and cultural requirements, while taking into account of economic and recreational requirements, or to adapt the population of these species to that level.*"

and Article 3 states that:

"*In the light of the requirements referred to in Article 2, Member States shall take the requisite measures to preserve, maintain or re-establish a sufficient diversity and area of habitats for all the species of birds referred to in Article 1.*"

To comply with these requirements, when assessing the impact of a project on a seabird population, it is therefore necessary to determine first whether the magnitude of the impact is such that a population will not be maintained at its current size, or that attempts to restore a population to a level corresponding to ecological, scientific and cultural requirements will not be impaired. Furthermore, it should be noted that populations of seabirds (or any other taxa) rarely occur in isolation, and movement between adjacent (or even more distant) populations may be non-negligible. Thus setting of thresholds of acceptable impact may need to take into account (indirect) impacts on other, demographically connected, populations.

Thresholds against which targets can be assessed cannot be thought of as biologically meaningful unless they are based on models which accurately depict population processes at the site concerned. Without these models it is not possible to determine, with confidence, how the impacts of an offshore wind farm on survival and productivity are likely to interact with one another at individual breeding colonies or at a meta-population level. However, such models ideally require a detailed knowledge of age-specific survival, productivity, immigration and emigration rates and any relevant density-dependent processes, all at a site-specific level. In practice, these data will not all exist and the models used to assess population level impacts will be generalisations, often based on data collected at regional or national levels, rather than a site specific level. As a consequence, these models will generally not be able to indicate with any degree of certainty whether or not the predicted impacts from offshore wind farms (or other developments) will have a significant effect at a population level. This means that thresholds applied to any metrics derived from these models will be based on a qualitative assessment of the evidence presented, rather than underlying biological processes, and hence there is a risk that inappropriate conclusions may be drawn. Therefore, any such threshold should be used as guidance by decision makers, rather than being viewed as firm predictions.

We believe that, at present, limitations in the data and models used to assess population level effects of offshore wind farms mean that biologically meaningful thresholds of impact cannot be set. Without improved data and more refined models, it is generally not possible to identify a threshold that is biologically relevant to the population concerned. For this reason, any threshold set will likely be largely subjective and based on a qualitative assessment of the evidence presented and should be acknowledged as such.

4. SPECIFIC CRITICISMS OF APPROACHES USED TO DATE

4.1 Inadequate knowledge of density dependence means models incorporating density dependence are inappropriate – *Section B.3 of Annex B of Green (2014); Paragraph 3.12 of the Hornsea Project One RSPB written representation*[5]

The risk-based metrics listed in Table 1 can either be derived from density-dependent or density-independent models. Density-dependence is also implicitly assumed by PBR (Wade 1998, Niel & Lebreton 2005, Dillingham & Fletcher 2008). Green (2012) raised concerns about the incorporation of density-dependence in population models previously in relation to the London Array Offshore Wind Farm Phase II Appropriate Assessment where a density-dependent response in wintering oystercatchers *Haematopus ostralegus* was extrapolated to red-throated divers *Gavia stellata*. In relation to the Hornsea One Offshore Wind Farm, RSPB suggested a review of density dependent relationships in seabirds. Only a single study, Cury *et al.* (2011), was presented in response to this suggestion[6]. However, ongoing work has highlighted a range of studies in which density-dependent responses in seabird populations have been identified (Horswill & Robinson 2015). These relate to both survival (e.g. Breton *et al.* 2006, Coulson 2001, Milne 1974) and productivity (Andersson & Eriksson 1982, Butler & Trivelpiece 1981, Kilpi 1989) and cover a range of species, e.g. Atlantic puffin *Fratercula arctica* (Harris 1980, Breton *et al.* 2006), black-legged kittiwake *Rissa tridactyla* (Coulson 2001), great black-backed gull *Larus marinus* (Butler & Trivelpiece 1981), herring gull *L. argentatus* (Kilpi 1989) and common guillemot *Uria aalge* (Birkhead *et al.* 1977). We believe that knowledge about the range of density-dependent responses in seabird populations may be greater than has been previously assumed, however, Horswill & Robinson (2015) suggest that density-dependent effects can vary markedly between colonies in relation to local conditions, and may result in the expected impacts being exacerbated or mitigated. Focussing on a single study, even one as comprehensive as Cury *et al.* (2011), therefore risks potentially over-looking important responses.

We believe that whilst evidence concerning density-dependent responses in seabird survival and productivity rates have not been routinely presented in impact assessment work, a detailed review of the topic suggests that there may be a useful evidence base to draw upon. With careful consideration of this evidence, it may be possible to consider models incorporating density-dependence when assessing the population-level impacts of offshore wind farms. However, in many cases, density-independent models are likely to represent a more precautionary approach where there is uncertainty about the shape or magnitude of any response, as they do not assume a compensatory increase in survival or productivity at low population sizes.

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[5] ***Summary written representation for the Royal Society for the Protection of Birds in the matter of planning application for the Hornsea Project One Offshore Wind Farm (Zone 4) located approximately 103km off the East Riding of Yorkshire Coast***

[6] http://infrastructure.planningportal.gov.uk/wp-content/ipc/uploads/projects/EN010033/2.%20Post-Submission/Representations/ExA%20Questions/Appendix%20X%20-%20PVA%20Note.pdf

4.2. The probability of a population decline cannot be reliably calculated without statistical bias – *Paragraphs 3.21 & 3.22 of the Hornsea Project One RSPB oral representation*[7]

The RSPB highlight that uncertainty surrounding both model input parameters and predicted impacts means that the probability of a population decline cannot be reliably calculated. Whilst it is acknowledged that uncertainty in the demographic parameters should be addressed through sensitivity analysis, the issue of uncertainty surrounding the predicted impacts remains. This issue also affects all probability outputs from PVAs, including those used by ABC and ruABC. Although impacts, such as collision risk and displacement, are routinely presented as a single value with no estimate of uncertainty, as outlined above (3.1) we believe that ongoing methodological advances and data collection are likely to facilitate the estimation of uncertainty, for example through the use of Monte Carlo simulations in the modelling process. However, it is important to present these estimates of uncertainty in a manner that can be easily interpreted, and to draw a distinction between uncertainty (i.e. we believe the impact will be in the range of x-y) and risk (i.e. there is an x% chance of the impact being greater than or equal to y).

Whilst it may be possible to estimate uncertainty surrounding predicted impacts, a second criticism of metrics related to the probability of a population declining remains. These metrics may be sensitive to assumptions about whether demographic rates predict future populations to be growing, stable or declining. If future populations are predicted to be growing, predicted population-level effects may be less significant than if populations are predicted to be stable or declining. This assumption is significant as policy and environmental changes including, but not limited to, climate change, fisheries discard policy and landfill closure are expected to impact populations in coming years. The exact magnitude of these changes is unknown, but may be substantial. Given the uncertain impacts of these changes on seabird population trends, understanding the sensitivity of metrics based on the probability of decline to assumptions about underlying population growth rate is necessary before they can be used with confidence.

It should also be noted that it may not be possible to use metrics related to population declines (probability of growth rate being <1, change in probability of growth rate being <1, probability of population decreasing by a given magnitude below its initial size, change in probability of a population decreasing by a given magnitude, probability of decreasing below baseline population size) to determine whether the conservation objectives of a site are being met, particularly if these are phrased relative to the population the site can potentially support. Thus, whilst a declining population is likely to reflect a site failing to reach its conservation objectives, if an impact caused a population that was previously increasing to stabilise, but at a lower level, then, although the population is not declining, the site may still not be achieving its conservation objectives. Similarly, PBR, which determines a level beyond which additional mortality will be unsustainable (i.e. the population is likely to become extinct), is generally not suited for defining acceptable population impacts. Aside from the practical challenges of defining a single sustainable level of impact in a continually changing environment, a population that is merely viable would have little resilience to any additional adverse factors that may be imposed.

We believe that in the light of ongoing methodological advances and data collection it is possible to provide some estimate of the uncertainty surrounding predicted impacts. We further suggest that the utility of metrics related to probability of population decline will depend on the context of existing population trajectories.

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[7] ***Summary of Oral Case presented at the April Issue Hearings by the Royal Society for the Protection of Birds in the matter of Planning Application for the Hornsea Project One Offshore Wind Farm (Zone 4) located approximately 103km off the East riding Of Yorkshire Coast***

4.3 Models do not incorporate additional sources of mortality (e.g. drowning in fishing gear) – *Section B.2 of Annex B of Green (2014); Paragraph 3.11 of Annex V of the Hornsea Project One RSPB written representation[6]; Paragraph 3.2 of the Hornsea Project One RSPB oral representation[7]*

It is clear that the impact of offshore wind farms is only one of a number of pressures facing seabirds in northern Europe (Burthe *et al.* 2014), each of which may contribute to additional mortality. Criticisms of a failure to incorporate all sources of additional mortality when assessing the population level impacts associated with offshore wind farms typically focus on the PBR method (Green 2014), but may be equally applicable to other metrics, for example those based on PVA.

When used to determine whether population declines are attributable to anthropogenic causes, Abraham & Richard (2013) highlight that PBR should be compared to total anthropogenic mortality, as opposed to just that associated with a single source, such as an offshore wind farm. Failure to do so may result in the total anthropogenic impact on a species being underestimated. For example, in the Baltic, Zydelis *et al.* (2009) highlight a lack of data on the number of long-tailed ducks *Clangula hyemalis* killed by Russian hunters and oil pollution as an explanation for why the population is declining despite their estimates of additional mortality not exceeding that allowable by PBR. This criticism is equally applicable to metrics derived from PVAs, especially those related to the likelihood of population declines. Unless the demographic parameters used in a PVA reflect the conditions at the site in question over the period of interest, it is likely that other anthropogenic impacts will not be accounted for. However, as PVA-based metrics can be used to compare population trends with and without an impact applied (rather than simply being a binary assessment of whether additional mortality is above or below a given value), this omission may be less important in the context of PVAs.

The impact of an offshore wind farm on its own may not exceed PBR, or trigger a population decline; however, in combination with existing sources of additional mortality not accounted for in the baseline demographic rates considered, this may not be the case. It may be possible to address this issue if it can be demonstrated that the demographic parameters used in PBR or PVA calculations already account for these additional sources of mortality. If this is not the case, then outputs must be interpreted and presented more carefully. For example, it may be possible to say that *on their own* impacts from offshore wind farms will not cause a population decline or exceed PBR, but this may not be true when considered in combination with the cumulative effects of other sources of anthropogenic mortality.

We believe that the failure to incorporate additional sources of anthropogenic mortality, for example drowning in fishing nets, is likely to impact both PBR and also PVA-based metrics, particularly those linked to the probability of decline. However, given that PBR focuses on whether a certain level of mortality is exceeded or not, we believe that failure to incorporate this additional mortality is a more significant issue for PBR than for PVA- based metrics.

4.4 The recovery factors used in PBR calculations are not based on empirical evidence – *Section B.4 of Annex B of Green (2014); Paragraph 3.12 of Annex V of Hornsea Project One RSPB written representation[6]*

A key parameter in the PBR method is the recovery rate (*f*). Wade (1998) states that *f* can be seen as, "*both an additional factor to hasten the recovery of a population and as a safety factor to account for additional uncertainties other than the precision of the abundance estimate*". As *f* is, at least partly, designed to allow for unknown biases it is difficult to choose a value objectively, however, it is typically assumed that *f* takes a value between 0.1 and 1, reflecting the ability of a population to recover, with lower values set to enable a population to recover more quickly (Wade 1998). Thus, Dillingham & Fletcher (2008) recommend that *f* values should be set with reference to the conservation status of a species as assessed by IUCN and Birdlife International (IUCN 2001). Following this guidance, a value of 0.1 is used for all threatened species, 0.3 for species assessed as near threatened, 0.5 for species of least concern and 1.0 for species of least concern whose populations are known to be increasing or stable. Such definitions are generally not suitable for use in relation to a species at a single colony as the population size and trend of the species at the site of interest may not represent the conservation status of the species population as a whole. However, a similar logic of assigning *f* values based on the population status of a species at a colony concerned could be applied. For example, where a population of one species at a site is declining, it would be appropriate to select a lower *f* value than is used for a population of another species which is stable or increasing. In some ways, the use of *f* mirrors the debate surrounding the avoidance rate used in collision risk modelling (Cook *et al.* 2014) which has (incorrectly) been used as a "fudge-factor" to account for error and uncertainty in the model input parameters. Simulation approaches have been applied to guide the choice of *f* values most appropriate given biases thought to exist in parameter estimates and tolerance for risk (Richard & Abraham 2013). Such an approach is consistent with using *f* as a safety factor to account for uncertainties in the model as suggested by Wade (1998). However, using *f* as a "fudge-factor" in this way is not desirable as it potentially obscures the influence that variation in other parameters may have on the final estimate of PBR.

The final PBR values are sensitive to the *f* value assumed, with an increase in *f* from 0.1 to 0.5 reflecting a five-fold increase in the PBR value estimated. However, the value selected is rarely based on empirical evidence. Indeed, without observing populations going extinct, it is debatable whether it is possible to obtain empirical evidence in support of a particular *f* value. Consequently, while Dillingham & Fletcher (2008) argue that the selection of *f* is a management decision which should be made with reference to conservation goals, stakeholder desires and the ability to monitor a population, it should be emphasised that ultimately the value selected will necessarily be subjective.

We agree that the f values are not based on empirical evidence. Indeed Wade (1998) emphasises that f should be taken as both the recovery factor and also as a safety factor to account for uncertainty in input parameters. Therefore, without detailed quantification of the uncertainty in the model parameters, it is difficult to see how f values could be based on empirical evidence.

4.5 PBR does not quantify the impact of additional mortality on population size – *Section B.5 of Annex B of Green (2014)*

If the aim of metrics is to test whether or not the conservation objectives of a site will be met, any approach used must typically be capable of assessing whether the resultant additional mortality will mean a population can be maintained at its current level. Both Wade (1998) and Niel & Lebreton (2005) make a distinction between additional mortality exceeding PBR and a population undergoing a significant decline. Niel & Lebreton (2005) explicitly state that "*It [PBR] could be used to predict whether an additional source of mortality is unsustainable, but it cannot be used the other way around (i.e. to predict that it is sustainable).*" Indeed, the simulations of Wade (1998) demonstrate that if the additional mortality resulting from a project is equal to that obtained from estimates of PBR, populations can reach equilibrium at a point well below the carrying capacity of the available habitat (see Figs. 2, 3 & 6 of Wade 1998).

We do not believe that PBR is suitable for use in quantifying the impact of additional mortality on population size.

4.6 PBR has not been adequately validated by empirical studies – *p17, Section B.4 of Annex B of Green (2014); Paragraph 3.13 of Hornsea Project One RSPB written representation*[6]

As with methodologies used to assess collision risk, displacement and barrier effects, a key criticism of PBR is that it has not been validated by empirical studies. Green (2014) suggests that this validation could be achieved by comparing reliably measured population trends in species where additional mortality was less than the PBR value with those in species where the additional mortality was greater than the PBR value. In order to show empirical support for PBR, these studies should demonstrate that the latter populations, where the additional mortality was greater than the PBR value, were declining whilst others were not.

There are relatively few studies available with which PBR could be validated. Thompson *et al.* (2007) found that the number of grey seals *Halichoerus grypus* shot by fisheries managers in the North Sea exceeded that allowable under PBR and that this was sufficient to explain a localised population decline. However, Zydelis *et al.* (2009) in a study of fishing mortality in the Baltic and North Seas found only limited support for the use of PBR. They investigated the population effect of the additional mortality in three species. The additional mortality only exceeded PBR for greater scaup *Aythya marila* and this population was indeed declining. The long-tailed duck population was also declining despite the mortality not exceeding PBR, although they attribute this to an underestimate of the additional mortality. Conversely, however, common guillemot populations remained stable, despite additional mortality exceeding PBR. These data would suggest that PBR does lack adequate validation by empirical studies and that those studies which do exist highlight the need for improved understanding of additional sources of mortality.

We believe that where PBR has been used to assess impacts on seabirds, results have been inconsistent and do not offer empirical support for the approach.

4.7 ABC uses an arbitrary and inappropriate threshold probability value for the acceptable population size – *Section A.2 of Annex A of Green (2014)*

As outlined above (3.3), any attempt to derive a threshold with which to assess the impact of additional mortality on a population is likely to be subjective (i.e. based on an individual perception of the evidence presented), as opposed to arbitrary (i.e. based on no obvious reasoning or system), rather than having a firm biological basis, and should be acknowledged as such. Bennet (2013) uses terminology from the IPCC to identify this threshold. However, the IPCC terminology regarding the likelihood of an event occurring is inconsistent with the way in which risk is assessed at other stages of the planning process for offshore wind farms, for example, in an EIA where impacts may be categorised in relation to their magnitude, rather than their likelihood. Masden *et al.* (2014) highlight the use of consistent language as a key step in reducing the uncertainty associated with environment assessment. As such, introducing new terminology is undesirable unless it facilitates greater clarity, or allows useful new insights. Thresholds could be related to those used to assess magnitude as part of an EIA, however, it should be emphasised that, given current knowledge, these would be subjective thresholds and generally not biologically meaningful; there is thus a risk that the consequences for the seabird population may not be fully considered. We therefore feel that the task of determining whether or not they are appropriate in a particular case should be the responsibility of decision makers, once they have carefully considered all of the evidence they are presented with, and that their subjective nature be made clear.

We agree that the thresholds used to define ABC are not biologically meaningful.

4.8 Use of ABC results in perverse consequences of measurement errors – *Section A.3 of Annex A of Green (2014)*

Under ABC, if the predicted size of the impacted population is equal to, or greater than, that 66.7% likely to be achieved in the absence of an impact, the development should be deemed ‘acceptable’. However, where there is greater uncertainty surrounding the demographic parameters, the distribution of predicted population sizes will have a wider spread around the median estimate. As a consequence, where there is greater uncertainty around the model input parameters, more substantial impacts will be deemed acceptable. Therefore, whether or not an impact is deemed acceptable is likely to reflect the limitations in the data rather than the ecology of the species concerned, potentially misrepresenting the population-level consequences of any impact. In recognition of this, JNCC and SNH (2014) proposed a modification to this approach, referred to as reduced uncertainty Acceptable Biological Change (ruABC). Using ruABC, uncertainty is reduced by using a regional population model which absorbs error resulting from sampling variation, but attempts to retain natural variation in demographic rates. The model prediction uncertainty from this model can then be applied to predicted population sizes at a colony level in order to identify an acceptable level of biological change. As seabird demographic parameters and population trends vary spatially (Frederiksen *et al.* 2005, Cook *et al.* 2011), it is important to ensure that the regional population is representative of the colonies concerned. If the regional populations are not representative of the colonies concerned, it may lead to bias and inaccurate estimates of the scale of impacts that a population could withstand. However, even with ruABC there is a significant risk that whether an impact is deemed acceptable or not will reflect limitations in the data, rather than the vulnerability of the species concerned.

We agree that using both ABC and ruABC, there is a significant risk that projects are deemed acceptable as a result of limitations in the data rather than the magnitude of predicted impacts. We believe that much better, colony-specific estimates of impacts and demographic parameters are needed if these approaches are to be used.

4.9 ABC uses the wrong exceedance probability distribution to define acceptable risk – *Section A.4 of Annex A of Green (2014)*

The ABC (and ruABC) approach uses a threshold determined by the uncertainty in the population size of a species predicted in the absence of any impact from an offshore wind farm. However, there is also likely to be significant uncertainty surrounding the magnitude of any predicted impact from an offshore wind farm. Disregarding this uncertainty risks giving a misleading impression of the confidence associated with the assessment of any impact and an incomplete picture of the likely risks of population-level impacts.

We agree that the ABC approach needs to consider the uncertainty in population size in both the presence and absence of a development.

5. CONCLUSIONS

The RSPB criticisms make reference to whether “the best scientific knowledge” has been used to assess the population level impacts of offshore wind farms on seabirds (Green 2014). We believe that, overall, the best scientific knowledge available is being used to assess the magnitudes of impact of individual effects on a species (e.g. collision, displacement, barrier effects). Work programmes, such as the COWRIE and SOSS programmes, other, methodological advances and reviews (Cook *et al.* 2012, 2014, Davies *et al.* 2013, Horswill & Robinson 2015, Johnston *et al.* 2014, 2015 Masden 2015) and ongoing data collection are helping to improve the knowledge base with which these impacts can be assessed, and better account for uncertainty in these impacts. With regards to the assessment of the consequences of these impacts at a population level, the situation is less clear. A range of different metrics have been proposed with which to assess population-level effects, and, as discussed above, we believe that some are more appropriate than others.

We agree with RSPB that PBR generally cannot be used to assess whether the population-level effects of offshore wind farms mean that the conservation objectives (whatever they may be) of protected sites are (or are not) being met. This is because PBR considers only whether a pre-determined level of mortality is exceeded, rather than the biological impact of any additional mortality at a population level. For similar reasons, we have concerns about metrics related to the probability of a population decline. However, these may have some merit, subject to the outcome of an assessment of their sensitivity to demographic parameters and projected population trends.

We also agree that neither ABC nor ruABC are suitable metrics given the risk that whether an impact is deemed ‘acceptable’ or not may reflect uncertainty in the data rather than the status of the population concerned. Of the remaining metrics, we believe that those linked to population size at a given point (e.g. the end of the lifetime of a project) and population growth rate have the most promise, subject to a careful consideration of the sensitivity of metrics to model input parameters, model assumptions and uncertainty about the population trajectory and demographic characteristics of the species concerned. Those linked to population growth rate may be of particular value in that predicted growth rates could be compared to observed growth rates from the outset of the project. This may allow a rapid assessment of the impact of an offshore wind farm on seabird populations. These metrics may be calculated from either stochastic or deterministic models, although the benefits of each of these approaches would need to be considered as part of a wider sensitivity analysis.

We agree with Green (2014) that any thresholds applied to the metrics described above are likely to be subjective, rather than biologically meaningful, and that, where they are defined, they need to be acknowledged as such, as should the risk that inappropriate conclusions may be drawn.

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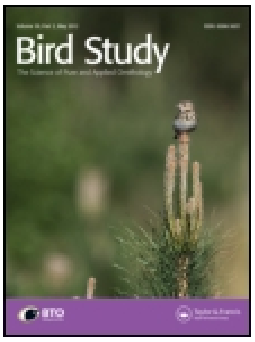

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Lesser Black-backed Gulls *Larus fuscus* nesting in an inland urban colony: the importance of earthworms (Lumbricidae) in their diet

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Lesser Black-backed Gulls *Larus fuscus* nesting in an inland urban colony: the importance of earthworms (Lumbricidae) in their diet

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Capsule Earthworms can be an important food of birds breeding inland and on the coast.
Aims To identify food and feeding sites of Lesser Black-backed Gulls *Larus fuscus* breeding on buildings in Dumfries, Scotland.
Methods Direct observations were made of feeding behaviour, and 181 regurgitated pellets, obtained from flat roofs where only Lesser Black-backed Gulls nested, were analysed. Food items were identified using standard techniques with microscopic examination used to identify the presence of earthworm chaetae.
Results Lesser Black-backed Gulls breeding in Dumfries fed extensively on earthworms and insects obtained from agricultural land (55% of pellets). They also fed on landfill (23% of pellets) and marine sites (12% of pellets), but they did not take fish or fish offal.
Conclusions Previous studies on the food of large gulls have often overlooked the importance of earthworms because of the lack of macro-remains in stomachs and pellets. New studies are required to establish the general extent of earthworm consumption in these large gulls.

Previous investigations of the food of Lesser Black-backed Gulls *Larus fuscus* were summarized by Cramp & Simmons (1983) and Del Hoyo *et al.* (1996). These indicate that a wide spectrum of food is taken. In particular, Cramp & Simmons mentioned that this species fed on agricultural land and consumed earthworms (Lumbricidae), but this is not mentioned by Del Hoyo *et al.* This difference is indicative of infrequent and inconsistent reports of earthworms in the food of Lesser Black-backed Gulls.

The increase of Lesser Black-backed Gulls during the 20th century resulted in both urban and more extensive inland breeding (Cramp 1971, Monaghan & Coulson 1977, Raven & Coulson 1997). This study reports on the feeding and food of Lesser Black-backed Gulls nesting in an inland, urban colony in SW Scotland, and presents new information on the food and feeding of this species, highlighting the importance of farmland and earthworms as a source of food in this colony.

METHODS

Dumfries is a town in SW Scotland with a population of 38000 people (2001 census). It is about 15 km from the Solway Firth and the Irish Sea. In 2003 and 2004, we made a detailed study of the feeding and food of the town-nesting gulls in Dumfries during the breeding season, and made general observations on gull feeding in the area in more recent years. We also made visits to the two landfill sites within gull feeding range of Dumfries, at Lochside (5 km away) and at Aucheninnis, near Dalbeattie (19 km away). Few Lesser Black-backed Gulls nested within 25 km of Dumfries, other than those that bred in the town and nearby on an industrial complex at Cargenbridge, and so adults seen feeding in the surrounding area were very likely to be from these colonies.

Our observations concentrated on Lesser Black-backed Gulls, which formed about 88% of the approximately 240 pairs of gulls nesting in Dumfries in 2002, and a similar number of pairs nesting nearby at Cargenbridge. Records of gulls feeding in the streets within the town boundaries were recorded and particular attention was given to those feeding in the town centre and on the riverside at Whitesands, a large car park area alongside the River Nith and adjacent to the town centre. Studies were made on flight lines and direction of departure of gulls from the colonies, and feeding beyond Dumfries in adjacent rural areas and at landfill.

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The nature and sources of food of Lesser Black-backed Gulls were investigated by analysis of the contents of regurgitated pellets. The pellets were obtained from flat roofs where only Lesser Black-backed Gulls nested and which were cleared of pellets of unknown age in mid-May. The sites were then revisited and pellets collected from early June to mid-July in 2002 and 2003. A total of 181 pellets were examined and, as no significant differences were found between the results obtained for the two years, the data have been pooled. Food items were identified from inspection of pellets after their contents had been dispersed in ethanol. Mammal remains were identified to species from skulls, jaws and bones and also from hair found in the pellets. Insect, mollusc and arthropod remains were identified to major taxa from hard parts such as jaws, head capsules, elytra, shells, limbs and spiracles.

We were aware of the problem of detecting the consumption of earthworms (Lumbricidae) from examination of pellets, as we had previously observed their consumption in fields, but failed to find evidence in preliminary examinations of pellets. In some pellets, we found insect remains that had almost certainly come from farmland, but we found no evidence of earthworms, presumably because worms did not leave macroscopic remains after digestion. However, the problem of detecting earthworm remains in the food was solved by examining samples from pellets under at least ×80 magnification, which revealed the presence of very large numbers of extremely small, golden, earthworm chaetae. In almost all cases when chaetae were found, hundreds and often thousands were present in a single pellet.

We made no attempt to count these chaetae to estimate the numbers of worms consumed, bearing in mind that a single earthworm has many chaetae, and we had no indication of the proportion of chaetae which were retained in a pellet rather than being passed through the intestine and excreted in faeces. Although we did not use the method, dispersing and agitating the pellet contents in ethanol, retaining the larger particles in a sieve and then centrifuging the liquid might be a useful way of separating the chaetae from larger items. Concurrent and independent of this investigation, Kubetski (2002) reported the presence and identification of chaetae in pellets from the Common Gull *Larus canus*.

In addition to animal remains, the pellets contained non-food items including plastic objects, aluminium ('tin') foil, small pieces of broken glass and small stones and pieces of brick. Vegetable material included grass shoots and roots and a few melon seeds.

Remains of flesh and bread-like materials were not found in the pellets, presumably because these had been totally digested and did not leave hard, undigested material in the pellet. This introduces a bias that should be considered when interpreting the results. In addition, food regurgitated by five chicks, each from different nests was examined.

In the following text, 'gulls' refers to Lesser Black-backed Gulls unless otherwise stated.

RESULTS

Flight lines

The directions of adult gulls leaving Dumfries, presumably on feeding trips, were recorded, as were flight lines of those leaving the Lochside landfill site, which was to the east of the town. Gulls departed from the town in all eight compass directions used, and no specific directions were favoured. Adults from Dumfries and Cargenbridge visiting the landfill were not sufficiently numerous to produce a regular and obvious flight line of gulls to and from the colonies. We detected only two fight lines, both formed by mainly immature gulls departing from the Lochside landfill in the evenings. The birds (including some Herring Gulls) flew either to the north (inland) or south (towards the coast), but neither involved birds settling or passing over Dumfries.

Feeding in the town

During the breeding season, a small number of Lesser Black-backed Gulls habitually fed in the main shopping area and also along the side of the river within Dumfries. These town-feeding gulls were 'specialists', and in 2003 we estimated that only eight individuals were involved (i.e. <3% of the gulls nesting in the town). Most of these specialists repeatedly patrolled the same area on the wing or stood on the same vantage points, such as the roof of a building or the top of a lamppost, waiting for food to be offered or accidentally dropped. This behaviour was restricted entirely to the central square in the main shopping area and to the riverside car park. Some of these gulls were bold when feeding and closely approached people, and this gave rise to complaints from the public. On the riverside, food thrown by the public attracted gulls from a regular roosting and bathing flock on a nearby shingle bank in the river. However, such feeding was opportunistic and infrequent, and these birds could not depend on handouts as a reliable source of food. Occasionally, individual gulls picked up food items from the River Nith where it passed through Dumfries, but there was no systematic or regular feeding on the river, other than occasionally taking food thrown there by the public.

Requests from the Dumfries and Galloway Council for the public not to feed the gulls (and pigeons) probably reduced the amount of food fed to the gulls in the town, but some feeding has continued. Elsewhere in Dumfries, gulls foraged infrequently in the streets and on school playgrounds. No gulls were seen to seek food from refuse containers in Dumfries, although this occurs in some other towns. It is evident from these observations that the great majority of Lesser Black-backed Gulls breeding in Dumfries obtained their food well beyond the town boundaries.

Use of landfill

In the breeding season, the landfill at Aucheninnes (19 km from Dumfries) was used almost entirely by adult Herring Gulls, and so was not an important feeding area for Lesser Black-backed Gulls from Dumfries. The landfill site close to Dumfries, at Lochside, was predominantly used by about 100 mainly immature Herring and Lesser Black-backed Gulls daily in May, June and July. Only small numbers of adults fed there at any one time. Gulls were deterred from feeding at these landfill sites for five days a week from late 2001 to 2004 by a hawk that was flown over the site during part of the working hours each day. This, and rapid covering of the refuse, probably reduced gull numbers during working hours, but both in the morning until the hawk was flown, and in the evening until 21:00 hours during May and June (nine hours of daylight), gulls were present at the Lochside site. Outside working hours, and because the refuse was covered, feeding consisted mainly of gulls walking over the site (secondary feeding; Sibly & McCleery 1983b), picking up items as and when encountered, and high density feeding by the gulls did not occur.

Use of agricultural land

Much of the area surrounding Dumfries is agricultural land. Small groups of gulls were seen feeding in pastures and in fields where silage had been cut. On two occasions, gulls were seen following behind machinery cutting grass, in a manner more usually associated with Black-headed Gulls *Larus ridibundus*.

Gulls were occasionally encountered on rural roads feeding on dead mammals, usually rabbits, killed by traffic.

Analysis of gull food pellets

Of the 181 pellets examined, the contents of all but four (2%) fell into one of four well-defined categories that we interpreted as representing four different feeding habitat sources.

1. *Agricultural land* (55% of pellets plus 2% partial). Ninety-six (53% of the total pellets, and 96% of the pellets in this category) contained earthworm chaetae. Sixty-five pellets (36% of total) also contained insect hard parts (head capsules, elytra, legs, spiracles) from adult ground beetles (Carabidae), dung beetles (Aphodidae), click beetles (Elyteridae), a ladybird beetle (Coccinellidae), and from the larvae of crane-flies (leatherjackets, Tipulidae). Three pellets contained vole bones and two had grain husks. Pieces of grass shoots and roots were frequently present in these pellets, but most were too long to be the remains of food ingested by earthworms or leatherjackets (as implied by Noordhuis & Spaans 1992). We suggest these pieces of grass were accidentally included while seizing and then swallowing worms and insects. The pellets placed in this category but lacking chaetae included a pellet containing the skull and bones of a Common Shrew *Sorex areneus*, a similar pellet containing the remains of a passerine, one pellet containing only grass and another made up totally of grain (oat?) husks.

2. *Road-kill* (4% of pellets). These seven pellets contained only fur (mainly rabbit) but no bones, chaetae or insect remains were present. While gulls are known occasionally to capture young rabbits, feeding on road-kills was a more likely source.

3. *Landfill* (23% of pellets). These 42 pellets were characterized by the consistent presence of several indigestible, mainly inorganic items in each pellet, including small stones, small pieces of brick, broken glass, a variety of plastic items, compacted pieces of 'tin' foil and, in two cases, melon seeds. Another characteristic of these pellets was the total absence of invertebrate remains, including earthworm chaetae. Occasionally, poultry feathers and small broken parts of chicken bones were present. Two large fish vertebrae in one pellet (probably from an adult salmonid which had been used as human food) were included in this

category, but no other evidence of vertebrate food was detected. These items were considered typical of landfill, where earthworms are absent and a wide range of indigestible materials occur, including building rubble, pieces of plastic and foil. Previous investigators of pellets have not recorded how they identified feeding on landfill from the pellet contents. It is likely that some food items consumed on landfill did not leave remnants or cause the production of pellets.

4. *Marine areas* (12% of pellets plus 2% partial). Evidence of marine animal remains were found in 22 pellets, and included the presence of claws and parts of the carapace of small shore crabs *Carcinus maenas*, bivalve mollusc shells (particularly *Tellina* sp., but only one *Macoma balthica* shell was recorded). These pellets were also typified by the absence of chaetae from earthworms or marine worms, and remains of insects. Surprisingly, small bones and otoliths from fish were absent from all samples. It is likely that these gulls had fed at intertidal areas, and had not fed over the open sea.

The consistency of the pellet data falling into four characteristic groups probably indicates that each pellet represented the consumption during a single foraging trip. The four pellets that did not fall into one of these categories contain remains of marine invertebrates, parts of insects and earthworm chaetae. This combination may have been produced when gulls fed for a period on tidal mud flats, and then moved onto grassland at high tide. Each of these pellets was allocated as half value to the marine and agricultural land categories.

When evaluating feeding at inland rural areas, the proportion identified as feeding on road-kills should be added to the proportion in the agricultural land category, which gives a total of 61% of the pellets.

Chick regurgitates

Five samples of regurgitated food were obtained from young gulls, each from a different brood. In all cases the samples were mainly liquid, rich brown in colour, almost totally lacked hard structures except for a few insect tergites, and had a highly unpleasant odour. Microscopic examination revealed the presence of vast numbers of earthworm chaetae. Presumably the colour was produced by soil that had been in the guts of consumed and digested worms. Similarly to the analyses of pellets, the presence of chaetae revealed the consumption of many earthworms, which otherwise would not have been detected.

DISCUSSION

Previous studies on the food of Lesser Black-backed Gulls

Early writers such as Seebohm (1883–85) and Morris (1898) listed fish, offal, insects and earthworms (Lumbricidae) as the food of Lesser Black-backed Gulls. Collinge (1924–27) examined 43 stomachs and reported that the great majority of the food by weight was marine organisms, mainly fish, but 14% was insects. He also recorded the presence of Annelida (worms), but he did not distinguish between marine worms and earthworms, and these taxa were grouped with marine Crustacea in the presentation of the quantitative consumption.

Nordberg (1950) examined pellets of Lesser Black-backed Gulls collected in Finland and reported that fish formed 58% of the food items and insects made up 7%, but the consumption of earthworms was not reported. Pearson (1964, 1968) examined 68 regurgitated food samples from chicks in two breeding seasons (incorrectly reported as stomach samples in Cramp & Simmons 1983) on the Farne Islands, Northumberland. On average, 77% of the food by weight was fish and the overall contribution of earthworms was low, but reached 19% of the food in July 1961. Conder (1952) and Davis (1973) made studies of the food and feeding habits of the Lesser Black-backed Gulls breeding on Skokholm, SW Wales. Both recorded fish as the main food, but Conder did not report the consumption of earthworms or feeding on agricultural land, while Davis concluded that a minority of individuals took earthworms from agricultural land. Harris (1965) examined 55 stomachs of adults culled on the neighbouring island of Skomer, and found earthworms in only one stomach, while 16 stomachs contained unidentified beetles, the same number as were found containing fish. In the summary of the paper (but not in the main text), Harris stated that, '*L. fuscus* obtains most of its food on the shore and on arable land', presumably a conclusion based on the presence of insects, 'corn' and the single earthworm record.

Considerable variation and anomalies in the reported food consumed by Lesser Black-backed Gulls are evident in a succession of research studies made at Walney, in NW England. The initial study by Brown (1967a, 1967b) mentioned that 'much of the potential food is garbage' and identified the source as landfill sites, fish docks, and 'pickings from holiday resorts'. He also recorded feeding in agricultural areas, emphasizing the importance of an area used for poultry farming. His records of the food regurgitated by males during courtship listed garbage, fish, crabs and mussels, but consumption of earthworms was not mentioned in either paper.

Two subsequent studies at Walney (Shaffer 1971, Verbeek 1977) used pellets and flight lines to identify food and feeding areas. These emphasized that fish and landfill were the main food sources, but neither mentioned the consumption of earthworms or extensive feeding on farmland. However, a few years later, Sibly & McCleery (1983a, 1983b) used telemetry to identify feeding areas of Herring Gulls *Larus argentatus* nesting at Walney. As a result, and from subsequent direct observations, they found that earthworms were collected extensively from farmland on the Furness peninsula by both Herring and Lesser Black-backed Gulls. From fig. 2 and the caption to fig. 4 in Sibly & McCleery (1983b) it can be calculated that up to 13 000 Lesser Black-backed Gulls were found at one time feeding on earthworms on the Furness peninsula.

Most recently, Kim & Monaghan (2006) used pellets and flight lines to identify the food and feeding of Lesser Black-backed Gulls at the Walney colony and compared their results with those of Verbeek (1977) and Shaffer (1971). They acknowledged that the use of pellets failed to reveal the consumption of soft-bodied prey, but again, the utilization of earthworms and extensive feeding on agricultural land was not specifically mentioned, although they indicated a flight line to the Furness peninsula area where Sibly & McCleery had previously reported extensive consumption of earthworms. Thus these five studies at one colony produce different, and sometimes contradictory, evidence of the extent of feeding on agricultural land and the consumption of earthworms. It seemed likely that the consumption of earthworms by the gulls at this colony was overlooked in several of the investigations.

On Rhum, in the Inner Hebrides, the examination of 45 pellets from Lesser Black-backed Gulls showed that fish and intertidal invertebrates dominated the food, and failed to find any evidence of feeding on terrestrial invertebrates (Furness *et al.* 1992). In the Netherlands, Noordhuis & Spaans (1992) found that fish dominated the food. However, 13% of the pellets they examined contained grass; they attributed this to earthworms and leatherjackets (Tipulidae) collected from farmland, the grass being released from the guts of these invertebrates during digestion.

Three general conclusions emerge from this review of the food and feeding areas of Lesser Black-backed Gulls. Firstly, fish is the main food of this species reported in almost all studies. Secondly, the majority of studies did not mention these gulls feeding on farmland, and of those that did, only one supplied quantitative estimates. Thirdly, an appreciable proportion of the studies did not report the consumption of earthworms, and only one study allowed quantitative estimates to be made of earthworm consumption.

In part, the variation in records of feeding on earthworms may have arisen from variability in the availability of worms between wet and dry years, as pointed out by Pearson (1968) and subsequently by Sibly & McCleery (1983b). But the main reason for the infrequent recognition of earthworms in the food has been the failure to detect earthworm remains in pellets when worms had been consumed, while rapid digestion often left few or no obvious macroscopic remains of earthworms in stomachs or in regurgitation samples.

As far as we can ascertain from previous studies or directly from the authors, identification of food items in pellets, regurgitation and stomach samples have been made unaided or by examination under low-power (e.g. ×10) magnification. In our study, we used higher-powered microscopic examination and were then able to detect the abundant presence of earthworm chaetae in both pellets and in regurgitation samples, although other earthworm remains were found.

Comparison with the results from Dumfries

The food consumed by Lesser Black-backed Gulls nesting in Dumfries differed markedly from those reported from other studies, including those that were based on the analysis of pellets.

Firstly, Dumfries birds did not consume fish or fish offal, despite being only 15 km from the sea and in contrast to their importance found in other studies. Secondly, the use of farmland as a feeding site and the consumption of earthworms were very much greater than all but one of the earlier studies (the exception being the investigation by Sibly & McCleery at the Walney colony, if their results on the use of earthworms by Herring Gulls also applied to Lesser Black-backed Gulls). An appreciable part of this difference in the frequency of consumption of earthworms arose from microscopic examination of pellets and regurgitation samples used in our study to detect the presence of chaetae. However, if we ignored the presence of chaetae, about 33% of all the pellets we examined would still have indicated feeding on farmland, because of the presence of beetle remains, grass or bones and fur of small mammals, a value that would then not be dissimilar from the results reported by Harris (1965), where 16 (29%) out of 55 stomachs contained beetle remains.

At the other extreme, Furness *et al.* (1992) obtained no evidence that gulls breeding on Rhum utilized farmland at all. We suggest that in most previous studies, the potential importance of earthworms as food of Lesser Black-backed Gulls has been either overlooked or much underestimated. One consequence of this oversight is that the use and importance of feeding on agricultural land by this species has probably not been fully appreciated. Because of the limitations of many of the previous studies, Monaghan (2007) is justified in saying that currently, 'very little is understood about the feeding ecology of the Lesser Black-backed Gull'.

We saw relatively few Lesser Black-backed Gulls feeding on agricultural land and this might be considered an anomaly between our observations and the results obtained from the analysis of pellets. However, there is about 1000 km^2 of agricultural land within a 20 km radius of Dumfries (30 minutes flying time and well within the feeding range of this species). Assuming that one of a pair of gulls was feeding at any one time and that half of these fed on agricultural land (i.e. about 200 adult gulls of the 800 nesting in Dumfries and at Cargenbridge), this would correspond to only one gull to every 5 km^2. Thus the low density of birds recorded by direct observations is not at variance with the pellet analyses.

The availability of earthworms to gulls is influenced by both time of day and the extent to which the ground is damp or wet (Sibly & McCleery 1983b). In periods of drought or semi-drought, the worms aestivate well below the surface and are unavailable to gulls. This is likely to produce day-to-day, month-to-month and year-to-year variation in the consumption of earthworms. However, the availability of beetles and leatherjackets on agricultural land is unlikely to be influenced to the same extent. In addition to this source of variation, there are likely to be marked geographical variations in the availability and therefore the use of farmland and earthworms, an effect hinted at by the data from Rhum (Furness *et al.* 1992). As a result, it is likely that Lesser Black-backed Gulls exhibit considerable flexibility in diet according to the position of the colony and the available feeding areas in the vicinity.

Should it be established that Lesser Black-backed Gulls in the same area fed on earthworms more than Herring Gulls, the higher rainfall in western regions of Britain would produce fewer periods of drought and may explain, at least in part, the more westerly distribution of the former species in Britain.

Use of landfill

The use of landfill was indicated by only 23% of the 181 pellets examined, but it is likely that some of the food obtained there did not leave identifiable traces in the pellets or did not result in pellet production, so the proportion of gulls using landfill is possibly underestimated. However, the abundance of non-food items in the pellets used to indicate feeding at landfill suggests that this bias might not be large. Further, direct observations indicated that there was not extensive use of landfill by the adult gulls nesting in Dumfries and certainly not enough to allow the detection of flight lines between the town and the landfill. At many (but not all) landfill sites, gulls rest nearby and wait long periods for feeding opportunities, which results in a low efficiency (Sibly & McCleery 1983a). Perhaps the importance of landfill as a major source of food for large gulls has been exaggerated. Coulson *et al.* (1987) showed that individually marked Herring Gulls on average fed at landfill on only one or two days in the week, despite large numbers being present daily, and obviously most obtained much of their food elsewhere.

While the deposition at landfill sites of industrial animal waste (such as from processing factories) provides high densities of good and quickly obtained food for gulls, such material occurs at few landfill sites. A great amount of household and industrial refuse contains material that is unsuitable as food for gulls and material enclosed in plastic bags is difficult to search. Greig *et al.* (1983) found that efficient feeding at landfill by gulls is an acquired skill and it takes several years before individuals become fully efficient. Further, reports of the 'high calorific value' of materials dumped at landfill (Sumner 1971) can be confusing. These high values come from the incineration of samples of waste and include energy released from appreciable amounts of plastic, wood and paper, all of which, obviously, are not available as sources of energy to gulls.

Explanations of the increase of the Lesser Black-backed Gull

The publications following the national census of seabirds in 1985–87 (Lloyd *et al.* 1991) and in 1998–2002 (Calladine 2004) both contain sections that discuss the reasons for the change in abundance of

Lesser Black-backed Gulls in Britain. Both emphasize the effects of food, particularly from landfill and fish discards, but neither comment on feeding in agricultural land or the use and availability of earthworms as food. There is a clear need for more detailed and more precise examination of the food of large gulls, and in particular the extent to which they obtain food from farmland. Until that is done, caution should be applied before assuming that fish, fish offal and landfill are always the key sources of food for Lesser Black-backed Gulls.

Researchers in the past may have been excessively influenced by the high density of gulls feeding in large numbers on fish waste and at landfills, but have ignored the large areas of agricultural land available as a source of food, which, even with a low density of gulls, could involve an appreciable proportion of the population. An analogy might be appropriate. People in the UK visit take-away and fast-food sources in large numbers, sometimes with many queuing to be served, but most people do not use this source of food every day. Would the closure of these shops result in a cessation of the increase in the human population? Gulls, too, have alternative sources of food and evaluation of the importance of these still awaits detailed research.

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The accuracy of urban nesting gull censuses

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The accuracy of urban nesting gull censuses

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Capsule The vantage point survey methods used in the 1999–2002 surveys of urban nesting large gulls in Britain and Ireland appreciably underestimated numbers, resulting in lower national totals and exaggerated national population changes, because they did not account for a substantial proportion of undetected nests.
Aims To evaluate the efficiency of various survey methods for urban nesting large gulls.
Methods Vantage point and street surveys of Herring and Lesser Black-backed Gulls *Larus argentatus* and *Larus fuscus* nests were made in six urban conurbations and used to compare detection efficiencies. Some nests missed by both methods were later identified by the presence of unfledged chicks. In later years, the numbers of nests which were actually missed was determined by using a 'cherry-picker' which allowed all nests to be located in Dumfries. The proportion of nests on industrial and commercial buildings was recorded.
Results Vantage point and street surveys missed an appreciable number of nests and had average maximum nest detection rates of 78% and 48%, respectively. Combining the two methods raised the efficiency to a maximum of 88%. The detection rate varied inversely with the proportion of nest sites that were on commercial or industrial sites. The complete census of nests in Dumfries in 2013–14 showed that vantage point surveys detected only 75% of nests, while the combination of the two survey methods increased the detection rate to 84%.
Conclusion Vantage point surveys markedly underestimate numbers of nesting large gulls. Surveys were less effective on conurbations where nesting gulls used a high proportion of industrial and commercial properties. Therefore the 1999–2002 national surveys in Britain and Ireland underestimated the numbers of urban nesting Herring and Lesser Black-backed Gulls and, as a result, may have exaggerated the national trends in abundance of both species. More intensive methods are required in surveys of urban nesting gulls and correction factors need to be established from very high vantage points by use of cherry-pickers or aerial surveillance.

The numbers of large gulls nesting on buildings in towns and cities in Britain have been increasing for over half a century (Cramp 1971). More recently, this habit has created problems in coastal areas of western Europe, particularly Britain (Calladine *et al.* 2006), and is also developing in parts of the USA and Canada (Paynter 1963, Dwyer *et al.* 1996). The spread and distribution of gulls nesting in urban areas within Britain and Ireland have been well tracked by records of the increase in numbers of conurbations with nesting large gulls, mainly Herring Gulls *Larus argentatus* and Lesser Black-backed Gulls *Larus fuscus*, but also small numbers of Great Black-backed Gulls *Larus marinus* and Common Gulls *Larus canus* occur in places (Cramp 1971, Monaghan & Coulson 1977, Raven & Coulson 1997, Mitchell *et al.* 2004, Calladine *et al.* 2006). While the spread of gulls to new urban areas has continued in Britain, this spread is currently increasing at a slower rate than that experienced in the last few decades of the last century, if for no other reason than there are now appreciably fewer new urban localities remaining to be colonized. As a result, measuring the changing status of urban nesting gulls is, or soon will be, increasingly dependent upon the numbers nesting at each location. Accurate census data of the numbers nesting in urban areas are particularly important, because nationally the numbers of Herring Gulls have declined on natural nesting sites since 1970, yet have continued to increase in urban areas (Mitchell *et al.* 2004). Although the proportion nesting in urban areas is a minority, the census made in 1999–2002 reported that 13.5% of Herring Gulls in

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Britain and Ireland were nesting on buildings in urban areas, and this proportion exceeded 26% in England (Mitchell *et al.* 2004).

The methods for census work on urban gulls recommended by Walsh *et al.* (1995) involve vantage points or aerial surveys and, for the 1999–2002 national census, these methods were made mandatory (Mitchell *et al.* 2004, in Appendix 1). In practice, aerial surveys were rarely used. In our studies, we used vantage point surveys and separate ground level visits coupled with access to specific roofs where this could be achieved. This latter method is subsequently referred to as 'street surveys' for conciseness. This paper compares the effectiveness of street and vantage points surveys and draws attention to the appreciable underestimates which have been revealed by comparing results from the two methods. Because of the particular difficulty in examining the roofs at many commercial and industrial sites, the level of the underestimate of numbers was examined in relation to the proportion of breeding gulls in each urban area utilizing industrial areas. Finally the efficiency of these gull survey methods was evaluated relative to an absolute census at a single site, Dumfries, where Dumfries and Galloway Council have used a mobile lift platform (subsequently called a 'cherry-picker') to search and reach otherwise inaccessible and hidden roof nesting sites.

METHODS

Surveys during a single breeding season were made of the numbers of urban nesting Herring and Lesser Black-backed Gulls between 1994 and 2005 at South Shields, Jarrow, Sunderland and Berwick-upon-Tweed in England, in response to requests from local councils. A survey was made in Durham City, NE England in 2009. Annual surveys were made in Dumfries, SW Scotland, between 1999 and 2014 in conjunction with the Dumfries and Galloway Council and numbers have been consistently kept separated between the town centre and outlying areas for comparative purposes. Dumfries data have only been presented in this paper for the years 2002 and 2013–14 because of the similarity of the distribution of gulls in other years. All of our surveys were made in the same way and were based on recording the nest locations from vantage points and street surveys made on the same days.

Visits were made to each conurbation on several dates spread between early May and early July to accommodate late laying and variations in the nesting season, while the late visits were made to detect young at sites where no nest had been detected previously, but nesting was suspected. Vantage point surveys were made from tall buildings and natural raised ground, using binoculars and a 20–60× telescope. Access to some otherwise suitable viewing sites could not be obtained and, in an appreciable minority of areas, no suitable vantage points existed. In addition, every street was visited at least twice during the 'street survey' to locate gull nests and, during these searches, visits were made to any specific flat roofs where gulls were nesting, where access could be obtained. No aerial surveys were made.

The position of each nest was recorded to specific parts of each building (denoted by the postal address or the name of the commercial firm) to ensure that the same nests detected by both street surveys and from vantage points were identified as such. The type of building on which each nest was located was recorded, including whether it was a residential property, commercial building or an industrial structure. An attempt was made to locate nesting sites missed by both survey methods by visits made in late June or July to record places where unfledged young could be seen, yet no nest had been detected. Numbers of such nests missed by both street and vantage point surveys were obviously underestimated, because they did not include those where the eggs had been removed or all of the eggs failed to hatch and presumably there were some nesting sites where neither nests nor chicks could be seen. A value we have called the 'maximum success rate' of detecting nests from vantage points, street surveys and by combining these methods was calculated for each survey, based on the total nests located. The use of 'maximum' denotes that a small number of nests were probably not detected, and these would have inflated the denominator (total number of nests) used in the calculation and so reducing the real success rate below that reported.

The problem of accurately determining the proportion of nests missed during surveys was eventually solved when Dumfries and Galloway Council staff used a 'cherry-picker' to view and visit potential nesting sites throughout Dumfries, including all sites that could not be effectively examined during surveys. As a result, and for the first time, in three breeding seasons the absolute efficiency of nest detection by street and vantage surveys in the town was determined.

Street surveys

In residential areas with terraced houses, semi-detached and detached properties, nests on top of chimney stacks were the most frequent and were readily detected during street visits. However, on residences where chimney stacks were not at the roof apex, but protruded through part of the sloping roof, some nests were concealed in the acute angle between the upper side of the stack and the roof. Complex roof systems of large buildings had areas which were not visible from ground level or vantage points, concealing some nests. Gulls nesting on roofs of dormer windows at the rear of residential properties could also be missed if the rear of the houses could not be examined. In industrial areas, street surveys were often ineffective. Flat roofs on many industrial or commercial sites could not be viewed from the street level, while multiple 'saw-tooth' roofs often had appreciable areas which could not be examined from ground level or vantage points. Where possible, visits were made to flat roofs where gulls were suspected of nesting in order to determine the number of nests and were included in the street surveys. In many cases, roof access was not possible (see below).

Rooftop visits

Rooftop visits served two purposes: to record nests on that specific flat roof as part of the street survey, or to use rooftops as vantage points to view neighbouring properties. Many buildings lacked internal access to the roofs and so access was not possible without ladders or exterior scaffolding (which we did not use). While management at some industrial sites were co-operative in giving access to their roofs, others required written requests and disclaimers, and others refused access on grounds of health and safety regulations, while banks would not allow roof access on grounds of security.

Surveys from vantage points

Searches using binoculars or a telescope from vantage points on tall buildings or nearby hills were particularly valuable in locating nests, but the availability and suitability of such sites varied considerably within and between towns. Counts of nests from specific vantage points were often incomplete because the low angle of view concealed areas behind parapets, skylights, ventilators, structures housing lift machinery and materials stored on the roofs. Such sites needed viewing from two or even three different angles to ensure that all nests were detected and this was not always possible. For example, in Jarrow and in parts of South Shields, there were no good vantage points available and values were obtained from a considerable distance. Four sites in Dumfries, each with several pairs of nesting gulls, could not be viewed from street level or from the available vantage points.

RESULTS

Many nests were located both during street surveys (including visits to specific roofs) and from vantage points, but some nests were located by only one of the methods and neither was highly effective (Table 1).

Table 1. The numbers of nests of large gulls detected in surveys in six urban areas, based on street level surveys, numbers detected by using high buildings or natural vantage points overlooking the breeding area, and the minimum number of nests missed as indicated by the appearance of chicks at sites where no nests were observed. The percentages are the minimum success rates of each survey method and are based on the total number of nests detected, i.e. A + B + C.

	A			B		C	A + B + C	
Location	Number (and percent) detected from street level survey	Number (and percent) detected from vantage points	Number recorded both from street and vantage points	Additional number from vantage points	Number (and percent) combining vantage and street surveys	Number (and percent) later detected by chicks	Estimated total	Percent on industrial or commercial sites
Berwick- on- Tweed	77 (68)	90 (79)	60	30	107 (94)	7 (6)	114	72
Sunderland	380 (76)	442 (89)	40	102	482 (97)	17 (3)	499	62
South Shields	340 (70)	431 (88)	309	122	462 (95)	26 (5)	488	54
Jarrow	10 (13)	37 (49)	0	37	47 (63)	28 (37)	75	98
Durham City	4 (4)	76 (75)	1	75	79 (78)	22 (22)	101	99
Dumfries (central area 2002)	155 (65)	218 (91)	143	75	230 (96)	10 (4)	240	76
Dumfries (outlying area 2002)	60 (40)	121 (81)	41	80	140 (94)	9 (6)	149	85

The later detection of well grown chicks still incapable of flight on some buildings added evidence of further nests missed by both census methods (and other broods were almost certainly missed) and accordingly success rates were underestimated. The efficiency of the survey methods varied considerably from area to area within a town, but vantage points surveys were invariably more successful than street counts (Table 1). Combining the two methods gave higher, but still incomplete detection rates in all cases.

Considering individual urban areas, the most successful street surveys achieved only a maximum of 76% success in locating nests. Vantage point surveys had maximum success rates ranging between 75% and 91% at five areas, with the exception of Jarrow, where it was below 50%. Combining ground surveys and those from vantage points, the maximum success rate exceeded 90% in four of the six conurbations, but at two localities (Jarrow and Durham City) it was only 63% and 78%, respectively, emphasizing that the success rates varied considerable between localities. It is likely that at all of the localities studied, the effectiveness of the survey methods, even when combined, always appreciably underestimated the numbers of nests present.

Commercial and industrial sites were more difficult to survey and the estimated maximum success rates of street surveys declined markedly in towns and cities where more nests were on industrial or commercial properties (Fig. 1a, $r_5 = -0.93$, $P < 0.01$), mainly caused by the difficulty in viewing nests on flat roofs. The negative linear correlation between the maximum success rates of vantage point surveys and the proportion of gulls nesting on industrial and commercial buildings was less marked and only reached significance (Fig. 1b, $r_5 = -0.754$, $P = 0.05$), but a significant correlation still existed when vantage and street surveys were combined (Fig. 1c, $r_5 = -0.78$, $P < 0.05$). Although statistically significant, these relationships are too imprecise to calculate an accurate correction factor for the negative effect of industrial and commercial properties on census accuracy. However as a preliminary indication, when commercial and industrial buildings exceeded 80% of those used by gulls, the maximum success rate of survey is decreased by about one percentage point for each percent point that this value of 80% commercial and industrial buildings is exceeded. More data from additional localities are needed before a more precise correction factor can be determined.

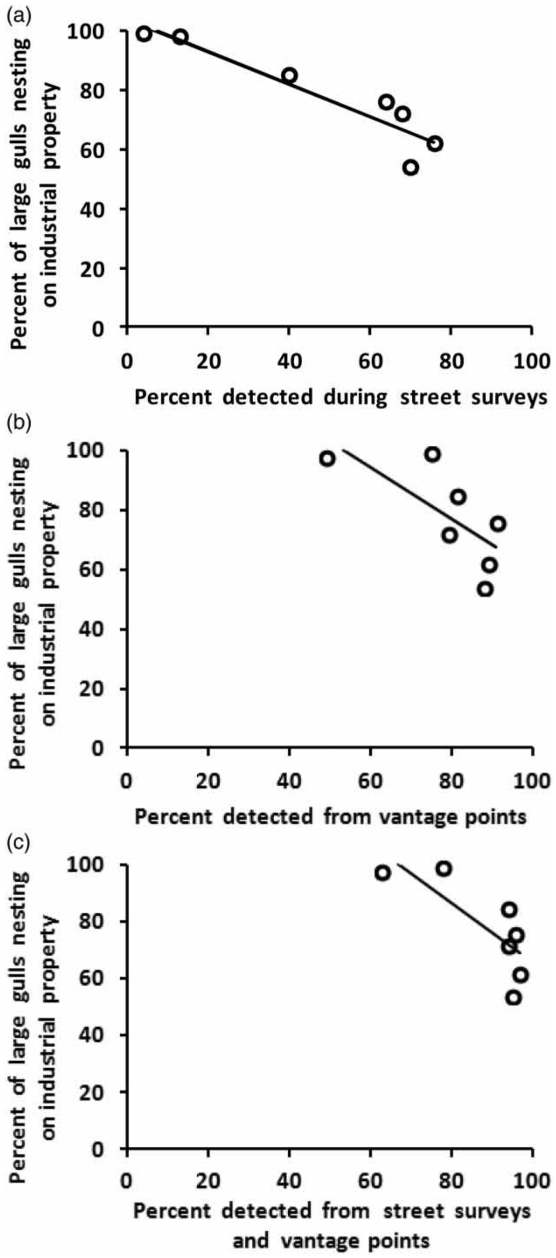

Figure 1. Correlations between the percentage of nests on industrial and commercial buildings and (a) percentage of nests detected by street surveys ($P < 0.01$, linear regression $y = 103 - 0.545x$), (b) percentage of nests detected from vantage points ($P = 0.05$, linear regression $y = 146 - 0.86x$) and (c) percentage of nests detected by both vantage point and street surveys ($P < 0.05$, linear regression $y = 170 - 0.78x$). Data from Table 1.

Absolute counts

The use of a cherry-picker throughout Dumfries increased the proportion of nests which had not been recorded by street or vantage point surveys by 15.5%

Table 2. The efficiency of street surveys, vantage point surveys, combined street and vantage point surveys and the percentage of nests detected only from the mobile platform (cherry-picker) in Dumfries, 2013 and 2014, of the likely absolute number of nests present.

Year	Total nests	Detected by street survey	Detected by vantage points	Additional detected by street survey	Additional detected by cherry-picker	Percent detected by street and vantage points
2013	687	57%	76%	9%	15%	85%
2014	563	55%	72%	12%	16%	84%
Average		56%	74%	10.5%	15.5%	84.5%

(Table 2). Those missed were mainly on industrial or commercial sites, particularly on roofs with a parapet around the edge or those screened by trees, which concealed nests from the street or vantage points. Flat and saw-tooth roofs, where many nests could not be observed or counted, were also present in South Shields, Jarrow, Sunderland, Durham City and Berwick-upon-Tweed and probably occur in most urban nesting areas, contributing to the under-recording of nests. In Dumfries, the recommended method, that of using vantage point surveys, found only 72% and 76% of nests present during each of two years. The combination of vantage and street surveys increased the detection rate in Dumfries to 84.5%. In earlier years in Dumfries, the presence of unfledged chicks at sites where nests were not otherwise identified allowed detection of only about a third of those nests otherwise missed by the survey methods.

DISCUSSION

Because of difficulties in access and viewing, nests counts used to record the numbers of adult large gulls breeding in urban areas have appreciably underestimated numbers. This study evaluated the accuracy of census work on urban nesting gulls using the method of counting from vantage points, which was used in the 1999–2002 national survey protocol (Mitchell *et al.* 2004, Appendix 1) and by comparing these with nests identified from street surveys made at the same time. The percentage of nests detected by both survey methods considerably underestimated of the total obtained by combining the two methods, and must also be an underestimate as the more recent cherry-picker data suggests. The use of vantage points detected only an average of 76% of nests, but was considerably lower at some localities. The success rate in Sunderland was highest because of the numerous vantage points available. Street surveys were consistently less successful than those made from vantage points, but they did detect numbers of nests missed by the latter method and combining the two methods enhanced the maximum success rate in all cases.

Precise measures of the success of survey methods require determining the absolute number of nests in an urban area, but has been difficult to achieve. This problem was overcome in Dumfries by the use of a cherry-picker to overlook and detect those sites where nests were concealed and had been missed by the vantage point and street surveys. This complete count of the numbers of large gull nests in the town revealed that, in Dumfries at least, the vantage point method was only 75% efficient. While the success of vantage point surveys undoubtedly vary from one area to another, Dumfries would appear to be typical of many towns and cities, a suggestion supported by the similar maximum success values at Dumfries and at the first three localities listed in Table 1. However, numbers detected at some conurbations were underestimated to a much greater extent (Jarrow and Durham City, Table 1) and this was related to the nests being mainly restricted to industrial and commercial buildings, which offered poor vantage points and frequently had roofs with concealed sections.

This study has identified marked weaknesses in the current methods of determining numbers of large gulls nesting in urban areas, but alternative methods are expensive. While it cannot be expected that all future counts of gulls nesting in urban areas can be accompanied by the use of a cherry-picker, we recommend that both vantage point and street surveys, at least, are used in future surveys. This study suggests that vantage point surveys should be multiplied by 1.33 and those combining both vantage and street surveys multiplied by 1.19, but these should be regarded only as preliminary recommendations. There is a need for the correction factors to be determined by data based on more conurbations where cherry-pickers are used in tandem with other survey methods. In addition, it would be valuable in future census studies to record the proportions of nests in each area that were detected on commercial or industrial sites, because this was the main variable we detected that affected efficiency of the vantage and street surveys and so this additional information could be used to improve correction factors.

The findings from this investigation have implications which extend beyond counts in individual conurbations and affect the estimated size of the national populations of large gulls and the numbers nesting at urban sites and consequently questions the endangered 'red status' currently applied to the Herring Gull. First, the total numbers of urban nesting gulls in Britain were likely appreciably underestimated in the 1999–2002 national census (Mitchell *et al.* 2004). A correction of the total nests in urban areas, obtained by vantage point surveys, requires these to be increased by a multiplying factor of 1.33 and would increase the estimated numbers of urban nesting Herring Gulls in 1999–2002 to at least 27 000 adult pairs, rather than the 20 170 recorded. Similarly, the 10 874 recorded pairs of urban nesting Lesser Black-backed Gulls was perhaps in reality approximately 14 600 pairs. These corrections raise the proportions of Lesser Black-backed Gulls nesting on urban sites in 1999–2002 to 12% of the British and Irish population, while the corresponding value for Herring Gulls breeding in urban areas increases to 18% (and to 29% in England). In the period 1999–2002, our revised estimates indicate that over 83 000 large gulls were nesting in urban areas of Britain. Since there are indications that urban nesting is still increasing at the present time, the numbers and proportions nesting in urban areas are now probably even greater than in 1999–2002. These higher numbers and proportions in urban areas will increase concerns of the public and local administrative councils, who are still searching for effective, non-lethal methods of reducing or even excluding large gulls from nesting in towns.

Second, the impact caused by underestimating numbers of urban nesting gulls would have had more effect on the national totals in the 1999–2002 census than in earlier years, when urban nesting was less extensive. The appreciable underestimation of the numbers of urban nesting Herring Gulls in Britain in the 1999–2002 census requires adjustment and an appreciable increase in the numbers presented in Mitchell *et al.* (2004). The reported decline of Herring Gulls in Britain (excluding Ireland) between 1985–88 and 1999–2002 was −2% (Mitchell *et al.* 2004), although the Joint Nature Conservancy Committee (JNCC 2014) using the same data set give slightly different numbers and changes in abundance. The correction to the number of urban nesting gulls would change this decline to an overall 2% *increase* over this 15 year period. Since the 1999–2002 census, the status of the Herring Gull in Britain has been monitored annually from samples based on a modest number colonies (but none at urban sites) and the results reported annually by the Joint Nature Conservancy Committee since 2000 (JNCC 2014). They show no significant change in Herring Gulls abundance at any time since the 2002 census and up to 2014. Including immature individuals, the summer numbers of Herring Gulls in Britain would seem to have exceeded a third of a million individuals (revised 1999–2002 census numbers, with 30% added for immature gulls; J. Coulson unpubl.) for at least the past 28 years.

Finally, Eaton *et al.* (2009), using data available since the 1999–2002 census and sample data up to that time, placed the Herring Gull in the red category status as a threatened species in Britain. Yet the evidence of major or consistent declines in abundance of this species in Britain since 1986 is equivocal and the endangered status given to the extremely abundant Herring Gull probably requires reconsideration.

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The role of industrial fisheries and oceanographic change in the decline of North Sea black-legged kittiwakes

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Summary

1. Marine predators can be adversely affected by human activities in several potentially interacting ways. Industrial fisheries can affect predator populations adversely through competition for shared prey, and marine environmental change has also been implicated in population declines. In the North Sea, black-legged kittiwakes *Rissa tridactyla* have declined by > 50% since 1990, a period during which a lesser sandeel *Ammodytes marinus* fishery was active and profound oceanographic changes occurred. We studied the role of fisheries and oceanography in kittiwake declines on the Isle of May, south-east Scotland, where sandeels are the main prey.

2. Demographic data were collected from 1986 to 2002. We examined changes over time and correlations between population parameters, the local sandeel fishery and environmental factors, and incorporated the results in a deterministic and a stochastic matrix population model.

3. Breeding success and adult survival were low when the sandeel fishery was active (1991–98) and were also negatively correlated with winter sea temperature, with a 1-year lag for breeding success. Modelling showed that the observed changes in population parameters could explain the change in population growth rate from +8% year^{-1} in the late 1980s to −11% year^{-1} in the late 1990s. An observed improvement in breeding success from 2000 onwards has been insufficient to arrest the population decline. To stabilize the population, breeding success must increase to unprecedented levels or survival needs to increase substantially. Stochastic modelling indicated that the population was unlikely to increase if the fishery was active or sea temperature increased, and that the population was almost certain to decrease if both occurred.

4. Sandeel recruitment is reduced in warm winters, and we propose that this explains the temperature effects on kittiwake survival and breeding success. The fishery also had a strong effect on demographic performance, although the exact mechanism is unclear as kittiwakes and fishermen target different sandeel age groups.

5. *Synthesis and applications*. Poor breeding success of kittiwakes was associated with warm winters and the presence of a local sandeel fishery. Reversing the trend towards warmer winters may be impossible and, at best, would be a very slow process. Therefore, to safeguard kittiwake populations we recommend that the current closure of the commercial sandeel fishery remain in place indefinitely. This study shows that adequate monitoring of the effect of a fishery or of environmental change on seabird populations needs to include survival as well as breeding success.

Key-words: adult survival, *Ammodytes marinus*, breeding success, oceanography, population model, sandeels

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Introduction

One of the main issues in fisheries management is assessing and reducing fishery effects on other predators such as marine mammals and seabirds (Ormerod 2003). Such effects may be direct, as in by-catch mortality (Tuck *et al*. 2001), or indirect through reductions in food supplies. During the 20th century, many seabird species showed large increases in population size in the north-east Atlantic (Mitchell *et al*. 2004). For some, these increases were probably linked to growing and changing human fisheries (Montevecchi 2002), either through provisioning of extra food in the form of discards and offal (Garthe, Camphuysen & Furness 1996) or through high fishing pressure on large predatory fish leading to increased availability of the smaller fish that usually form the main prey of seabirds (Sherman *et al*. 1981; Furness 1982). However, towards the end of the last century, the development of industrial fisheries harvesting small forage fish has led to direct competition between humans and seabirds. Regional declines have become apparent in some seabird species (Anker-Nilssen, Barrett & Krasnov 1997; Heubeck *et al*. 1999) and some have been related to human fisheries (Anker-Nilssen, Barrett & Krasnov 1997). Overexploitation has been suggested as the cause of population crashes of some forage fish species and concomitant declines in seabird populations during the last 50 years (Newton 1998).

The largest single-species industrial fishery in the North Sea harvests the lesser sandeel *Ammodytes marinus* Raitt, up to a million tonnes of which are landed each year (ICES 2003). Lesser sandeels are an important prey for most species of seabirds around the North Sea during the breeding season (Monaghan 1992; Wanless, Harris & Greenstreet 1998). Among these is the black-legged kittiwake *Rissa tridactyla* (L.), which in this region feeds almost exclusively on lesser sandeels (Harris & Wanless 1997; Lewis *et al*. 2001). Adult kittiwakes eat mostly older (1+ year group) sandeels during April and May, but switch to juvenile (0 year group) sandeels for both themselves and their young in June and July (Harris & Wanless 1997; Lewis *et al*. 2001). The number of kittiwakes breeding along the British North Sea coast declined by 20–50% between *c*. 1986 and 2000, with even stronger declines in Shetland (Heubeck *et al*. 1999; Heubeck 2004). The cause and mechanism of these declines are unclear, although consistently low breeding success was observed in Shetland and south-east Scotland (Mavor *et al*. 2003). The underlying cause is likely to be low availability of sandeels, but in the absence of long-term sandeel monitoring outside Shetland correlations have to be sought between kittiwake performance and proxies thought to be affecting sandeel abundance.

In 1990, a summer industrial fishery for sandeels started around the Wee Bankie off the Firth of Forth, south-east Scotland. Landings from this fishery quickly grew to 100 000 tonnes in 1993 but then stabilized at a lower level (see Appendix 1). Subsequent research indicated that the Wee Bankie fishery almost certainly caused local depletion of sandeel numbers, which coincided with reduced breeding success of kittiwakes (Rindorf, Wanless & Harris 2000). In 2000, following advice from the International Council for the Exploration of the Sea (ICES) Advisory Committee on the Marine Environment, summer fishing for sandeels was banned in 20 000 km^2 of sea to the east of Scotland and north-east England, an area that included the Wee Bankie (Wright *et al*. 2002). A limited scientific fishery was maintained in the exclusion zone in 2000–01 (see Appendix 1).

Changes in oceanographic conditions can also exert a profound influence on ecosystem structure and hence top predator populations (Hunt *et al*. 2002). Physical and biological oceanographic conditions in the North Sea have changed markedly in recent decades (Beaugrand 2004), and high winter sea temperatures have been shown to reduce recruitment of sandeels, which spawn in winter (Arnott & Ruxton 2002). There is some evidence that seabird population dynamics in the region may be linked to ocean climate fluctuations (Thompson & Ollason 2001; Durant, Anker-Nilssen & Stenseth 2003). Previous studies of kittiwakes have shown a link between breeding success and prevailing weather patterns (Aebischer, Coulson & Colebrook 1990), but no study so far has investigated the combined effects of oceanography and fisheries on a North Sea seabird.

Since 1986, the Isle of May in the outer Firth of Forth has been part of the UK Joint Nature Conservation Committee (JNCC) seabird monitoring programme. Numbers, breeding success and adult survival of kittiwakes and other seabird species are monitored annually. Kittiwakes breeding on the Isle of May forage largely within the area targeted by the Wee Bankie sandeel fishery (Wanless, Harris & Greenstreet 1998; Daunt *et al*. 2002) and are thus potentially vulnerable to declines in sandeel abundance caused by the fishery. We used data collected between 1986 and 2002 to (i) estimate population parameters and model the growth of the Isle of May kittiwake population over this period; (ii) investigate to what extent the Wee Bankie sandeel fishery and oceanographic change are likely to have affected population growth; and (iii) predict future population growth under various scenarios of fishery and oceanographic conditions.

Methods

STUDY AREA AND FIELD METHODS

The Isle of May (56°11′N, 2°33′W) is one of the most important seabird colonies along the British North Sea coast. We have monitored population size, breeding success and adult survival of kittiwakes at the Isle of May since 1986. Single total counts of the number of completed kittiwake nests (Harris 1987) were carried out in late May or early June, when egg laying was completed, every year from 1986 to 2002. In 15 plots distributed to give a good geographical spread within the whole colony, active nests were identified and counted during early incubation, and several visits were made during the fledging period to determine the success of each nest. Plot limits have only been constant since 1992, so earlier plot counts of nests are not used here. The unit used for quantifying annual mean breeding success was the number of fledged chicks per completed nest (Harris 1987). Annual estimates presented are means [with 95% confidence limits (CL)] across plots, weighted by plot sample size. These means were then used in a multiple regression analysis (weighted by inverse standard error) to identify environmental covariates of breeding success (see below).

Breeding kittiwakes have been ringed with unique colour-ring combinations and metal rings every year since 1986. Initially, all birds were ringed in two study plots, but as the population declined it became difficult to maintain a satisfactory sample of marked birds, so from 1996 birds were also ringed in five other plots dispersed around the island. Searches for colour-ringed birds throughout the breeding season were concentrated in these areas, but regular searches have also been made elsewhere and some birds (maximum 15 year^{-1}) have been located as breeders elsewhere on the island. A total of 470 breeding adults was colour-ringed in 1986–2001; 69 of these had originally been ringed as chicks (1970–89) so were of known age. The total number of colour-ringed birds observed annually (including newly ringed birds) ranged from 129 to 181 (mean 148).

ADULT SURVIVAL ANALYSIS

We used MARK (White & Burnham 1999) to estimate annual probabilities of survival and resighting of breeding kittiwakes, and U-CARE (Choquet & Pradel 2002) to test goodness-of-fit of the initial model. Because birds were less likely to be seen if they moved outside the main study plots, at least until first detection at a new site, we expected to find evidence of 'trap-happiness' (i.e. higher resighting probability of birds seen the previous year). We therefore analysed capture histories using a recent multistate approach to modelling trap dependence (L. Crespin, personal communication). In this approach, two states were defined: 'seen' and 'not seen'. Resighting probabilities were then estimated as transition probabilities to the seen state, separately for birds seen and those not seen in the previous year, whereas the nominal resighting probabilities in MARK were fixed to 1 for the seen state and 0 for the not seen state (Frederiksen, Wanless & Harris 2004). In contrast to previous techniques for dealing with trap dependence (Pradel 1993), the multistate approach allows simultaneous modelling of age and trap dependence in survival and resighting probabilities.

We tested the effect of relative age (time since colour-ringing, hereafter referred to as 'age') and time on survival and resighting probabilities by fitting models including either constancy, categorical effects, linear or quadratic trends over 'age' or time. Model notation follows the principles of Lebreton *et al*. (1992): survival and resighting probabilities are denoted ϕ and p, respectively. Subscripts indicate predictors included in the model (see Appendix 2), with capital letters used for linear trends over 'age' or time. An asterisk between two terms indicates that an interaction term is included, and a plus sign indicates an additive model without interactions. Specific hypotheses about environmental effects on survival were tested, with models constraining variation over time to be a function of one or more covariates (see below). The most appropriate model to describe the data was selected using AIC$_c$, Akaike's information criterion adjusted for sample size (Burnham & Anderson 1998), and the importance of covariates was assessed as the proportion of between-year variation explained (analogous to R^2 in regression).

ENVIRONMENTAL CORRELATES OF BREEDING SUCCESS AND SURVIVAL

To explore how kittiwake breeding success and survival was influenced by environmental factors, we tested for correlations with four covariates. (i) Kittiwakes winter throughout the North Atlantic (Coulson 2002), and survival as well as breeding success might be affected by winter weather conditions in the same or the previous year. We tested for a correlation with the winter North Atlantic Oscillation index (NAO; obtained from the US National Center for Atmospheric Research, http://www.cgd.ucar.edu/~jhurrell), a widely used summary measure of winter conditions throughout the kittiwake range (Drinkwater *et al*. 2003; Hurrell *et al*. 2003), for either the current or previous winter. (ii) Previous studies (Rindorf, Wanless & Harris 2000) found a correlation between kittiwake breeding success and the local abundance of sandeels in June, measured as catch per unit effort (CPUE), during the period when the fishery operated. CPUE data were only available for 1991–2001 (Wright *et al*. 2002), and for survival we used the mean of those years as the value for years without data; for breeding success, we report results restricting analysis to those years when CPUE data were available. (iii) The Wee Bankie sandeel fishery might affect kittiwake breeding success (Rindorf, Wanless & Harris 2000) or survival, either directly through food depletion or indirectly through disturbance, and we tested this hypothesis by including the presence or absence of a fishery in the models. We scored the fishery as present from 1991 to 1998, the period when landings were substantial (see Appendix 1). Despite some fishing taking place, 1990 and 1999 were scored as non-fishery years (but see the Results). (iv) Winter sea surface temperature (SST) has been found to affect sandeel recruitment (Arnott & Ruxton 2002), so we tested for correlations between breeding success and survival and inshore SST in February/March (obtained from the German Bundesamt für Seeschifffahrt und Hydrographie, http://www.bsh.de). A negative correlation with SST in the immediately preceding winter would indicate reliance on 0 group sandeels, whereas a negative correlation with SST 1 year earlier would imply that 1+ group sandeels were more important.

POPULATION MODELLING

A fully parameterized population model for Isle of May kittiwakes would need estimates of juvenile (first-year) survival, emigration and immigration (mostly by pre-breeders) and the age of recruitment to the breeding population. Our data were insufficient to estimate these parameters, but we had recorded emigration to, and immigration from, other colonies. Net recruitment (first-year survival corrected for net dispersal) was therefore estimated by fitting the model to the observed time series of population counts.

The first step was to develop a model to describe the growth of the kittiwake breeding population on the Isle of May 1986–2002 and assess whether the observed changes in growth rate could be explained by observed changes in breeding success and adult survival. We constructed a deterministic, time-specific matrix model (Caswell 2001) with 15 age classes. The initial age distribution was based on asymptotic properties of a model, with mean parameter values as observed from 1986 to 1989, when breeding success and survival were high (see the Results). The model was then parameterized with our annual estimates of breeding success and adult survival. A separate analysis of survival of the subsample of known-age birds indicated a strong quadratic relationship between true age, from age 2 onwards, and survival, with pronounced senescence (Frederiksen, Wanless & Harris 2004), and we used this relationship in the model rather than the observed relationship with 'age'. Age- and year-specific survival were thus modelled as a combination of results from the two analyses:

$$\mathrm{logit}(\phi_{a,t}) = 1{\cdot}2 + 0{\cdot}334a - 0.020a^2 + (\beta_t - \bar{\beta})$$

where a indicates true age, t year, β_t annual effects on the logit scale from model ϕ_{A+t}, p_{t+m} (see the Results) and $\bar{\beta}$ the mean of annual effects from the same model. Under this model, mean survival increases from 0·86 at age 2 to 0·93 at age 7–10 years [we refer to this life stage, corresponding to the mean age at colour-ringing (8·3 years, $n = 69$) as prime age], and declines to 0·85 at age 15. Second-year survival (from age 1–2 years) was estimated by extrapolating this relationship. Previous studies have shown that most kittiwakes first breed when aged 3–5 years (Porter & Coulson 1987; Cam *et al.* 2002); we included this in the model by assuming that 25% of 3 year olds, 75% of 4 year olds and all birds 5 years or older are breeders. We then ran the model with varying values of first-year survival to (i) assess whether observed changes in adult survival and breeding success could explain population growth, or whether changes in recruitment also needed to be invoked, and (ii) find the value for net recruitment (first-year survival including net dispersal) that provided the best fit to the data.

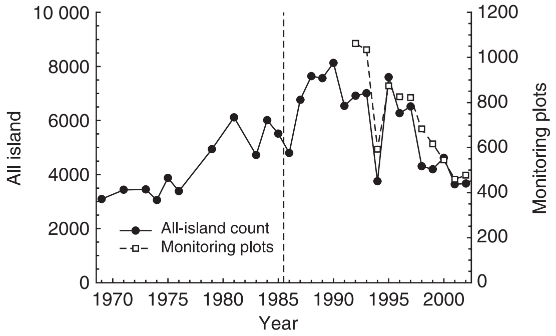

Fig. 1. The number of completed kittiwake nests counted on the Isle of May 1969–2002 (left axis), and in the 15 productivity monitoring plots 1992–2002 (right axis). The dashed line indicates the onset of annual monitoring of population size and parameters. Data from this study and Isle of May Bird Observatory and Scottish Natural Heritage records.

As a second step, we modelled how the population might recover from the observed severe decline since 1995 (Fig. 1). We took the predicted age distribution from the first modelling exercise in 2000, i.e. after 10 bad breeding years, as a starting point and constructed a constant, deterministic model in ULM (Legendre & Clobert 1995). Within this model, we ran scenarios with varying values of breeding success, adult and first-year survival, in order to evaluate which improvements in these parameters would be necessary to induce stability in the population. Finally, we used a stochastic matrix model to predict population growth under various scenarios of fishery and oceanographic change (see Appendix 3).

Results

POPULATION SIZE

Continuing a trend observed since 1969, the number of completed kittiwake nests counted increased from 4801 in 1986 to a peak of 8129 in 1990; subsequently numbers declined to 3666 in 2002 (Fig. 1). The pronounced dip in 1994 corresponded to a documented non-breeding event; *c*. 40% of all birds attending the colony early in the season did not build nests. Both the all-island counts and the monitoring plots showed a decline of about 50% from 1995 to 2001 (Fig. 1), or approximately 11% year^{-1}.

BREEDING SUCCESS

Breeding success varied widely, from 1·24 fledged chicks per completed nest in 1986 to 0·02 in 1998. During 1986–89, breeding success was high, with a mean of 1·07 (95% CL 1·00–1·14; Fig. 2). The 10 subsequent seasons (1990–99), i.e. the period when the Wee Bankie sandeel fishery was active, were characterized by low and variable breeding success [mean 0·30 (0·26–0·34)], followed

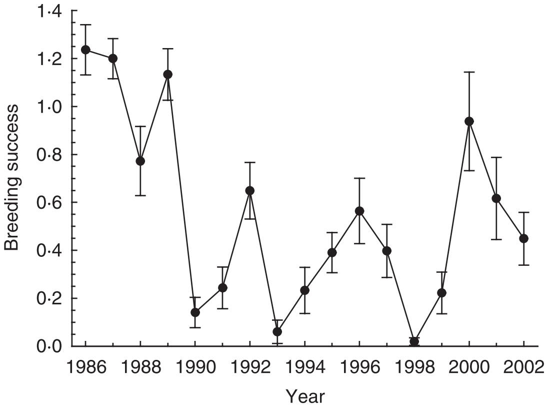

Fig. 2. Kittiwake breeding success on the Isle of May 1986–2002, expressed as the number of fledged chicks per completed nest. Weighted means and 95% confidence limits of 15 monitoring plots (10 in 1986) are shown.

Table 1. Percentages of the total between-year variation in breeding success and adult survival explained by linear and quadratic trends, as well as several environmental covariates. Values shown are R^2 for breeding success and proportion of deviance explained for survival. Results for survival are shown for models with 'age' dependence in either survival or resighting probability (see text). For breeding success, the amount of variation explained by CPUE refers to an analysis using only years when this covariate was available (1991–2001)

		Survival	
Covariate	Breeding success	'Age'-specific survival	'Age'-specific resighting
Year (linear trend)	33	30	28
Year (quadratic trend)	64	42	50
Fishery (1991–98)	40	14	24
NAO	0·6	0·1	0·2
NAO (lagged)	22	9	5
SST	22	14	26
SST (lagged)	45	6	12
CPUE	17	0·5	0
Fishery + SST	50	35	52
Fishery + SST (lagged)	81	21	35

by a partial recovery in 2000–02 after the closure of the fishery [mean 0·68 (0·57–0·79)]. When all fishery years (1990–99) were included, the presence or absence of a fishery explained 70% of the observed between-year variation in breeding success, but if fishery was only scored as present in 1991–98, i.e. when landings were substantially higher than in the scientific fishery 2000–02 (see Appendix 1), this proportion dropped to 40% (Table 1). Breeding success was also highly correlated with winter SST in the previous year, and together these two variables explained 81% of the variation (Table 1 and Fig. 3) (type III multiple regression: fishery $F_{1,14} = 26{\cdot}6$, $P = 0{\cdot}0001$; SST_{t-1} $F_{1,14} = 30{\cdot}3$, $P < 0{\cdot}0001$; interaction $F_{1,13} = 0{\cdot}23$, $P > 0{\cdot}5$). Other environmental covariates explained only minor proportions of the variation, whereas a quadratic trend summarized the data well (Table 1).

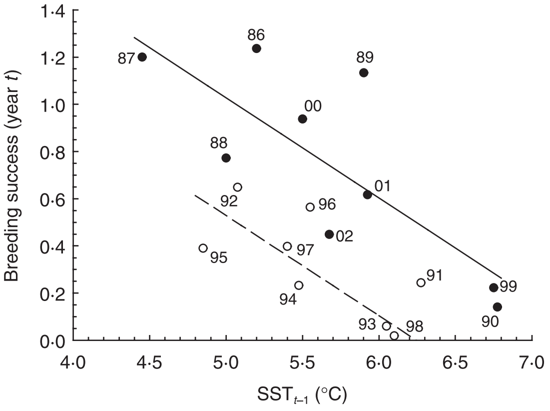

Fig. 3. Kittiwake breeding success as a function of local SST in February–March of the previous year and presence/absence of the Wee Bankie sandeel fishery. Data labels indicate current year. Regression lines estimated from weighted multiple regression (see text). Filled circles and solid line, non-fishery years; open symbols and dashed line, fishery years.

Table 2. Model selection for the analysis of kittiwake adult survival. Models are ranked according to ΔAIC_c, i.e. the difference in AIC_c between the model in question and the best model. For explanation of model subscripts, see Appendix 2

	Model	Deviance	Number of parameters	ΔAIC_c
1	ϕ_{A+T2}, p_{t+m}	807·04	21	0
2	ϕ_{F+SST}, p_{A*m}	835·64	7	0·26
3	ϕ_{A+T}, p_{t+m}	810·05	20	0·97
4	ϕ_{T2}, p_{A*m}	836·42	7	1·03
5	$\phi_{A+F+SST}, p_{t+m}$	808·81	21	1·77
6	ϕ_{A+T2}, p_{t*m}	780·67	35	2·32
7	ϕ_{A+T2}, p_{A*m}	836·30	8	2·93
8	ϕ_{A+SST}, p_{t+m}	814·03	20	4·96
9	ϕ_{A+F}, p_{t+m}	814·16	20	5·09
10	$\phi_{A+F+SST-1}, p_{t+m}$	812·24	21	5·20
11	$\phi_{A+NAO-1}, p_{t+m}$	815·28	20	6·20
12	ϕ_{A}, p_{t+m}	817·69	19	6·58
13	$\phi_{A+SST-1}, p_{t+m}$	816·19	20	7·11
14	ϕ_{A+CPUE}, p_{t+m}	817·57	20	8·50
15	ϕ_{A+NAO}, p_{t+m}	817·65	20	8·58
16	ϕ_{A+t}, p_{t+m}	792·25	33	9·77
17	ϕ_{a+t}, p_{t+m}	776·23	47	22·73
18	ϕ_{t}, p_{t+m}	821·37	32	36·84
19	ϕ_{t}, p_{t*m}	795·54	45	37·88

ADULT SURVIVAL

Initial goodness-of-fit tests of model ϕ_t, p_t with time-dependence in survival and resighting probabilities showed a highly significant lack of fit in one component (directional test for trap-happiness: $z = -14{\cdot}2$, $P \ll 0{\cdot}001$), indicating capture heterogeneity in the form of immediate trap dependence (Pradel 1993). The remaining test components showed no evidence of lack of fit ($\chi^2 = 47{\cdot}8$, d.f. = 43, $P = 0{\cdot}28$), indicating satisfactory fit of model ϕ_t, p_{t*m} including trap dependence, which we therefore used as a starting point for the analysis. Table 2 illustrates the process of model selection for the survival analysis. Temporal variation in resighting

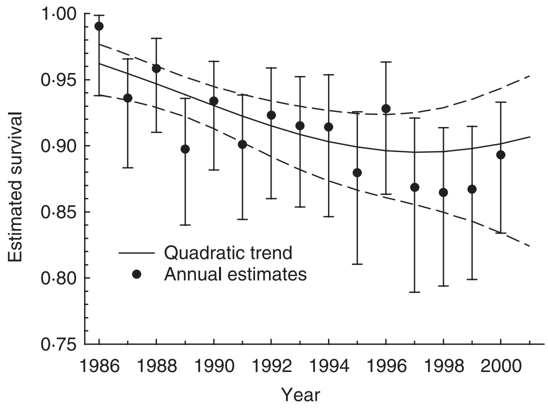

Fig. 4. Survival estimates of Isle of May kittiwakes (from model 1 in Table 2), with 95% confidence limits. The line shows a quadratic trend over time in survival during the year immediately after colour-ringing (prime-age survival). For comparison, annual estimates from the model with full variation over time (model 16 in Table 2) are also shown.

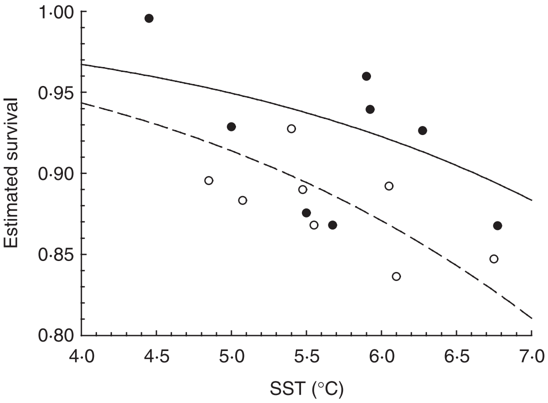

Fig. 5. Kittiwake survival from year *t* to *t* + 1 as a function of local SST in February–March of year *t* and presence/absence of the Wee Bankie sandeel fishery. Regression lines are estimated from a statistical model and are parallel on a logit scale. For clarity, results are shown from a model with 'age'-dependence in resighting probability rather than survival (model 2 in Table 2). Also shown are point estimates from the equivalent model with full variation over time. Filled circles and solid line, non-fishery years; open symbols and dashed line, fishery years.

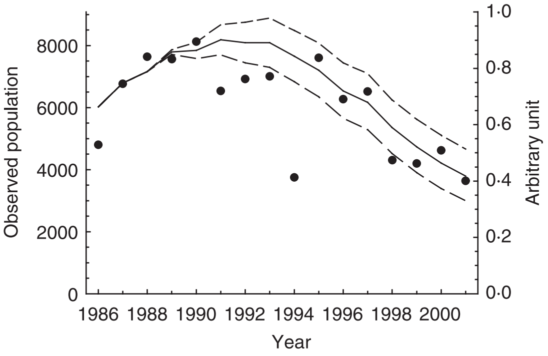

Fig. 6. Observed (symbols) and modelled (lines) size of the kittiwake breeding population. Model results are shown for three values of first-year survival (from the top 0·5, 0·4 and 0·3) and are scaled to be comparable with the counts.

probability p was similar for individuals seen or not seen in the previous year (model 18 vs. model 19). The fit of the model was improved by the addition of a categorical 'age' effect in survival (model 17 vs. model 18), and the 'age' effect was well summarized by a linear trend (on the logit scale; model 16 vs. model 17). Between-year variation in survival was not pronounced (categorical year effect not needed; model 12 vs. model 16) but could be summarized by a quadratic trend (model 1 vs. model 12), explaining 42% of total between-year variation (Table 1). Environmental covariates were less successful at explaining between-year variation in survival (Table 1; models 8–15 in Table 2), although a model with fishery and SST was almost as good as the quadratic trend (model 5). An alternative parameterization, with 'age'-dependence in resighting probability rather than in survival, had a very similar $\mathrm{AIC_c}$ (Table 2, model 4); under this parameterization, 'age'-dependence in survival was not supported (model 7 vs. model 4). All environmental effects were more pronounced under this parameterization, i.e. they explained a larger proportion of the annual variation (Table 1; model 2 in Table 2, other results not shown). Prime age survival estimates from the highest ranked model showed a decline during the first 10 years of the study, from about 0·96 to 0·90, followed by stabilization or a slight improvement (Fig. 4). At the same time, survival declined with 'age' from about 0·9 to about 0·7 after 15 years. Both fishery and SST had negative effects on survival (Fig. 5; likelihood ratio test with 'age' dependence in survival: fishery $\chi^2_1 = 5{\cdot}23$, $P = 0{\cdot}022$; SST $\chi^2_1 = 5{\cdot}36$, $P = 0{\cdot}021$; interaction $\chi^2_1 = 1{\cdot}63$, $P = 0{\cdot}20$; both effects were more pronounced with 'age' dependence in resighting probability). As expected from their respective dependence on SST, survival was positively correlated with breeding success in the following year ($r = 0{\cdot}67$, $P = 0{\cdot}0042$; see Appendix 4). Resighting probabilities varied between years, but were generally high for birds seen in the previous year (weighted mean from model 1: 0·93) and much lower for those not seen in the previous year (weighted mean: 0·37).

POPULATION MODELLING

The starting age distribution of the simulation model was obtained from the asymptotic properties of a model using mean parameter values 1986–89 and a first-year survival of 0·4 (see below); the asymptotic population multiplication rate λ predicted from this model was 1·076. Using annual estimates of breeding success and survival, we attempted to find the value of first-year survival that would provide the best agreement between the observed and modelled size of the breeding population. A good fit was obtained for a value of 0·4 (Fig. 6); taking this value as constant, the model described well both the population increase during the late 1980s and the subsequent decline (Fig. 6). As expected from the observed high incidence of

Table 3. Improvements in mean population performance needed to stabilize the Isle of May kittiwake population, according to a deterministic matrix model. Basic population parameters were: breeding success, 0·68; age-dependent survival with prime age adult survival, 0·908; first-year survival, 0·4. Also shown is the initial decline expected from the 2000 population size (see also Appendix 5)

Parameter	Value needed	Improvement needed (%)	Initial decline (first 10 years) (%)
Breeding success (BS)	1·17	72	31
Prime-age survival (ϕ_{ad})	0·941	3·6 (36% decline in mortality)	19
First-year survival	0·688	72	31
Improvement in several parameters (for example)	*BS* 0·9 ϕ_{ad} 0·925	32 1·9 (18% decline in mortality)	25

non-breeding in 1994, the count for that year was much lower than predicted by the model. In 2000, after 10 bad breeding years (Fig. 2), the modelled age structure of the population had changed considerably; the proportion of the population more than 10 years old was predicted to be 61%, in contrast to 25% in the starting age distribution.

To investigate whether the change in population growth rate was linked to changes in survival and/or breeding success, we ran the model with one or both of these sets of parameters fixed at the observed mean for 1986–89. Predicted breeding population size in 2001 was 2·43 times as high as under the best-fit model if no change in survival was assumed, 2·03 times as high if no change in breeding success was assumed, and 4·85 times as high if both survival and breeding success were assumed to have remained at the favourable levels recorded during the early years of the study.

In the next step, we used the predicted age distribution in 2000 as a starting point for prospective modelling, aiming to investigate how likely the population would be to stabilize and recover. The growth of the population was simulated under several scenarios for improvement in various population parameters, starting in 2000. The basic model, using the observed mean breeding success 1990–99 (0·30 fledged chicks pair^{-1}) and a prime age survival as observed in the late 1990s (0·908), predicted a λ of 0·894, i.e. close to the observed 11% decline per year. Incorporating the observed improvement in breeding success 2000–02 (0·68) changed λ to 0·950 and was thus not sufficient to stabilize the population. Table 3 lists the improvements needed to achieve $\lambda = 1$, i.e. asymptotic population stability, as well as the initial decline due to the unbalanced age structure (assuming improvements take effect from 2000). All scenarios predicted that even if further improvements took effect from 2000, a decline of 19–31% would still be expected over the first few years; this accords well with population counts 2000–03 (see Appendix 5).

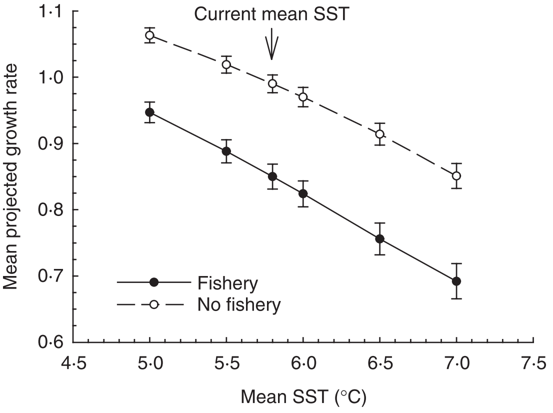

Fig. 7. Mean projected growth rate λ (± 2 SD) of the Isle of May kittiwake population from a stochastic matrix model (see Appendix 3 for details), as a function of mean winter SST and the presence/absence of the Wee Bankie sandeel fishery. $\lambda = 1$ indicates population stability.

The stochastic model indicated that the projected mean growth rate was strongly dependent both on SST and on whether a fishery was operating (Fig. 7). In the presence of the fishery, population increase was unlikely even if SST was substantially lower than today. It was also apparent that a continued decline in the kittiwake population was likely if winter sea temperatures in the North Sea increased in the future, even if there was no fishery. If the fishery was active and SST increased, a catastrophic decline of 25% year^{-1} or more was predicted (Fig. 7).

Discussion

CAUSES OF LOW BREEDING SUCCESS AND SURVIVAL

Breeding success at the Isle of May was high during the late 1980s, very low during the 1990s and showed a partial recovery 2000–02 (Fig. 2). Very similar patterns occurred at many British kittiwake colonies along the North Sea during this period (Mavor *et al.* 2003). The high breeding success observed during 1986–89 was comparable with the long-term mean at the very productive North Shields colony in north-east England (Thomas & Coulson 1988), whereas the worst years during the 1990s were similar to observed values in Alaska (Dragoo, Byrd & Irons 2001). The range observed over our 17-year study was thus comparable with that found for the species over its entire range. The period of low breeding success coincided with the activity of the Wee Bankie sandeel fishery (compare Appendix 1 and Fig. 2), but when the 2 years with very low catches (1990 and 1999) were excluded the correlation was less strong (Table 1). However, when winter SST during the previous year was included in the model, a very large part of the variation was explained (Table 1 and Fig. 3).

The highest ranked models for adult survival all included a decline with 'age' in either survival or resighting probabilities (Table 2). This could indicate senescence of either birds or colour-rings. However, an analysis of the subset of known-age birds provided strong evidence in favour of a quadratic trend in survival with true age rather than any effect of the age of the colour-ring (Frederiksen, Wanless & Harris 2004), and we used this result in conjunction with year-specific estimates from the model presented here as input for the population model. Adult survival declined substantially from the high values observed early in the study and subsequently stabilized (Fig. 4). The decline was less pronounced than in a previous analysis where the effect of 'age' was not taken into account (Harris, Wanless & Rothery 2000) but nevertheless substantial. The population model provided an estimate of the mean survival of all adults (aged ≥ 2 years) in the population, ranging from 0·98 in 1986–87 to 0·82 in 1998–99. Again, this range contains most estimates of kittiwake survival from other populations, from very high in Alaska (Hatch, Roberts & Fadely 1993; Golet, Irons & Estes 1998) to low in Brittany (Danchin & Monnat 1992) and at North Shields (Aebischer & Coulson 1990). Survival was low when the Wee Bankie fishery was active and when breeding success was also low (compare Appendix 1 and Figs 2 and 4), and when the previous winter's SST was included in the model a high proportion of the variation was explained (Table 1 and Fig. 5). Furthermore, there was a strong correlation between survival and breeding success in the following season (see Appendix 4).

It seems clear that the effect of SST on kittiwake breeding success and survival must be mediated through sandeel availability, and the changing importance of sandeel age classes to kittiwakes through the breeding season provides a clue to the mechanism. We propose the following scenario: warm winters lead to low sandeel recruitment, perhaps by reducing the food supplies available to larval sandeels (Arnott & Ruxton 2002). Kittiwakes start feeding on this weak year class of 0 group sandeel during chick rearing in June. This is too late to have a major effect on breeding success in the current year, but adults (and presumably fledglings) are in poor condition at the end of the breeding season and overwinter survival is correspondingly low. Next spring, breeding adults again feed mainly on the same weak year class in the early part of the breeding season (sandeels older than 1 group constitute a small proportion of kittiwake diet; Lewis *et al.* 2001) and their poor condition leads to low reproductive investment and correspondingly low breeding success. After a cold winter, the opposite situation occurs. The observed correlation between independent estimates of survival and breeding success in the following season ($r = 0{\cdot}67$; see Appendix 4) was sufficiently strong to lend some credence to this hypothesis. The sharp decline in breeding success in 1990 occurred soon after a 'regime shift' in the North Sea, when an increased influx of warm Atlantic water had major ecological consequences, including a change in plankton community composition (Beaugrand 2004). Sandeel size and quality declined markedly over the study period (Wanless *et al.* 2004), perhaps as a response to changes in ocean climate, and this may also have affected kittiwakes adversely.

The Wee Bankie sandeel fishery also appears to have had a large effect on kittiwake performance (Figs 3 and 5). The exact mechanism is unclear, because the fishery is only likely to have had a strong direct effect on sandeel abundance in 1993, the peak year (Rindorf, Wanless & Harris 2000). Also, the fishery mainly took 1+ group sandeel in June and July, i.e. after the kittiwakes had switched to 0 group as their main prey (Lewis *et al.* 2001). However, it is possible that the fishery affected the behaviour of all sandeel age classes, so that they spent less time near the surface and thus became less available for surface feeders such as kittiwakes. Clearly, detailed studies of the behavioural responses of sandeels to fishing activity are required to elucidate the precise mechanism.

Kittiwake performance and population size declined at the same time as other sandeel-dependent seabirds were more successful on the Isle of May. As an extreme example, numbers of Atlantic puffins *Fratercula arctica* (L.), which to a large extent also feed their young on 0 group sandeels, increased by about 10% year^{-1} during the 1990s (Wanless *et al.* 2003). However, kittiwakes appear more sensitive to food shortages than many seabirds (Furness & Tasker 2000).

At Foula in Shetland, Oro & Furness (2002) found that kittiwake survival was associated with sandeel abundance, kittiwake body condition and breeding success of great skuas *Catharacta skua* Brünnich, an important predator of kittiwakes. The two first findings are consistent with our results, but great skuas are absent from the Isle of May and predation by great black-backed gulls *Larus marinus* L., for example, is rare. At the same time, kittiwake survival may also be affected by factors operating outside the breeding season. Identifying such factors is problematic, because only fragmentary data exist about the wintering areas of British kittiwakes: some individuals winter in the North Sea while others are dispersed in the North Atlantic (Coulson 2002). We found practically no correlation between adult survival and the winter NAO index (Table 1), a widely used indicator of winter conditions in the North Atlantic (Hurrell *et al.* 2003). In the future, application of lightweight data loggers (Wilson *et al.* 2002) may allow us to identify wintering areas more precisely and test whether survival is correlated with environmental conditions in these areas.

PREDICTIONS OF FUTURE POPULATION GROWTH

The dynamics of the Isle of May kittiwake population have changed dramatically over the study period, from an approximately 8% increase year^{-1} in the late 1980s to a 11% decline year^{-1} in the late 1990s (Fig. 1). Our model showed that this change could be explained by the observed declines in adult survival and breeding success (Fig. 6), and that both factors were important. There was no need to invoke changes in net recruitment, i.e. juvenile survival including emigration/immigration, as the model was rather insensitive to variation in this parameter (Fig. 6). Large changes in emigration or immigration also seem unlikely if kittiwakes, as evidence from other studies suggest, preferentially recruit to successful colonies (Danchin, Boulinier & Massot 1998): breeding success for all major colonies within approximately 150 km of the Isle of May was closely correlated with our study colony (Harris & Wanless 1997), and these colonies have also shown similar declining trends (Heubeck 2004). Both the deterministic and stochastic models also indicated that the kittiwake population was unlikely to increase under current environmental conditions, and that a reopening of the Wee Bankie fishery would probably lead to a marked decline in numbers (see Appendix 5 and Fig. 7). Furthermore, if climate change, as predicted by climate models, should lead to increases in North Sea temperatures in winter, this is predicted to exacerbate the already critical situation for kittiwakes, particularly if the fishery is active.

IMPLICATIONS FOR MANAGEMENT

In the North Sea, kittiwakes are almost completely dependent on sandeels during the breeding season. Therefore, any factor reducing the abundance or availability of sandeels is likely to have a negative effect and lead to population declines. We have identified two factors associated with low performance in kittiwakes at the Isle of May: high winter sea temperatures and an industrial sandeel fishery. It is unclear exactly how the fishery would affect the availability of sandeels to kittiwakes. Nevertheless, in a situation where kittiwakes are declining not only in the North Sea but throughout most of their range (Hatch, Roberts & Fadely 1993; Barrett 2001), any activities that endanger their main food supply should be avoided. Although sea temperatures are also to some extent within human control, any policy changes aimed at reversing the warming trend will be very slow to take effect. Thus, it seems prudent and in accordance with the precautionary principle to extend indefinitely the closure of sandeel fisheries within the feeding range of kittiwake colonies in the western North Sea.

Our results suggest that kittiwake performance is strongly dependent on sandeel availability, and therefore reflects the general health of marine ecosystems in terms of their ability to produce sufficient amounts of forage fish to sustain thriving populations of top predators. Monitoring kittiwake performance (breeding success and survival) could therefore be an efficient way of monitoring marine ecosystem health. Furthermore, intensive long-term studies remain the only source of consistent data allowing evaluation of both population trends and extensive monitoring programmes, and maintaining such studies should be a high priority for research and management.

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Supplementary material

The following material is available from http://www.blackwellpublishing.com/products/journals/suppmat/JPE/JPE966/JPE966sm.htm.

Appendix S1. Annual sandeel landings from the industrial fishery in ICES squares 41E7 and 41E8.

Appendix S2. List of factors and covariates used as predictors in the analysis of survival and resighting probabilities, and the subscripts used to identify them.

Appendix S3. Details of the stochastic matrix model.

Appendix S4. The correlation between kittiwake survival and breeding success in the following season.

Appendix S5. Predicted population growth of Isle of May kittiwakes 2000–2010 under various scenarios.

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Lesser Black-backed Gulls *Larus fuscus* thriving on a non-marine diet

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ABSTRACT

Capsule: Lesser Black-backed Gulls *Larus fuscus* breeding 30 km from the coast in the Netherlands focussed entirely on terrestrial food sources and reached relatively high breeding success.

Aim: To gain insight in the foraging ecology, habitat use and breeding performance of inland-breeding Lesser Black-backed Gulls.

Methods: We received data from seven birds fitted with global positioning system (GPS) loggers. The colony was frequently visited to collect pellets and boluses and to monitor reproductive success, mortality and growth rate of chicks.

Results: The GPS data revealed that mainly terrestrial habitats were used, 98% of these GPS positions were within 25 km of the colony. Refuse dumps were the most preferred sites, but also agricultural fields and freshwater bodies were often visited. Only two of the 710 recorded trips were directed to the North Sea. The pellet and bolus analyses confirmed the GPS data: no marine food remains were found. Breeding success of birds in the enclosure was relatively high, with 90% of eggs hatched and 51% of chicks fledged (1.6 chicks/pair).

Conclusions: Relying on terrestrial food is feasible when sources are available in the vicinity of the colony. We conclude that Lesser Black-backed Gulls could theoretically shift towards inland breeding after a fishery discards ban.

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Large gulls *Laridae*, among them the Lesser Black-backed Gull *Larus fuscus*, are considered to be generalists in their selection of prey (Harris 1965, Cramp & Simmons 1978, Kubetzki & Garthe 2003, Camphuysen 2013). In most coastal colonies around the North Sea, however, Lesser Black-backed Gulls are primarily marine-orientated, conducting long foraging flights offshore and feeding both on discards of fishing vessels as well as on natural fish prey (Furness *et al.* 1992, Camphuysen 1995, Garthe *et al.* 1996, Camphuysen 2013).

Many coastal colonies in the dune area of the mainland North Sea coast of the Netherlands were abandoned due to predation pressure by Red Foxes *Vulpes vulpes* (Spaans 1998a). Consequently, Lesser Black-backed Gulls in the Netherlands started to colonize areas farther inland (especially in the Delta area in the south-western part of the Netherlands), and the number of birds breeding inland has been increasing ever since (Camphuysen 2013). However, the largest colonies of this species and the majority of the Dutch breeding population are still located at the coast (van der Helm 1992, Spaans 1998b, Camphuysen 2013).

In terms of reproductive output, foraging at the open sea was shown to be advantageous compared with feeding on land for this species and other Larids (Annett & Pierotti 1989, Spaans *et al.* 1994, Bukacinska *et al.* 1996). The energetic quality of fish is relatively high, assuring a high reproductive performance when consumed in sufficient quantity and quality in the breeding period (Garthe *et al.* 1996, Camphuysen 2013). Therefore, it has been assumed that the inland distribution of gulls is limited by the distance to accessible fish prey (Hüppop & Hüppop 1999).

However, a recently accepted European Union ban on discards (European Parliament 2013) is expected to diminish the offshore feeding opportunities of large gulls (stepwise from 2014 onwards), which will likely reduce the food availability in the marine environment and cause a shift in diet (Bicknell *et al.* 2013). Lesser Black-backed Gulls can readily switch to terrestrial food resources (Oro 1996) and recent studies showed that there is a large inter-annual, inter-individual and even within-week variation in foraging destinations between terrestrial and marine sites at coastal colonies

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(Thaxter *et al.* 2015). Also inland-breeding birds in the Netherlands are known to exploit alternative terrestrial food sources extensively (Spaans 1998b, Camphuysen *et al.* 2005). Nevertheless, despite the increasing number of inland Lesser Black-backed Gulls colonies in the Netherlands, there have been few field studies on them. Studies on food choice, habitat use and corresponding reproductive output have mainly been studied in coastal colonies.

In order to gain insight in the foraging site selection, diet and corresponding reproductive performance of Lesser Black-backed Gulls breeding in a non-coastal colony, we conducted our study in a colony in Lake Volkerak, The Netherlands, situated approximately 30 km from the North Sea. The research was carried out within the framework of the Shortlist Masterplan of Rijkswaterstaat Waterdienst and Rijkswaterstaat Directie Noordzee to gather relevant ecological data on, for example, foraging ranges, flight heights, activity patterns, proportion of floaters (i.e. sexually mature but not yet breeding individuals) in the population, and figures on annual survival, especially for the fraction of birds that forage offshore and can potentially be impacted by future windfarm developments. The study was based on global positioning system (GPS)-tracking and pellet analysis, and was extended by an enclosure study to reveal the reproductive performance that birds could achieve by their foraging decisions.

Materials and methods

Study site

Lake Volkerak, together with Lake Krammer (with a total surface area of 6450 ha) is a former arm of the North Sea, which was closed off in 1987 and became a freshwater lake. Lesser Black-backed Gulls breed at five different sites within the lake (Strucker *et al.* 2011). Our study focused on the colony at the Noordplaat (N51.644029°, E4.237396°), a group of three islands with a surface area of 13 ha. As part of the management plan, vegetation is systematically removed from a strip of 15–50 m at the western and southern side of the most western island, in order to prevent plant succession and preserve the Lesser Black-backed Gull colony. In 2010, 41 pairs of Lesser Black-backed Gulls, 18 pairs of Herring Gulls *Larus argentatus* and 112 pairs of Barnacle Geese *Branta leucopsis* bred on the Noordplaat (Strucker *et al.* 2011).

Data collection and analysis

Breeding ecology

In 2010, in the colony on the Noordplaat, 40 nests (of which six turned out to be of Herring Gulls) were marked with wooden poles and their geographical position was recorded with a handheld Garmin GPS unit. During the breeding phase, 31 adult Lesser Black-backed Gulls were trapped on nests, using walk-in cages. Based on morphometric measurements and other phenotypic characteristics, birds were sexed (Muusse *et al.* 2011) and subsequently colour-ringed. On 18 May, an enclosure was built with 0.5 m high, 0.02 m mesh size chicken wire to fence off 18 nests (including eight nests of ringed birds) in the middle of the colony. The lower 0.4 m of the chicken wire was covered with green hard plastic to prevent chicks from forcing their head in the mesh and injuring themselves. The enclosure was subdivided into two sections (an eastern and western part) with roughly equal surface areas. There were 13 Lesser Black-backed Gull nests (comprising 32% of the whole colony) in the enclosure, with four nests in the western part (plus two Herring Gull nests), and nine nests in the eastern part (plus three Herring Gull nests).

The colony was visited twice a week (three to four days interval between two visits) between mid-May (breeding phase) and Mid-July (fledging of the last young) in 2010. Altogether 16 visits were made to the colony. During the visits, all marked nests on the island were assessed for the presence of eggs or chicks. Eggs were sequentially numbered with a permanent marker to record laying order, were weighed to the nearest 0.1 g, and length (L) and width (W) were measured to the nearest 0.001 m. Subsequently, egg volume (V) was calculated by the formula (Stonehouse 1966):

$$V = K_v * L * W^2,$$

where the constant K_v was set at 0.5035 (Camphuysen 2013). On the following visits, the pipping date (appearance of star-like bursts in the shell) and the actual hatching date of the eggs were recorded. Wet chicks were registered as hatchlings of that day; dry chicks were assigned to one of the previous days according to their size. As the incubation of eggs had already started by the first visit to the colony, median laying date in the colony was calculated by subtracting the documented incubation period of Lesser Black-backed Gulls (28 days: MacRoberts & MacRoberts 1972b, Camphuysen 2013) from the mean hatching date observed in the field.

Upon the first encounter, young chicks within the enclosure were temporarily marked with coloured cable ties. These were cut short so as not to get entangled in vegetation or the enclosure. When the tibias of the chicks were more developed, the cable ties were replaced by a permanent aluminium ring on the tarsus and a colour-ring on the tibia. During each visit, chick weight (to the nearest g), total head length (from the tip of the bill to the back of the head to the nearest 0.1 mm) and from the onset of feather growth also stretched wing length (from the bend of the wing to the tip of the longest primary to the nearest mm) were measured. In order to minimize disturbance to the colony, searching for chicks was terminated after 45 minutes, whether or not all individuals were found. Chicks missing before the age of 30 days were considered dead. Chicks seen in the enclosure at the age of 30 days but not afterwards were considered to be fledged.

In order to compare the fledging rate of chicks of treated adults (i.e. those ringed or equipped with GPS; see below) and a control group of the adults (not ringed or equipped with GPS) within the enclosure, the number of dead and fledged chicks was ordered in a two by two contingency table and a V-square statistic was carried out (a Chi-square test for small sample sizes). The fledging rates were also compared based on the number of hatched eggs and the number of chicks fledged. Dealing with dependent samples, this comparison was carried out using a McNemar chi-square test. All statistics of the study were carried out using SPSS 15.0 for Windows.

Dietary analysis

Boluses of partly digested stomach contents regurgitated during the handling of the adults or chicks were collected, labelled with the date, location and originating individual, and later kept frozen in the laboratory until further analysis. On nine days, pellets of regurgitated indigestible prey material were also collected in the enclosure. If applicable, the originating nest number was recorded, otherwise the appropriate enclosure side was recorded (east or west). There were no pellets collected in the immediate surroundings of Herring Gull nests. Nevertheless, Lesser Black-backed Gull and Herring Gull nests were in close vicinity of each other, and in most cases pellets could not be attributed to species.

Altogether 18 boluses and 67 pellets were collected, which were analysed under a 10x binocular magnification. Food remains were identified to species group or to species if possible. Subsequently, pellets and boluses were qualitatively ordered in the main categories of terrestrial, aquatic or mixed sources. Results are presented as frequency of occurrence (in percentages) of food items in pellets and boluses (Barrett *et al.* 2007).

Habitat use

Of the 31 birds trapped on nests, 9 birds (4 of them breeding within the enclosure) received an 18 g, solar-powered GPS logger, (UvA-BiTS, Bouten *et al.* 2013). The loggers were attached as backpacks with a flexible harness of synthetic elastic inner lining and teflon outer lining (Bally Ribbon Mills, USA). Loggers were only deployed on birds in good condition and with a minimum weight of 700 g. The system was set to obtain the geographical position twice per hour in May, four times per hour in June and again twice per hour afterwards. In addition, ground speed and altitude above sea level were also recorded. Data were automatically downloaded from the GPS devices via a wireless network to one of two antennas placed on the island. The antennas transmitted the data to a laptop that was also positioned on the island and powered by two solar panels. The laptop had a wireless Internet-connection, and thus data could be remotely downloaded and new configurations uploaded.

GPS logger 321 stopped working on the first day and logger 323 after one week (Table 2). Therefore, data analysis was restricted to seven GPS-transmitters (representing 8.5% of all birds in the colony). GPS measurements showed that a few birds were conducting only a few trips in the first two days after deployment. Therefore, in order to avoid biased data after the placement of the loggers, the first three days of tracking were excluded from the analysis. Visited destinations were qualitatively categorized based on publicly available satellite images. The position of the colony was specified as a 1 km (the approximate distance to the nearest shore of Lake Volkerak) radius circle around the centre of the colony, to include birds floating on the water around the island. A kernel density analysis on the measured GPS positions was conducted (Thaxter *et al.* 2015). Search radius to fit the kernel densities was 1.5 km and positions within a 1 km zone around the colony were excluded. The kernel density analysis was carried out using the Spatial Analyst Tools within ArcGIS 10.1. Based on the resulting kernel rasters, isopleths lines of 50%, 75% and 95% were extracted using the Geospatial Modelling Environment (version 0.7.2.1) to visualize where birds were found most often.

Results

Feeding ecology

Although most of the pellets comprised multiple food sources, in most cases a main category of origin could be identified. All in all, there were no food remains of marine origin found in the pellet or bolus samples. Both sample types indicated that the Lesser Black-backed Gulls mainly relied on terrestrial food sources. For example, most of the pellets (84%; Table 1, see an overview of all pellets in Supplementary Online Appendix 1) contained at least a small quantity of beetle remains.

Table 1. Source of origin of the pellets ($n = 67$). % of all pellets provides the main categorization of pellets into terrestrial, mixed and aquatic origin. Mixed source regards samples with both terrestrial and aquatic food remains within. Prey types occurring in the pellets are grouped per source of origin and provided in a descending order based on the percentage of occurrence in all pellets.

	% of all pellets	Prey type	Occurrence of prey type (%) within all pellets
Terrestrial	69	Beetles	84
		Mole	36
		Refuse	31
		Ants	28
		Seeds	27
		Bird remains	10
		Larvae	9
		Mouse	1
		Eggshell	1
Mixed	19	Shell remains	40
Aquatic	12	Freshwater fish	28
		Crustacea	10
		Zebra mussel	6

Based on the frequency of occurrence in pellets, Moles *Talpa europaea* also seemed to be important prey, occurring in 36% of all pellets. Furthermore, 31% of the pellets contained food remains from refuse dumps (indicated by a large amount of paper, glass, plastic or a mixture of these). The frequency of occurrence of human waste in pellets was low, however, compared with boluses: 61% of the boluses contained food remains of refuse dumps.

Regarding food items from undoubtedly aquatic sources, fish otoliths were found most often (in 28% of pellets; Table 1). All of these originated from freshwater fish, mostly from *Cyprinidae* species. Furthermore, 10% of the pellets contained remains of freshwater crayfish or crabs (Spiny-cheeked Crayfish *Orconectes limosus* and Chinese Mitten Crab *Eriocheir sinensis*), while another 6% contained shells of Zebra Mussels *Dreissena polymorpha*. Small pieces of shells from unknown species were, however, found in many more pellets (40%).

Eighty-three per cent of the boluses ($n = 18$) had a terrestrial origin. Only 11% was categorized as having a mixed origin and 6% as having an aquatic origin. Most of the boluses contained food items from domestic waste. Mainly bread or waste meat was encountered, for instance chicken skin or pure fat. In boluses, earthworms (on two occasions), leatherjackets (larvae of crane flies *Tipulidae* spp.) and a beetle species were also identifiable (Garden Chafer *Phyllopertha horticola*).

Habitat use

Altogether 710 foraging trips were identified. There were only two occasions (0.3% of all flights) when a bird flew to the North Sea. One of these trips (conducted by a female, ID 316) lasted only one hour, on 5 June 2014, during the chick-rearing phase. The bird had just lost its last chick around the time of this trip. A few days later data transition from this bird stopped (Table 2). Another female (ID 331), conducted a nearly day-long trip to the North Sea at the end of the chick-rearing phase, on 27 June. The farthest measured locations during this trip were 15–20 km away from the coast, with the farthest point being 64 km from the colony. After returning to land, the bird did not enter the colony but flew straight to the refuse dump of the town Bergen op Zoom. The last signal from this GPS logger was received a few days later (3 July; Table 2). The last egg in the nest of this individual was found on 22 June and on later visits the nest was empty.

Table 2. Data transmission period and number of fledglings raised by Lesser Black-backed Gulls. Number of fledglings outside the enclosure could not reliably estimated (indicated by '–').

GPS-ID	Sex	Period GPS data	Number of fledglings
189	Male	22/5–11/7	2
299	Male	22/5–10/7	1
316	Female	22/5–8/6	0
322	Female	22/5–8/6	–
330	Male	22/5–16/6	–
331	Female	22/5–3/7	–
332	Male	22/5–11/7	–
[a]321	Male	22/5	2
[a]323	Female	22/5–29/5	–

[a]Due to technical problems these tags were not included in the analysis.

All other foraging flights of Lesser Black-backed Gulls were directed inland, mostly to the south-southeast (Figure 1). Nearly all of these were directed to terrestrial foraging sites and only some to freshwater sites. Ninety-seven per cent of the measurements occurred at a distance less than 25 km from the colony. Fifty-one per cent of these points were in or around the colony at less than 1 km distance (Figure 2). A second peak (19% of all measurements) occurred between 15 and 20 km from the colony (Figure 2), corresponding to foraging locations in or nearby two towns (Bergen op Zoom and Roosendaal).

Individuals showed a high personal preference for specific foraging sites (Figure 1) while nearly completely neglecting sites that were frequented by other individuals. The main foraging sites included agricultural fields to the south-southeast of the colony (Figure 1, location 1, almost exclusively used by ID 332 and often visited by most of the birds), the town of Bergen op Zoom (Figure 1, location 2, visited by ID 316 and 330), a refuse dump near to Bergen op Zoom (Figure 1, location 3, almost exclusively used by IDs 189 and 331 and regularly by IDs 299, 316 and 322), the town of Roosendaal (Figure 1, location 4, visited by ID 299, 316 and 322). When Dutch refuse dumps were closed on Sundays, the birds

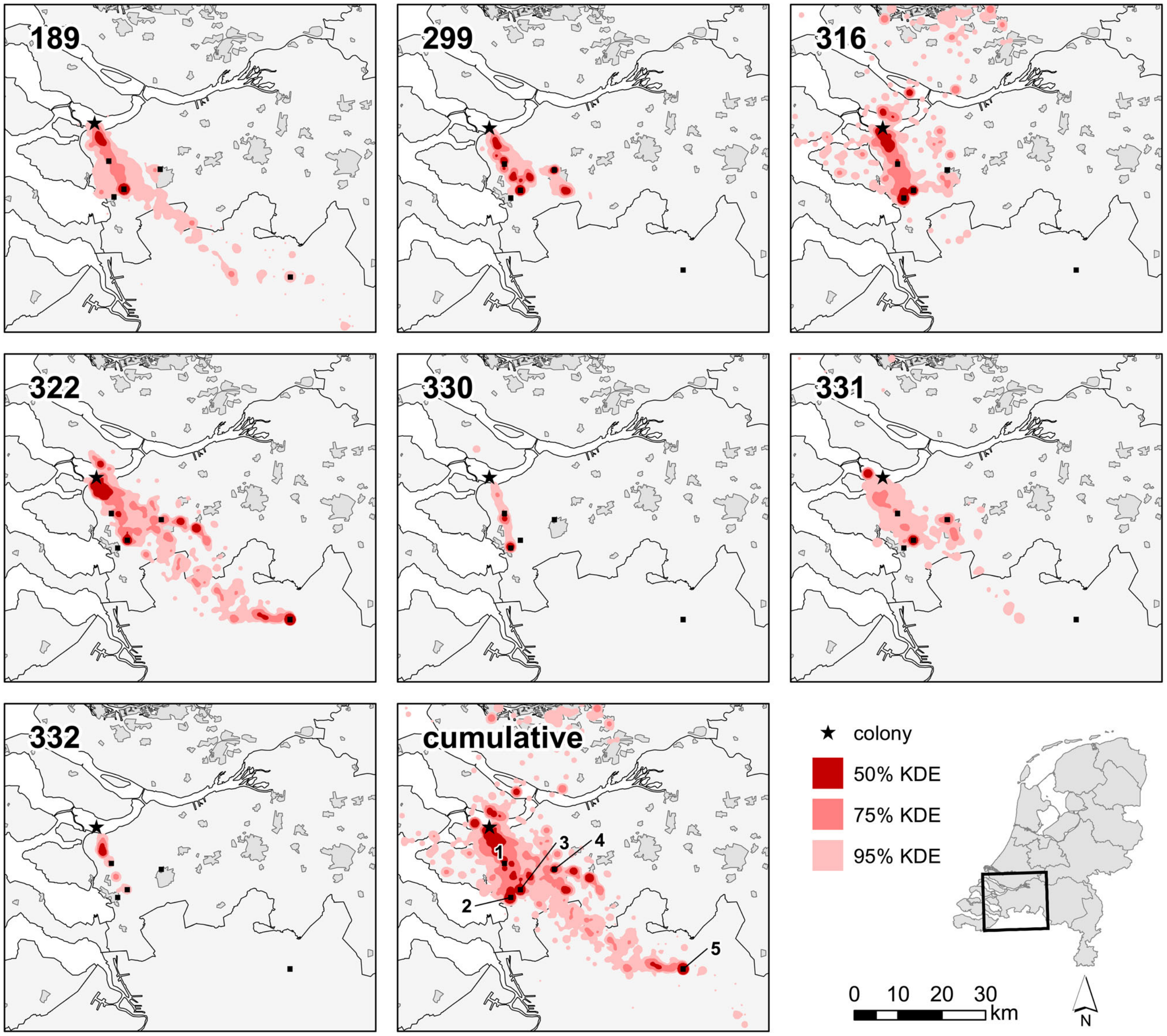

Figure 1. Kernel density estimates of space use by seven Lesser Black-backed Gulls and a cumulative map based on GPS measurements outside a 1 km zone around the colony (location depicted by a black star). Dark red areas provide the 50% kernel density estimates, clearly showing the individual preferences of the birds. The most important foraging areas are numbered: (1) Agricultural areas; (2) Town of Bergen op Zoom; (3) Refuse dump of Bergen op Zoom; (4) Town of Roosendaal and (5) Refuse dump in Belgium. Darker grey shaded areas are human settlements.

switched to alternative food sources, while one individual (ID 322) often flew farther to a Belgian refuse dump that was not covered up (Figure 1, location 5).

Breeding ecology

Quantification of hatching success outside the enclosure was not reliably possible (Table 3) because chicks could wander freely around and could not be confidently linked to a nest. Hatching of the eggs within the enclosure ($n = 36$) occurred in the period 22 May–4 June, with a median date of 28 May 2010. Considering an incubation period of 28 days, the calculated median laying date of these eggs was 1 May. Hatching success was high in the enclosure (94.2%): at least one egg hatched in each nest.

The first fledged chick left the enclosure on 25 June. The peak fledging period occurred in the first days of July. Fledging success of chicks within the enclosure was generally high (51%), resulting in a relatively high, 1.6 fledged young per nest (Table 3). The number of fledged young of the treated birds (ringed or GPS-tagged) was lower, but still relatively high (1.3 fledged young per nest). 18.3% of all chicks were found dead but intact within the enclosure, and hence died likely of other causes than predation. Another 24.2% of all young disappeared without trace before the fledging period.

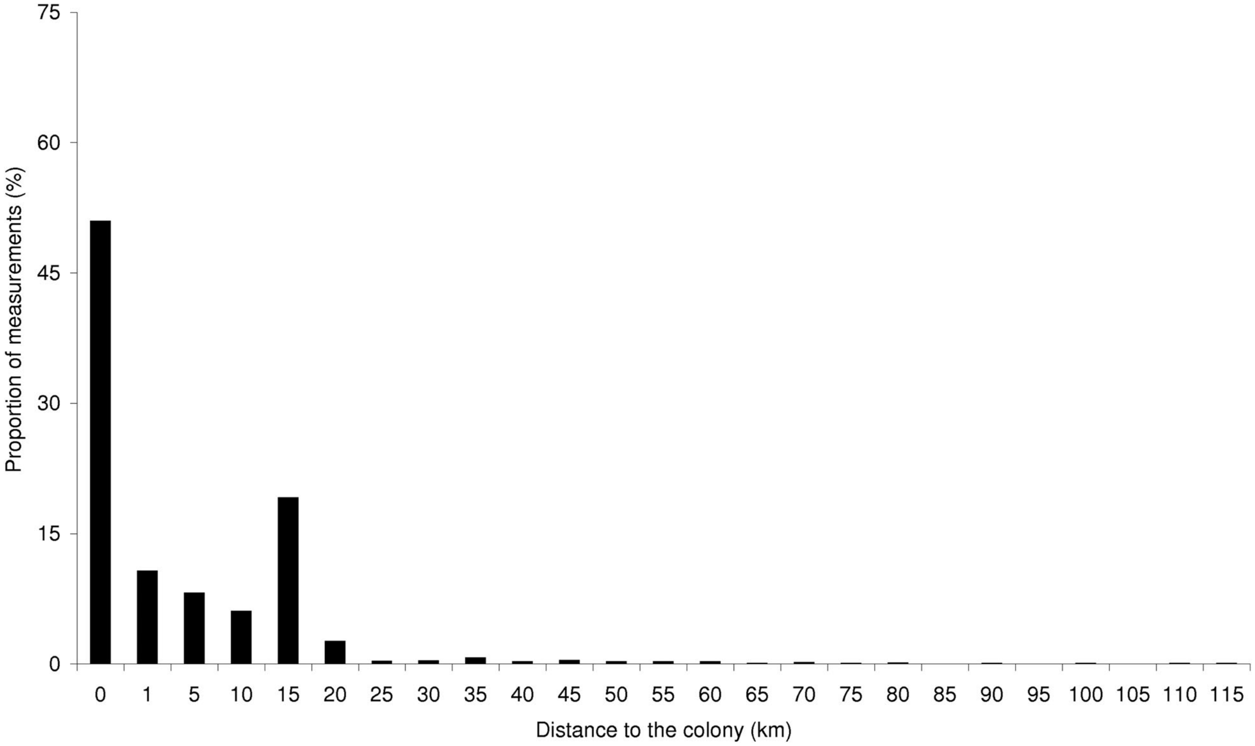

Figure 2. Distance of the GPS measurements (in 5 km categories) measured from the centre of the colony. Category '0' refers to locations in or around (up to 1 km) the colony. The vertical axis shows the proportion of a certain category from all measurements.

Table 3. Overview of reproductive parameters measured inside and outside the enclosure of the Noordplaat. Hatching and fledging success outside the enclosure was not reliably estimable due to chicks leaving the nest and wandering freely around.

	No. nests	Clutch size mean (±sd)	Hatching success %	Fledging success %	No. fledglings /pair
Outside enclosure	21	2.9 (0.3)	–	–	–
Encl. with GPS logger	4	3.0 (0.0)	92	33	1.3
Encl. with colour-ring	4	2.8 (0.5)	91	45	1.3
Encl. control	5	3.0 (0.3)	87	73	2.0
Total enclosure	13	2.9 (0.3)	90	51	1.6

There was no significant difference in the number of chicks died and fledged per adult between the treated and control groups within the enclosure (V-square statistic: df = 1, $\chi^2 = 2.76$, $P = 0.96$). Similarly, the number of hatched eggs and the number of resulting chicks fledged did not differ significantly between the treated and control groups (McNemar $\chi^2 = 3.23$, $P = 0.72$).

Discussion

The results provided insights into the flight patterns, habitat use and foraging site selection of Lesser Black-backed Gulls breeding in a colony 30 km from the coast. Based on the GPS measurements and the diet analysis, we believe that the birds did not forage in the North Sea and subsequently did not rely on marine food sources, as birds breeding in colonies directly on the coast mainly typically do (Garthe *et al.* 1996, Camphuysen 2013). Our research revealed that birds within the enclosure relied on a terrestrial diet, yet still reached a relatively high reproductive output.

Although the sample size was small, the fledging success at Lake Volkerak was high compared with traditional natural sites, such as coastal colonies in the Netherlands (0.49 fledglings/nest; Camphuysen 2013), Sweden (0.02–0.16 fledglings/nest; Lif *et al.* 2005) or England (1.21 chicks/pair; Sellers & Shackleton 2011). The productivity rates we measured are more comparable with that of urban inland nesting Lesser Black-backed Gulls, which might be foraging on similar resources (2–3 fledglings/nest; Rock 2005, Sellers & Shackleton 2011). Also, inland colonies in the Netherlands were shown to have higher reproductive output than that of stable or decreasing colonies on the coast (Spaans 1998b), which likely reach low breeding success and growth rates due to food shortages (Bukacinski *et al.* 1998, Spaans 1998a). Compared with these natural coastal colonies, even the fledging success of our tagged birds (1.3 fledglings per nest) was higher.

Outcomes of the diet analysis correspond with previous indications that Lesser Black-backed Gulls breeding more inland consume predominantly terrestrial food (Camphuysen *et al.* 2005, Camphuysen *et al.* 2010). Nevertheless, compared to results of other studies, the complete lack of marine food items in the diet of Lesser Black-backed Gulls was unexpected (Camphuysen *et al.* 2015). Except for two trips directed to the North Sea, conducted by two different individuals that had just lost their eggs or chicks, all other trips were directed inland. Shortly after these trips to sea, GPS data transmission from these two birds stopped, possibly because they left the colony. Failed breeders in a coastal colony in the Netherlands also performed trips to divergent destinations compared with active breeding birds (Camphuysen 2013, Camphuysen *et al.* 2015).

The bolus analysis and the recorded flight movements corresponded with each other; refuse dumps were one of the most important foraging locations for Lesser Black-backed Gulls of the Noordplaat. In contrast, at a coastal colony in the Netherlands domestic waste materials were found in none of the regurgitated boluses of male gulls and only in 16.7% of boluses from females (Camphuysen 2010). Visits of Lesser Black-backed Gulls to the refuse dump of Bergen op Zoom have already been documented, where they were observed to forage on urban domicile – and organic waste (especially meat waste), and the insects concentrated within (Buijs 1998). During the past decade, the importance of this refuse dump for Lesser Black-backed Gulls has grown (R.-J. Buijs, unpubl. data). However, the gulls also proved their opportunistic feeding character by switching to alternative food resources on Sundays when the refuse dump was closed (compare with Tyson *et al.* 2015).

The pellet analysis, on the other hand, suggested a minor importance of the refuse dump. Food pellets give a representation of the diet of a larger fraction of the colony (approximately a third of all birds), in comparison with the few individuals equipped with GPS loggers. However, pellets contain only the non-digestible part of the food (Barrett *et al.* 2007), while highly digestible food, such as bread or meat collected at a refuse dump, leaves nearly no remains in pellets (Ottoni *et al.* 2009). Moreover, fish consumption based on the number of otoliths in pellets may be underestimated (Barret *et al.* 2007). In this sense, pellets are only representative of the non-digestible matter from the consumed food items. For instance, the large proportion of Moles found in the pellets is remarkable. Moles are known to occur in pellets of Lesser Black-backed Gulls with a terrestrial diet (Camphuysen *et al.* 2005) and in an inland colony in the Netherlands a similar share of Moles was found in pellets (Camphuysen *et al.* 2010). Nevertheless, these proportions in pellets might be biased, due to Mole remains being large and easily recognizable.

The diet analysis and the GPS data both verified the main finding of the study: the gulls from this colony apparently did not forage at sea. While the sample size of birds with GPS loggers was small, the combination of tracking data and diet analysis provide complementary results on foraging site selection in terrestrial as opposed to marine habitats. The relatively high breeding success in this colony could be due to the low breeding density, or an artefact of the position of the enclosure, that is, around the highest concentration of nests. Such concentrations suggest a preferred position within the colony (MacRoberts & MacRoberts 1972a, Savoca *et al.* 2011) that are commonly occupied by higher-quality individuals (Kim & Monaghan 2005). The high breeding success could also be a positive effect of the enclosure itself, by making nests less accessible for ground predators, although a 0.5 m high chicken wire should not pose a serious barrier to most ground predators. Nevertheless, our main message holds also in this case, Lesser Black-backed Gulls do not necessarily need to rely on marine food sources, but may also reach (at least temporarily) high breeding performance by consuming food generally considered of lower quality and labelled as ‘junk food’ (Oro 1996, Annett & Pierotti 1999, Camphuysen 2013).

Foraging on discards at sea was shown to be advantageous for Lesser Black-backed Gulls compared with feeding on land, due to the higher energetic value of fish (Bolton *et al.* 1992, Spaans *et al.* 1994). The availability of natural prey species, like Herring *Clupea harengus* and Sprat *Sprattus sprattus*, may be crucially important for a high breeding success (Noordhuis & Spaans 1992, Spaans *et al.* 1994). At sea, fishing boats or fish shoals can offer a foraging hotspot for gulls by providing a large amount of readily available food (Camphuysen & Webb 1999, Bartumeus *et al.* 2010). However, due to the high competition levels, this holds only for the more dominant individuals (Hudson & Furness 1989). Moreover, due to their mobility, the location of fish shoals or trawlers is rather unpredictable. For birds breeding farther inland, searching for such food sources would also be preceded by a flight to the sea (in our case 30 km to the coast), with all its accompanying time and energy costs (Weimerskirch 2007), which are especially crucial during the chick-rearing period. Decisions in foraging site selection are not only essential for the individual itself, but also for the offspring; staying longer away for food means a lower prey delivery rate and a lower nest attendance that might lead to higher predation rates (Spaans *et al.* 1994, Bukacinski *et al.* 1998). Therefore, foraging offshore, where Lesser Black-backed Gulls have a mean foraging range of 181 km (Thaxter *et al.* 2012), can be considered for inland breeders rather risk-prone (Annett & Pierotti 1999, Camphuysen 2013).

In contrast, all of our studied individuals exploited predictable, readily available food sources at inland locations mainly within a distance of 25 km, providing a potentially high prey delivery rate and nest attendance. Terrestrial food sources often involve stationary food supplies and the birds can revisit the same foraging location on repeated occasions. Even for females that might be outcompeted at refuse dumps (Monaghan 1980, Greig *et al.* 1985) and hence need to fly longer distances for food, it still seemed to be worth choosing terrestrial foraging sites. To our knowledge, our study is the first to provide detailed information on the foraging site selection, diet and corresponding reproductive output of a non-coastal Lesser Black-backed Gull colony. Despite the limited sample size of our study, based on our results we expect that other inland-breeding Lesser Black-backed Gull colonies may also primarily focus on terrestrial food sources (Camphuysen *et al.* 2005). In view of the recently accepted European Union ban on discards, more and more Lesser Black-backed Gulls are expected to shift to alternative, likely terrestrial, food sources, eventually accompanied by breeding farther inland (Bicknell *et al.* 2013, Camphuysen 2013). Whether an individual can promptly shift its diet or if the frequency of specialists in a population can slowly transition over time, needs to be investigated in the near future, especially as the number of refuse dumps in the Netherlands has declined in recent decades (Camphuysen 2013). Moreover, in 2014, the European Commission adopted a legislative proposal that aims at phasing out landfilling by 2025 in favour of recycling waste, including bio-waste (European Commission 2014). How many large gulls will find their nutritional needs in such a quickly changing world remains to be seen.

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EFFECTS OF TAGGING ON BEHAVIOR, PROVISIONING, AND REPRODUCTION IN THE COMMON MURRE (URIA AALGE), A DIVING SEABIRD

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EFFECTS OF TAGGING ON BEHAVIOR, PROVISIONING, AND REPRODUCTION IN THE COMMON MURRE (*URIA AALGE*), A DIVING SEABIRD

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Abstract.—Telemetry devices used on seabirds have been shown to affect their behavior, reproduction, and survival. However, studies have primarily been limited to larger species, and results are occasionally equivocal. We quantified tag retention time and effects of attaching subcutaneous anchor radiotransmitters on behavior, provisioning, and reproduction in the Common Murre (*Uria aalge*), a medium-bodied seabird. Between 1999 and 2001, we tagged 48 Common Murres on Tatoosh Island, Washington. Subsequently, 46 birds were detected during colony observation, radiotracking, or both. Activity budgets and time spent at the nest site did not differ between tagged birds and their untagged mates. However, tagged birds made fewer but longer trips away from the nest and provisioned their chicks significantly less than their mates did (0.07 ± 0.02 fish h^{-1} and 0.18 ± 0.02 fish h^{-1}, respectively). Prey size and energy content did not differ. Despite the disparity in provisioning rates at the individual level, tagged pairs and control pairs had equivalent energy-delivery rates and reproductive success. Tagging effects did not persist in the long term, because percentage of tagged birds returning to the colony the next breeding season was similar to percentage of birds without tags, and tagged birds' reproductive success was comparable to that of the rest of the colony. Subcutaneous anchor attachments compared favorably to glued tags and implanted transmitters in terms of retention time and survival, respectively. We recommend using subcutaneous anchor transmitters for medium-bodied seabirds that employ flexible foraging strategies. Possible exceptions are years of poor food availability, when the capacity to absorb tag effects may be lower. *Received 23 June 2003, accepted 30 June 2004.*

Résumé.—Il a été démontré que les appareils de télémétrie fixés sur les oiseaux marins affectent leur comportement, leur reproduction et leur survie. Cependant, les études, limitées principalement aux espèces de grande taille, ont donné des résultats parfois équivoques. Nous avons quantifié les effets de radio-émetteurs à ancrage sous-cutané sur le comportement, l'approvisionnement et la reproduction du Guillemot Marmette (*Uria aalge*), oiseau marin de taille moyenne. Entre 1999 et 2001, nous avons équipé 48 guillemots avec des émetteurs sur l'Île Tatoosh, Washington. Par la suite, 46 oiseaux ont été repérés visuellement durant nos observations de la colonie et/ou par télémétrie. Les bilans d'activités et le temps passé sur le site de nidification ne différaient pas entre les oiseaux munis d'émetteur et leur partenaire (sans émetteur). Par contre, les oiseaux avec émetteur ont quitté le nid moins fréquemment mais étaient absents plus longtemps et ont significativement moins nourri leur progéniture que leur partenaire (0.07 ± 0.02 et 0.18 ± 0.02 poissons h^{-1} respectivement). La taille et la valeur énergétique de leur proie n'étaient pas différentes. Malgré la disparité entre les taux d'approvisionnement individuels, les taux d'énergie délivrés et le succès de reproduction des couples avec émetteur étaient les mêmes que les couples contrôles. Les émetteurs n'ont pas eu d'effet à long terme puisque le pourcentage d'oiseaux marqués qui sont retournés à la colonie la saison suivante était comparable à celui des oiseaux sans émetteur. De plus, le succès de reproduction des oiseaux marqués était comparable à celui du reste de la colonie. L'efficacité des émetteurs à ancrage sous-cutané se compare favorablement à celui des émetteurs fixés à l'aide de colle ou implantés en terme de durée de maintien et de survie des oiseaux. Nous recommandons l'emploi d'émetteurs à ancrage sous-cutané pour les oiseaux de taille moyenne caractérisés par des stratégies de quête de nourriture flexibles. On pourrait s'abstenir de marquer les oiseaux durant les années où la disponibilité de nourriture suppress et la capacité de supporter les effets des émetteurs sont moindres.

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Telemetry devices, such as transmitters and data-loggers, are used to address a wide range of ecological and conservation questions (Kenward 2001). Such instruments are of particular relevance to seabirds, which range widely and are difficult to find or follow (Wilson et al. 1986, Jouventin and Weimerskirch 1990). Success in telemetry studies depends on instrumentation not altering the behavior being measured, either in time (e.g. time spent foraging) or in space (e.g. foraging locations). However, a range of behavioral and demographic effects of carrying devices have been documented in seabirds (Calvo and Furness 1992, Withey et al. 2001)—albeit in studies mostly limited to larger species (Phillips et al. 2003) and with occasionally equivocal results (Fraser and Trivelpiece 1994).

Devices have been deployed on smaller seabirds (<1 kg) by mounting them externally with adhesive, cable ties, or harnesses (Wanless et al. 1989, Klaassen et al. 1992, Falk and Møller 1995), with reported effects on behavioral and demographic parameters (Wanless et al. 1988, Falk and Møller 1995, Meyers et al. 1998, Söhle et al. 2000). Implantation is thought to be preferable to external attachment because of reduced drag and energetic costs, and increased transmitter retention time and battery life (Olsen et al. 1992, Hatch et al. 2000, Murray and Fuller 2000, Ropert-Coudert et al. 2000). However, that tagging procedure has had detrimental effects on nesting and survival when used on alcids (Meyers et al. 1998, Hatch et al. 2000). Subcutaneous anchors have been proposed as a less intrusive method, with better retention time than other attachment types (Pietz et al. 1995, Newman et al. 1999); however, their behavioral and demographic effects have not yet been evaluated.

Here, we examined effects of deploying subcutaneous anchor radiotransmitters (tags) on behavior, provisioning, and reproduction in the Common Murre (*Uria aalge*), a medium-bodied, pursuit-diving alcid. Our objectives were to determine whether this type of tag attachment impaired birds as compared with other attachment types (e.g. glue; Wanless et al. 1988) and to quantify tag retention time. Disturbances associated with tagging may operate at the level of the individual (the tagged bird), but effects may extend to the mate, chick (pair-level effect), or both. In addition, tagging can have immediate (days) as well as long-term (months to years) effects (Kenward 2001). Therefore, we quantified tag effects by assessing differences between tagged and untagged individuals, and between pairs with and without tagged birds, within days, weeks, and two years of tagging.

Methods

Study site.—The present study was part of a larger project using radiotelemetry to investigate the risk of bycatch in coastal gillnet fisheries to Common Murres nesting on Tatoosh Island (48°23′N, 124°44′W), located at the confluence of the Strait of Juan de Fuca and the Pacific Ocean in Washington State. Demographic parameters, including colony size, reproductive success, and mortality have been monitored there since 1991. Colony attendance estimates during the 1990s ranged from 2,700 to 4,200 individuals (Parrish et al. 2001). Common Murres nest on ledges on vertical cliffs, and on cliff-tops under salmonberry (*Rubus spectabilis*), the predominant vegetation (Parrish 1995). Those cliff-top locations are relatively large (hundreds of nesting birds) and allow close observation via a series of blinds fitted with one-way mirror glass, situated at the edge of the nesting areas and accessed via landscape-cloth tunnels. In all years, Common Murres did not startle during entry, even though they nested against the blinds.

Capture and tagging.—Blinds allowed easy capture of birds. Common Murres were noosed with a 1-m-long telescoping pole and pulled through window holes blocked by flexible curtains. From 1999 to 2001, we captured both nonbreeders (individuals with no reproductive investment and no obvious mate), and breeders (but only one adult per breeding pair). To minimize chance of desertion, we chose breeders with chicks rather than eggs (Wanless et al. 1985). Whenever possible, we selected birds whose mates were present at the nest to ensure that chicks were not left alone. In 1999, 2000, and 2001, we captured 5 (all breeders), 18 (6 nonbreeders; 12 breeders) and 26 (4 nonbreeders; 22 breeders) Common Murres, respectively.

We used radiotransmitters, made by Advanced Telemetry Systems in 1999 (Isanti, Minnesota; model 2032-prong) and Holohil Systems in 2000–2001 (Carp, Ontario; model R1-2B), fitted with subcutaneous anchors. Transmitters were disk-shaped, approximately 2 cm in diameter and 0.8 cm in height (~1.6 cm^2 in cross-sectional area, which is <1% of Common Murres' cross-sectional area), and weighed 6–8 g (<1% of body weight). A 1.5 × 2 cm, 16-gauge surgical steel anchor protruded from the transmitter's leading edge; a suture channel was located along the trailing edge. Each device had a 20-cm whip antenna, projecting at a 30° angle from the trailing end, and battery life of ~3 months.

Within 5 min after capture, birds were restrained, hooded, and transported to a nearby building. After

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anesthetization with 2–5% isofluorane, the sterilized anchor was inserted through the skin at the intersection of the cervical and thoracic vertebrae, dorsally between the shoulder blades. A drop of cyanoacrylate glue (Bird Epoxy, Titan Corporation, Washington) was placed on the bottom of the transmitter, and the trailing edge was sutured to the skin to hold it centered and to prevent excessive wear of the anchor puncture. Contour feathers lapped over the transmitter, concealing its presence and preserving the bird's hydrodynamic profile as much as possible. Anesthesia and surgery were performed and monitored by experienced wildlife veterinarians. We drew a 1–2 cc blood sample from the medial metatarsal vein for later sexing via DNA analysis (Griffiths et al. 1998). In 2000 and 2001, Common Murres were weighed and then banded with a combination of one federal stainless steel band and three Darvic plastic color bands for individual recognition. Birds were released within 5 m of their nesting area, and walked back to their nest site. The entire procedure, from capture to release, took an average of 70 min (range: 28–153 min; average anesthesia time = 15 min).

Data on nonbreeders were collected in all years but restricted to (1) whether tagged birds subsequently abandoned the colony and (2) whether tagged birds were resighted in the two following years. All other analyses consisted of data on breeders only. We recorded five types of information about breeders: (1) subsequent presence at the colony, (2) time spent at the nest site, (3) behavioral activity at the nest site, (4) chick provisioning, (5) reproductive success, and (6) resightings in the following two years. Data in categories (2), (3), and (4) were collected in 1999 and 2000 only. Two types of controls were used: (1) mates of tagged breeders and (2) control pairs chosen randomly from a pool of pairs with chicks nesting within 2 m of tagged-pair nest sites (7, 12, and 21 control pairs in 1999, 2000, and 2001, respectively). None of the control birds were captured. Therefore, it is possible that any effect was attributable to handling, the tag *per se*, or both.

Presence at the colony.—Presence (or absence) of all tagged birds was recorded on a daily basis. Birds absent for three days and never seen again were considered deserters. Presence–absence observations were confirmed by daily on-island radiotracking surveys conducted from a fixed station at the highest point on the island. Those surveys allowed us to verify the presence–absence of tagged birds within a 10-km radius of the island. We did not attempt to record desertions by control pair birds, because they were not individually marked. In addition to presence–absence, we recorded amount of watch time (in minutes) each bird of a tagged pair spent on the nest site as a function of total observation time.

Activity.—We compared behaviors at the nest site of tagged breeders, their untagged mates, and birds in control pairs. We classified nest-site behavior as routine or incidental. Routine behaviors were long-lasting activities categorized as sitting–sleeping (sitting, with or without closed eyes), standing, and preening. Preening was further defined as preening feathers, preening chick, and preening tag (in the case of tagged birds). Incidental behaviors were discrete bouts of short-term activity or interactions with mates and neighbors, including allopreening (receiving preening from or preening another adult), fighting (repeated bill jabbing and occasionally grabbing), and mud-flinging (a stereotypical behavior of ritualized picking and tossing of mud or debris). In cases where both birds of the pair were present, we monitored the activity of the parent attending the chick.

Using a modified scan-sample technique, we recorded routine behavior of each brooding adult for the majority of each contiguous 5-min interval (the dominant behavior) over a 30-min period (i.e. six samples of dominant behavior). We created a distribution of dominant behaviors for each treatment group by averaging proportions of total time each bird spent in each behavior category (i.e. sitting–sleeping, standing, preening self, and preening chick). Because control birds could not be uniquely identified, we summarized activity data at the pair level. Although that may have ameliorated individual variation to some degree, we found no systematic gender-bias in behavior of tagged birds and their mates; therefore, we had no reason to suspect that pooling across pairs was unwarranted.

Incidental behavior bouts lasted much less than 2.5 min on average and would thus never be incorporated into an activity budget as a dominant behavior, but nevertheless may be important. Therefore, we recorded instances of incidental behavior, rather than their duration. We summed bouts of incidental behavior over each 30-min period and calculated an average number of bouts per watch within each treatment group.

Provisioning.—During brooding, one parent is usually present at the nest site (Gaston and Jones 1998). When the other mate returns to the nest site, it usually brings a single prey item for the chick (Ainley et al. 2002) and switches parental-care role with the brooding mate (Wanless and Harris 1986). At the individual level (tagged birds vs. mates) and pair level (tagged pairs vs. control pairs), we recorded (1) instances of return with or without fish, (2) fish identification to lowest possible taxa, (3) fish length relative to bill length (to the nearest one-quarter), and (4) instances of departure as a function of switching opportunities.

We used proportion of returns with or without fish as a measure of provisioning efficiency. To examine provisioning from the chick's point-of-view, we estimated the energy delivery rate as follows. We converted fish length to mass (Vermeer and Devito 1986, FishBase 2001), and mass to energy content (Vermeer

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and Devito 1986, Van Pelt et al. 1997). Because species-specific conversion factors may vary within as well as between years (Anthony et al. 2000, Pedersen and Hislop 2001), our calculations of fish mass and energy content are approximations. Energy delivery rate was calculated as the sum of fish energy content, divided by total observation hours.

It was logistically unfeasible to simultaneously observe all tagged birds and relevant controls. Therefore, behavioral data were recorded for a subset of 8–10 birds per blind (including birds from both tagged pairs and control pairs) during 30-min activity watches spread evenly throughout the day (0600–2000 hours) between 27 July and 20 August. Provisioning data (instances of departure and of return with and without fish) were also collected during activity watches. Other provisioning data (instances of returns with fish, fish taxa, and fish size) were collected during 1-h fish watches for a much larger sample of pairs (80–100 per blind). Observations started the day after tagging and generally lasted until chicks fledged, ~15 days after tagging.

Reproductive success.—Reproductive success was measured as fledging rate, defined as the proportion of chicks that survived to ≥18 days, were actually observed fledging, or both. A more restrictive measure of reproductive success considered additional information known for tagged birds, including sex of each parent and specific timing of premature departure of parents from the nest site (i.e. desertions). If the males (the parent who cares for Common Murre fledglings; Gaston and Jones 1998) deserted and the chick was still at the nest site two days later, we scored chick fate as not fledging. However, if the female deserted, the chick was >18 days of age, and no chick death was observed thereafter, we scored the chick as having fledged. We did not apply this latter measure to controls because we did not know the individual identity or sex of the birds.

Sample sizes for all analyses varied as a function of bird breeding status (nonbreeder vs. breeder), treatment (tagged bird, mate, control pair) and data type, as follows. We captured 49 Common Murres over three years (39 breeders: 13 males, 26 females; 10 nonbreeders: 5 males, 5 females). All survived capture and handling, except for one female which died during surgery. Out of 17 tagged pairs in the 1999 and 2000 samples, one chick died immediately, one pair nested at a blind not used for activity watches, and two pairs were unable to be seen easily, reducing the total sample used for activity watches to 13. Of those, one tagged bird had two potential mates, reducing the mate sample size to 12. Birds returning with fish were recorded in both fish (n = 16 pairs) and activity watches (n = 13 pairs), but birds returning without fish were recorded only in activity watches. Finally, for fish species composition, we restricted the analysis to only those birds that returned with fish (11 tagged birds).

Colony returns.—During routine colony monitoring, we recorded presence–absence of any previously tagged bird (identified by leg bands) in the two breeding seasons following tagging. No control birds were marked; therefore, they could not be used to compare the proportion of returning tagged with untagged birds. Despite that difficulty, we developed a proxy comparison of long-term tag effect as follows. Because all but one returning bird had shed its tag, we assumed that the probability of return in the second year was affected by factors other than tagging. Therefore, as a control, we calculated proportion of previously tagged birds (now without tags) that returned between years one and two after tagging. When possible, we recorded the breeding status of returning birds and the fate of their nesting attempt.

Tag retention time.—We monitored tagged birds dispersing after the breeding season with weekly aerial tracking surveys off the coasts of Washington and British Columbia. Detections collected during radiotracking surveys allowed us to derive a proxy measure for tag retention time: number of days between tagging and when the tag was last detected during radiotracking surveys. Tag retention is a minimum time, given that birds may have retained the transmitter after the last detection.

Statistical analysis.—We used two-way ANOVA models to test for main effects and interactions of tagging treatment and year, and of tagging treatment and sex, on behavior, nest-site attendance, and provisioning measures. With the exception of fish composition, there were no statistical differences between years; therefore, data were pooled across years.

Distribution of routine behavior was compared among tagged birds, their mates, and control pairs using compositional data analysis, a statistical technique appropriate for data sets, such as time budgets, where variables are not independent of each other (Aitchison 1986). Each bird or control pair was represented by a vector of five components, each consisting of percentage of time spent in each routine activity category. Compositional data were returned to a sample space suitable for conventional statistics through the use of additive log-ratio transformations. We divided all but one of the components by the remaining one. Any component can be used as this common denominator (Aitchison 1986); we used time spent in the "sit–sleep" category. Ratios equaling zero were increased by a small quantity (0.5) to accommodate a logarithmic transformation. We used Hotelling's Trace multivariate analysis of variance (MANOVA; Morrison 1967) to test for effects of tagging treatment.

We tested for differences in the three incidental behaviors between tagged birds, their mates, and control pairs using Kruskal-Wallis tests. Attendance and provisioning (percentage of time at nest, percentage of departures by brooder, percentage of returns with

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fish, rate of fish return, prey wet mass, energy content, and energy delivery rate) measures were compared within tagged pairs (tag birds vs. mates) and between pairs (tagged vs. control pairs), using ANOVA models or Student's *t*-tests. Proportional data that followed a binomial distribution were normalized with arcsine square-root transformations (Zar 1999). Prey species composition was analyzed using compositional data analysis as described above, except that vector components were percentages of fish in each taxon. We compared fledging success of tagged to control pairs, as well as percentage of returns to the colony, using a chi-square test. We used Tukey HSD for *post-hoc* pairwise comparisons. Means are accompanied by ±1 standard error (SE). All tests were two-tailed and differences were considered significant at $P = 0.05$. All statistical analyses were run in SPSS version 10.0.5 (SPSS, Chicago, Illinois) for WINDOWS.

RESULTS

In 1999 and 2000, activity behaviors were collected for a total of 40 h of activity watches ($\bar{x} = 7.7 \pm 0.7$ h individual^{-1}) and 195 h of fish watches ($\bar{x} = 40.4 \pm 2.9$ h pair^{-1}). Colony monitoring lasted 209 and 252 h in 2001 and 2002, respectively, during which we recorded breeding pair egg-and-chick fate, as well as banded bird returns. Island-based radiotracking lasted 111 h (37.0 ± 9.1 h year^{-1}).

Presence at the colony.—Of all tagged birds, only one—a nonbreeder in 2000—was never seen again at the colony after tagging (i.e. deserted), nor was it detected by on-island radiotracking. However, the bird was detected later during aerial surveys. Five tagged breeders (two in 1999 and three in 2001) deserted 8 ± 1 days after tagging (range: 6–13 days), while their chicks were still at the nest site. Three of those five birds were detected after they abandoned the colony: one during island-based radiotracking surveys, and two during subsequent aerial-tracking surveys. The fate of the remaining two breeders (4% of tag sample) is unknown. Desertion was not dependent on Common Murre mass (deserters: $\bar{x} = 949.5 \pm 12.4$ g, $n = 4$; nondeserters: $\bar{x} = 936.6 \pm 9.7$ g, $n = 39$; $t = 0.42$, df = 41, $P = 0.68$) or on handling time (deserters: $\bar{x} = 57.9 \pm 11.0$ min, $n = 4$; nondeserters: $\bar{x} = 71.7 \pm 3.9$ min, $n = 42$; $t = -1.23$, df = 46, $P = 0.22$).

Activity.—Males and females in our tag sample behaved similarly, regardless of treatment (MANOVA: $F = 0.67$, df = 3 and 19, $P = 0.572$); therefore, sexes were combined within tagged and mate treatment groups. There was no difference in time budgets of routine activities among tagged birds, their mates, and controls (MANOVA: $F = 0.98$, df = 6 and 64, $P = 0.45$; Fig. 1A). Birds spent the majority of their time sitting–sleeping and relatively little time in more alert activities (e.g. standing, preening). Tagged birds spent 3 ± 1% of their time preening the tag attachment site. Despite that additional attention to preening, time spent self-preening did not differ between tagged and untagged birds (ANOVA: $F = 0.63$, df = 2 and 35, $P = 0.54$). None of the incidental behavior comparisons were significant (allopreening: $\chi^2 = 3.89$, df = 2, $P = 0.14$; fighting: $\chi^2 = 0.60$, df = 2, $P = 0.74$; mud flinging: $\chi^2 = 0.38$, df = 2, $P = 0.83$; Fig. 1B).

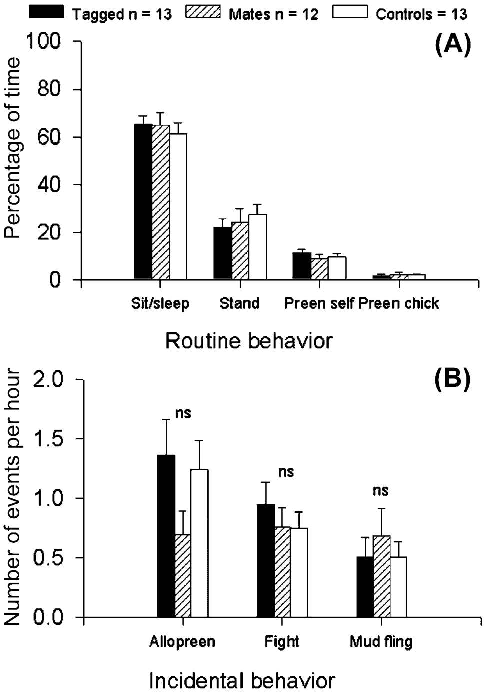

FIG. 1. Comparison of activities of tagged birds, their mates, and control birds. (A) Activity budget of routine behavior and (B) frequency of incidental behavior.

Time spent at nest site.—Tagged breeders and their mates were each present at the nest site during roughly half of our observation time, and that pattern held regardless of sex (ANOVA tagging treatment: $F = 2.69$, df = 1 and 26, $P = 0.1$; sex: $F = 1.51$, df = 1 and 26, $P = 2.31$; Fig. 2A). However, tagged birds were less likely than their mates to depart the nest site (ANOVA: $F = 32.5$, df = 1

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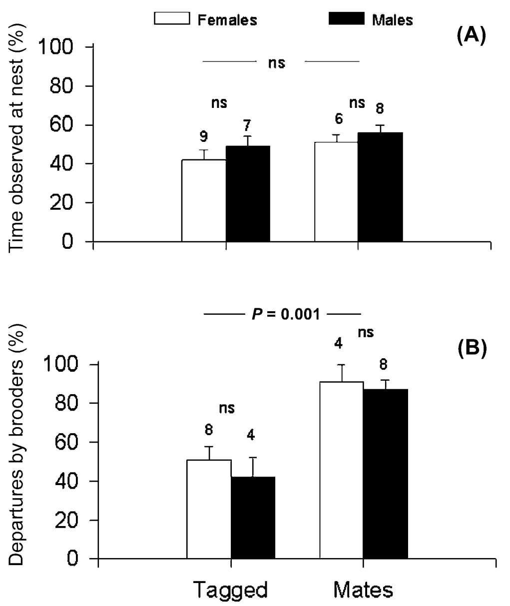

Fig. 2. (A) Time spent at nest site as percentage of total observation time. (B) Departure frequency expressed as percentage of all departures by brooders. Outcome of statistical tests between and within groups is indicated above graphs (ns = no significant difference). Sample sizes directly above error bars are number of birds.

and 21, $P < 0.001$; Fig. 2B); that difference was not a function of sex ($F = 2.58$, df = 1 and 21, $P = 0.123$). In fact, mates departed at almost every opportunity, regardless of whether they were brooders or had just returned to the nest site. Given that tagged birds were on the nest site an equal amount of time but switched to depart much less often, tagged bird trips away from the nest site were necessarily less frequent and much longer.

Provisioning.—Tagged birds returned to the nest with fish only half the time, whereas mates returned with a fish ~90% of the time (ANOVA: $F = 8.4$, df = 1 and 21, $P = 0.009$; Fig. 3A). There were no differences between sexes (ANOVA: $F = 0.44$, df = 1 and 21, $P = 0.52$; Fig. 3A). The pattern of lower chick provisioning by tagged birds relative to their mates was consistent throughout the observation period (repeated measures ANOVA: $F = 0.68$, df = 1 and 15, $P = 0.42$; Fig. 3B). As a result, fish delivery rate by tagged birds was less than half that of their mates (tagged birds: $\bar{x} = 0.07 \pm 0.02$ fish h^{-1}, $n = 16$; mates: $\bar{x} = 0.18 \pm 0.02$ fish h^{-1}, $n = 14$; $t = 4.09$, df = 28, $P < 0.001$).

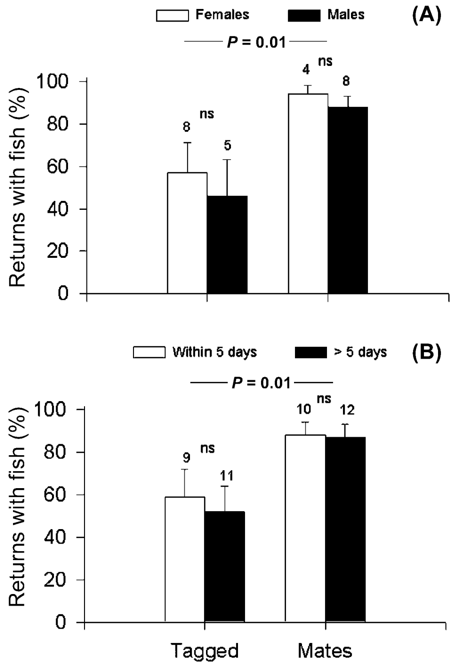

Fig. 3. Chick provisioning of male and female tagged birds and their mates. (A) Percentage of returns with fish over all days and (B) percentage of returns with fish within five days of tagging compared with percentage after fifth day of tagging. Outcome of statistical tests between and within groups is indicated above graphs (ns = no significant difference). Sample sizes directly above error bars are number of birds.

Fish taxonomic composition did not differ between tagged birds and their mates (MANOVA: $F = 1.09$, df = 4 and 20, $P = 0.39$; Fig. 4). In both 1999 and 2000, Pacific herring (*Clupea harengus*) and smelts (*Hypomesus pretiosus* and *Thaleichthys pacificus*) were the most-favored prey items. Mean approximated fish mass was also equal between tagged birds and their mates ($t = -0.52$, df = 22, $P = 0.61$). In sum, tagged birds were able to find and capture fish of the same type and size captured by their mates. However, because of lower prey-delivery rates, the amount of energy tagged birds delivered to their chicks was significantly lower than that delivered by their mates (tagged birds: $\bar{x} = 8.3 \pm 2.2$ kJ h^{-1}, $n = 16$; mates: $\bar{x} = 21.9 \pm 3.1$ kJ h^{-1}, $n = 14$; $t = 3.64$, df = 28, $P = 0.001$).

Although tagged birds fed their chick less often than their mates did, chicks of tagged pairs did not experience different provisioning rates as compared with chicks of control pairs

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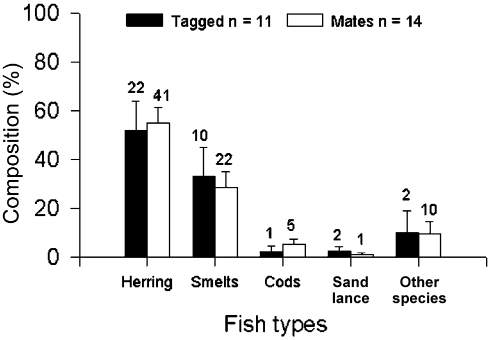

FIG. 4. Average distribution of fish types delivered to chicks by tagged birds and their mates: herring (*Clupea harengus*), smelt (*Thaleichthys pacificus*, *Hypomesus pretiosus*), cods (*Gadus macrocephalus*, *Microgadus proximus*), sand lance (*Ammodytes hexapterus*), and other species (making up <4% of the diet). Sample sizes of treatment groups are in the legend; fish sample sizes are above error bars. Outcome of statistical tests between and within group was not significant.

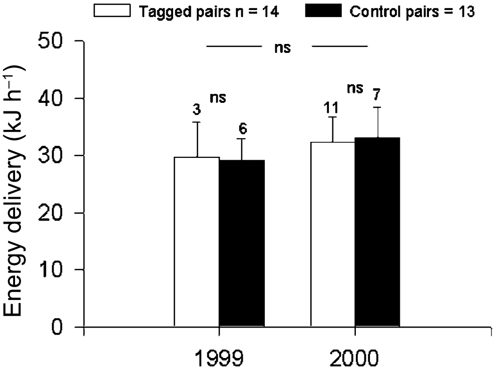

FIG. 5. Estimated average rate of fish energy delivery to chicks of tagged and control pairs in each of two years. Outcome of statistical tests between and within groups is shown above graph (ns = no significant difference).

(tagged pairs: $\bar{x} = 0.26 \pm 0.02$ fish h^{-1}, $n = 14$; control pairs: $\bar{x} = 0.32 \pm 0.03$ fish h^{-1}, $n = 13$; $t = 1.6$, df = 25, $P = 0.12$). There was an effect of year and a marginal difference in fish taxonomic composition between tagged ($n = 12$) and control ($n = 13$) pairs (MANOVA, tagging treatment: $F = 2.93$, df = 4 and 18, $P = 0.05$; year: $F = 6.31$, df = 4 and 18, $P = 0.002$), though the interaction was not significant ($F = 1.32$, df = 4 and 18, $P = 0.30$). However, energy delivery rate to chicks in tagged pairs was not significantly different from delivery rate to chicks of control pairs (tagged pairs: $\bar{x} = 31.8 \pm 3.6$ kJ, $n = 14$; control pairs: $\bar{x} = 31.3 \pm 3.3$ kJ, $n = 13$; $t = -0.12$, df = 25, $P = 0.91$; Fig. 5).

Reproductive success.—Three tagged-pair chicks and one control-pair chick died in the colony. Despite that disparity, the difference in fledging rate (percentage of chicks that fledged) between tagged and control pairs was not significant (tagged pairs: 92%, $n = 38$; control pairs: 98%, $n = 40$; $\chi^2 = 1.17$, df = 1, $P = 0.28$). However, it is likely that two additional tagged pairs failed to fledge their chicks—on the basis of our more restrictive measure that incorporates known sex and desertion dates of parents—which increases the total tagged-pair chick mortality to five (fledge rate = 87%). Even with that larger reduction in tagged-pair success, the apparent difference was not significant ($\chi^2 = 3.12$, df = 1, $P = 0.08$).

Colony returns.—For birds tagged overall, annual attrition was ~50%, and that rate did not appear to be solely a function of tagging. Percentage of tagged birds resighted the year after tagging was 56% and 31% for birds tagged in 2000 and 2001, respectively. There was no effect of year ($\chi^2 = 1.87$, df = 1, $P = 0.17$; Table 1); therefore, data were pooled. Breeders and nonbreeders did not differ in percentage returning the following year (46% and 43%, respectively; $\chi^2 = 0.02$, df = 1, $P = 0.89$; Table 1). Finally, percentage of previously tagged birds that returned again (i.e. in the second year after tagging) was not significantly different from the proportion returning just the first year (57% vs. 45%; $\chi^2 = 0.55$, df = 1, $P = 0.47$; Table 1). All Common Murres returning the second year were also seen the previous year. Effort, measured as observation hours, was not correlated with return probabilities (Pearson $r = 0.49$, $P = 0.18$).

Of nonbreeders returning in the breeding season following tagging and for which we could determine nesting status, 75% (3 of 4) became breeders. All returning breeders with known fates (12) maintained their breeding status. Of returning birds that bred, 73% successfully produced chicks, a percentage comparable to colony-wide estimates of pair reproductive success of untagged breeders at Tatoosh Island (77%, 2001; 71%, 2002; J. K. Parrish unpubl. data).

Tag retention time.—After release, one bird lost its transmitter, and a second transmitter failed, both within two weeks of tagging and

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Table 1. Return percentages of tagged birds to the colony for the first and second breeding seasons after tagging.

		Number of returns (rate)	
Tagging years	Number of tagged birds[a]	First breeding season after tagging	Second breeding season after tagging
		All tagged birds	
2000	18	10 (56%)	5 (50%)
2001	13	4 (31%)	3 (75%)
		Tagged breeders	
2000	12	7 (58%)	4 (57%)
2001	12	4 (30%)	3 (75%)
		Tagged nonbreeders	
2000	6	3 (50%)	1 (33%)
2001	1	0 (0%)	0 (0%)

[a] Birds tagged in 1999 were not banded and, therefore, not included here. In addition, 12 birds (3 nonbreeders and 9 breeders) tagged in 2001 were excluded because regular monitoring was not conducted at their subcolony the next year.

before the birds dispersed from the island for the season. We tracked postbreeding Common Murres for 200 h of aerial-survey flights. Transmitters were retained a minimum of 51 ± 9 days (n = 10) by nonbreeders and 47 ± 4 days (n = 38) by breeders.

Discussion

The majority of Common Murres in our study survived tagging with subcutaneous anchor transmitters and continued to attend the colony. In the colony, the behavioral repertoire of tagged and untagged breeders did not differ. However, nest-site attendance patterns and provisioning efficiency were altered by tags or tagging, though prey were of similar size and species distribution. Those effects did not translate into reduced chick provisioning or reproductive success. Effects of tags and handling did not persist into subsequent years, because return rates to the colony one year after tagging were similar to return rates two years after tagging, even though the birds were no longer carrying tags in the second year. Breeding success the following year was comparable to that of the rest of the Tatoosh Island colony. Finally, Common Murres retained their tags a minimum of 51 days.

Although others have studied the effects of tags and tagging on alcids (Wanless et al. 1985, 1988, 1989; Meyers et al. 1998; Newman et al. 1999), the present study is the first to report behavior at the colony in detail. Other studies of non-alcid species have reported increased preening (Perry 1981, Garrettson et al. 2000), a behavior associated with discomfort (Calvo and Furness 1992). We found no significant change in preening or in any of the on-colony routine or incidental activities of tagged birds in comparison with their mates or control-pair birds (Fig. 1). Therefore, tags did not appear to substantially alter activity patterns and level of social interaction of tagged individuals at the nest site.

In a similar study, Wanless et al. (1988) found an approximately three-fold difference in fish returns between tagged birds and their untagged mates (tagged birds' percentage of arrivals with fish: $\bar{x}$ = 19.7 ± 13.7% [n = 3] vs. their mates: $\bar{x}$ = 70.0 ± 6.1% [n = 3]). We found only an approximately two-fold difference in the same measure (tagged birds' percentage of arrivals with fish: $\bar{x}$ = 52.5 ± 10.2% [n = 13] vs. their mates: $\bar{x}$ = 90.1 ± 3.8% [n = 12]; Fig. 3). Those differences suggest that foraging conditions were easier for our study birds (indicated by differences in provisioning by mates of ~20%), that there was a difference as a function of attachment type (indicated by differences in tagged birds standardized by mate differences, ~30%), or both.

Several studies have suggested that depressed foraging efficiency in diving birds may be mediated by tag mass or cross-sectional area (Wilson et al. 1986, Hull 1997, Obrecht et al. 1998). External devices create hydrodynamic drag that may impose an additional energetic cost on swimming (Bannasch et al. 1994, Hull 1997). King Penguins (*Aptenodytes patagonica*) wearing

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transmitters (<1% of cross-sectional area) made fewer deep dives and paused longer at the surface (Ropert-Coudert et al. 2000), which suggests that even small devices can be energetically costly. It is conceivable that the energetic cost associated with drag may have limited dive depth, preventing Common Murres from fully using their normal foraging-depth range. However, in our study, prey returned by tagged birds did not differ in taxonomic composition (Fig. 4) or mass as compared with those returned by mates, which suggests that tagged and untagged Common Murres were fishing in similar areas. Instead, energetic constraints imposed by the tag may have reduced frequency of dives, successful dives, or both, as suggested by the lower rate of return with fish (fish per hour).

Both the present study and Wanless et al. (1988) document the compensatory effect of mates in chick provisioning. Flexible time-budgets and foraging habits allow seabirds to survive in variable environments (Davoren 2000, Barrett 2002). Above a certain threshold of food abundance, Common Murre pairs can maintain a relatively constant chick-feeding rate by varying amounts of time spent on- and off-colony (Burger and Piatt 1990, Zador and Piatt 1999). In a good year, that behavioral flexibility may accommodate and absorb tag effects on parental foraging success. However, the capacity of a mate to compensate for decreases in the other parent's provisioning may disappear during years of poor food conditions (Fraser and Trivelpiece 1994), given that the compensator is likely already maximizing foraging effort (Monaghan et al. 1994, Uttley et al. 1994). Therefore, tagging effects may translate into decreased reproductive success during poor food years (e.g. during El Niño events).

In the longer term, tagging did not appear to have an effect, at least as measured by annual return to the colony. Annual return percentages of tagged birds were similar both the year after tagging and in the subsequent year, well after the tag and anchor had been shed (Table 1). However, those percentages were substantially lower than annual return rates (to the same breeding sites) published for other colonies (e.g. Isle of May, Scotland: 91%, Harris et al. 1996; Great Island, Newfoundland: 95%, Ainley 2002) and may reflect frequent predator disturbance (Parrish 1995, Parrish and Paine 1996, Parrish et al. 2001) or anthropogenic factors (Melvin et al. 1999).

An important component of a tagging study is length of time that the tag remains attached to the animal in relation to the time interval of data collection (White and Garrott 1990). In our study, tags were retained for an average minimum of 51 days before radio signals were no longer detected. Newman et al. (1999) reported somewhat shorter retention times for subcutaneous anchor radiotransmitters attached to smaller alcids (Xantus's Murrelet [*Synthliboramphus hypoleucus*], mean = 15 days, maximum = 51 days; Marbled Murrelet [*Brachyramphus marmoratus*], mean = 45 days, maximum = 78 days). In dabbling ducks with average mass similar to that of Common Murres (~1000 g), subcutaneous anchor attachments have lasted from many months to a year without any sign of feather wear or infection, and anchors were eventually expelled as epidermal tissue was replaced (Pietz et al. 1995, Paquette et al. 1997). In contrast, radiotransmitters similar to the ones used here were lost within 10–20 days when glued to the back feathers of Common Murres (Wanless 1985, J. K. Parrish unpubl. data). Abdominal implants potentially have longer retention times; however, that attachment technique caused significant nest abandonment and ≤40% mortality in alcids within 20 days (Meyers et al. 1998, Hatch et al. 2000). Therefore, at least for medium-bodied diving species such as Common Murres, subcutaneous anchor attachments appear to have superior retention times as compared with most other attachment types, without causing overt harm to the birds or their chicks. We recommend using subcutaneous anchor transmitters on diving seabirds that employ flexible foraging strategies, except during years of low food availability when the capacity to absorb tag effects may be lower.

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reviewers improved the manuscript. Washington Sea Grant (award no. NA76RG0019, project R/F-131) and the Pacific Northwest Coastal Ecosystems Regional Study (PNCERS; grant no. NA96OP0238) funded the research.

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No: 552

Review of Seabird Demographic Rates and Density Dependence

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Summary

Constructing realistic population models is the first step towards reliably assessing how infrastructure developments, such as offshore wind farms, impact the population trends of different species. The construction of these models requires the individual demographic processes that influence the size of a population to be well understood. However, it is currently unclear how many UK seabird species have sufficient data to support the development of species-specific models.

Density-dependent regulation of demographic rates has been documented in a number of different seabird species. However, the majority of the population models used to assess the potential impacts of wind farms do not consider it. Models that incorporate such effects are more complex, and there is also a lack of clear expectation as to what form such regulation might take.

We surveyed the published literature in order to collate available estimates of seabird and sea duck demographic rates. Where sufficient data could not be gathered using UK examples, data from colonies outside of the UK or proxy species are presented. We assessed each estimate's quality and representativeness. Estimates are usually only available for a limited number of colonies, and there may be substantial inter-colony variation. Therefore we also indicate the extent to which estimates may be applied to different colonies.

This report and the accompanying material details demographic information on the 32 species of seabird and sea duck thought to be most vulnerable to off-shore renewable developments in the UK. The species covered are (in taxonomic order): greater scaup (*Aythya marila*), common eider (*Somateria mollissima*), goldeneye (*Bucephala clangula),* long-tailed duck (*Clangula hyemalis)*, common scoter (*Melanitta nigra*), velvet scoter (*Melanitta fusca*), red-throated diver (*Gavia stellata*), black-throated diver (*Gavia arctica*), great northern diver (*Gavia immer*), Manx shearwater (*Puffinus puffinus*), northern fulmar (*Fulmarus glacialis*), great cormorant (*Phalacrocorax carbo*), European shag (*Phalacrocorax aristotelis*), northern gannet (*Morus bassanus*), great crested grebe (*Podiceps cristatus*), Arctic skua (*Stercorarius parasiticus*), great skua (*Stercorarius skua*), black-legged kittiwake (*Rissa tridactyla*), little gull (*Larus minutus*), black-headed gull (*Chroicocephalus ridibundus*), common gull (*Larus canus*), lesser black-backed gull (*Larus fuscus*), herring gull (*Larus argentatus*), great black-backed gull (*Larus marinus*), Sandwich tern (*Sterna sandvicensis*), common tern (*Sterna hirundo*), Arctic tern (*Sterna paradisaea*), little tern (*Sternula albifrons*), common guillemot (*Uria aalge*), razorbill (*Alca torda*), black guillemot (*Cepphus grylle*), Atlantic puffin (*Fratercula arctica*).

The drivers that cause year-to-year variation in survival and productivity rates and the direction of their influence are also presented. In the majority of species this demonstrates that information on a specific colony cannot necessarily be extrapolated to multiple colonies without applying some degree of qualitative interpretation.

The majority of the seabird species considered received high and intermediate data quality and representation scores for adult survival rates. The exceptions were great black-backed gull and little gull. There was considerably less information available on juvenile and immature survival rates, and great northern diver, northern fulmar, Arctic skua, common tern, razorbill and Atlantic puffin were only available as return rates between fledging and recruitment. Juvenile and immature survival rates were not available for goldeneye, long-tailed duck, velvet scoter, Manx shearwater, little gull, black-headed gull, great black-backed gull, Arctic tern and little tern. An estimate of productivity from a long-term monitoring study was identified for the majority of species, with the exception of little gull. However, in some

species this was only available for a limited number of species. The survival and productivity rates of sea ducks were largely lacking, with the exception of common eider.

An estimate for age of recruitment was available for all of the seabird and sea duck species considered. There was considerably less information identified for the incidence of missed breeding and the rate of breeding dispersal (both of juveniles and adults). Elevated rates of natal dispersal were identified in northern fulmar, black-legged kittiwake, common gull, lesser black-backed gull, herring gull, arctic tern, common guillemot and black guillemot. However, these processes are likely to vary between colonies. Therefore the reported rates should not be used without applying some degree of qualitative interpretation.

There was substantial evidence that populations of seabirds and sea ducks exhibit compensatory density-dependent regulation on survival, productivity, recruitment and dispersal processes. However, in specific species and populations there was also clear evidence that depensatory density-dependent regulation operates on the rate of productivity. Depensation was reported in almost two times the number of studies that reported compensation as a mechanism regulating productivity rates. This positive feedback mechanism on the population size has the potential to be highly destabilising.

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Abbreviations

BAP – British Action Plan

CR – constant recapture probability with time or age class

MR – mark-recapture

OSPAR - The Convention for the Protection of the Marine Environment of the North-East Atlantic

RR – ring-recovery

SD_p – Standard deviation of the demographic process

SE_e – Standard error of the confidence

SMP – Seabird Monitoring Program

VR – variable recapture probability with time or age class

Glossary

Age-class – A group of individuals from the same species that are of similar age. Age-specific demographic parameters are detailed within each species account and accompanying material where possible. The age classes considered in this report are as follows; juvenile (fledging year), immature (years between the fledging year and recruitment), adult (breeding age bird) and senior.

Demographic parameter - A factor that determines the population size. The most important demographic parameters for seabirds and sea ducks are; survival, productivity, recruitment, dispersal and incidence of breeding.

Density-dependence - The influence of population size or density on one or more demographic parameters. This report considers density-dependent regulation acting as either *compensatory* (a negative feedback with population size) or *depensatory* (a positive feedback with population size).

Dispersal – Annual rate of permanent migration from the natal (hatching) or breeding population.

Estimation method – The modelling structure used to estimate recapture rates. This can be constant (CR) or varying (VR) through time. In long-term studies, recovery or re-sighting rates may change (e.g. caused by changes in reporting, protection laws or re-sighting effort). The influence of this process on the estimation of survival will depend on whether the rates were modelled as constant over time or time-dependent.

Family – A scientific taxonomy grouping level, indicating species that typically share similar demographic and ecological traits. The family is detailed for each species in the accompanying material to aid matching of demographic rates between similar species.

Incidence of missed breeding – Annual rate of breeding age birds that do not attempt to breed.

Key Site - The Seabird Monitoring Programme (SMP) has established a UK and Ireland-wide network of four 'Key Site' colonies; Skomer, Canna, Fair Isle and the Isle of May.

Mark-recapture – A technique used to estimate survival rates by re-capturing or re-sighting the identification rings on live birds.

Productivity - The annual population estimate of number of chicks fledged per pair.

Recruitment – The mean age that an individual from a population finishes deferred reproduction and recruits into the breeding population.

Ring-recovery – A technique used to estimate survival rates by recovering of identification ring from dead birds.

Standard deviation of the demographic process – When a demographic parameter is estimated to vary through time either with or without a trend, an estimation of the range is given as the standard deviation of the process (SD_p). For a single study this is the reported standard deviation, and for multiple studies this is the mean of the standard deviations or the standard deviation of the point estimates, whichever is larger.

Standard error of the confidence - When a demographic parameter is estimated to be constant through time, the confidence associated with this point estimate is given as the standard error of the estimate (SE_e).

Survival - The annual population estimate of number of individuals that survive from one breeding season to the next.

1. Introduction

The interaction between seabirds and offshore renewable energy developments has received considerable attention in recent years. Negative consequences include birds colliding with turbines, being displaced from their feeding grounds and wind farms acting as barriers (e.g. Drewitt & Langston 2006; Everaert & Stienen 2007; Johnston *et al* 2014). In order to assess the potential impacts of proposed new developments, population models are used to predict how a population is likely to change following their construction. The type of population model employed has largely depended on the quantity and precision of the demographic data available, and it is currently unclear how many UK seabird and sea duck species have sufficient data to support the development of species specific models.

In some cases, a stochastic Leslie matrix model (Caswell 2001) has been used to assess the potential impacts of offshore renewable developments on seabird and sea duck populations. However, for populations that have been intensively studied, more recent methods have also been used. A limited number of population studies have also included a compensatory density-dependent response (e.g. Freeman *et al* 2014). This negative feedback on population size operates to offset the losses of individuals from the population, for example, a lowered population density may cause a temporary increase in survival or productivity in the remaining individuals, supporting an increase in the population size back towards the long-term average. The use of density-independent population models has generally been considered to be a precautionary approach. However, among certain species, particularly gulls, the influence of density-dependent regulation on demographic rates can also appear as a positive feedback on population size, i.e. it acts as a depensatory mechanism. This mechanism can act to accelerate further population decline and has the potential to be highly destabilising.

This report presents individual species accounts for a selection of British seabirds, sea ducks, divers and grebes. Each account gathers the most up to date published estimates on the following demographic parameters: age-specific survival, age-specific productivity, age of recruitment, incidence of missed breeding, and natal and adult breeding dispersal. Particular attention has been given to regional variation in demographic rates, indicating the extent to which estimates may be applied to other less-well studied colonies. Where possible, the intrinsic and extrinsic factors that influence demographic rates are also detailed. The reported rates should enable population models that assess the impacts of offshore wind farms to be developed as reliably and realistically as possible. Where sufficient data could not be gathered using UK examples, data from colonies outside of the UK have been presented, or a proxy species has been identified. The evidence for density-dependent regulation of seabird demographic rates is also reviewed using examples from the UK, as well as non-UK studies on similar species.

2. Methodology

The project steering group included representatives from the Statutory Nature Conservation Bodies and Marine Scotland Science. The species considered in this report were identified by the Project Steering Group as those most likely to be impacted by offshore renewable developments. An extensive review was conducted for each species of peer-reviewed, scientific literature using the ISI Web of Science database. Grey literature was also explored using the Google Scholar search engine. Each search was conducted using the species scientific name, common name and the demographic terms: survival, productivity OR breeding success, and recruitment. The search was focused on studies from the UK, but when UK information was lacking, the geographic scope was widened. The productivity of several species is monitored annually at four Seabird Monitoring Program (SMP) Key Sites: Canna, Isle of May, Fair Isle and Skomer. Annual reports for the SMP key sites were obtained from http://jncc.defra.gov.uk/page-4467. For species where insufficient information was obtained through these channels, secondary literature sources were used: Cramp and Simmons (1977, 1983) Cramp (1985) and Baldassarre (2014). Where possible, all relevant articles were obtained in order to extract the relevant information from the text. To facilitate the comparison of demographic studies with different study periods the national population trajectory is detailed for each species. Trends were described based on national census data accessed from http://jncc.defra.gov.uk/page-3201 (JNCC 2014).

2.1. Demographic rates for seabirds in the UK

Survival rates are generally presented as annual estimates. When estimates for more than one colony were identified, the mean was estimated (weighted by the duration of each study) to represent survival across a broader geographic area. Age-specific survival rates are detailed where possible using the following age classes; juvenile (fledging year), immature (years between the fledging year and recruitment in to the breeding population), adult (breeding age bird) and senior. Where immature survival rates have been published as a single value between fledging and recruitment, these rates have not been converted into annual estimates. To construct age-specific models these rates will need to be split into annual rates that reflect the relevant age of recruitment. Caution should be applied when using these estimates to parameterise population models because survival rates of birds typically increase following the first year of life (Caughley 1977). A direct division of the immature survival rate into the recruitment time period is likely to be unrealistic. In the accompanying material, each survival estimate is presented with the methodology used for data collection. This is because the assumptions and biases associated with estimating survival rates from live recapture and dead recovery data differ (see Robinson & Ratcliffe 2010 for review). The biggest difference being that estimates of survival from live recapture models will typically be confounded by permanent emigration (which is equivalent to death within the study, since the individual is no longer available for recapture). This means that survival estimates generated from these models will underestimate the ‘true’ survival rate by an amount proportional to the degree of dispersal. For breeding adults of some species (e.g. guillemot, puffin) sufficient effort to re-encounter individuals (e.g. Harris *et al* 2000a) will reduce this bias since there is a high degree of site fidelity once individuals recruit to the breeding population. For others species (e.g. Sandwich tern), a high degree of breeding dispersal is likely and this bias may be large (e.g. Ratcliffe *et al* 2008). Each survival estimate in the accompanying material is also presented with the methodology used for estimation. This is because the recovery or re-sighting rates in long-term studies can change with time (e.g. caused by changes in reporting, protection laws or re-sighting effort). The bias on the survival estimates will depend on whether the rates were modelled as constant or varying over time. Estimates modelled using variable recapture rates are likely to be more reliable.

Productivity rates are expressed as the annual population estimate of number of chicks fledged per pair. Where rates of productivity were reported to increase with parental age or experience, annual estimates are also given for each age class separately. Regional trends in productivity rates were assessed using the analysis carried out by Cook and Robinson (2010). A regional mean and variance was estimated by using the grouping highlighted by Cook and Robinson (2010) to categorise the colony-specific rates detailed by Mavor *et al* (2008) and any more recent studies. For species where it was not possible to identify consistent regional trends, but reported large variation between colonies, colony-specific means were presented from Mavor *et al* (2008) and any more recent studies.

The incidence of missed breeding is the proportion of breeding adults that may skip an individual breeding attempt, but continue to breed at the same colony in subsequent years. The rate of breeding dispersal represents the proportion of breeding age individuals that migrate to breed at other colonies each year, and natal dispersal represents the proportion of individuals in a cohort that recruit into a breeding colony separate from their birth colony. These parameters are given as annual means to illustrate the potential biases associated with the specified estimates of survival and productivity. There is considerably less information available on both of these demographic processes, and they are likely to vary markedly between colonies depending on the local population trend and resource availability. To incorporate these mechanisms into population models would require some degree of qualitative interpretation. Unless specified, recruitment into the breeding population is given as the rounded modal (i.e. most common) value.

All of the demographic parameters identified for each species of seabird and sea duck are detailed in the accompanying material. To assist the accessibility of this information the best available parameters for each species are presented in this report. When a process is estimated to vary temporally either with or without a trend, an estimation of the range is given as the standard deviation of the process (SD_p). For a single study this is the reported standard deviation, and for multiple studies this is the mean of the standard deviations or the standard deviation of the point estimates, whichever is larger. The SD_p of multiple studies will represent the average variability. When a process is estimated to be constant through time, the confidence associated with the point estimate is given as the standard error of the estimate (SE_e). We suggest that, for population modelling purposes, values of $\pm 2*SD_p$ should approximately encompass much of the likely range of variation in each demographic parameter.

Each demographic parameter is presented with a quality and representation score. To assess quality, the estimate is scored on the number of years considered by the study, the number of individuals included per year, and whether an estimation of the range or error is available with the point estimate. Representation is assessed at the national scale; the estimate is scored on whether the data is from the UK, includes recent data (<10 years old), and whether the trajectory of the study colony reflects the current UK population trend. For example, if a study is based on more than 5 years but the population trajectory changed during the study period or does not match the current UK trend, the estimate may receive a good data quality score but a poor representation score (conditional on the other criteria). Each criterion receives a 0 for “no”, 1 for “partially/unknown and therefore requiring further evaluation”, and 2 for “yes”; scoring quality and representation individually out of 6. The itemisation of the scores for each species is detailed on the second worksheet of the accompanying material.

3. How to use this report

When interpreting the individual species accounts, it is worth considering that long-term studies and those that include several locations are likely to give more reliable parameter values and be more representative of the inherent variability in these. Estimates based on more than one study are identified in bold in the summary table (Table 33). When selecting values to parameterise population models some qualitative interpretation of the local population trends should also be employed. For species where regionally specific values are recommended in the species account, preference should be given to rates from colonies that have the same population trajectory, as opposed to colonies that are closely distributed.

The assessment of model validity is an essential part of model parameterisation. The model output should be heavily evaluated against the current population trend. If the model does not realistically recreate the local population trend alternative parameter values should be considered. Different modelling frameworks (e.g. stochastic) should also be explored if the quantity and precision of the demographic data available are sufficient.

4. Quality Assurance

An extensive and consistent search of the relevant literature was undertaken by using standardised search terms that were applied to all species. The data in the accompanying material and the body of text presented in this report have been subjected to an internal review by a senior BTO scientist. An earlier version of the report was commented on by the project steering group and all comments were incorporated in the final version. Additional spot-checking between the accompanying material and the report was conducted by Sue O’Brien (JNCC).

5. Species-specific accounts of demographic rates in seabirds

5.1 Sea ducks

5.1.1. Greater scaup (*Aythya marila*)

Table 1. Recommended estimates of demographic rates for the greater scaup.

Demographic parameter	Mean	SD_p	SE_e	Data quality	Data representation
Age-specific survival					
Juvenile survival (0-1 year) [1]	0.400			3	1
Adult survival [1]	0.810		0.038	5	1
Productivity [1]					
	0.570		0.120	5	1
Age of recruitment [2]					
	2			4	2
Incidence of missed breeding					
	-			-	-
Breeding dispersal					
Natal	-			-	-
Adult	-			-	-

[1]Flint *et al* 2006; [2]Dementiev & Gladkov 1952, cited by Cramp & Simmons 1977.

Recommended demographic rates

Published estimates of survival are largely from breeding populations in North America and Iceland. These include a mark-recapture study (Flint *et al* 2006) and two other studies where the methods were not detailed (Boyd 1962; Austin *et al* 2000). Juvenile survival rates have not been empirically measured but have been estimated based on productivity and population size data (Flint *et al* 2006). At present there are not sufficient data to explore regional variation in survival rates, therefore population models will have to assume the reported estimates of survival to be representative more broadly.

Estimates of local productivity are also limited to a few non-UK studies (see accompanying material greater scaup). Further matching of local population trends is recommended in order to assess the suitability of these estimates for modelling UK birds. At present there are not sufficient data to explore regional variation in productivity rates, and population models will have to assume the reported estimates to be representative more broadly.

Published information on the age of recruitment for greater scaup is relatively old and not based on UK data. Furthermore, the reported estimates differ; recruitment occurs at age 1 (Delacour 1959) or 2 years (Dementiev & Gladkov 1952, cited by Cramp & Simmons 1977). It was not possible to verify the corresponding population trend for these estimates. The older age is presented in Table 1 because it matches the estimates reported for other sea ducks (Tables 1-6). Information on natal and adult dispersal and the incidence of missed breeding is lacking. As a proxy for the incidence of missed breeding it may be appropriate to use estimates based on common eider (Table 2), however further matching of local population trends is recommended in order to assess the suitability of these estimates.

There is large disparity between the rates of breeding dispersal identified for the sea ducks (Tables 1-6). The application of these rates across multiple different species is not recommended.

Population trend in UK

The greater scaup is a UK BAP species and a red-list Bird of Conservation Concern. In the UK they breed in low numbers, but the winter population is supplemented from Iceland, Fennoscandia and Russia (Wernham *et al* 2002). Winter densities are highest in Northern Ireland and Scotland (Kirby *et al* 1993). Numbers in the UK decreased between the late 1980s and early 2010s (Austin *et al* 2014).

Factors influencing survival rates

Populations of greater scaup in Canada are declining and the possible drivers of local survival are reviewed by Austin *et al* (2000).

Factors influencing productivity rates

Populations of scaup in Canada are declining and the possible drivers of local productivity are reviewed by Austin *et al* (2000) and Baldassarre (2014).

Knowledge gaps

The demographic parameters of greater scaup are poorly resolved for the UK and information on the incidence of missed breeding and rates of dispersal are lacking. In the absence of a focused ringing or monitoring program in the UK it will be difficult to improve these estimates based on local data.

5.1.2. Common eider (*Somateria mollissima*)

Table 2. Recommended estimates of demographic rates for the eider.

Demographic parameter	Mean	SD_p	SE_e	Data quality	Data representation
Age-specific survival					
Juvenile survival (0-1 year) [1]	0.200			2	2
Adult survival ($\geq$2 year) [2-3]	0.886	0.009		6	4
Productivity [4-6]					
	0.379	0.470		5	1
Age of recruitment [1,7]					
	3			3	1
Incidence of missed breeding [2]					
	0.200	0.141		4	2
Breeding dispersal					
Natal	-			-	-
Adult [3]	0.012			4	3

[1]Sperduto *et al* 2003; [2]Coulson 1984; [3]Hario *et al* 2009; [4]Sweenen 1991; [5]Hario & Rintala 2006; [6]Lehikoinen *et al* 2006; [7]Christensen 1999.

Recommended demographic rates

The published estimates of survival for UK common eiders are relatively old (Coulson 1984). There are a number of more recent studies on populations outside of the UK (see accompanying material for common eider), however the local population trends for these studies could not be identified, and further evaluation is needed to assess their suitability for modelling UK eiders. The estimate given for Finland (Hario *et al* 2009) was similar to the UK estimate, and therefore these studies were combined to give the estimate of adult survival in Table 2. At present there are not sufficient data to explore regional variation in survival rates, therefore population models will have to assume the reported estimates to be representative more broadly. Sperduto *et al* (2003) estimated juvenile survival by combining data on similar species. There is some disparity between the estimate of adult survival given by Sperduto *et al* (2003) and the estimate in Table 2. Therefore caution should be applied when using their estimate of juvenile survival to model UK common eider.

Estimates of productivity are also limited to a few non-UK studies that indicate productivity can vary substantially between breeding sites (see accompanying material for common eider). Further matching of local population trends is recommended in order to assess the suitability of these estimates for modelling UK birds.

Eiders are thought to start breeding between the age of 2 and 4 years (Christensen 1999; Sperduto *et al* 2003). It was not possible to verify the corresponding population trend for these estimates; however the mean age is similar to the other estimates reported for sea ducks (Tables 1-6). Outside of the UK, females are thought to remain site faithful once they have recruited (Hario *et al* 2009). Information on natal dispersal is lacking. The incidence of intermittent breeding among eiders was found to increase in relation to population size (Coulson 1984).

Population trend in UK

The density of eiders breeding in the UK is greatest along coastal Scotland and Northern Ireland. During the winter, densities are supplemented by birds of continental origin (Baillie & Milne 1989), and concentrations are highest in south-east Scotland (especially the Firths of Tay and Forth), Cumbria and eastern UK, with smaller numbers also occurring in sheltered locations in western Scotland. Breeding and non-breeding numbers increased markedly during the last century (Kirby *et al* 1993), but more recently, numbers have declined (Austin *et al* 2014).

Factors influencing survival rates

Outside of the UK, adult survival does not show spatial or temporal variation (Alaska, Wilson *et al* 2007). However, large scale mortality events have been linked to the over exploitation of food resources (Camphuysen *et al* 2002), as well as epidemic disease (i.e. avian cholera) (Tjørnløv *et al* 2013).

Factors influencing productivity rates

Productivity increases with age (Baillie & Milne 1982). Local productivity rates have also been linked to winter climate (Lehikoinen *et al* 2006), parent quality, nest location (Bolduc *et al* 2005) and salinity (DeVink *et al* 2005). Productivity rates are also reported to decrease as local population densities increase (Hario & Rintala 2006), and levels of nest predation by gulls increases (Mendenhall & Milne 1985; Mawhinney *et al* 1999; Donehower & Bird 2008). Clutch size has also been negatively linked to population size (Coulson 1999).

Knowledge gaps

Information on breeding dispersal and incidence of missed breeding is lacking. To improve the data quality scores on the age of recruitment and the incidence of missed breeding would require intensive monitoring.

5.1.3. Goldeneye (*Bucephala clangula*)

Table 3. Recommended estimates of demographic rates for the goldeneye.

Demographic parameter	Mean	SD_p	SE_e	Data quality	Data representation
Age-specific survival					
Adult survival [1]	0.770			3	1
Productivity [2]					
	0.365			2	1
Age of recruitment [1-2]					
	3			5	1
Incidence of missed breeding [1-3]					
	Some			-	-
Breeding dispersal					
Natal	-			-	-
Adult [3-5]	0.564	0.125		5	1

[1]Dow & Fredga 1984; [2]Eadie *et al* 1995; [3]Milonoff *et al* 2002; [4]Dow & Fredga 1983; [5]Johnson 1967.

Recommended demographic rates

There are no published estimates of survival for goldeneye in the UK. The estimates of survival published by Dow and Fredga (1984) are based on Swedish breeding populations, and the local population trend could not be identified. Therefore further matching of local population trends is recommended in order to assess the suitability of these estimates for modelling UK goldeneye. At present there are not sufficient data to explore regional variation in survival rates, therefore population models will have to assume the estimate of adult survival from Sweden to be representative more broadly.

Estimates of productivity are lacking because it is difficult to follow chicks after the brood has left the nest site. Chick mortality is highest in the first week following hatching (Paasivaara & Pöysä 2007) and two non-UK studies estimate productivity to 15 days (Milonoff & Paananen 1993; Eadie *et al* 1995). Nest success rates are also reviewed in Baldassarre (2014). Brood survival is low, but it is thought to vary with geographic location and year (Baldassarre 2014). Therefore, colony-specific rates may be more suitable than a national average when parameterising population models for this species. Further matching of local population trends is recommended in order to assess the suitability of the non-UK estimates for modelling UK goldeneye. The values in Table 3 are presented for consideration as national values when parameterising population models for this species.

Published information on the age of recruitment for goldeneye is relatively old and not based on UK data (Dow & Fredga 1984). It was not possible to verify the corresponding population trend for this estimate; however the age is similar to the other estimates reported for sea ducks (Tables 1-6). Some incidence of missed breeding is reported by Dow and Fredga (1984), Eadie *et al* (1995) and Milonoff *et al* (2002), however the rate has not been quantified at the population-level. Breeding dispersal for females is relatively high (Johnson 1967; Dow & Fredga 1983; Eadie *et al* 1995). Furthermore, females that did not breed successfully were more likely to change nesting sites, although distances moved were typically less than 1km (Dow & Fredga 1983). Information on natal dispersal is lacking but thought to be lower for females than males (Dow & Fredga 1983). For the incidence of missed breeding it may be appropriate to use estimates based on eider. Further matching of local population trends is recommended in order to assess the suitability of these estimates. There is large disparity between the rates of breeding dispersal identified for the sea ducks (Tables 1-6). Therefore the application of these rates across multiple different species is not recommended.

Population trend in UK

A small population of goldeneye breed in Scotland (Musgrove *et al* 2013). Birds winter in both freshwater and coastal habitats, and coastal, winter densities are highest in Scotland (Campbell *et al* 1986). Numbers in the UK declined between 1986 and 2012 (Austin *et al* 2014).

Factors influencing survival rates

The drivers of survival for goldeneye have not been identified.

Factors influencing productivity rates

Outside of the UK, first time breeders have smaller clutches (Dow & Fredga 1984; Milonoff *et al* 2002). Local productivity has been linked to lay date (Dow & Fredga 1984) and availability of nest sites (Pöysä & Pöysä 2002). Productivity rates are reported to be negatively influenced by predation (Andersson & Eriksson 1982; Dow & Fredga 1984), and clutch size (Eriksson 1979), but increase with chick body mass (Paasivaara & Pöysä 2007). There was no evidence for an effect of temperature and rainfall (Paasivaara & Pöysä 2007).

Knowledge gaps

The demographic rates for the UK population of goldeneye are largely unknown. Outside of the UK, productivity and survival rates are also poorly resolved and estimates of breeding dispersal and the incidence of missed breeding are lacking. In the absence of a focused ringing or monitoring program in the UK it will be difficult to improve these estimates based on local data.

5.1.4. Long-tailed duck (*Clangula hyemalis*)

Table 4. Recommended estimates of demographic rates for the long-tailed duck.

Demographic parameter	Mean	SD_p	SE_e	Data quality	Data representation
Age-specific survival					
Adult survival [1-2]	0.730	0.014		5	1
Productivity [3]					
	1.900	0.660		5	1
Age of recruitment [4]					
	2			2	1
Incidence of missed breeding					
	-			-	-
Breeding dispersal					
Natal	-			-	-
Adult	-			-	-

[1]Boyd 1962; [2]Schamber *et al* 2009; [3]Bengtson 1972; [4]Cramp & Simmons 1977.

Recommended demographic rates

In the absence of a focused UK program on long-tailed duck, local survival and productivity rates are largely unknown. Published estimates are from breeding populations in Iceland (survival - Boyd 1962; productivity - Bengtson 1972; see accompanying material for long-tailed duck). The local population trend could not be identified for these studies, and there is no evidence to suggest that the UK population and the Icelandic population overlap on wintering grounds (Wernham *et al* 2002). Estimates based on the other sea duck species may therefore be more appropriate for parameterising population models on this species (Tables 1-6). Further matching of local population trends is recommended in order to assess the suitability of these estimates.

Published information on the age of recruitment for long-tailed duck is relatively old and not based on UK data (Cramp & Simmons 1977). It was not possible to verify the corresponding population trend for this estimate; however the age is similar to the other estimates reported for sea ducks (Tables 1-6). Information on breeding dispersal and the incidence of missed breeding is lacking. The paucity of ring-recoveries and lack of reliable offshore counts means that little is known about site fidelity in this species between winters. As a proxy for the incidence of missed breeding it may be appropriate to use estimates based on eider, however further matching of local population trends is recommended in order to assess the suitability of these estimates. There is large disparity between the rates of breeding dispersal reported for the other sea ducks. Therefore the application of these rates across multiple different species is not recommended.

Population trend in UK

The long-tailed duck is a winter visitor to the UK, breeding in northern Eurasia and North America. Winter densities are highest in the east coast firths of Scotland, but birds also overwinter on Shetland, Orkney and the Outer Hebrides (Campbell *et al* 1986). In the absence of targeted monitoring, exact population trends are unknown (Austin *et al* 2014). Numbers were thought to be relatively stable in the 1980s (Kirby *et al* 1993) but have shown signs of decline in recent decades (Austin *et al* 2014).

Factors influencing survival rates

Mortality in fixed fishing nets can be substantial (Bengtson 1972).

Factors influencing productivity rates

Local productivity rates have been linked to predation (Alison 1975).

Knowledge gaps

Information on age of recruitment, incidence of missed breeding and rates of dispersal are lacking. In the absence of a focused ringing or monitoring program in the UK it will be difficult to improve these estimates based on local data.

5.1.5. Common scoter (*Melanitta nigra*)

Table 5. Recommended estimates of demographic rates for the common scoter.

Demographic parameter	Mean	SD_p	SE_e	Data quality	Data representation
Age-specific survival					
Juvenile survival (0-1 year) [1]	0.749		0.056	3	1
Adult survival [1]	0.783		0.032	3	1
Productivity [2]					
	1.838	1.184		5	1
Age of recruitment [3]					
	3			3	2
Incidence of missed breeding					
	-			-	-
Breeding dispersal					
Natal	-			-	-
Adult	0.123		0.040	4	1

[1]Fox *et al* 2003; [2]Gardarsson and Einarsson 2004; [3]Bengtson 1972; [4]Cramp and Simmons 1977.

Recommended demographic rates

In the absence of a focused UK program on common scoter, local survival and productivity rates are largely unknown. Published estimates are from breeding populations in Iceland (survival - Fox *et al* 2003; productivity - Bengtson 1972; Gardarsson & Einarsson 2004; see accompanying material for common scoter), where a male-biased sex ratio was noted (Bengtson 1972). The local population trend could not be identified for these studies, and consequently the estimates given in Table 5 received low data quality and representation scores. The Icelandic population considered by Bengtson (1972) are likely to overlap with the UK breeding population on wintering grounds (Wernham *et al* 2002), and therefore both populations may experience the same drivers influencing their survival rates. The estimate of juvenile survival appeared very high compared to the rates identified for the other sea duck species (Tables 1-6). Therefore when parameterising population models for common scoter it may be more appropriate to use estimates of juvenile survival from similar species, such as common eider (Table 2).

Published information on the age of recruitment for common scoter is relatively old and not based on UK data (Cramp & Simmons 1977). It was not possible to verify the corresponding population trend for this estimate; however the age is similar to the other estimates reported for sea ducks (Tables 1-6). Female breeding dispersal is thought to be relatively low (Fox *et al* 2003), but information on natal dispersal and the incidence of missed breeding is lacking. As a proxy for the incidence of missed breeding it may be possible to use estimates based on common eider (Table 2), however further matching of local population trends is recommended in order to assess the suitability of these estimates. There is large disparity between the rates of breeding dispersal reported for the other sea ducks (Tables 1-6). Therefore the application of these rates across multiple different species is not recommended.

Population trend in UK
The common scoter is a UK BAP species and a red-list Bird of Conservation Concern. They breed in low numbers in western and northern Scotland (Gibbons *et al* 1993), but the wintering population is inflated with birds that breed in western Siberia and western and northern Europe. Winter densities are highest in coastal areas; the Carmarthen Bay, Liverpool Bay, the Moray and Dornoch Firths and the Firth of Forth (Kirby *et al* 1993). During the 1980s numbers were thought to be relatively stable, although declines were noted in the Moray Firth (Kirby *et al* 1993). More recent trends were not identified.

Factors influencing survival rates
The drivers of survival for common scoter have not been identified.

Factors influencing productivity rates
Outside of the UK, local productivity rates are negatively influenced by decreased food availability, severe weather and avian predation (Cramp & Simmons 1977; Gardarsson & Einarsson 2004).

Knowledge gaps
Information on age of recruitment, incidence of missed breeding and rates of dispersal are lacking for this species. In the absence of a focused ringing or monitoring program in the UK it will be difficult to improve these estimates based on local data.

5.1.6. Velvet scoter (*Melanitta fusca*)

Table 6. Recommended estimates of demographic rates for the velvet scoter.

Demographic parameter	Mean	SD_p	SE_e	Data quality	Data representation
Age-specific survival					
Adult survival [1]	0.773		0.018	4	1
Productivity [2]					
	0.350	0.058		3	1
Age of recruitment [3]					
	2			2	2
Incidence of missed breeding					
	-			-	-
Breeding dispersal					
Natal	-			-	-
Adult [4]	0.688			2	1

[1]Krementz *et al* 1997; [2]Traylor *et al* 2004; [3]Hartman *et al* 2013 [4]Brown & Brown 1981.

Recommended demographic rates

In the absence of a focused UK program on velvet scoter, local survival and productivity rates are largely unknown. Published estimates of adult survival are based on similar species (Garthe & Hüppop 2004), or were not published with the methodology used for estimation (Krementz *et al* 1997). Similarly, the majority of published estimates for productivity are relatively old and not based on UK data (Hildén 1964; Waaramaki 1968, cited by Cramp & Simmons 1977; see accompanying material for velvet scoter). The value of productivity given in Table 6 reflects estimates from Canada. Further matching of local population trends is recommended in order to assess the suitability of these estimates for modelling the UK population. The estimates of survival and productivity identified for the other sea duck species should also be considered when parameterising population models for this species (Tables 1-6).

The published estimates for the age of recruitment differ; birds recruit to the breeding population at age 2 (Hartman *et al* 2013) or 3 years (Cramp & Simmons 1977). Both of these values were identified as the modal age for the other sea duck species (Tables 1-6). It was not possible to verify the corresponding population trend for these estimates and therefore the more recent estimate is reported in Table 6. There is some relatively old information on the dispersal of breeding adults from outside of the UK (Brown & Brown 1981). However, information on natal dispersal and the incidence of missed breeding are lacking. As a proxy for the incidence of missed breeding it may be appropriate to use estimates based on common eider (Table 2), however further matching of local population trends is recommended in order to assess suitability.

Population trend in UK

The velvet scoter is a winter visitor to the UK, breeding in Scandinavia and Siberia. Winter densities are highest in the Moray Firth and St Andrews Bay (Kirby *et al* 1993). However, in the absence of targeted monitoring, exact population trends are unknown (Austin *et al* 2014). Regional differences in population change emerged in the 1980s (Kirby *et al* 1993), and signs of decline have been noted in recent decades (Austin *et al* 2014).

Factors influencing survival rates

Survival rates were not related to concentrations of heavy metals (Wayland *et al* 2008), but mortality in fixed fishing nets can be locally substantial (Stempniewicz 1994).

Factors influencing productivity rates

Local productivity rates have been linked to lay dates and brood size (Traylor & Alisauskas 2006). Productivity rates increase during years with favourable weather conditions (Hildén 1964; Traylor & Alisauskas 2006), but decrease in response to predation from mink (Nordström *et al* 2002) and gulls (Finland, Mikola *et al* 1994). Local productivity may also decline at high population densities (Hartman *et al* 2013).

Knowledge gaps

Information on juvenile and immature survival, age of recruitment, incidence of missed breeding and rates of dispersal are lacking. In the absence of a focused ringing or monitoring program in the UK it will be difficult to improve these estimates based on local data.

5.2 Divers

5.2.1. Red-throated diver (*Gavia stellata*)

Table 7. Recommended estimates of demographic rates for the red-throated diver.

Demographic parameter	Mean	SD_p	SE_e	Data quality	Data representation
Age-specific survival					
Juvenile survival (0-1 year) [1]	0.600			2	1
Immature survival (1-2 year) [1]	0.620			2	1
Adult survival (≥3 year) [1-2]	0.840		0.074	4	2
National-average productivity [3-4]					
	0.571	0.222		6	3
Age of recruitment [5]					
	3			4	3
Incidence of missed breeding					
	-			-	-
Breeding dispersal					
Natal	-			-	-
Adult [6]	0.250			2	3

[1]Hemmingsson & Eriksson 2002; [2]Schmutz 2014; [3]Gomersall 1986; [4]Booth 1999; [5]Okill 1994; [6]Okill 1992.

Recommended demographic rates

In the absence of a focused UK ringing program on red-throated diver, there are no published estimates of local survival rate. Outside of the UK, published estimates include a ring-recovery study from Sweden (Hemmingsson & Eriksson 2002) and a mark–recapture study from Alaska (Schmutz 2014). Both of these studies are based on stable populations. The Swedish population considered by Hemmingsson and Eriksson (2002) is likely to overlap with the UK breeding population on wintering grounds (Wernham *et al* 2002). Therefore both populations may experience the same drivers influencing their survival rates. The estimate of adult survival given in Table 7 combined both of these studies and is recommended for modelling stable populations of red-throated diver, such as Shetland and the Hebrides (Dillon *et al* 2009). When constructing population models for populations that are increasing (i.e. Scottish Mainland and the Orkneys) these survival estimates may not be suitable. Regional differences in population trend should be considered when interpreting the outputs of any models parameterised with these values. Juvenile and immature survival rates are poorly resolved due to small sample sizes. Hemmingsson and Eriksson (2002) did not exclude birds ringed as non-fledged chicks, which may cause values to be slightly underestimated. However, the quoted rates appear to be higher than those identified for the black-throated and great northern diver (Table 8; Table 9). These rates should be used with caution in population models, and it may be more suitable to use values estimated for great northern diver (Table 9).

The rate of productivity given in Table 7 is calculated from all the UK estimates listed in the accompanying material for red-throated diver. Although this estimate is derived from relatively old studies where the underlying population trajectory could not be verified, the value is similar to the estimate reported for the stable population breeding in Alaska (Rizzolo *et al* 2014). In the UK, regional differences in population trends have been reported (Dillon *et* *al* 2009), but regional trends in productivity have not been investigated (e.g. Cook & Robinson 2010). Colony-specific rates may be more suitable than a national average when constructing population models. These should be selected based on the local population trajectory. For example, the estimates given by Booth (1999) may be more suitable for modelling populations with increasing trends than those given by Gomersall (1986) (see accompanying material for red-throated diver).

Based on a small sample size, birds first breed at age 3 although some may also attempt to breed at age 2 (Okill 1994). This age of recruitment is lower than the estimates identified for black-throated and great northern diver (Table 8; Table 9). Some qualitative interpretation of the local population trends should be employed when parameterising this value in population models, and the age of recruitment reported for black-throated or great northern divers may be more suitable (Table 8; Table 9). In the UK, dispersal of breeding adults is high (Okill 1992), but information on natal dispersal is generally lacking. Hemmingsson and Eriksson (2002) report some fidelity to the natal colony, however small sample sizes prevented reliable estimation of emigration rates.

Population trend in UK

The number of breeding pairs in Orkney and north and west Scotland increased between 1994 and 2006. During the same time period, populations in Shetland and the Hebrides numbers remained relatively stable (Dillon *et al* 2009). Birds overwinter in coastal and near offshore waters around the UK and the population are inflated by individuals from Scandinavia and Greenland (Wernham *et al* 2002). The number of red-throated divers wintering in the UK was relatively stable between the mid-1990s and 2005, but increased between 2005 and 2012 (Austin *et al* 2014).

Factors influencing survival rates

Outside of the UK, local survival rates have been linked to the marine environment, and decrease at high values of the Pacific Decadal Oscillation (i.e. survival rates are lower following warmer sea temperatures and stronger winds, Schmutz 2014).

Factors influencing productivity rates

Outside of the UK, local productivity rates have been linked to flooding risk (Eberl & Picman 1993; Douglas & Reimchen 1988), human disturbance (Rizzolo *et al* 2014) and predation by foxes (Douglas & Reimchen 1988; Rizzolo *et al* 2014).

Knowledge gaps

Adult survival rates for the UK population of red-throated diver are currently unknown, and are poorly resolved for populations outside the UK. Juvenile and immature survival rates are also poorly resolved. There is little information on the incidence of missed breeding and natal dispersal. The estimate for age of recruitment is also based on a small sample size. Further analysis of colour-marking data from Shetland and Orkney may help improve these estimates.

5.2.2. Black-throated diver (*Gavia arctica*)

Table 8. Recommended estimates of demographic rates for the black throated-diver.

Demographic parameter	Mean	SD_p	SE_e	Data quality	Data representation
Age-specific survival					
Juvenile survival (0-1 year) [1]	0.400			2	1
Adult survival (≥2 year) [1-2]	0.817	0.064		4	1
National-average productivity [3-8]					
	0.425	0.292		6	4
Age of recruitment [9]					
	5			3	1
Incidence of missed breeding				-	-
	-			-	-
Breeding dispersal					
Natal	-			-	-
Adult	-			-	-

[1]Hemmingsson & Eriksson 2002; [2]Nilsson 1977; [3]Mudge & Talbot 1993; [4]Hancock 2000; [5]Sharrock 1976; [6]Jackson 2003; [7]Bundy 1979; [8]Thom 1986; [9]Lehtonen 1970.

Recommended demographic rates

In the absence of a focused UK ringing program on black-throated diver, there are no published estimates of local survival rate. The estimates of survival published by Hemmingsson and Eriksson (2002) and Nilsson (1977) are based on Swedish breeding populations. Although the local population trend could not be identified for either of these studies, the wider European and UK breeding populations were both increasing during the 1990s (Hemmingsson & Eriksson 2002; Austin *et al* 2014). The rates of survival estimated by Hemmingsson and Eriksson (2002) are constrained by very small sample sizes; therefore, despite the study being considerably older, the estimate reported by Nilsson (1977) was also included when calculating the estimate of adult survival given in Table 8. Regional variation in the survival rates of black-throated diver has not been investigated. At present there are not sufficient data to model colony-specific survival rates, therefore population models will have to assume that the estimate of adult survival from Sweden is representative more broadly. Juvenile survival rates are poorly resolved due to small sample sizes. Therefore it may be more appropriate to use survival estimates based on the other diver species to build population models for black-throated diver.

Regional trends in population size and productivity rates have not been investigated (e.g. Cook & Robinson 2010). The average rate of productivity given in Table 8 is calculated from all of the UK estimates listed in the accompanying material for black-throated diver. Although these studies are relatively old, the UK population trend is not thought to have changed since the mid-1990s (Austin *et al* 2014). Furthermore, these studies considered large geographic areas and therefore should be representative more broadly.

Published information on the age of recruitment for black-throated diver is also relatively old and not based on UK data (Lehtonen 1970, cited by Nilsson 1977). It was not possible to verify the corresponding population trend for this estimate; however the age is similar to the estimate reported for the great northern diver (Table 9). Hemmingsson and Eriksson (2002) report some natal and adult site-fidelity, however very small sample sizes (n=3 birds tagged as adults; n=2 birds tagged as chicks) prevented reliable estimation of dispersal rates. The rates of dispersal given for red-throated or great northern diver should be considered when parameterising population models for black-throated diver (Table 7; Table 9). Further matching of local population trends is recommended in order to assess the suitability of these estimates for modelling UK birds.

Population trend in UK

The black-throated diver is a UK BAP species. The European population of black-throated divers increased during the 1990s (Hemmingsson & Eriksson 2002), and this trend has continued in the UK over the last decade (Austin *et al* 2014). More recent trends for outside of the UK could not be identified for this publication.

Factors influencing survival rates

The drivers of survival for black-throated diver have not been identified.

Factors influencing productivity rates

In the UK, local productivity rates have been linked to prey diversity (Jackson 2003, 2005) and floating artificial nesting rafts (Hancock 2000)

Knowledge gaps

The survival rates of black-throated diver in the UK are currently unknown, and are poorly resolved outside of the UK. Information on the incidence of missed breeding and dispersal is lacking, and the age of recruitment is based on relatively old data. In the absence of a focused UK ringing program on black-throated diver it will be difficult to improve these estimates using local data.

5.2.3. Great northern diver (*Gavia immer*)

Table 9. Recommended estimates of demographic rates for the great northern diver.

Demographic parameter	Mean	SD_p	SE_e	Data quality	Data representation
Age-specific survival					
Immature survival (0-3 year) [1]	0.770		0.020	6	2
Adult survival (≥4 year) [1-2]	0.870		0.078	6	2
Productivity [3-6]					
	0.543	0.170		6	1
Age of recruitment [7]					
	6			3	1
Incidence of missed breeding [7]					
	0.307			3	1
Breeding dispersal					
Natal [1]	0.069			2	2
Adult [2]	0.090	0.166		6	1

[1]Piper *et al* 2012; [2]Mitro *et al* 2008; [3]Belant & Anderson 1991; [4]Croskery 1991; [5]Timmermans *et al* 2005; [6]Grear *et al* 2009; [7]Evers 2004.

Recommended demographic rates

The great northern diver is a winter visitor to the UK that breeds in Iceland. In the absence of a focused ringing program in Iceland, the local survival rates are largely unknown. Published estimates are limited to breeding populations in New England and Wisconsin; including a ring-recovery (Mitro *et al* 2008) and two mark–recapture studies (Mitro *et al* 2008; Piper *et al* 2012). These studies identified drivers of survival that were similar to those identified to influence the productivity of black-throated diver in the UK; e.g. fluctuating water levels (UK – Hancock 2000; US – Piper *et al* 2002) and water chemistry (UK – Jackson 2005; US – Piper *et al* 2012). The estimates of survival in Table 9 appear to be higher than those identified for the other two species of diver. However, the higher data quality score attained by these studies indicates that they are likely to be more robust. The rates for juvenile and immature survival given by Piper *et al* (2012) are adjusted for natal dispersal and therefore the bias on true survival rate may be low. In America, adult survival rates are similar across different regions (Mitro *et al* 2008). Therefore, these rates may be suitable for modelling the population of great northern divers that winter in the UK.

There is little published information on the productivity of great northern divers breeding in Iceland. Therefore the rate of productivity given in Table 9 reflects estimates from America and Canada (see accompanying material for great northern diver). One of the identified drivers of productivity in America and Canada is similar to that identified to influence black-throated diver breeding in the UK; water chemistry (UK – Jackson 2005; US – Piper *et al* 2012). Therefore this value may be suitable for modelling the productivity of UK birds.

The mean age of first breeding is higher than identified for the other two species of diver considered (range 4-11 years; Evers *et al* 2000, cited by Evers 2004). In America, adult and natal breeding dispersal is thought to be low (Mitro *et al* 2008; Piper *et al* 2012), and the incidence of missed breeding is thought to be high (Yonge 1981, cited by Evers 2004; Taylor & Vogel 2003, cited by Evers 2004).

Population trend in UK

The wintering population increased during the late 1990s, but stabilised between 2000 and 2012 (Austin *et al* 2014).

Factors influencing survival rates

Outside of the UK, local survival rates have been linked to water chemistry and lake size (Piper *et al* 2012), nest location and mercury levels (Mitro *et al* 2008). In America, survival rates do not appear to differ between the sexes (Mitro *et al* 2008).

Factors influencing productivity rates

Outside of the UK, local productivity rates have been linked to water chemistry, lake-size (Piper *et al* 2012) and population size (Evers 2004).

Knowledge gaps

The demography of great northern divers wintering in the UK is poorly resolved. In the absence of a focused ringing or monitoring program in Iceland it will be difficult to improve these estimates based on local data.

5.3 Procellariiformes

5.3.1. Manx shearwater (*Puffinus puffinus*)

Table 10. Recommended estimates of demographic rates for the Manx shearwater.

Demographic parameter	Mean	SD_p	SE_e	Data quality	Data representation
Age-specific survival					
Adult survival [1]	0.870	0.080		6	4
Experience-specific productivity					
1st attempt [2]	0.630	0.141		2	3
>1 attempt [2]	0.781	0.118		2	3
Colony-specific productivity					
Rum [3]	0.680	1.442		6	4
Sanda [3]	0.890	0.134		6	4
Skomer [1]	0.598	0.098		6	4
Bardsey [3]	0.810	0.032		6	4
National-average productivity [1,3]					
	0.697	0.426		6	4
Age of recruitment [4]					
	5			2	3
Incidence of missed breeding [5]					
	0.157			2	3
Breeding dispersal					
Natal [6-7]	Low			-	-
Adult [4]	Low			-	-

[1]Büche *et al* 2013; [2]Brooke 1978a; [3]Mavor *et al* 2008; [4]Harris 1966; [5]Perrins *et al* 1973; [6]Harris 1972; [7]Brooke 1978b.

Recommended demographic rates

The UK estimates of survival for Manx shearwater are largely limited to Wales (see accompanying material for Manx shearwater). They include a relatively old ring-recovery and mark-recapture study from Skokholm that adjusted the estimates of survival for the occurrence of skipped breeding (Perrins *et al* 1973). Therefore the bias on true survival rate may be low. More recent estimates include a mark-recapture analysis from Skomer (Büche *et al* 2013; Table 10). Regional variation in survival rates has not been investigated; therefore population models will have to assume the estimates of adult survival from Wales to be representative more broadly. Juvenile and immature survival rates are more difficult to resolve due to lower recovery rates. In Perrins *et al* (1973), the sample of years and individuals was too small for reliable conclusions to be drawn. More recent attempts to estimate survival during this life stage have not been made.

Regional trends in productivity have not been investigated for this species (e.g. Cook & Robinson 2010). Consequently, it may be more suitable to use colony-specific rates of productivity to parameterise population models on this species (Table 10). The productivity of Manx shearwater is monitored annually at one SMP Key Site: Skomer, and long-term productivity estimates are given for three additional UK localities by Mavor *et al* (2008).

The age of first breeding has been explored by aging Manx shearwaters ringed as young and later found with an egg (Harris 1966). The modal age was 5 years, however a large proportion were also 6 or 7 years old at first breeding. Because some of the older birds in this study might have bred one or more seasons before being discovered, these results cannot be taken as definite ages of first breeding. Harris (1966) also reports that the modal age of first return following deferred reproduction occurs at 4 years old. Given that seabirds often prospect breeding sites during the year of first return, a modal age of first breeding at 5 years appears to be sensible. Local incidence of missed breeding has been estimated on Skokholm (Perrins *et al* 1973), but this study is based on one year and is therefore unlikely to be representative more widely. Breeding dispersal of adult Manx shearwaters is low and any movements within a colony are usually short (Harris 1966; Perrins *et al* 1973). Natal dispersal also appears to be low (Skokholm, Harris 1972), especially for males (Skokholm, Brooke 1978b).

Population trend in UK
Manx shearwaters breed exclusively on islands, and of the UK population, 40% breed on Rum, and 50% in Pembrokeshire on the adjacent islands of Skomer, Skokholm and Middleholm. Exact population trends are difficult to verify because of the nocturnal and subterranean habit of this species (JNCC 2014).

Factors influencing survival rates
The drivers of survival for Manx shearwater have not been identified.

Factors influencing productivity rates
On Rhum, Manx shearwater showed a preference for burrows less susceptible to flooding (Thompson & Furness 1991). Although fledging success was not influenced by rainfall, an increase in total rainfall and the incidence of heavy rainfall events during incubation were linked to a decrease in hatching success (Thompson & Furness 1991). On Skokholm, productivity of first time breeders was relatively low, stabilising from the second attempt (Brooke 1978a).

Knowledge gaps
Existing ring-recovery and sighting data should be sufficient to estimate adult survival rates away from Wales and investigate regional trends. Other active mark-recapture studies are on Rum and Copeland. There is little information on juvenile and immature survival rates, and breeding dispersal rates have not been quantified. Assuming low levels of natal dispersal, one approach would be to solve for juvenile survival based on other demographic parameters and population trend data from a regularly monitored site (e.g. Skomer). To improve the data quality scores on the incidence of missed breeding and the age of recruitment would require intensive monitoring.

5.3.2. Northern fulmar (*Fulmarus glacialis*)

Table 11. Recommended estimates of demographic rates for the northern fulmar.

Demographic parameter	Mean	SD_p	SE_e	Data quality	Data representation
Age-specific survival					
Immature survival (0-8 years)[1]	0.260	0.150		6	1
Adult survival [2]	0.936	0.055		6	3
Colony specific productivity					
Ailsa Craig [3]	0.740	0.112		6	3
Handa [3]	0.460	0.124		6	3
St. Kilda [3]	0.280	0.072		6	3
Canna [3]	0.400	0.089		6	3
Hermaness [3]	0.410	0.124		6	3
Burravoe [3]	0.320	0.017		4	3
Eshaness [3]	0.420	0.089		6	3
Noss [3]	0.400	0.108		6	3
Foula [3]	0.390	0.000		4	3
Troswick Ness [3]	0.390	0.089		6	3
Sumburgh Head [3]	0.450	0.089		6	3
Fair Isle [4]	0.426	0.087		6	4
North Ronaldsay [3]	0.230	0.170		6	3
Papa Westray [3]	0.480	0.190		6	3
Rousay [3]	0.520	0.150		6	3
Eynhallow [5]	0.414	0.138		6	3
Costa Head [3]	0.470	0.124		6	3
Mull Head [3]	0.400	0.124		6	3
Gultak [3]	0.400	0.124		6	3
Old Man, Hoy [3]	0.450	0.133		6	3
Wilkhaven [3]	0.220	0.210		6	3
Easter Ross [3]	0.160	0.139		6	3
Isle of May [6]	0.346	0.101		6	4
Tantallon [3]	0.310	0.218		6	3
Farne Islands [3]	0.540	0.089		6	3
Coquet Island [3]	0.460	0.150		6	3
Marsden Cliffs [3]	0.230	-		4	3
Peel Headlands [3]	0.340	0.246		6	3
Glen Maye [3]	0.350	0.174		6	3
Bardsey [3]	0.740	0.112		6	3
Skomer [7]	0.327	0.131		6	4
National-average productivity [3-7]					
	0.419	0.127		6	4

Demographic parameter	Mean	SD_p	SE_e	Data quality	Data representation
Age of recruitment [8]					
	9			4	3
Incidence of missed breeding					
	-			-	-
Breeding dispersal					
Natal [9]	0.915	0.035		5	3
Adult [10-11]	Low			-	-

[1]Jenouvrier *et al* 2003; [2]Grosbois & Thompson 2005; [3]Mavor *et al* 2008; [4]Shaw *et al* 2010; [5]Lewis *et al* 2009; [6]Newell *et al* 2010; [7]Büche *et al* 2013; [8]Ollason & Dunnet 1978b; [9]Dunnet *et al* 1979; [10]Macdonald 1977; [11]Hatch 1987.

Recommended demographic rates

The UK estimates of survival for northern fulmar are largely limited to Eynhallow, Orkney (see accompanying material for northern fulmar). They include two mark-recapture studies, conducted almost 30-years apart, that report similar estimates of adult survival (Dunnet & Ollason 1978a; Grosbois & Thompson 2005). The estimate of adult survival given in Table 11 reflects the more recent study that utilises a longer time series (Grosbois & Thompson 2005). Juvenile and immature survival rates are more difficult to resolve because few rings are recovered for these age-classes. As a proxy for survival between fledging and first return to the colony, it may be appropriate to use estimates based on southern fulmars (Jenouvrier *et al* 2003). Converted into an annual rate of survival (using the age of recruitment for southern petrel, 11±4 years, Jenouvrier *et al* 2003), the quoted rates for immature survival appear to be higher than the rates identified for the other species of seabird considered in this report (0.884). Therefore it may be more suitable to use juvenile survival estimates from other Procellariiform species. Regional variation in survival rates has not been investigated; however regional differences in population trends could not be wholly attributed to local differences in breeding success (Cook & Robinson 2010). At present there are not sufficient data to model colony-specific survival rates for northern fulmar, therefore population models will have to assume the estimates from Orkney are representative more broadly. It will be important to consider the regional differences in population trend as caveats of population models that are developed for colonies away from Orkney.

Cook and Robinson (2010) could not identify consistent trends in the productivity of northern fulmar within the SMP and OSPAR regions. Consequently, it may be more suitable to use colony-specific rates of productivity to parameterise population models on this species (Table 11). The productivity of northern fulmars is monitored annually at all four SMP Key Sites: Isle of May, Canna, Fair Isle and Skomer, and long-term productivity estimates are given for 27 additional UK localities by Mavor *et al* (2008).

Published information on the age of recruitment for northern fulmar is relatively old (Ollason & Dunnet 1978), and national census data indicate that the trajectory of the UK population has changed since this study; from increasing to decreasing (Austin *et al* 2014). Therefore caution should be applied when using the reported age to model current population trends (Table 11). The reported age is considerably higher than reported for other UK seabirds, but is similar to the closely related southern fulmar (11±4 years, Jenouvrier *et al* 2003). Consequently, the difference to the other seabirds may reflect life history strategy, as opposed to population trajectory. Some missed breeding is reported by Carrick (1954) and Ollason and Dunnet (1983), however small sample sizes prevented population rates being reliably quantified. Some tendency for individuals to skip breeding in response to environmental factors is reported for southern (Jenouvrier *et al* 2003) and northern fulmars

(Thompson & Ollason 2001). In the UK, the proposed mechanism is lower recruitment of sand eels during the previous winter (Thompson & Ollason 2001). Adult dispersal has not been quantified, but very high mate and site-fidelity indicates that this parameter is likely to be low (MacDonald 1977; Hatch 1987). In contrast, natal breeding dispersal is thought to be extremely high (Dunnet *et al* 1979). Although this estimate comes from a relatively old study, high levels of natal dispersal will strongly influence the subsequent population trajectory. This process requires consideration when building population models for this species.

Population trend in UK

The northern fulmar breeds throughout the British coastline, and national census data indicate that UK colonies rapidly increased between 1969 and 1998, stabilised between the late 1980s and early 2000s, and declined between 2000 and 2013 (JNCC 2014). The recent decline was attributed in part to a decline in discards from trawler fisheries. Regional differences in the rate of decline were identified in three discrete geographic areas; one surrounding the Irish Sea, one on the West coast of Scotland and one covering Orkney, Shetland and the East Coast of Scotland and Northern England (Cook & Robinson 2010).

Factors influencing survival rates

Local survival has been linked to the marine environment, and decrease at high values of the winter Northern Atlantic Oscillation (i.e. survival rates are lower following warmer sea temperatures and stronger winds, Grosbois & Thompson 2005). Survival does not appear to differ between the sexes (Dunnet & Ollason 1978a).

Factors influencing productivity rates

Local productivity rates have been linked to parental age and quality (Dunnet & Ollason 1978b; Ollason & Dunnet 1978; Lewis *et al* 2009), as well as lay dates (Hatch 1990). Productivity rates are also reported to be negatively influenced by a decrease in prey biomass (Grey *et al* 2003), and also decrease at high values of the winter Northern Atlantic Oscillation (i.e. productivity rates are lower following warmer sea temperatures and stronger winds, Thompson & Ollason 2001; Lewis *et al* 2009). There is no evidence that productivity depends on nesting density (Lewis *et al* 2009).

Knowledge gaps

Juvenile and immature survival rates are largely unknown, and high levels of natal dispersal will make this parameter difficult to deduce without multi-colony studies. However, this would also permit adult breeding dispersal to be quantified. The reported age of recruitment reflects a period when the population trajectory differed from the present day, and the incidence of missed breeding is not resolved. These parameters could potentially be examined using the long-term monitoring data from Orkney.

5.4 Gannet

5.4.1. Northern gannet (*Morus bassanus*)

Table 12. Recommended estimates of demographic rates for the northern gannet.

Demographic parameter	Mean	SD_p	SE_e	Data quality	Data representation
Age-specific survival					
Juvenile survival (0-1 year) [1]	0.424		0.007	6	3
Immature survival (1-2 years) [1]	0.829		0.004	6	3
Immature survival (2-3 years) [1]	0.891		0.003	6	3
Immature survival (3-4 years) [1]	0.895		0.003	6	3
Adult survival (≥5 year) [1]	0.919	0.042		6	3
Experience-specific productivity					
1st attempt [2]	0.553		0.019	3	3
2nd attempt [2]	0.645			1	3
3rd attempt [2]	0.770			1	3
≥4th attempt [2]	0.817			1	3
Regional-specific productivity					
East [1-5]	0.698	0.071		6	4
West [3-4]	0.710	0.105		6	4
National-average productivity [1-3, 5]					
	0.700	0.082		6	4
Age of recruitment [1-2]					
	5			5	3
Incidence of missed breeding					
	-			-	-
Breeding dispersal					
Natal	-			-	-
Adult [6]	Low			-	-

[1]Wanless *et al* 2006; [2]Nelson 1966; [3]Mavor *et al* 2008; [4]Cook & Robinson 2010; [5]Shaw 2010; [6]Nelson 1965.

Recommended demographic rates

The survival rates of northern gannet in the UK are relatively well resolved and recent estimates include a ring-recovery analysis that considers regional differences in survival (see accompanying material for northern gannet). The identified differences were attributed to the relatively low number of individuals ringed as adults at colonies other than Bass Rock (Wanless *et al* 2006). Furthermore, regional differences in population trends were largely attributed to local differences in breeding success (Cook & Robinson 2010). Therefore, it may be sufficient to employ a national average for survival when parameterising population models for this species. The study by Wanless *et al* (2006) examined ten colonies in the UK and excluded ring-recoveries of birds found dead or recaptured alive at the colony. Therefore, the estimates of survival given in Table 12 are unlikely to be biased by natal or adult dispersal.

Regional trends in productivity rates were identified in two discrete geographic areas (Cook & Robinson 2010; Appendix S1 for map). The productivity of northern gannets is monitored annually at one SMP Key Site: Fair Isle, and long-term productivity is given for seven additional UK localities by Nelson (1966), Wanless *et al* (2006), Mavor *et al* (2008) and Shaw *et al* (2010) (see accompanying material for northern gannet). The range of productivity values given in Table 12 were calculated using the regional groupings highlighted by Cook and Robinson (2010) to categorise the values specified by Nelson (1966), Wanless *et al* (2006), Mavor *et al* (2008) and Shaw *et al* (2010). The published information on experience-specific productivity rates is relatively old (Table 12). These values should be used in population models to guide the proportional change that can be expected with age, as opposed to absolute values of productivity.

Published information on the modal age of recruitment for northern gannet is available from two studies conducted 40 years apart (Nelson 1966; Wanless *et al* 2006). Although the rate of population growth has changed during this time (Wanless *et al* 2006; Murray *et al* 2006; JNCC 2014), the reported age of recruitment was similar in both studies (Table 12). Adult breeding dispersal is thought to be low (Nelson 1965) but has not been quantified at the population level. Information on natal breeding dispersal and the incidence of missed breeding is also lacking. It may be necessary to identify proxy species from outside of the UK to parameterise these processes in population models.

Population trend in UK

There are 21 gannet colonies around the British Isles, and the majority of these occur on remote offshore islands and stacks, with two on mainland cliffs. National census data indicate that the majority of colonies in the UK increased rapidly between 1969 and 2002 (JNCC 2014). More recently, this rate of increase is thought to have slowed (Wanless *et al* 2006; Murray *et al* 2006). The rate of increase varies consistently within two discrete geographic areas; the first covering Orkney, the West Coast of the United Kingdom and Ireland and the second covering Shetland and the East Coast of the United Kingdom (Cook & Robinson 2010).

Factors influencing survival rates

The drivers of survival for northern gannet have not been identified.

Factors influencing productivity rates

Local productivity rates have been linked to parental experience and increase sequentially between the first and the fourth breeding attempt (Nelson 1966).

Knowledge gaps

The juvenile and immature survival rates for northern gannets are poorly resolved, and breeding dispersal and the incidence of non-breeding are largely unknown. The incidence of missed breeding could potentially be explored using long-term monitoring data from Fair Isle, however the estimation of other demographic parameters would require intensive monitoring and multi-colony studies. Assuming that natal dispersal is low, an alternative approach would be to solve for juvenile survival rates based on other demographic parameters and population trend data collected from a regularly monitored site (e.g. Isle of May).

5.5 Grebe

5.5.1. Great crested grebe (*Podiceps cristatus*)

Table 13. Recommended estimates of demographic rates for the great crested grebe.

Demographic parameter	Mean	SD_p	SE_e	Data quality	Data representation
Age-specific survival					
Adult survival (≥4 years) [1-2]	0.725		0.028	4	2
Productivity [1,3]					
	1.275	0.035		4	3
Age of recruitment [1,4]					
	2			2	3
Incidence of missed breeding					
	-			-	-
Breeding dispersal					
Natal	-			-	-
Adult	-			-	-

[1]Abt & Konter 2009; [2]Fuchs 1982; [3]Prestt & Jefferies 1969; [4]Cramp & Simmons 1977.

Recommended demographic rates

There are no published estimates of survival for the great crested grebe in the UK. Published estimates of adult survival include two ring-recovery studies that examine data collected across extensive areas (Germany, Fuchs 1982, cited by Garthe & Hüppop 2004; and Europe, Abt & Konter 2009). Further matching of local population trends is recommended in order to assess the suitability of these estimates for modelling the UK population. At present there are not sufficient data to model colony-specific survival rates, therefore population models will have to assume that these estimates are representative more broadly. The estimate of immature survival reported by Abt and Konter (2009) is not listed in Table 13 because it extends past the age of recruitment (see accompanying material for great crested grebe). It may be necessary to identify proxy species to parameterise the survival rates of great crested grebe in population models.

Estimates of productivity are also limited to a few studies (see accompanying material for great crested grebe). These include an old estimate for the UK (Prestt & Jefferies 1969) and a more recent estimate for Europe (Abt & Konter 2009). Further matching of local population trends is recommended in order to assess the suitability of these estimates for modelling UK birds. A limited proportion of great crested grebes can also successfully fledge two broods within one breeding season (4.3%, Harrison & Hollom 1932; 4.6%, Simmons 1974). This mechanism should be considered when constructing population models for this species.

The age of first breeding is thought to occur at age 2 (Abt & Konter 2009; Cramp & Simmons 1977), although birds may pair and hold territory from their first year (Cramp & Simmons 1977). Information on natal and adult dispersal and the incidence of missed breeding is lacking. Consequently, it may be more appropriate to use estimates based on the diver species to parameterise these processes in population models. Further matching of local population trends and demographic processes is recommended in order to assess the suitability of other species for modelling great crested grebe.

Population trend in UK

In the UK, the great crested grebe is widely, although thinly distributed. They breed on large, shallow inland waters, and some move to coastal waters during winter. The UK population declined between 2002 and 2012 (Austin *et al* 2014).

Survival rates

The drivers of survival for great crested grebe have not been identified.

Productivity rates

Outside of the UK, local productivity rates are reported to be positively influenced by increased prey availability (Ulenaers & Dhont 1994; Ulfvens 1988), but are negatively influenced by predation levels associated with human disturbance (Keller 1989).

Knowledge gaps

Juvenile and immature survival rates are poorly resolved. Information on natal and adult dispersal, as well as missed breeding is also lacking. In the absence of a focused ringing or monitoring program in the UK it will be difficult to improve these estimates based on local data.

5.6 Cormorants

5.6.1 Great cormorant (*Phalacrocorax carbo*)

Table 14. Recommended estimates of demographic rates for the great cormorant.

Demographic parameter	Mean	SD_p	SE_e	Data quality	Data representation
Age-specific survival					
Juvenile survival (0-1 year) [1-2]	0.540		0.090	6	1
Adult survival (≥3 year) [1-2]	0.868	0.055		6	1
Experience-specific productivity					
1st attempt [3]	1.445			4	1
2nd attempt [3]	1.758			4	1
3rd attempt [3]	2.076			4	1
≥4th attempt [3]	2.274			4	1
Regional-specific productivity					
An Glas Eilean [4]	1.730	1.010		6	4
North Sutor [4]	2.060	0.581		6	4
Ballard Cliff [4]	1.340	0.201		6	4
Ynysoedd Gwylan [4]	2.970	0.759		6	4
Skomer [4]	1.590	0.710		6	4
South Solway "B" [4]	1.090	0.980		4	4
Will's Strand [4]	2.480	0.106		6	4
Caithness [4]	2.652	0.752		4	4
National-average productivity [4-5]					
	1.985	0.666		6	4
Age of recruitment [6-7]					
	3			5	1
Incidence of missed breeding					
	-			-	-
Breeding dispersal					
Natal [1-2]	0.240	0.127		6	1
Adult [2,7]	0.123	0.042		6	1

[1]Frederiksen & Bregnballe 2000a; [2]Hénaux *et al* 2007; [3]Bregnballe 2006; [4]Mavor *et al* 2008; [5]Budworth *et al* 2000; [6]Kortlandt 1942; [7]Frederiksen & Bregnballe 2000b.

Recommended demographic rates

The UK estimates of adult survival for *P. carbo* are relatively old and based on ring-recovery studies from Caithness, NE Scotland (Wernham & Peach 1999; Budworth *et al* 2000). The reported values are considerably lower than those identified for the other seabird species, possibly due to hunting activities changing levels of survival and emigration (Wernham & Peach 1999; see accompanying material for great cormorant). The breeding biology of *P. carbo* and *P. c. sinensis* (the continental race) is highly similar; therefore demographic rates may be interchangeable for these populations. The values of adult survival given in Table 14 are estimated from mark-recapture studies on the continental race (Frederiksen & Bregnballe 2000a; Hénaux *et al* 2007). In the UK, regional differences in population trends could not be wholly attributed to local differences in breeding success (Cook & Robinson 2010). However, at present there are not sufficient data to model colony-specific survival rates. It will be important to consider these regional differences in population trend as caveats of any population models developed. Furthermore, *P. c. sinensis* has increased rapidly in recent decades (Frederiksen & Bregnballe 2000a), and local population trajectories should be matched before extrapolating these rates across multiple colonies.

Cook and Robinson (2010) could not identify regional trends in the productivity of cormorants within the SMP and OSPAR regions. Consequently, it may be more suitable to use colony-specific rates of productivity to parameterise population models on this species (Table 14). The productivity of great cormorant (*P. carbo*) is monitored annually at one SMP Key Site: Skomer, and long-term productivity is given for six additional UK localities by Mavor *et al* (2008). Published information on experience-dependent productivity for great cormorant is not from the UK (Table 14; Bregnballe 2006). These values should be used in population models to guide the proportional change that can be expected with age, as opposed to absolute values of productivity.

The majority of birds breed for the first time at age 3 (range 2-5, *P. c. sinensis* - Kortlandt 1942; Frederiksen & Bregnballe 2000b). Although Budworth *et al* (2000) provide some information on inter-colony movements of *P. carbo* in Scotland, information on natal and adult dispersal is largely lacking for the UK. The natal and adult dispersal rates detailed in Table 14 are for the continental race (Frederiksen & Bregnballe 2000a; Hénaux *et al* 2007). Information on the incidence of missed breeding is lacking.

Population trend in UK

National census data indicate that great cormorant populations increased slightly between the 1970s and 2000, but have declined between 2000 and 2013 (JNCC 2014). Regional differences in population trends were identified in seven discrete geographic areas; Shetland, Orkney and North Scotland, East Scotland, East England, South East England, South and West England and East Ireland and West Scotland. Colonies in Orkney, North Scotland, East Ireland and West Scotland were declining, whilst the others were increasing (Cook & Robinson 2010). Furthermore, there is an increasing number of the continental race *P. c. sinensis* now overwintering in the UK (Newson *et al* 2005).

Factors influencing survival rates

Local survival rates of *P. c. sinensis* decrease when winter sea temperatures increase and this effect is greater at high population densities (Frederiksen & Bregnballe 2000a). Survival rates do not appear to differ between the sexes (Frederiksen & Bregnballe 2000a).

Factors influencing productivity rates

Local productivity is higher at inland colonies compared to coastal colonies (Newson *et al* 2005). Outside of the UK, local productivity rates of *P. c. sinensis* have been linked to concentrations of contaminants (Dirksen 1995), and in Denmark, productivity increases with parental experience up the fourth breeding attempt (Bregnballe 2006).

Knowledge gaps

There is a large quantity of data on colour-ringed cormorants that could be used to assess regional differences in survival. To resolve the incidence of missed breeding would require intensive monitoring of a marked population.

5.6.2 European shag (*Phalacrocorax aristotelis*)

Table 15. Recommended estimates of demographic rates for the European shag.

Demographic parameter	Mean	SD_p	SE_e	Data quality	Data representation
Age-specific survival					
Juvenile survival (0-1 year) [1]	0.513	0.246		6	4
Immature survival (1-2 year) [1]	0.737	0.181		6	4
Adult survival (≥3 year) [1]	0.858	0.194		6	4
Age-specific productivity					
1st attempt [2]	0.532	0.049		3	4
≥2nd attempt [2]	0.785	0.042		3	4
Regional-specific productivity					
West [3-4]	2.135	0.322		6	4
North [3-4]	1.277	0.456		6	4
East [3-5]	1.132	0.511		6	5
National-average productivity [3,5]					
	1.303	0.483		6	5
Age of recruitment [6]					
	2			4	2
Incidence of missed breeding					
	-			-	-
Breeding dispersal					
Natal [7]	0.049			2	2
Adult [8]	0.100			4	5

[1]Frederiksen *et al* 2008; [2]Daunt *et al* 1999; [3]Mavor *et al* 2008; [4]Cook & Robinson 2010; [5]Newell *et al* 2010; [6]Aebischer 1986; [7]Aebischer 1995; [8]Barlow *et al* 2013.

Recommended demographic rates

The UK estimates of survival for European shag are largely limited to the Isle of May, SE Scotland (see accompanying material for European shag). There is also a relatively old ring-recovery study from the Farne Islands, NE England (Coulson & White 1957). The rates reported from this older study were considerably lower than those reported from the Isle of May using mark-capture analysis. This is most likely due to the old-style rings becoming worn and illegible with time (Coulson & White 1957), and therefore these estimates are likely to be unreliable. The estimates of adult survival from the Isle of May were largely similar. The survival rates detailed in Table 15 are taken from Frederiksen *et al* (2008) because the time series considered spanned most of the other studies. Regional differences in population trends could be largely attributed to local differences in breeding success (Cook & Robinson 2010). Therefore, it may be sufficient to assume that the Isle of May estimate of survival are representative more broadly.

Regional trends in productivity rates were identified in three discrete geographic areas (Cook & Robinson 2010; Appendix S2 for map). The productivity of the European shag is monitored annually at three SMP Key Sites: Isle of May, Canna and Skomer, and long-term productivity estimates are given for 18 additional UK localities by Mavor *et al* (2008). The range of productivity values given in Table 15 were calculated using the regional groupings highlighted by Cook and Robinson (2010) to categorise the values specified by Mavor *et al* (2008) and Newell *et al* (2010). These estimates indicate large variability to occur within each region. Published information on experience-dependent productivity for European shag is relatively old (Table 15; Daunt *et al* 1999). These values should be used in population models to guide the proportional change that can be expected with age, as opposed to absolute values of productivity.

Published information on the age of recruitment for the European shag is relatively old. The European shag typically breeds for the first time at age 2, although individuals may also defer breeding until they are 3 or 4 years old (Aebischer 1986). National census data indicate that the trajectory of the UK population has changed since this estimate was published; from increasing to decreasing (JNCC 2014). Caution should be applied when using the reported age to model current population trends (Table 15). Adult breeding dispersal is thought to be low (Barlow *et al* 2013). Similarly, natal dispersal is also thought to be low, but may vary between colonies (Aebischer 1995). Information on the incidence of missed breeding is lacking. It may be appropriate to substitute this parameter with estimates from great cormorant (Table 14), although further matching of population trends is recommended.

Population trend in UK

The European shag is distributed throughout UK coastal waters. National census data indicate that the majority of colonies in the UK increased between 1969 and 1988, but numbers declined between the late 1980s and 2012 (JNCC 2014). The rate of decline varies within four discrete geographic areas; the first covering the West Coast of Scotland, the second covering the West Coast of England and Wales, the third covering the East Coast of Scotland and Orkney and the fourth covering Shetland (Cook & Robinson 2010).

Factors influencing survival rates

Decreased levels of survival have been linked to toxic poisoning (Coulson *et al* 1968), as well as offshore winds and rainfall (Frederiksen *et al* 2008).

Factors influencing productivity rates

Initially, productivity rates depend on parental age and quality (Daunt *et al* 1999). However, after the first attempt there is no evidence that productivity depends on age, nest density, pair status, and site or mate fidelity (Potts *et al* 1980). Large-scale non-breeding events have been associated with poor feeding conditions (Aebischer & Wanless 1992).

Knowledge gaps

The age of recruitment was estimated during a period where the population trajectory differed from the present day. Information on the average incidence of missed breeding is also lacking. Estimation of these demographic parameters could potentially be achieved using the long-term monitoring data from the Isle of May. There is also a large quantity of mark-recapture data collected as part of the British and Irish ringing scheme that could provide estimates of regional survival.

5.7. Skuas

5.7.1. Arctic skua (*Stercorarius parasiticus*)

Table 16. Recommended estimates of demographic rates for Arctic skua.

Demographic parameter	Mean	SD_p	SE_e	Data quality	Data representation
Age-specific survival					
Immature survival (0-4 years) [1]	0.346			2	3
Adult survival [2]	0.910			4	3
Regional-specific productivity					
North – Shetland [3-4]	0.465	0.427		6	4
South – Orkney [3-4]	0.620	0.412		6	4
National-average productivity [3]					
	0.487	0.436		6	4
Age of recruitment [5]					
	4			3	2
Incidence of missed breeding [6]					
	0.030			4	4
Breeding dispersal					
Natal	-			-	-
Adult [2]	Low			-	-

[1]O'Donald 1983; [2]Phillips & Furness 1998; [3]Mavor *et al* 2008; [4]Cook & Robinson 2010; [5]O'Donald & Davis 1975; [6]Catry *et al* 1998.

Recommended demographic rates

The UK estimates of survival for Arctic skua are limited to Shetland (Phillips & Furness 1998, see accompanying material for Arctic skua). The resighting rate in the study by Phillips and Furness (1998) was considered to be high, therefore the bias on true survival rate may be low (Table 16). In the UK, regional differences in population trends could be largely attributed to local differences in breeding success (Cook & Robinson 2010). Consequently, it may be sufficient to assume that the Shetland estimate is representative more broadly (Phillips & Furness 1998; Table 16). Immature survival rates have been estimated based on relatively old productivity, age of maturity and population size data (O'Donald 1983). It may be appropriate to substitute these estimates of survival with those identified for great skua (Table 17), although further matching of local population trends is recommended in order to assess suitability.

Regional trends in productivity rates were identified in two discrete geographic areas (Cook & Robinson 2010; Appendix S3 for map). The productivity of Arctic skua is monitored annually at one SMP Key Site: Fair Isle, and long-term productivity estimates are given for six additional UK localities by Mavor *et al* (2008). The range of productivity values given in Table 16 were calculated using the regional groupings highlighted by Cook and Robinson (2010) to categorise the values specified by Mavor *et al* (2008). These estimates indicate large variability to occur within each region.

Published information on the age of recruitment for Arctic skua is relatively old (O'Donald & Davis 1975), and national census data indicate that the trajectory of the UK population has changed since this estimate was published; from increasing to decreasing (JNCC 2014). Therefore caution should be applied when using the reported age to model current population trends (Table 16). Published information on natal breeding dispersal is conflicting and therefore neither study is included in Table 16 (O'Donald & Davis 1975; Phillips & Furness 1998; see accompanying material for Arctic skua). Adult breeding dispersal has not been quantified, but high mate and site-fidelity indicates that it is low (Phillips & Furness 1998). Local incidence of missed breeding is also low (Catry *et al* 1998).

Population trend in UK

The Arctic skua is a UK BAP species and a red-list Bird of Conservation Concern. The breeding range is restricted to north and west Scotland, and national census data indicate that numbers increased rapidly between 1969 and 1988, but declined between the late 1980s and 2013, possibly more than any other seabird species in the UK (JNCC 2014). Regional differences in population trends were identified in three discrete geographic areas; Shetland, Orkney and North Scotland (Cook & Robinson 2010).

Factors influencing survival rates

Local survival rates are reported to be negatively influenced by decreased prey availability (Davis 2005).

Factors influencing productivity rates

Local productivity rates have been linked to predation from great skuas (Phillips *et al* 1998). They are also relatively low for first time breeders, stabilising from the second breeding attempt (Davis 1976).

Knowledge gaps

The age of recruitment was estimated during a period when the population trajectory differed from the present day. Re-estimation of this parameter could potentially be achieved using the long-term monitoring data from Shetland. Juvenile survival rates are also poorly resolved and rates of adult and natal breeding dispersal are largely unknown. To estimate these demographic parameters would require intensive monitoring and multi-colony studies. Assuming a low rate of natal dispersal, an alternative approach would be to solve for juvenile survival rates based on more recent demographic parameters and population trend data than considered by O’Donald (1983).

5.7.2. Great skua (*Stercorarius skua*)

Table 17. Recommended estimates of demographic rates for the great skua.

Demographic parameter	Mean	SD_p	SE_e	Data quality	Data representation
Age-specific survival					
Immature survival (0-5 years) [1]	0.730			4	4
Adult survival [1-2]	0.882	0.038		6	4
Colony-specific productivity					
Hermaness [3]	0.800	0.060		6	4
Fetlar [3]	0.740	0.100		6	4
Noss [3]	0.500	0.060		6	4
Mousa [3]	0.670	0.100		6	4
Fair Isle [3]	0.650	0.090		6	4
Foula [3]	0.550	0.080		6	4
North Hill, Papa Westray [3]	0.640	0.080		6	4
Stourdale, Hoy [3]	0.330	0.110		4	4
Handa [3]	0.880	0.200		6	4
National-average productivity [3]					
	0.651	0.308		6	4
Age of recruitment [4]					
	7			2	3
Incidence of missed breeding [5]					
	0.089	0.040		6	4
Breeding dispersal					
Natal [6]	0.015			4	4
Adult [4,7-8]	Low			-	-

[1]Ratcliffe *et al* 2002; [2]Votier *et al* 2004; [3]Mavor *et al* 2008; [4]Furness 1987; [5]Catry *et al* 1998; [6]Klomp & Furness 1992a; [7]Catry & Furness 1997; [8]Catry *et al* 1998.

Recommended demographic rates
The UK estimates of survival for great skua are largely limited to Shetland. However, long term mark-recapture studies are also conducted on Handa and St Kilda. Recently published studies from Foula (Ratcliffe *et al* 2002) and Hermaness (Votier *et al* 2004) reported similar estimates of adult survival. Regional variation in the survival rates of great skuas has not been investigated; therefore the estimate given in Table 17 assumes that the Shetland estimates are representative more broadly. Juvenile and immature survival rates are poorly resolved because few recoveries or resightings are made for these age-classes. The rate provided in Table 17 reflects the total survival between fledging and recruitment (Ratcliffe *et al* 2002).

Regional trends in productivity have been examined using a short-term time series (n=1 year, Phillips *et al* 1997) and therefore would benefit from further analysis (e.g. Cook & Robinson 2010). Consequently, it may be more suitable to use colony-specific rates of productivity to parameterise population models on this species (Table 17). The productivity of great skua is monitored annually at one SMP Key Site: Fair Isle, and long-term productivity is given for eight additional UK localities by Mavor *et al* (2008).

Published information on the modal age of recruitment for great skua is relatively old and refers to a time when the population was increasing at a faster rate compared to the current trends (Furness 1987; JNCC 2014). The estimate is also considerably older than the estimate for Arctic skua (Table 16). Further consideration is therefore required when parameterising this value in population models. Local incidence of missed breeding is low (Catry *et al* 1998). Breeding dispersal of adults is also very low and any movements within a colony are usually short (Furness *et al* 1987; Catry & Furness 1997; Catry *et al* 1998). Natal breeding dispersal is also reported to be low (Klomp & Furness 1992a).

Population trend in UK
The breeding range of the great skua is restricted to Scotland and concentrated in the Northern Isles. National census data indicate that numbers increased rapidly between 1969 and 1988, this rate of increase then slowed between 1989 and 2002 (JNCC 2014). Regional differences in population trend were noted by Phillips *et al* (1997).

Factors influencing survival rates
The drivers of survival for great skua have not been identified.

Factors influencing productivity rates
Local productivity rates are reported to be negatively influenced by a decrease in prey biomass (Hamer *et al* 1991). The incidence of missed breeding is also higher in years with reduced food availability (Hamer *et al* 1991). There is no evidence that productivity depends on age (Furness 1984; Ratcliffe *et al* 1998).

Knowledge gaps
Information on juvenile and immature survival rates is lacking. Given the low rates of natal dispersal reported (Klomp & Furness 1992a), existing ring-recovery and sighting data should be sufficient to estimate these rates. An alternative approach would be to solve for juvenile survival rates based on other demographic parameters and population trend data collected from a regularly monitored site. Regional variation in survival and productivity rates could also be examined using long-term monitoring data. The age of recruitment would also benefit from re-evaluation using data from key monitoring sites.

5.8. Gulls

5.8.1. Black-legged kittiwake (*Rissa tridactyla*)

Table 18. Recommended estimates of demographic rates for the black-legged kittiwake.

Demographic parameter	Mean	SD_p	SE_e	Data quality	Data representation
Age-specific survival					
Juvenile survival (0-1 year) [1]	0.790			1	3
Adult survival (≥2 year) [2-4]	0.854	0.051		6	5
Experience-specific productivity					
1st attempt [5-7]	0.898	0.206		6	4
≥2nd attempt [5-7]	1.379	0.185		6	4
Regional-specific productivity					
Shetland [8-10]	0.408	0.477		6	4
East [8-9,11]	0.819	0.332		6	4
West [8-9]	0.643	0.313		6	4
National-average productivity [8,10-11]					
	0.690	0.296		6	5
Age of recruitment [7,12]					
	4			6	4
Incidence of missed breeding					
Increasing population [13]	0.180	0.188		4	2
Decreasing population [13]	0.208	0.207		4	2
Breeding dispersal					
Natal [12]	0.890			3	3
Adult					
Increasing population [13]	0.012	0.018		6	2
Decreasing population [13]	0.062	0.045		6	2

[1]Coulson & White 1959; [2]Frederiksen *et al* 2004a; [3]Oro & Furness 2002; [4]Taylor *et al* 2010; [5]Coulson & White 1958; [6]Porter 1990; [7]Wooller & Coulson 1977; [8]Mavor *et al* 2008; [9]Frederiksen *et al* 2005; [10]Shaw *et al* 2010; [11]Newell *et al* 2010; [12]Porter & Coulson 1987; [13]Danchin & Monnat 1992.

Recommended demographic rates

The UK estimates of survival for black-legged kittiwake are largely limited to Foula, the Isle of May and Skomer. There are also some older estimates of adult survival from North Shields, NE England (Coulson & Wooller 1976; Aebischer & Coulson 1990; Coulson & Strowger 1999). The most recent colony-specific estimates of survival are from Oro and Furness 2002 (Foula, Shetland), Frederiksen *et al* 2004a (Isle of May, SE Scotland) and Taylor *et al* 2010 (Skomer, Wales). Regional differences in population trends could be largely attributed to local differences in breeding success (Cook & Robinson 2010). Therefore, it may be sufficient to employ a national average for survival when parameterising population models. Juvenile and immature survival rates are poorly resolved because few recoveries or resightings are made for these age-classes. The estimate for juvenile survival in Table 18 is taken from a short-term ring-recovery study from the 1950s (Coulson & White 1959). National census data indicate that the trajectory of the UK population has changed since this study; from increasing to decreasing (JNCC 2014). Therefore caution should be applied when using this rate to model current population trends, and it may be more appropriate to use estimates based on other gull species (Tables 18-24).

Regional trends in productivity rates were identified in three discrete geographic areas by Cook and Robinson (2010) (Appendix S4 for map), and six discrete geographic areas by Frederiksen *et al* (2005). The productivity of kittiwakes is monitored annually at all four SMP Key Sites: Isle of May, Canna, Fair Isle and Skomer, and long-term productivity estimates are given for 41 additional UK localities by Mavor *et al* (2008). The range of productivity values given in Table 18 were calculated using the regional groupings highlighted by Cook and Robinson (2010) to categorise the values specified by Mavor *et al* (2008). The published information on experience-specific productivity for kittiwake is relatively old (Table 18; Coulson & White 1958; Wooller & Coulson 1977; Porter 1990). These values should be used in population models to guide the proportional change that can be expected with age, as opposed to absolute values of productivity.

Published information on the age of recruitment for black-legged kittiwake is relatively old (Wooller & Coulson 1977), and national census data indicate that the trajectory of the UK population has changed since this study; from increasing to decreasing (JNCC 2014). Therefore caution should be applied when using the reported age to model current population trends (Table 18). The majority of birds breed for the first time at age 4, although a small proportion will start earlier (from age 2), or later (up to age 7; Wooller & Coulson 1977). The incidence of missed breeding in the black-legged kittiwakes has been quantified for two UK colonies; North Shields, NE England (Coulson & White 1956; Coulson & Nève de Mévergnies 1992) and Shetland (Richardson 1985). However, these studies include all non-breeding birds observed in the colony, such as sub-adults, and therefore estimation of non-breeding amongst established breeding pairs is likely to be overestimated. Danchin and Monnat (1992) demonstrate that the incidence of missed breeding in France may reflect the colony trajectory. The values reported in Table 18 are given for colonies with increasing and decreasing population trajectories (Danchin & Monnat 1992). Published estimates of breeding dispersal for adult black-legged kittiwakes in the UK are lacking, although it is expected to be low (Coulson & Strowger 1999; Coulson & Wooller 1976). For populations breeding in France adult dispersal is low, but may vary depending on the local population trajectory and extrinsic conditions (Danchin & Monnat 1992). In contrast, natal dispersal is thought to be high, more so in females (Porter & Coulson 1987; Coulson & Nève de Mévergnies 1992). Although this estimate comes from a relatively old study, high levels of natal dispersal will strongly influence the subsequent population trajectory. This process will require consideration when building population models for this species. Dispersal patterns are also examined by McCoy *et al* (2005) using population genetics.

Population trend in UK

In the UK, the largest black-legged kittiwake colonies are found along the North Sea coasts, around Orkney and Shetland, and off north-west Scotland. National census data indicate that UK colonies increased between 1969 and 1988, but have declined rapidly between the late 1980s and 2013 (JNCC 2014). The national trend was attributed in part to reduced productivity rates (Taylor *et al* 2010). Regional differences in population trends were identified in six discrete geographic areas; Orkney and Shetland, East Scotland and North East England, South East England, South West England, Wales and North East Ireland and West Scotland (Cook & Robinson 2010).

Factors influencing survival rates

Local survival is reported to be positively influenced by prey availability, but negatively influenced by predation from great skuas (Oro & Furness 2002) and winter sea surface temperatures (Frederiksen *et al* 2004a). Local survival was lower when the sand eel fisheries were active (Frederiksen *et al* 2004a), and was significantly influenced by nesting density (Coulson & Wooller 1976), nest position (Aebischer & Coulson 1990) and algal blooms (Coulson & Strowger 1999). Survival rates do not appear to differ between the sexes (Aebischer & Coulson 1990). For older birds, survival rates apparently decrease, but this could be an artefact of territory-loss, or in earlier studies, ring-loss causing heterogeneity in recapture rates (Aebischer & Coulson 1990; Frederiksen *et al* 2004b).

Factors influencing productivity rates

Local productivity rates are reported to be negatively influenced by a decrease in prey biomass (Hamer *et al* 1993; Harris & Wanless 1990; Lewis *et al* 2001; Oro & Furness 2002; Frederiksen *et al* 2005). Productivity may also decrease following an increase in fishing effort (Lewis *et al* 2001; Frederiksen *et al* 2004a), and increased winter sea surface temperatures (Frederiksen *et al* 2004a). Outside of the UK, studies have reported that local productivity can also be reduced by predation from species such as ravens *Corvus corax* (Danchin & Monnat 1992). There is no evidence that productivity depends on local population size (Frederiksen *et al* 2005). In North Shields, NE England, the productivity of first time breeders was relatively low, stabilising from the second (Wooller & Coulson 1977; Porter 1990) or third breeding attempt (Coulson & White 1958).

Knowledge gaps

The age of recruitment was estimated during a period when the population trajectory differed from the present day. Re-estimation could potentially be achieved using the long-term monitoring data from Foula, Isle of May or Skomer. Juvenile and immature survival rates are largely unknown. High levels of natal dispersal will make this parameter difficult to deduce without multi-colony studies, which would also permit the breeding dispersal of adults to be quantified.

5.8.2. Little gull (*Larus minutus*)

Table 19. Recommended estimates of demographic rates for the little gull.

Demographic parameter	Mean	SD_p	SE_e	Data quality	Data representation
Age-specific survival					
Adult survival [1]	0.800			1	1
Productivity					
	-			-	-
Age of recruitment [2]					
	2-3			4	1
Incidence of missed breeding					
	-			-	-
Breeding dispersal					
Natal	-			-	-
Adult	-			-	-

[1]Garthe & Hüppop 2004; [2]Berg 1937, cited by Cramp & Simmons 1983.

Recommended demographic rates
In the absence of a focused UK program on little gulls, local survival and productivity rates are largely unknown. Published estimates of adult survival are based on similar species (Garthe & Hüppop 2004; Table 19), and published estimates for productivity are lacking completely. Population models should be constructed based on the demographic parameters reported for the other gull species. However, without detailed knowledge on the population trend of this species it will be difficult to assess the suitability of these estimates.

Published information on the age of recruitment for the little gull is relatively old and not based on UK data (Berg 1937, cited by Cramp & Simmons 1983). It was not possible to verify the corresponding population trend for this estimate; however the age is similar to the other estimates reported for small gull species (Tables 19-21). Information on natal and adult dispersal and the incidence of missed breeding is lacking. It may be appropriate to use estimates of dispersal based on black-headed or common gull for population models (Tables 20-21).

Population trend in UK
The little gull is a passage migrant occurring in the UK in both the autumn and the spring (Stone *et al* 1995). During this time they occur mainly offshore, only coming close to shore during strong gales (Wernham *et al* 2002). Therefore exact population trends are unknown.

Factors influencing survival rates
The drivers of survival for little gull have not been identified.

Factors influencing productivity rates
The drivers of productivity for little gull have not been identified.

Knowledge gaps
The demographic parameters of the little gull are largely unknown. Although a small number of birds have been ringed in the UK, more intensive efforts at the breeding grounds would be needed to resolve the local demography of this species.

5.8.3. Black-headed gull (*Chroicocephalus ridibundus*)

Table 20. Recommended estimates of demographic rates for the black-headed gull.

Demographic parameter	Mean	SD_p	SE_e	Data quality	Data representation
Age-specific survival					
Adult survival (≥2 year) [1-2]	0.825	0.028		6	2
Age-specific productivity					
≤4 breeding attempts [3]	0.683	0.105		6	1
>4 breeding attempts [3]	1.643	0.302		6	1
Colony-specific productivity					
Eilean Inshaig [4]	0.560	0.564		6	4
Sgeir na Caillich [4]	0.490	0.312		6	4
Eilean Ruisg [4]	1.010	0.503		6	4
Airds Islet [4]	0.820	0.791		6	4
Eilean nan Gabhar [4]	0.230	0.325		4	4
Fetla [4]	0.260	0.316		6	4
Egilsay [4]	0.230	0.294		6	4
Shapinsay [4]	0.500	0.380		4	4
Coquet Island [4]	1.230	0.379		6	4
Havergate [4]	0.250	0.014		4	4
Blakeney Point [4]	0.320	0.200		4	4
Scolt Head [4]	0.990	0.191		4	4
Rye Harbour [4]	0.840	0.589		6	4
Langstone Harbour [4]	0.650	0.416		6	4
Hayling Oysterbeds [4]	0.500	0.468		4	4
Brownsea Island [4]	0.510	0.268		6	4
Rockcliffe Marsh [4]	0.200	0.201		6	4
National-average productivity [4]					
	0.625	0.365		6	4
Age of recruitment [3,5]					
	2			6	1
Incidence of missed breeding					
	-			-	-
Breeding dispersal					
Natal	-			-	-
Adult [6]	0.167	0.086		6	1

[1] Majoor *et al* 2005; [2] Péron *et al* 2010; [3] Lebreton *et al* 1990; [4] Mavor *et al* 2008; [5] Prevot-Julliard *et al* 2001; [6] Grosbois & Tavecchia 2003.

Recommended demographic rates

Although a large number of black-headed gulls have been ringed in the UK, ringing often takes place in the winter and away from breeding colonies. These birds may originate from several different breeding colonies, and therefore colony-specific estimates of survival rates are lacking. In France and the Netherlands local adult survival is thought to be relatively constant, although the two estimates differ (Majoor *et al* 2005; Péron *et al* 2010; see accompanying material for black-headed gull). In the absence of UK studies the estimate of adult survival given in Table 20 combined the studies from France and the Netherlands. The local population trends could not be identified, and consequently further consideration is recommended in order to assess the suitability of these estimates. Immature and juvenile survival is poorly resolved because few rings are recovered for this age-class (Robinson & Ratcliffe 2010).

Regional consistency in productivity rates has not been investigated for the black-headed gull (e.g. Cook & Robinson 2010). Consequently, it may be more suitable to use colony-specific rates of productivity to parameterise population models on this species. Long-term productivity is given for 17 UK localities by Mavor *et al* (2008) and there is a marked degree of variation in the reported rates (Table 20). Published information on experience-specific productivity is relatively old and not from the UK (Table 20; Lebreton *et al* 1990). Consequently, these values should be used in population models to guide the proportional change that can be expected with age, as opposed to absolute values of productivity.

Published information on the age of recruitment for black-headed gull is relatively old and not based on UK data (Lebreton *et al* 1990). It was not possible to verify the corresponding population trend for this estimate, however the age is similar to other small gulls (Tables 19-21). Information on the incidence of non-breeding is lacking, but it may be possible to substitute this parameter with values based on the black-legged kittiwake (Table 18). Adult breeding dispersal is thought to be high (Grosbois & Tavecchia 2003), and therefore natal dispersal is also likely to be high.

Population trend in UK

The majority of the UK black-headed gull population are resident throughout the year, and in winter the population are inflated by individuals from Northern and Eastern Europe (Wernham *et al* 2002). National census data indicate that the population was relatively stable between 1985 and 2002, increasing between 2000 and 2013 (JNCC 2014). Regional patterns of change emerged between the 1980s and early 2000s, possibly reflecting the redistribution of birds from coastal to inland colonies (Mitchell *et al* 2004).

Factors influencing survival rates

In France and the Netherlands adult survival rates show little year-to-year variation (Majoor *et al* 2005; Péron *et al* 2010).

Factors influencing productivity rates

Outside of the UK, local productivity rates increase with increased rainfall because this decreases brood predation (Thyen & Becker 2006). The nesting habitat can also be significantly influential (Patterson 1965). The productivity of birds in France was lower during the first three breeding attempts (Lebreton *et al* 1990).

Knowledge gaps

Information on juvenile survival rates, natal dispersal and the incidence of non-breeding is lacking. High levels of natal dispersal will make juvenile survival rates difficult to deduce without multi-colony studies.

5.8.4. Common (or mew) gull (*Larus canus*)

Table 21. Recommended estimates of demographic rates for the common gull.

Demographic parameter	Mean	SD_p	SE_e	Data quality	Data representation
Age-specific survival					
Juvenile survival (0-1 year) [1]	0.410			2	1
Immature survival (1-2 years) [1]	0.710			2	1
Adult survival (≥3 year) [2]	0.828	0.050		6	1
Colony specific productivity					
Handa [3]	1.060			4	4
Fair Isle [4]	0.486	0.349		6	5
Mill Dam, Shapinsay [3]	0.010			4	4
Whaness [3]	0.550	0.485		4	4
Sandy Loch [3]	0.950			4	4
Nigg [3]	0.850	0.474		6	4
Bonar Bridge [3]	0.590			4	4
Blakeney Point [3]	0.000	0.000		4	4
Scolt Head [3]	0.200	0.346		4	4
National-average productivity [3-4]					
	0.543	0.391		6	5
Age of recruitment [1]					
	3			4	1
Incidence of missed breeding					
	-			-	-
Breeding dispersal					
Natal					
Male [2]	0.500			4	1
Female [2]	0.900			4	1
Adult [2]	<0.030			4	1

[1] Rattiste & Lilleleht 1987; [2] Rattiste 2004; [3] Mavor *et al* 2008; [4] Shaw *et al* 2010.

Recommended demographic rates

Although a large number of common gulls have been ringed in the UK, ringing often takes place in the winter and away from breeding colonies. These birds may originate from several different breeding colonies, and therefore colony-specific estimates of survival rates are lacking. Published estimates are largely based on mark-recapture studies from Estonia (Rattiste & Lilleleht 1987, 1995; Rattiste 2004). There is also a relatively old ring-recovery study from Denmark (Halling Sørensen 1977). This estimate is considerably lower than those estimated using mark-capture analysis and is therefore likely to be unreliable. Although the values of survival given in Table 21 reflect the most recent estimates, the estimate of juvenile survival is still relatively old (see accompanying material for common gull). The local population trends for these studies could not be identified, consequently further consideration is recommended in order to assess the suitability of these estimates for modelling UK common gulls.

Regional trends in productivity have not been investigated for this species (e.g. Cook & Robinson 2010). Consequently, it may be more suitable to use colony-specific rates of productivity to parameterise population models on this species. The productivity of the common gull is monitored annually at one SMP Key Sites: Fair Isle, and long-term productivity is given is given for 16 additional UK localities by Mavor *et al* (2008). There is a marked degree of variation in the reported rates (Table 21).

Published information on the age of recruitment for common gull is relatively old and not based on UK data (Rattiste & Lilleleht 1987). It was not possible to verify the corresponding population trend for this estimate, however the age is similar to other small gulls (Tables 19-21). Information on the incidence of non-breeding is lacking but it may be possible to substitute this parameter with values based on the black-legged kittiwake (Table 18). Outside of the UK, adult breeding dispersal is thought to be low (Rattiste 2004), although pairs with less breeding experience are more likely to divorce and disperse after 1 year (*c.* 17–21%), compared to birds with longer breeding experience (*c.* 8–12%). Natal dispersal is high, more so for females (Rattiste 2004). The estimate of natal dispersal comes from a relatively old study, however high levels will strongly influence the subsequent population trajectory. This process requires consideration when building population models for this species.

Population trend in UK

In the UK, breeding populations of common gulls are concentrated in Scotland and Northern Ireland. They breed on coasts and inland sites, dispersing during the winter. Exact population trends are difficult to verify because different methods were used to conduct censuses. Numbers are thought to have increased between 1969 and 2002 (JNCC 2014).

Factors influencing survival rates

Survival of first time breeders is lower during cold winters compared to normal and warm winters (Rattiste & Lilleleht 1995). For older birds (>5 breeding years), survival of males appeared to be slightly higher than for females, but this could reflect lower site fidelity in females (Rattiste & Lilleleht 1995).

Factors influencing productivity rates

Rates of productivity increase with experience (Rattiste 2004), and local productivity rates have been reported to decrease due to predation from aerial predators (Kilpi 1995).

Knowledge gaps

The survival rates of UK common gulls are poorly resolved, and information on the incidence of non-breeding is lacking. In the absence of a colony-specific ringing program in the UK it will be difficult to improve these estimates using local data. High levels of natal dispersal will make juvenile survival rates difficult to deduce without multi-colony studies.

5.8.5. Lesser black-backed gull (*Larus fuscus*)

Table 22. Recommended estimates of demographic rates for the lesser-black backed gull.

Demographic parameter	Mean	SD_p	SE_e	Data quality	Data representation
Age-specific survival					
Juvenile survival (0-1 year) [1]	0.820			2	2
Adult survival (≥2 year) [2-3]	0.885	0.022		6	4
Colony-specific productivity					
North Hill [4]	0.800	0.566		4	3
Isle of May [4]	0.880	0.289		6	3
Skomer [4]	0.270	0.165		6	3
Skokholm [4]	0.170	0.066		6	3
Bardsey [4]	0.720	0.221		6	3
National-average productivity [4]					
	0.530	0.325		6	3
Age of recruitment [1]					
	5			4	2
Incidence of missed breeding [5-6]					
	0.337	0.065		4	2
Breeding dispersal					
Natal [7]	0.470			2	2
Adult	-			-	-

[1]Harris 1970; [2]Wanless *et al* 1996; [3]Taylor *et al* 2010; [4]Mavor *et al* 2008; [5]O'Connell *et al* 1997; [6]Calladine & Harris 1997; [7]Coulson 1991.

Recommended demographic rates

The UK estimates of survival for lesser black-backed gull are largely limited to the Isle of May and Skomer. Colony specific estimates of survival appear to differ (see accompanying material for lesser black-backed gull), however regional differences have not been reliably tested. Therefore the estimate of adult survival reported in Table 22 combined the most recent colony-specific studies (Wanless *et al* 1996; Taylor *et al* 2010). Further analysis of regional trends may support the use of colony specific survival rates when parameterising population models for this species. Juvenile and immature survival rates are more difficult to resolve due to lower recovery rates of these age classes. Furthermore, published estimates are relatively old and may not reflect current population trends. It may be more appropriate to use survival estimates based on herring gulls for this age class (Table 23). Further matching of local population trends is recommended in order to assess the suitability of these estimates.

Regional trends in productivity rates have not been investigated for this species (e.g. Cook & Robinson 2010). Consequently, it may be more suitable to use colony-specific rates of productivity to parameterise population models on this species (Table 22). The productivity of the lesser black-backed gull is monitored annually at one SMP Key Site: Skomer, and long-term productivity estimates are given for four additional UK localities by Mavor *et al* (2008).

Published information on the age of recruitment for lesser black-backed gull is relatively old (Harris 1970), and national census data indicate that the trajectory of the UK population has changed since this study; from increasing to decreasing (JNCC 2014). Therefore caution should be applied when using this age to model current population trends (Table 22). The majority of birds breed for the first time at age 5, although a small proportion also breed for the first time at age 3 and 4, and a few individuals may delay breeding until age 6 (Harris 1970). Local incidence of missed breeding is relatively high (Calladine & Harris 1997; O'Connell *et al* 1997). Birds are thought to change breeding locations following failed breeding attempts (Perrins & Smith 2000) and disturbance (O'Connell 1995), indicating that adult breeding dispersal may be high (Ross-Smith *et al* 2014). Natal dispersal is also thought to be high (Coulson 1991).

Population trend in UK

The lesser black-backed gull breeds throughout the UK, and national census data indicate that numbers increased between the late 1960s and early 2000s, but declined during the last decade (JNCC 2014). This trend was attributed in part to a decline in fishery discards (Furness *et al* 1992), and may also reflect the redistribution of birds from coastal to inland colonies (Ross-Smith *et al* 2014).

Factors influencing survival rates

Survival rates do not appear to differ between the sexes (Wanless *et al* 1996), however Bradbury and Griffiths (1999) report a skewed sex-ratio at fledging that could result in more females recruiting to the breeding population.

Factors influencing productivity rates

Local productivity rates have been linked to nesting habitat (Calladine 1997; Kim & Monaghan 2005b), parental condition (Nager *et al* 2000) and fishery discards (Oro 1996).

Knowledge gaps

Information on juvenile survival rates is lacking. High levels of natal dispersal will make this parameter difficult to deduce without multi-colony studies. As with herring gull, a good number of rings are recovered for lesser black-backed gulls. Although this provides a means to estimate annual survival rates, this analysis has yet to be done. There are also a number of colour-marking studies which would provide regional estimates of survival, although these data would need to be collated first, as recommended by Ross-Smith *et al* (2014). The data representation scores on the incidence of missed breeding could also potentially be improved using these data.

5.8.6. Herring gull (*Larus argentatus*)

Table 23. Recommended estimates of demographic rates for the herring gull.

Demographic parameter	Mean	SD_p	SE_e	Data quality	Data representation
Age-specific survival					
Juvenile survival (0-1 year) [1-2]	0.798	0.092		6	3
Adult survival (≥2 year) [3-4]	0.834	0.034		6	4
Age-specific productivity					
Age 4 [1]	0.000			1	3
Age 5 [1]	0.510			1	3
Age 6 [1]	0.710			1	3
>Age 6 [1]	0.990			1	3
Colony-specific productivity					
Canna [5]	0.740	0.640		6	3
Noss [5]	0.840	0.396		6	3
Isle of May [5]	0.980	0.330		6	3
Skomer [5]	0.770	0.298		6	3
Skokholm [5]	0.700	0.398		6	3
Bardsey [5]	1.220	0.316		6	3
Ynysoedd Gwylan [5]	1.150	0.185		6	3
Walney Island [6]	1.993			4	4
Farne Islands [7]	1.833			4	4
National-average productivity [5-7]					
	0.920	0.477		6	4
Age of recruitment [8]					
	5			4	3
Incidence of missed breeding [9]					
	0.350	0.028		4	3
Dispersal					
Natal [1,10]	0.629	0.156		5	3
Adult	-	-		-	-

[1]Chabrzyk & Coulson 1976; [2]Harris 1970; [3]Wanless *et al* 1996; [4]Taylor *et al* 2010; [5]Mavor *et al* 2008; [6]Kim & Monaghan 2005a; [7]Swann 2010; [8]Harris 1970; [9]Calladine & Harris 1997; [10]Coulson 1991.

Recommended demographic rates

The UK estimates of survival for herring gull are largely limited to the Isle of May and Skomer. Older estimates of adult survival include mark-recapture and ring-recovery studies from Lizard Point, Skokholm and Grassholm (Harris 1970; Chabrzyk & Coulson 1976; Coulson & Butterfield 1986). Regional differences in population trends could not be wholly attributed to local differences in breeding success (Cook & Robinson 2010). Regional variation in survival rates is indicated from the colony-specific estimates (see accompanying material for herring gull); however this difference has not been reliably tested. Therefore the estimate of adult survival reported in Table 23 combined estimates from the most recent colony-specific studies (Wanless *et al* 1996; Taylor *et al* 2010). Further analysis of regional trends may support the use of colony specific survival rates when parameterising population models for this species. Estimates of juvenile and immature survival rates are based on relatively old studies (Table 23; see accompanying material for herring gull). It may be more appropriate to use values based on other large gull species to parameterise the survival rates of these age classes.

Cook and Robinson (2010) could not identify consistent trends in the productivity of herring gulls within the SMP and OSPAR regions. Consequently, it may be more suitable to use colony-specific rates of productivity to parameterise population models on this species (Table 23). The productivity of the herring gull is monitored annually at two SMP Key Sites: Canna and Skomer, and long-term productivity estimates are given for five additional UK localities by Mavor *et al* (2008).

Published information on the modal age of recruitment for the herring gull is available from two studies that were conducted 40 years apart (Harris 1970; Ross-Smith *et al* 2014). The rate of population decline continued during this time (JNCC 2014), and both studies reported a similar age of recruitment (see accompanying material for herring gull). The majority of birds are thought to breed for the first time at age 5, although a small proportion will breed at age 4, and a few individuals may delay until age 6 or 7 (Harris 1970; Ross-Smith *et al* 2014). Local incidence of missed breeding has been estimated using one year of data that is unlikely to be representative more broadly (Calladine & Harris 1997). Information on adult breeding dispersal is lacking, however short-term studies suggest high levels of natal dispersal that may differ between colonies (Chabrzyk & Coulson 1976; Duncan & Monaghan 1977; Coulson 1991). Although this estimate comes from a relatively old study, high levels of natal dispersal will strongly influence the subsequent population trajectory. This process requires consideration when building population models for this species.

Population trend in UK

The herring gull is a UK BAP species and a red-list Bird of Conservation Concern. They are widely distributed around the coasts of the British Isles, and national census data indicate that the UK population continued to decline between 1969 and 2013 (JNCC 2014). During this period the proportion of individuals breeding in the urban environment increased dramatically (Raven & Coulson 1997). More recently, specific colonies have stabilised (e.g. Canna, Swann 2010), and regional patterns of change were identified in four discrete geographic areas; Northern Ireland and Western Scotland, Wales and Western England, Eastern England and Eastern Scotland and North Eastern England (Cook & Robinson 2010).

Factors influencing survival rates

Survival rates do not appear to differ between the sexes (Wanless *et al* 1996).

Factors influencing productivity rates

Local productivity rates have been linked to nesting habitat and parent quality (Kim & Monaghan 2005a), and decrease at low and high nesting densities (Parsons 1976). Productivity increases with age, stabilising from age 6 onwards (Chabrzyk & Coulson 1976).

Knowledge gaps

Regional trends in survival rates could be investigated using the long-term mark-recapture studies from the Isle of May and Skomer. There are also a good amount of ring-recovery data (from birds ringed during the breeding season) that could be used to estimate national, or broad regional, survival rates. A small number of colour-marking studies are also available that could provide an alternative dataset for estimating annual survival rates. Information on adult breeding dispersal is lacking, and the incidence of missed breeding would also benefit from being re-estimated. Estimation of the incidence of missed breeding could potentially be achieved using the long-term monitoring data from the Isle of May or Skomer.

5.8.7. Great black-backed gull (*Larus marinus*)

Table 24. Recommended estimates of demographic rates for the great black-backed gull.

Demographic parameter	Mean	SD_p	SE_e	Data quality	Data representation
Age-specific survival					
Adult survival [1]	0.930			2	1
Colony-specific productivity					
Canna [2-3]	0.808	0.622		6	5
Noss [2]	0.650	0.268		6	4
North Hill [2]	0.810	0.420		5	4
Brownsea [2]	0.440	0.502		5	4
Skomer [2]	1.290	0.190		6	4
Skokholm [2]	1.150	0.288		6	4
Bardsey [2]	1.840	0.840		6	4
Ynysoedd Gwylan [2]	1.520	0.265		6	4
Rockcliffe Marsh [2]	0.670	0.134		6	4
Big Copeland Island [2]	2.000	-		5	4
National-average productivity [2-3]					
	1.139	0.533		6	5
Age of recruitment [4]					
	5			4	2
Incidence of missed breeding					
	-			-	-
Dispersal					
Natal	-			-	-
Adult	-			-	-

[1]Glutz von Blotzheim & Bauer 1982, cited by Garthe & Hüppop 2004; [2]Mavor *et al* 2008; [3]Swann 2010; [4]Cramp 1985.

Recommended demographic rates

In the absence of a focused UK ringing program on great black-backed gulls, local survival rates are largely unknown. Published estimates of adult survival include a relatively old study from Germany (Glutz von Blotzheim & Bauer 1982, cited in Garthe & Hüppop 2004; see accompanying material for great black-backed gull). Juvenile and immature survival rates are also unknown. Population models should be constructed based on the demographic estimates reported for the other large gull species. However, further matching of local population trends is recommended in order to assess the suitability of these estimates.

Regional trends in productivity rates have not been investigated for this species (e.g. Cook & Robinson 2010). Consequently, it may be more suitable to use colony-specific rates of productivity to parameterise population models on this species (Table 24). The productivity of great black-backed gulls is monitored annually at two SMP Key Sites: Canna and Skomer, and long-term productivity estimates are given for seven additional UK localities by Mavor *et al* (2008).

Published information on the age of recruitment for great black-backed gulls is relatively old and may not be based on UK data (Cramp 1985). The age of first breeding is thought to be between 4 and 5 years (Cramp 1985). National census data indicate that the rate of decrease for the UK population has changed since this estimate was published (JNCC 2014). Therefore caution should be applied when using this age to model current population trends (Table 24). Information on natal and adult dispersal and the incidence of missed breeding is lacking and estimates based on herring gull (Table 23) could be used to represent these. Matching of local population trends is recommended to assess the suitability of these estimates.

Population trend in UK

Great black-backed gulls breed mainly in the Outer and Inner Hebrides and the Northern Isles of Scotland. National census data indicate that colonies were declining slowly between 1960 and 2002, and more quickly between 2000 and 2013 (JNCC 2014). This earlier trend was partially attributed to a decline in fishery discards (Furness *et al* 1992).

Factors influencing survival rates

The drivers of survival for great black-backed gulls have not been identified.

Factors influencing productivity rates

Local productivity rates have been reported to decrease due to predation from mink, albeit to a lesser extent than observed in the smaller gull species (Craik 2013). Outside of the UK, productivity rates have been linked to nesting density (Butler & Trivelpiece 1981; Ellis & Good 2006), laying dates (Regehr & Rodway 1999), nesting habitat (Ellis & Grid 2006) and contaminants (Helberg *et al* 2005).

Knowledge gaps

Information on juvenile and immature survival rates, natal and adult dispersal, and the incidence of missed breeding are all lacking. There are some ring-recovery data for birds ringed during the breeding season (primarily as chicks) that have the potential to provide estimates of annual survival rates, but these data have not yet been analysed in this way. The ability to resolve juvenile survival from these estimates will depend on levels of natal dispersal.

5.9. Terns

5.9.1. Sandwich tern (*Sterna sandvicensis*)

Table 25. Recommended estimates of demographic rates for the Sandwich tern.

Demographic parameter	Mean	SD_p	SE_e	Data quality	Data representation
Age-specific survival					
Juvenile survival (0-1 year) [1]	0.358		0.219	4	4
Immature survival (2-3-4 years)[1]	0.741		0.206	4	4
Adult survival [1]	0.898		0.029	4	4
Age-specific productivity					
Age 3 years [2]	*0.167			2	1
Age 4 years [2]	*0.235			2	1
Age 5-11 years [2]	*0.649			2	1
Colony-specific productivity					
Sands of Forvie [3]	0.560	0.440		6	3
Coquet [3]	0.600	0.240		6	3
Scolt Head [3]	0.690	0.400		6	3
Blakeney [3]	0.690	0.320		6	3
Rye Harbour [3]	0.570	0.620		6	3
Langstone Harbour [3]	0.470	0.330		6	3
Brownsea [3]	0.670	0.392		6	3
Anglesey [3]	0.870	0.271		6	3
Hodbarrow [3]	0.480	0.480		6	3
Lough Swilly [3]	1.030	0.313		6	3
Mulroy Bay [3]	0.920	0.397		6	3
Lady's Island Lake [3]	0.800	0.265		6	3
National-average productivity [3]					
	0.702	0.372		6	3
Age of recruitment [2]					
	3			2	1
Incidence of missed breeding					
	-			-	-
Breeding dispersal					
Natal	-			-	-
Adult	-			-	-

[1]Robinson 2010; [2]Veen 1977; [3]Mavor *et al* 2008. * Maximum values (see text).

Recommended demographic rates
The size of Sandwich tern colonies can fluctuate significantly between years as birds redistribute between adjacent colonies (Lloyd *et al* 2001). High adult dispersal complicates

the ability to reliably resolve population trends and survival rates without multi-colony studies. In the absence of a focused UK ringing program on Sandwich terns, colony-specific survival rates are not available. Published estimates are based on similar species (Garthe & Hüppop 2004), and two ring-recovery studies from the UK (Green *et al* 1990; Robinson 2010) (see accompanying material for Sandwich tern). National census data indicate that the trajectory of the UK population has changed since the Green *et al* (1990); from increasing to decreasing (JNCC 2014). Therefore the estimate for adult survival in Table 25 is based on Robinson (2010). Regional variation in survival rates has not been investigated; however regional differences in population trends could not be wholly attributed to local differences in breeding success (Cook & Robinson 2010). At present there are not sufficient data to model colony-specific survival rates for Sandwich tern. It is important to consider these regional differences in population trend as caveats of any population models developed.

Cook and Robinson (2010) could not identify consistent regional trends in the productivity of Sandwich terns within the SMP and OSPAR regions. Consequently, it may be more suitable to use colony-specific rates of productivity to parameterise population models on this species. Long-term productivity is given for 12 localities in the UK by Mavor *et al* (2008). Published information on experience-specific productivity for Sandwich tern is relatively old (Table 25; Veen 1977). These values should be used in population models to guide the proportional change that can be expected with age, as opposed to absolute values of productivity. These estimates are maximum values only because they do not account for multiple chicks being fledged per pair.

Published information on the modal age that Sandwich terns breed for the first time is lacking. In the Netherlands, 3 year old but not 2 year old birds were reported in the breeding population (Veen 1977). This study is relatively old and it was not possible to verify the corresponding population trend, however the reported age is similar to other tern species (Tables 25-28). Although adult dispersal is known to be high (Lloyd *et al* 2001), rates have not been quantified. Information on the incidence of missed breeding and natal breeding dispersal are also currently lacking. It may be possible to substitute these parameters with values from the other tern species. Further matching of local population trends is recommended in order to assess the suitability of these estimates.

Population trend in UK
Sandwich tern colonies occur in a patchy distribution throughout the British coastline. National census data indicate that colonies increased between 1969 and 1988, decreased between the late 1980s and early 2000s and stabilised between 2000 and 2013 (JNCC 2014). Regional patterns of change were identified in five discrete geographic areas; the East of Scotland, the East of England, the South East of England, Wales, South and South West of England and the North Irish Sea (Cook & Robinson 2010).

Factors influencing survival rates
The drivers of survival for Sandwich tern have not been identified.

Factors influencing productivity rates
In the Netherlands, local productivity rates have been reported to decrease due to predation from gulls (Veen 1977). Furthermore, mean clutch size and hatching success increase with age (Veen 1977).

Knowledge gaps
The survival rates of Sandwich terns are not reliably resolved, and information on natal and adult dispersal and the incidence of missing breeding is also lacking. In the absence of a focused UK ringing program it will be difficult to improve these estimates using local data.

5.9.2. Common tern (*Sterna hirundo*)

Table 26. Recommended estimates of demographic rates for the common tern.

Demographic parameter	Mean	SD_p	SE_e	Data quality	Data representation
Age-specific survival					
Immature survival (0-2 years) [1]	0.441		0.004	4	1
Immature survival (3-4 years) [2]	0.850			4	1
Adult survival (≥6 year) [2-3]	0.883	0.014		6	1
Colony-specific productivity					
E an Ruisg, L Feochan [4]	1.820	0.594		6	3
Sgeir na Caillich, L Melfort [4]	0.670	0.688		6	3
G. Eileanan, Sd Mull [4]	0.530	0.529		6	3
Avoch Fish Farm [4]	0.770	0.563		6	3
St Fergus [4]	0.150	0.173		6	3
Forth [4]	0.910	0.381		6	3
Coquet [4]	1.030	0.626		6	3
Rye Meads [4]	0.530	0.411		6	3
Hoveton Great Broad [4]	0.260	0.291		6	3
Snettisham [4]	0.280	0.374		6	3
Holkham [4]	0.400	0.503		6	3
Blakeney [4]	0.440	0.469		6	3
Breydon Water [4]	0.990	0.252		6	3
Pitts Deep – Hurst [4]	0.370	0.311		6	3
Langstone Harbour [4]	0.540	0.552		6	3
Rye Harbour [4]	1.050	0.716		6	3
Brownsea Is. [4]	0.590	0.289		6	3
Lodmoor [4]	1.460	0.509		6	3
Shotton [4]	1.300	0.495		6	3
Cemlyn [4]	0.590	0.720		6	3
Seaforth [4]	0.610	0.313		6	3
Rockcliffe [4]	0.110	0.180		6	3
Belfast Lough [4]	1.350	0.313		6	3
Rockabill [4]	1.590	0.465		6	3
National-average productivity [4]					
	0.764	0.470		6	3
Age of recruitment [5-6]					
	3-4			6	1
Incidence of missed breeding [7]					
	Low			-	-

Demographic parameter	Mean	SD_p	SE_e	Data quality	Data representation
Breeding dispersal					
Natal [3]	0.070			2	1
Adult [2-3]	0.103	0.077		6	1

[1]Braasch *et al* 2008; [2]Breton *et al* 2014; [3]Nisbet & Cam 2002; [4]Mavor *et al* 2008; [5]Nisbet 1978; [6]Ludwigs & Becker 2002a; [7]Dittmann *et al* 2002.

Recommended demographic rates

In the absence of a focused UK ringing program on common terns, local survival rates are largely unknown. There are relatively old published estimates from North America; including three mark–recapture studies based on constant recapture rates (Austin & Austin 1956; Nisbet 1978; DiCostanzo 1980). More recent studies from North America model survival rates using variable recapture rates (Nisbet & Cam 2002; Breton *et al* 2014). Regional variation in the survival rates of common terns has not been investigated. Therefore the estimate of adult survival in Table 26 combined estimates from the two most recent colony-specific studies (see accompanying material for common tern). The local population trend could not be identified for these studies, therefore further matching of local population trends is recommended in order to assess their suitability for modelling the UK population. Estimates of juvenile survival are limited to relatively short-term studies (DiCostanzo 1980; Braasch *et al* 2008). The more recent of these was used to represent juvenile survival in Table 26.

Regional trends in productivity have not been investigated for this species (e.g. Cook & Robinson 2010). Consequently, it may be more suitable to use colony-specific rates of productivity to parameterise population models on this species (Table 26). Long-term productivity is given for 24 localities in the UK by Mavor *et al* (2008).

Published information on the age of recruitment is regionally specific. In Germany, common terns are thought to breed for the first time at age 3 (Ludwigs & Becker 2002a), and in North America the age of first breeding is thought to occur at age 4 (Nisbet 1978). Further matching of local population trends is recommended in order to assess the suitability of these estimates for modelling the UK population. The incidence of missed breeding is thought to be low (Dittmann *et al* 2002), although population rates have not been quantified. The dispersal of adult common terns is predominantly low, but may vary between colonies (González-Solís *et al* 1999; Nisbet & Cam 2002) and increase between colonies that are closely distributed (Breton *et al* 2014). Natal dispersal is also thought to be low (Nisbet & Cam 2002). The rates of dispersal listed in Table 26 reflect the number of studies that quantify this parameter. Dispersal is reported to be higher for adults compared to the natal age class, but this is unlikely to occur in reality. For the local difference between these age classes see Nisbet and Cam (2002).

Population trend in UK

Common terns breed throughout much of the British coastline and larger inland waterbodies. National census data indicate that colonies were relatively stable between 1969 and 2002, but declined between 2000 and 2013 (JNCC 2014).

Factors influencing survival rates

Outside of the UK, survival rates were relatively constant through time (Breton *et al* 2014), and do not appear to differ between the sexes (Braasch *et al* 2008; Breton *et al* 2014). Breton *et al* (2014) report that survival rates follow a quadratic relationship with age; increasing between fledging and recruitment, stabilising for a number of years, and then declining when individuals are >12 years old. There was no evidence that local survival from fledging to first breeding depends on hatching order, hatching date, number of fledglings per brood or parental age (Nisbet 1996; Dittmann *et al* 2001). First year survival increased with fledging mass (Ludwigs & Becker 2002b).

Factors influencing productivity rates
Local productivity rates have been reported to decrease due to predation (Nisbet & Welton 1984; Becker 1995; Mavor *et al* 2008), but increase with nesting density (Becker 1995). Productivity also depends on parental age or quality (Bollinger 1994; Wendeln & Becker 1999; Nisbet *et al* 2002; Arnold *et al* 2006).

Knowledge gaps
Age-specific survival rates and natal breeding dispersal are largely unknown for the UK population. The incidence of missed breeding is also poorly resolved. In the absence of a focused UK ringing program on common terns it will be difficult to improve these estimates using local data.

5.9.3. Arctic tern (*Sterna paradisaea*)

Table 27. Recommended estimates of demographic rates for the Arctic tern.

Demographic parameter	Mean	SD_p	SE_e	Data quality	Data representation
Age-specific survival					
Adult survival (≥2 year) [1-2]	0.837	0.035		6	2
Colony-specific productivity					
Ardullie [3]	0.780	0.156		4	3
Coquet [3]	0.770	0.320		6	3
Eigg [3]	0.120	0.170		6	3
Fair Isle [4]	0.125	0.210		6	4
Farne Islands [3]	0.570	0.348		6	3
Fetlar [3]	0.130	0.194		6	3
Foula [5]	0.238	0.288		6	4
Foulney [3]	0.350	0.224		6	3
Isle of May [3]	0.310	0.277		6	3
Kinloss [3]	0.160	0.232		6	3
Long Nanny [3]	0.660	0.480		6	3
Papa Westray [3]	0.160	0.312		6	3
Rockabill [3]	0.870	0.379		6	3
Sands of Forview [3]	0.260	0.310		6	3
Skerries [3]	1.080	0.400		6	3
St Fergus [3]	0.090	0.108		6	3
Tiree [3]	0.030	0.073		6	3
National-average productivity [3-5]					
	0.380	0.325		6	4
Age of recruitment [1,2]					
	4			6	3
Incidence of missed breeding					
	-			-	-
Breeding dispersal					
Natal [6]	0.610			4	1
Adult [6-7]	0.089	0.115		6	1

[1] Cullen 1957; [2] Coulson & Horobin 1976; [3] Mavor *et al* 2008; [4] Shaw *et al* 2010; [5] Furness 2007; [6] Devlin *et al* 2008; [7] Møller *et al* 2006.

Recommended demographic rates

The UK estimates of adult survival for Arctic tern are largely limited to the Farne Islands (Cullen 1957; Coulson & Horobin 1976). More recent estimates of survival are from Canada (Devlin *et al* 2008). Regional differences in survival have not been investigated; however the rates of survival reported for the UK and Canada are relatively similar (see accompanying material for Arctic tern). The estimate given in Table 27 combined the UK estimates. National census data indicate that the trajectory of the UK population has changed since these publications, from increasing to decreasing (Shaw *et al* 2010; JNCC 2014). Therefore caution should be applied when using these rates to model current population trends.

Regional trends in productivity rates have not been investigated for this species (e.g. Cook & Robinson 2010). Consequently, it may be more suitable to use colony-specific rates of productivity to parameterise population models on this species (Table 27). The productivity of Arctic tern is monitored annually at one SMP Key Site: Fair Isle, and long-term productivity is given for 15 additional UK localities by Mavor *et al* (2008).

Information on the age of recruitment for Arctic tern is relatively old, and the national population trend is thought to have changed since it was published (Cullen 1957; Coulson & Horobin 1976). The majority of Arctic terns breed for the first time at age 4, although a small proportion breed for the first time at age 5, and a few individuals may also attempt breeding at 2 or 3 years old (Cullen 1957; Coulson & Horobin 1976). The reported modal age is older than for the other terns (Tables 25-28), and therefore some caution should be applied when using this estimate to model current population trends. Information on the incidence of missed breeding is limited, although it is thought to occur primarily in response to food shortage (Monaghan *et al* 1992). Outside of the UK, breeding dispersal of adult Arctic terns is low (Møller *et al* 2006; Devlin *et al* 2008), although it is higher between colonies that are closely distributed (Brindley *et al* 1999; Møller *et al* 2006; Devlin *et al* 2008). Adult and natal dispersal are also thought to reflect local conditions at a colony, e.g. low levels of food availability can increase dispersal (Møller *et al* 2006). Further matching of local population trends is recommended in order to assess the suitability of these estimates for modelling UK birds. In Denmark, natal dispersal is thought to be high (Møller *et al* 2006). This process will strongly influence the subsequent population trajectory, and therefore requires consideration when building population models for this species.

Population trend in UK

Breeding colonies of Arctic terns are concentrated in the Northern Isles. Exact population trends are difficult to verify because different methods were used to conduct censuses. Numbers are thought to have increased between 1969 and 1988, decreased between 1988 and 2002, and stabilised between 2000 and 2013 (Shaw *et al* 2010; JNCC 2014).

Factors influencing survival rates

The drivers of survival for Arctic tern have not been identified.

Factors influencing productivity rates

Local productivity rates are reported to be positively influenced by increased prey abundance (Monaghan *et al* 1989; Suddaby & Ratcliffe 1997; Robinson *et al* 2001; Furness 2007) and negatively influenced by predation (Nordström *et al* 2004; Mavor *et al* 2008). Nest location can also be significantly influential (Bunin & Boates 1994). Mean clutch size may increase in relation to age, although small sample sizes prevented this from being reliably quantified (Coulson & Horobin 1976). Monaghan *et al* (1989) demonstrated that the productivity of Arctic terns differed between colonies in relation to foraging conditions.

Knowledge gaps

Estimates of juvenile and immature survival are lacking. High levels of natal dispersal will make these parameters difficult to deduce without multi-colony studies. A population rate for the incidence of missed breeding has also not been quantified. More intensive efforts at the breeding ground would be needed to resolve this parameter.

5.9.4. Little tern (*Sternula albifrons*)

Table 28. Recommended estimates of demographic rates for the little tern.

Demographic parameter	Mean	SD_p	SE_e	Data quality	Data representation
Age-specific survival					
Adult survival [1]	0.800			3	2
Regional-specific productivity					
North [2-3]	0.456	0.482		6	4
South [2-3]	0.464	0.506		6	4
National-average productivity [2]					
	0.518	0.518		6	4
Age of recruitment [4]					
	>2			2	2
Incidence of missed breeding					
	-			-	-
Breeding dispersal					
Natal	-			-	-
Adult	-			-	-

[1]Grosskopf 1964, cited by Robinson & Ratcliffe 2010; [2]Mavor *et al* 2008; [3]Cook & Robinson 2010; [4]Schönert 1961, cited by Cramp 1985.

Recommended demographic rates

In the absence of a focused UK ringing program on little terns, local survival rates are largely unknown (see accompanying material for little tern). Outside of the UK, published estimates include a relatively old ring-recovery study (Grosskopf 1964, cited by Robinson & Ratcliffe 2010). Regional differences in population trends could not be wholly attributed to local differences in breeding success (Cook & Robinson 2010). At present there are not sufficient data to model colony-specific survival rates, and therefore it will be important to consider regional differences in population trend as caveats of population models that are developed.

Regional trends in productivity rates were identified by Cook and Robinson (2010) in two discrete geographic areas (see Appendix S5 for map). Long-term productivity is given for 21 localities in the UK by Mavor *et al* (2008). The range of productivity values given in Table 28 were calculated using the regional groupings highlighted by Cook and Robinson (2010) to categorise the values specified by Mavor *et al* (2008). These estimates indicate large variability to occur within each region.

Published estimates for the age of recruitment of little tern is relatively old and not based on UK data. Little terns are thought to start breeding at a minimum of 2 years, usually older (Schönert 1961, cited by Cramp 1985). This minimum age is lower than the modal ages that were identified for the other tern species (Tables 25-28). It therefore may not be representative of the modal age for little tern. Information on natal and adult dispersal and the incidence of missed breeding is largely lacking. It may be possible to use estimates based on common tern or Arctic tern (Table 26; Table 27) to represent these demographic parameters. Matching of local population trends is recommended in order to assess the suitability of these estimates.

Population trend in UK

Little tern colonies are scattered along much of the British coastline, but the main concentration is in the south and east of England. In the UK, abundance generally declined after the late 1980s but stabilised in the 1990s (Ratcliffe *et al* 2000). The decline was attributed to low rates of productivity (Ratcliffe *et al* 2000). The rate of decline varies regionally in four discrete geographic areas; East Scotland and North East England, East and South England, Wales and West England and West Scotland and East Ireland (Cook & Robinson 2010).

Factors influencing survival rates

The drivers of survival for little tern have not been identified.

Factors influencing productivity rates

Local productivity rates have been linked to human disturbance (Medeiros *et al* 2007; Perrow *et al* 2011) and nest location (Oro *et al* 2004).

Knowledge gaps

Rates of survival and the age of recruitment are poorly resolved. The incidence of missed breeding and rates of breeding dispersal are also largely unknown. In the absence of a focused UK ringing program on little tern it will be difficult to improve these estimates using local data.

5.10. Auks

5.10.1. Common guillemot (*Uria aalge*)

Table 29. Recommended estimates of demographic rates for common guillemot.

Demographic parameter	Mean	SD_p	SE_e	Data quality	Data representation
Age-specific survival					
Juvenile survival (0-1 year) [1]	0.560		0.013	6	3
Immature survival (1-2 year) [1]	0.792		0.034	6	3
Immature survival (2-3 year) [1]	0.917		0.022	6	3
Adult survival (≥4th year) [2-4]	0.939	0.015		6	6
Experience-specific productivity					
1st year [5]	0.620			6	3
≥2nd year [5]	0.686	0.068		6	3
Regional-specific productivity					
North [6-8]	0.629	0.174		6	5
East [6-7,9]	0.659	0.133		6	5
West [4,6-7]	0.823	0.056		6	5
National average productivity [4,6,8-9]					
	0.672	0.147		6	5
Age of recruitment [10]					
	6			6	4
Incidence of missed breeding [11]					
	0.079			6	3
Dispersal					
Natal [1,12]	0.580	0.212		6	3
Adult [13]	0.007			4	3

[1]Harris *et al* 2007; [2]Lahoz-Monfort *et al* 2011; [3]Reynolds *et al* 2011; [4]Meade *et al* 2013; [5]Crespin *et al* 2006a; [6]Mavor *et al* 2008; [7]Cook & Robinson 2010; [8]Shaw *et al* 2010; [9]Newell *et al* 2010; [10]Halley & Harris 1993; [11]Harris & Wanless 1995; [12]Harris *et al* 1996; [13]Swann & Ramsay 1983.

Recommended demographic rates

The UK estimates of survival for common guillemot are largely limited to the Isle of May, Canna, Colonsay and Skomer. Regional differences in abundance trends were largely attributed to local differences in breeding success (Cook & Robinson 2010) and colonies with increasing and decreasing trends report adult survival rates to be similar (e.g. Isle of May – Reynolds *et al* 2011; Skomer - Meade *et al* 2013) (see accompanying material for common guillemot). Therefore, it may be sufficient to employ a national average for survival when parameterising population models of common guillemot. However, it should be noted that inter-colony comparisons of survival rates demonstrate some evidence of regional differences (Harris *et al* 2000b). A high correlation is thought to occur between breeding colonies that share wintering areas (Reynolds *et al* 2011). To apply colony-specific survival rates to population models, preference should be given to survival rates from colonies that overlap in winter distribution. The estimate of adult survival reported in Table 29 combined estimates from the most recent colony-specific studies. Stronger metal rings were introduced in Britain and Ireland in 1983 to reduce ring-loss and wear to the inscription associated with standing on rocky cliff-ledges. Therefore, unless survival estimates have been corrected for ring loss, only more recent estimates can be reliably used. Immature guillemots tend to revisit their natal colony at least once before recruiting to a breeding population. Therefore estimates of immature survival rate derived from mark-recapture data should be reliable when accompanied by extensive recapture effort (e.g. Harris *et al* 2007). Other estimates of juvenile survival from the Isle of May (Crespin *et al* 2006a) and Canna (Reynolds *et al* 2011) are detailed in the accompanying material for common guillemot.

Regional trends in productivity rates were identified in three discrete geographic areas (Cook & Robinson 2010; Appendix S6 for map). The productivity of common guillemot is monitored annually at three SMP Key Sites: Isle of May, Fair Isle and Skomer, and long-term productivity estimates are given for eight additional UK localities by Mavor *et al* (2008). The range of productivity values given in Table 29 were calculated using the regional groupings highlighted by Cook and Robinson (2010) to categorise the values specified by Mavor *et al* (2008), Shaw *et al* (2010), Newell *et al* (2010) and Meade *et al* (2013). The estimates of experience-specific productivity in Table 29 should be used to guide the proportional change that can be expected with age, as opposed to absolute values of productivity.

Published information on the age of recruitment for common guillemot is relatively old, and differs regionally. On Skomer and the Isle of May the median age of first breeding appeared to be one or two years earlier than on Canna (Birkhead & Hudson 1977; Halley & Harris 1993; Harris *et al* 1994a). This may reflect differences in methodology. Daily monitoring on the Isle of May minimises the bias caused by birds losing eggs soon after laying (Harris *et al* 1994a). Therefore, the value in Table 29 is based on the Isle of May estimate only. The incidence of missed breeding in the common guillemot was explored extensively by Harris and Wanless (1995). The behaviour seemed to vary between individuals, with 6% of individuals accounting for 47% of non-breeding cases (n=146). The occurrence of adult dispersal is low (Swann & Ramsay 1983) but increases between closely distributed colonies (Southern *et al* 1965). Natal dispersal is considerably higher (Swann & Ramsay 1983; Harris *et al* 1996; Harris *et al* 2007). The rates of natal dispersal in Table 29 are from the multi-colony study by Harris *et al* (1996). High levels of natal dispersal will strongly influence the subsequent population trajectory, and requires consideration when building population models for this species.

Population trend in UK

The common guillemot is the most abundant breeding seabird in the UK (Mitchell *et al* 2004). Populations increased rapidly in all regions of the UK between 1969 and 1985. This rate of increase slowed between 1985 and 2002, and then slowed again between 2000 and 2013 (JNCC 2014). In the early 1980s, regional differences emerged in two discrete geographic areas; the East Coast of the UK and the second on the West Coast of the UK (Cook & Robinson 2010; Heubeck & Mellor 2014). On Skomer Island numbers have increased at an almost constant rate of 5% per annum in the last 30 years (Meade *et al* 2013). In contrast, colonies on the Isle of May, Fair Isle and Canna are presently in decline (Harris *et al* 1996; Newell *et al* 2010; Swann 2010).

Factors influencing survival rates

Local survival rates have been linked to oil pollution and also the marine environment; decreasing at high values of winter Northern Atlantic Oscillation (i.e. survival rates are lower following warmer sea temperatures and stronger winds, Votier *et al* 2005, 2008). On the Isle of May (SE Scotland) there is no evidence that survival depends on prey density or local population size (Harris *et al* 2007). For older birds, survival begins to decline at >19 years (Crespin *et al* 2006a).

Factors influencing productivity rates

Local productivity rates are reported to be negatively influenced by decreased prey biomass (Rindorf *et al* 2000; Wanless *et al* 2005). Productivity also increases with greater synchrony of lay dates, although there was no evidence for an effect of the actual lay date (Birkhead 1977a). Productivity of first time breeders was relatively low, stabilising from the fifth attempt and decreasing when birds breed for >13 years (Crespin *et al* 2006a).

Knowledge gaps

The demography of guillemots in the UK is relatively well resolved.

5.10.2. Razorbill (*Alca torda*)

Table 30. Recommended estimates of demographic rates for the razorbill.

Demographic parameter	Mean	SD_p	SE_e	Data quality	Data representation
Age-specific survival					
Immature survival (0-2 years) [1]	0.630	0.209		6	2
Adult survival (≥3 year) [2-3]	0.895	0.067		6	6
Regional-specific productivity					
South [4-6]	0.642	0.271		6	4
Mid [4-5,7]	0.643	0.057		6	5
North [4-5,8]	0.459	0.236		6	5
National-average productivity [4,6-8]					
	0.570	0.247		6	5
Age of recruitment [9]					
	5			3	3
Incidence of missed breeding [10]					
	0.030			4	3
Dispersal					
Natal [9,11]	0.088	0.116		5	3
Adult [10-12]	0.047	0.046		5	3

[1]Lavers *et al* 2008; [2]Taylor *et al* 2010; [3]Lahoz-Monfort *et al* 2011; [4]Mavor *et al* 2008; [5]Cook & Robinson 2010; [6]Lloyd 1979; [7]Newell *et al* 2010; [8]Shaw *et al* 2010; [9]Lloyd & Perrins 1977; [10]Harris & Wanless 1989; [11]Lavers *et al* 2007; [12]Lloyd 1976.

Recommended demographic rates

The UK estimates of survival for razorbill are largely limited to the Isle of May and Skomer. There is also an older estimate based on shorter-term data from the Shiant Islands, NW Scotland (Chapdelaine 1997, see accompanying material for razorbill). Estimates of adult survival from outside of the UK appear to be higher (Sandvik *et al* 2005; Lavers *et al* 2008). Regional differences in population trends could be largely attributed to local differences in breeding success (Cook & Robinson 2010), therefore it may be sufficient to employ a national average for survival when parameterising population models of this species. Although the estimate given in Table 30 combined the most recent UK studies, information on juvenile and immature survival rates is limited to a relatively old ring-recovery study (Lloyd 1974). National census data indicate that the trajectory of the UK population has changed since this study; therefore the estimate of immature survival in Table 30 is taken from a more recent mark-recapture study from Canada (Lavers *et al* 2008). Although razorbills in Canada were following a similar population trend to the current UK population; the estimate of adult survival by Lavers *et al* (2008) is higher than the estimate reported for the UK by Lahoz-Monfort *et al* (2011) and Taylor *et al* (2010). Therefore caution should be applied when using these estimates to model the current UK population trend.

Regional differences in productivity were identified by Cook and Robinson (2010) in three discrete geographic areas (Appendix S7 for map). The productivity of razorbill is monitored annually at three SMP Key Sites: Isle of May, Skomer and Fair Isle, and long-term productivity estimates are given for four additional UK localities by Mavor *et al* (2008). The range of productivity values given in Table 30 were calculated using the regional groupings highlighted by Cook and Robinson (2010) to categorise the values specified by Lloyd (1979), Mavor *et al* (2008), Newell *et al* (2010) and Shaw *et al* (2010).

Published information on the age of recruitment for the razorbill is relatively old and may vary regionally in relation to nest site availability (anecdotal evidence reviewed by Lloyd & Perrins 1977). The reported age is similar to the other auk species (Tables 29-32), and consequently may be suitable for modelling current population trends. Local incidence of missed breeding is low (Harris & Wanless 1989), but the behaviour is more prevalent in certain individuals (Harris & Wanless 1989). Adult dispersal is also thought to be very low (Lloyd 1976; Harris & Wanless 1989; Lavers *et al* 2007). At a stable UK colony, natal dispersal was reported to be very low (Lloyd & Perrins 1977), but in Canada rates were higher (Lavers *et al* 2008). To represent the range of adult and natal dispersal, the value given in Table 30 combined estimates from the UK and Canada (see accompanying material for razorbill).

Population trend in UK

The UK population increased between 1969 and 2002 (JNCC 2014), however this trend varies regionally, for example particular colonies declined during the 1990s, before stabilising or increasing during the last decade (Shaw *et al* 2010; Swann 2010). On Canna this change in trajectory followed the successful eradication of rats (Swann 2010). Regional differences in abundance trends were identified in four discrete geographic areas; the East Coast of Scotland, the South Coast of Wales, the North Coast of Wales and the West Coast of Scotland (Cook & Robinson 2010).

Factors influencing survival rates

Survival rates have been linked to the marine environment, and decrease at high values of winter Northern Atlantic Oscillation (i.e. survival rates are lower following warmer sea temperatures and stronger winds, Sandvik *et al* 2005). The survival rates of razorbills do not appear to differ between the sexes (Harris *et al* 2000a).

Factors influencing productivity rates

Local productivity rates decline in response to predation from gulls (Hudson 1982).

Knowledge gaps

Juvenile and immature survival rates are poorly resolved for UK populations. Assuming low levels of natal dispersal (Lloyd & Perrins 1977), it should be possible to estimate these rates using existing ring-recovery and sighting data. An alternative approach would be to solve for juvenile survival rates based on other demographic parameters and population trend data collected from a regularly monitored site (e.g. Skomer or Isle of May).

5.10.3. Black guillemot (*Cepphus grylle*)

Table 31. Recommended estimates of demographic rates for black guillemot.

Demographic parameter	Mean	SD_p	SE_e	Data quality	Data representation
Age-specific survival					
Juvenile survival (0-1 year) [1]	0.731	0.239		6	2
Adult survival [1]	0.870	0.034		6	2
Colony-specific productivity					
North Ronaldsay [2]	1.420	0.316		6	4
Auskerry [2]	1.310	0.166		6	4
Swona [2]	1.500	0.201		6	4
Old Lighthouse Island [2]	1.080	0.280		6	4
Mew Island [2]	2.000			4	4
Rockabill [2]	1.260	0.226		6	4
National-average productivity [2]					
	1.298	0.315		6	4
Age of recruitment [3]					
	5			4	2
Incidence of missed breeding					
	-			-	-
Dispersal					
Natal [4]	0.657			2	2
Adult [4]	0.010			4	2

[1]Frederiksen & Petersen 1999a; [2]Mavor *et al* 2008; [3]Frederiksen 1998; [4]Frederiksen & Petersen 1999b.

Recommended demographic rates

Estimating the survival rates of black guillemots is complicated by nests occurring in hard to reach locations, such as small rocky islands, low-lying rocky coasts, and rock crevices. The UK estimates of survival are relatively old and based on short time series (1-2 years; Ewins 1988, 1989). More recent estimates are from Iceland. The population trajectory of black guillemots in Iceland between 1977 and 1991 was similar to the present UK trend; stable (Frederiksen & Petersen 2000; JNCC 2014). There are not sufficient data to explore regional variation in age-specific survival rates, therefore population models based on these values will have to assume that the reported estimates from Iceland are representative more broadly (see accompanying material for black guillemot). In Iceland, the survival rate of black guillemot are best described in two age classes; juvenile and birds from age one onwards.

Regional differences in productivity rates have not been investigated for this species (e.g. Cook & Robinson 2010). Consequently, it may be more suitable to use colony-specific rates of productivity to parameterise population models on this species (Table 31). Long-term productivity is given for six localities in the UK by Mavor *et al* (2008).

Although published information on the age of recruitment for the black guillemot is not based on UK data (Frederiksen 1999), the corresponding population trend for this estimate was similar to the present UK trajectory. There is no information available on the incidence of missed breeding. It may be appropriate to use estimates based on common guillemot or puffin (Table 29 and Table 32), however further matching of local population trends is recommended in order to assess suitability. Dispersal of breeding adults is low (Preston 1968; Asbirk 1979; Frederiksen & Petersen 1999b) but natal dispersal is thought to be high (Frederiksen & Petersen 1999b, 2000). The dispersal rates in Table 31 are from the multi-colony study by Frederiksen and Petersen (1999b). High levels of natal dispersal will strongly influence the subsequent population trajectory, and requires consideration when building population models for this species.

Population trend in UK

The black guillemot is largely restricted to the north and west coasts of Scotland and the coast of Ireland. Colonies in the UK increased slightly between 1982 and 2002, and are now considered to be relatively stable (JNCC 2014). However, there are regional differences in population trends (Mavor *et al* 2008; Shaw *et al* 2010).

Factors influencing survival rates

Local survival rates decrease following an increase in fishing effort (Frederiksen & Petersen 1999a).

Factors influencing productivity rates

Local productivity rates have been linked to nest quality (Ewins 1989).

Knowledge gaps

The survival rates of the UK population are poorly resolved, and information on the incidence of missed breeding is lacking. Intensive efforts at the breeding grounds would be needed to estimate these parameters.

5.10.4. Atlantic puffin (*Fratercula arctica*)

Table 32. Recommended estimates of demographic rates for Atlantic puffin.

Demographic parameter	Mean	SD_p	SE_e	Data quality	Data representation
Age-specific survival					
Immature survival (0-3 years) [1]	0.709		0.022	6	1
Immature survival (3-4 years) [1]	0.760		0.019	6	1
Immature survival (4-5 years) [1]	0.805		0.017	6	1
Adult survival (≥6 year) [2-4]	0.906	0.083		6	5
Colony-specific productivity					
Fair Isle [5]	0.570	0.141		6	5
Farne Islands [6]	0.415	0.219		6	4
Isle of May [7]	0.642	0.135		6	5
Skomer [6]	0.870	0.028		6	4
St Kilda [6]	0.785	0.078		6	4
National-average productivity [4-9]					
	0.617	0.151		6	5
Age of recruitment [8,10]					
	5			6	3
Incidence of missed breeding [8]					
	0.078			4	3
Dispersal					
Natal [10]	0.230			4	2
Adult [1,8,11]	0.042	0.040		6	3

[1]Breton *et al* 2006; [2]Harris *et al* 2005; [3]Lahoz-Monfort *et al* 2011; [4]Taylor *et al* 2010; [5]Shaw *et al* 2010; [6]Mavor *et al* 2008; [7]Newell *et al* 2010; [8]Ashcroft 1979; [9]Harris 1980; [10]Harris 1983; [11]Kress & Nettleship 1988.

Recommended demographic rates

The UK estimates of survival for Atlantic puffin are largely limited to the Isle of May, Fair Isle and Skomer. A number of studies have examined these data over different time scales (see accompanying material for Atlantic puffin). Harris *et al* (2005) demonstrated that the survival rates of puffins were highly similar across different colonies. Therefore, if may be sufficient to employ a national average for survival when parameterising population models of Atlantic puffin. The estimate of adult survival in Table 32 combined the most recent colony-specific studies. Estimates of adult survival from outside of the UK also appear to be similar (Harris *et al* 2005; Breton *et al* 2006; see accompanying material for Atlantic puffin). The UK estimates of juvenile and immature survival rates are limited to relatively old mark-recapture studies based on return rates only (Harris 1983; Harris & Rothery 1985). National census data indicate that the trajectory of the UK population has changed since these studies. Furthermore, these estimates are based on data collected before the introduction of hard rings and therefore are likely to be biased by ring-loss. The values of immature survival given in Table 32 are from a recent mark-recapture study on Atlantic puffins from Canada (Breton *et al* 2006). It was not possible to verify the corresponding population trend for this estimate, and therefore caution should be applied when using these estimates to model current UK population trends.

Regional trends in productivity rates have not been investigated for this species (e.g. Cook & Robinson 2010). Consequently, it may be more suitable to use colony-specific rates of productivity to parameterise population models on this species (Table 32). The productivity of Atlantic puffin is monitored annually at three SMP Key Sites: Isle of May, Skomer and Fair Isle, and long-term productivity estimates are given for two additional UK localities by Mavor *et al* (2008).

Published information on the age of recruitment for Atlantic puffin is relatively old. The national population trend is thought to have changed since their publication (Ashcroft 1979; Harris 1984; JNCC 2014); however the reported age is similar to the estimates identified for the other species of auk considered (Tables 29-32). The majority of birds breed for the first time at age 4, but rates of productivity are usually lower than at age 5 (Ashcroft 1979; Harris 1984). Adult breeding dispersal is thought to be low (Ashcroft 1979; Harris 1983; Kress & Nettleship 1988; Breton *et al* 2006) and natal dispersal is thought to be high (Harris 1983; Breton *et al* 2006). To represent the range of dispersal rates, the values given in Table 32 are estimated from all of the listed studies (see accompanying material for Atlantic puffin). Published estimates of the incidence of missed breeding are based on short time series (3 years; Ashcroft 1979) and are therefore unlikely to be representative more broadly.

Population trend in UK

The puffin is the second most abundant seabird breeding in the UK. Population trend information is based on long-term monitoring of burrow densities on St Kilda, the Shiant Islands, the Isle of May and the Shetlands. Numbers were stable between the early 1970s and late 1980s (Lloyd *et al* 2001), with the exception of the Isle of May where there was an annual increase of 22% between 1973 and 1981 that was partially attributed to immigration (Harris & Rothery 1985). Overall, the UK population is thought to have increased between 1969 and 2002 (JNCC 2014).

Factors influencing survival rates

There is no evidence that local survival from fledging to first breeding is dependent on fledging mass or fledging date (Harris & Rothery 1985). Local survival rates of adults have been reported to decrease at high population densities (Breton *et al* 2006), and due to predation by great black-backed gulls (Harris & Rothery 1985). Survival rates are also negatively influenced by a decrease in prey biomass and availability (Harris & Bailey 1992; Harris *et al* 2000a). The influence of sea surface temperatures and the Northern Atlantic Oscillation varied between colonies (Harris *et al* 2005), and survival rates do not appear to differ between the sexes (Harris *et al* 2000a).

Factors influencing productivity rates

Local productivity rates have been linked to parental age and disturbance from other burrowing animals (Ashcroft 1979), predation, nesting density (Harris 1980), prey abundance (Regehr & Rodway 1999) and levels of kleptoparasitism (Finney *et al* 2001). Outside of the UK, local productivity rates have been reported to decline following increased rainfall flooding burrows (Rodway *et al* 1998).

Knowledge gaps

The incidence of missed breeding would also benefit from further analysis across a longer time series. High levels of natal dispersal will make estimates of juvenile and immature survival difficult to deduce without multi-colony studies.

6. Summary of demographic rates available

Table 33. National estimates of each demographic parameter by species.

Estimates are weighted means of the available demographic data, and focused to the UK where possible. Where sufficient data could not be gathered using UK examples studies from outside of the UK have been included. Estimates are colour coded to reflect an overall score associated with the data quality and data representation as follows: Red=Low (poor [0-2] quality score and poor [0-2] or good [3-6] representation score); Orange=Intermediate (Intermediate [3-4] or good [5-6] quality score and poor [0-2] representation score; OR Intermediate [3-4] quality score and good [3-6] representation score); Green=High (good quality score 5-6 and good representation score 3-6). The itemisation of these scores is detailed on the second worksheet of the accompanying material for each species. Entries based on more than one study site are highlighted in bold, and estimates of total survival from fledging to recruitment are marked with *. If a species is thought to experience survival rates similar to an adult before age 5, the intermediate boxes are shaded grey.

Common name	**Survival**							**Productivity**	**Age of recruit-ment**	**Missed breeding**	**Dispersal**	
	Juv.	**Imm. (1-2)**	**Imm. (2-3)**	**Imm. (3-4)**	**Imm. (4-5)**	**Imm. (5-8)**	**Adult**				**Natal**	**Adult**
Greater scaup	0.400						0.810	0.570	2			
Common eider	0.200						**0.886**	**0.379**	**3**	0.200		0.012
Goldeneye							0.770	0.365	3			**0.564**
Long-tailed duck							**0.730**	1.900	2			
Common scoter	0.749						0.783	1.838	3			0.123
Velvet scoter							0.773	0.350	2			0.688
Red-throated diver	0.600	0.620					**0.840**	**0.571**	3			0.250
Black-throated diver	0.400						**0.817**	**0.425**	5			
Great northern diver	*	*	0.770				**0.870**	**0.543**	6	0.307	0.069	0.090
Manx shearwater							0.870	**0.697**	5	0.157	**Low**	**Low**
Northern fulmar	*	*	*	*	*	0.26	0.936	**0.419**	9		**0.915**	Low
Northern gannet	0.424	0.829	0.891	0.895			0.919	**0.700**	**5**			Low
Great crested grebe							**0.725**	**1.275**	2			
Great cormorant	**0.540**						**0.868**	**1.985**	**3**		**0.240**	0.123
European shag	0.513	0.737					0.858	**1.303**	2		0.049	0.010
Arctic skua	*	*	*	0.346			0.910	**0.487**	4	0.030		Low

Common name	**Survival**							**Productivity**	**Age of recruit-ment**	**Missed breeding**	**Dispersal**	
	Juv.	**Imm. (1-2)**	**Imm. (2-3)**	**Imm. (3-4)**	**Imm. (4-5)**	**Imm. (5-8)**	**Adult**				**Natal**	**Adult**
Great skua	0.730						**0.882**	**0.651**	7	0.089	0.015	**Low**
Black-legged kittiwake	0.790						**0.854**	**0.690**	**4**	0.194	0.890	0.037
Little gull							0.800		2-3			
Black-headed gull							**0.825**	**0.625**	2			0.167
Common gull	0.410	0.710					0.828	**0.543**	3		0.700	<0.030
Lesser black-backed gull	0.820						**0.885**	0.530	5	**0.337**	0.470	
Herring gull	**0.798**						**0.834**	**0.920**	5	0.350	**0.629**	
Great black-backed gull							0.930	**1.139**	5			
Sandwich tern	0.358	0.741	0.741	0.741			0.898	**0.702**	3			
Common tern	*	0.441		0.850			**0.883**	**0.764**	**3-4**	Low	0.070	**0.103**
Arctic tern							**0.837**	**0.380**	**4**		0.610	**0.089**
Little tern							0.800	**0.518**	2			
Common guillemot	0.560	0.792	0.917				**0.939**	**0.672**	6	0.079	0.580	0.007
Razorbill	*	0.630					**0.895**	**0.570**	5	0.030	**0.088**	**0.047**
Black guillemot	0.731						0.870	**1.298**	5		0.657	0.010
Atlantic Puffin	*	*	0.709	0.760	0.805		**0.906**	**0.617**	**5**	0.078	0.230	**0.042**

The majority of the seabird species received high and intermediate data scores for adult survival rates, with the exception of little gull and great black-backed gull. Excluding the common eider, the survival rates of the sea duck species were less well resolved, especially for the juvenile and immature age class. The juvenile and immature survival rates for great northern diver, northern fulmar, Arctic skua, common tern, razorbill and Atlantic puffin were only available as return rates between fledging and recruitment. Juvenile and immature survival rates were not available for goldeneye, long-tailed duck, velvet scoter, Manx shearwater, little gull, black-headed gull, great black-backed gull, Arctic tern and little tern. For a number of species regional differences in population trends could not be wholly attributed to local differences in productivity (Cook & Robinson 2010). In these cases it may be important to consider regional variation in recruitment or survival rates when parameterising population models. To investigate regional differences in survival rates would require long-term mark-recapture studies from several colonies (as recommended by Robinson & Baillie 2012); alternatively this could be explored by delineating non-breeding season populations (e.g. Furness 2015).

For all of the seabird species, with the exception of little gull, there was good information on productivity. In contrast, the productivity rates of the sea duck species were less well resolved. An estimate for the modal age of recruitment was available for all of the species considered, although the variability between individuals was less well resolved. The majority of these estimates received a high or intermediate data score, with the exception of long-tailed duck, velvet scoter, Manx shearwater, great crested grebe, great skua, Sandwich tern and little tern. However, many of the reported estimates are from relatively old studies and in a number of cases national census data indicate that the trajectory of the UK population has changed since their publication. It will be important to consider any temporal or regional differences in population trend as caveats of population models that are developed using the reported values. For the majority of species there is considerably less information on the incidence of missed breeding and breeding dispersal. These variables are likely to vary markedly between colonies, depending on local population trend and resource availability. Elevated rates of natal dispersal were identified in northern fulmar, black-legged kittiwake, common gull, lesser black-backed gull, herring gull, arctic tern, common guillemot and black guillemot. This process is likely to influence the reliability of population models if it is not considered.

7. Review of density dependence in seabirds

Demographic processes such as growth, survival, productivity and recruitment are density-dependent if their rates change as a function of the density or number of individuals in a population. Density-dependent processes are said to be compensatory if they stabilise the population around the long-term average, i.e. they reduce the rate of population growth as numbers rise and increase the rate as numbers fall (Lack 1954). This negative feedback on population size is important to management because it operates to offset the losses of individuals from the population, for example, a lowered population density may cause a temporary increase in survival or productivity in the remaining individuals, supporting an increase in the population size back towards the long-term average. However, density-dependent processes may also be depensatory, slowing the rate of population growth at lower population densities. This positive feedback on population size is especially important for depleted populations and endangered species, because it acts to accelerate further population decline and can delay recovery. The expression of these opposing processes in a population may also change in relation to population size, for example a population can be depensatory at low population levels and exactly compensating at higher population levels. This mechanism may result in an optimal population density, where survival and productivity rates decrease at high and low densities (Figure 1, Allee 1931, 1938), as documented in herring gull (Parsons 1976) and common tern (Becker & Finck 1986; Sudmann 1998).

In terms of population modelling, these effects can be captured by making the demographic parameters functions of population density. Consequently, the resulting frameworks are nonlinear because populations can no longer grow exponentially. At the time of publication, the majority of population models used to assess the potential impacts of wind farms on seabirds did not consider density dependent mechanisms. Existing models have included a density-dependent adjustment on juvenile survival rates because this will also capture any associated changes in natal dispersal and recruitment (e.g. Freeman *et al* 2014). However, the contexts for including density-dependent mechanisms in population models are not well understood. This review consolidates the empirical evidence of compensatory and depensatory regulation in seabirds and sea ducks. The focus is primarily on the species considered in the individual species accounts listed in Section B, but also includes studies on similar species.

7.1. Empirical evidence of density-dependence

Depensation

Depensation is typically exhibited in populations that have been significantly depleted in size. The underlying mechanism is related to benefits associated with conspecific presence such as: anti-predator vigilance or aggression; cooperative feeding or resource defence; social thermoregulation; collective improvement of the environment; increased availability of mates; nonspecific enhancement of reproduction; and reduction of inbreeding, genetic drift, or loss of integrity by hybridisation (Stephens *et al* 1999). The majority of empirical examples of depensation in seabirds relate to anti-predator vigilance (or colonial defence) increasing rates of productivity. We identified this relationship in 14 studies that considered eight different species of seabird and sea duck (Table 34; Table 35). However, there were also twelve studies that did not identify a relationship between productivity and colony size (Table 34; Table 35). Therefore, the presence of depensation is likely to be context dependent, for example anti-predator vigilance will only be important if a population is influenced by predation (Ryder & Ryder 1981; Harris & Wanless 1988). For species and populations that are influenced by depensation, the influence of colonial defence is likely to select for higher levels of breeding synchrony in order to align colonial nest defence with specific points in the breeding season (e.g. Patterson 1965). Depensation is likely to follow a similar function shape to the Allee effect (Figure 1).

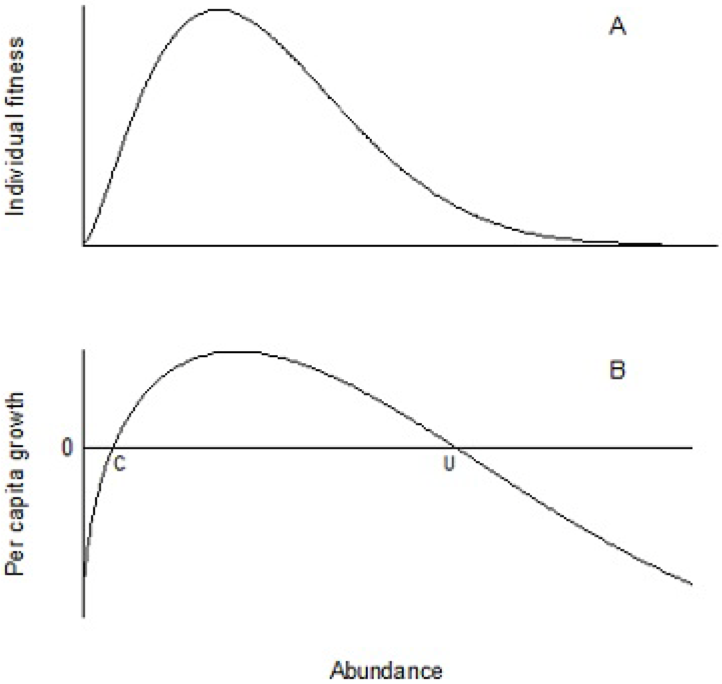

Figure 1. The Allee effect (replotted from Stephens *et al* 1999). A. At low densities, the benefits from the addition of each successive individual outweigh the costs, such that there is a net gain in individual fitness, and fitness is highest at intermediate numbers or densities. B. In this case, population growth rate may also be low at low levels of abundance, resulting in two equilibria, a lower unstable equilibrium (C) and an upper, stable equilibrium (U).

Compensation

Compensation is characterised by demographic changes that stabilise the population around the long-term average. We identified nine studies on five different species of seabirds and sea ducks that demonstrated a decrease in colony growth rates as a colony increases in size (Table 34; Table 35). For example, Porter and Coulson (1987) found the growth of a kittiwake colony to be limited by the number of attractive sites in the dense centre (also see Coulson 1983). Similarly, Chabrzyk and Coulson (1976) found that herring gulls have greater difficulty establishing a territory in a dense colony (also see Duncan 1978; Coulson *et al* 1982; Raven & Coulson 1997). Coulson and Wooller (1976) attributed this relationship to reduced survival rates. A decrease in survival rates associated with an increase in colony size was identified in seven species of seabird and sea duck across eight studies (Table 34; Table 35). However, the influence of density-dependent regulation on survival rates can also be age-specific, such that juvenile birds are more impacted by limited resources than birds older than age one (Marvelde *et al* 2009). An alternative explanation is that chicks fledge from denser colonies at significantly lighter weights (Gaston *et al* 1983). Other studies indicate that density-independent factors may have a stronger impact on survival rates at higher colony densities (Frederiksen & Bregnballe 2000b; Barbraud & Weimerskirch 2003). Here, a saturation of high quality wintering habitat may force some birds to winter in less good habitat where they are more vulnerable to poor environmental conditions. A decrease in population growth rate as the number of individuals increased can also be attributed to reduced productivity rates. This relationship was identified in seven species of seabird and sea duck across eight studies (Table 34; Table 35). In eiders this relationship was attributed to increased transmission of disease (Pöysä & Pöysä 2002) and a decrease in clutch size (Coulson 1984, 1999). In the other identified studies, this relationship was attributed to elevated disturbance from non-breeding conspecifics and predation of eggs and chicks by neighbouring pairs. However, there are also twelve studies that did not identify a relationship between productivity and colony size (Table 34; Table 35). This result may reflect a colony that is not limited by resources. Alternatively it may indicate that density-dependence operates through variation in recruitment or immigration.

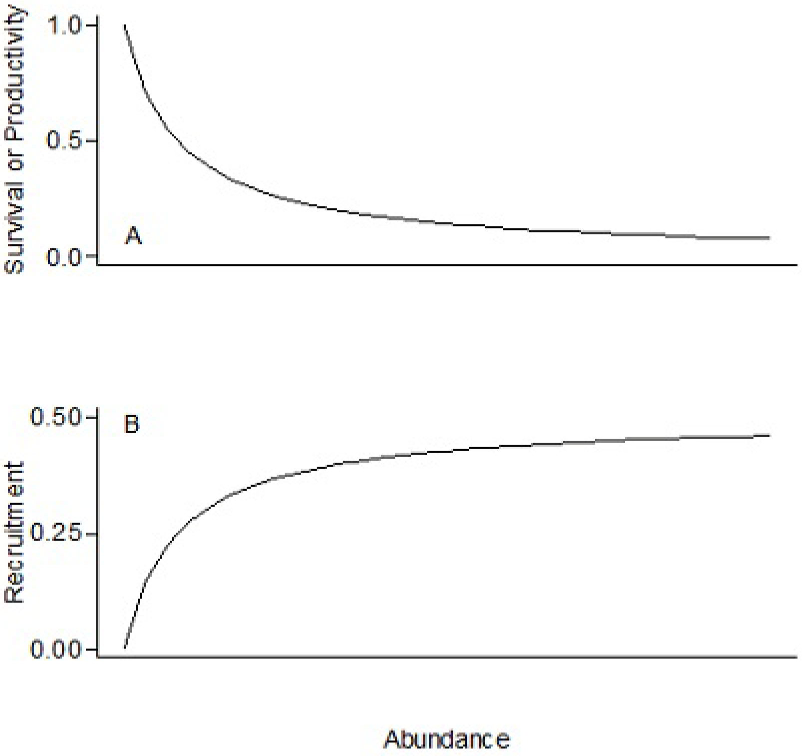

Figure 2. The compensatory (Beverton-Holt function) effect. A. At high population densities, increasing competition and depletion of resources will often result in decreased survival or productivity rates. B. Recruitment increases as the survival or productivity rates decrease resulting in a population in equilibrium (compensatory).

Elevated mortality rates in the breeding population may not influence the total colony size because new recruits take over high quality nesting sites left vacant, and thus achieve productivity rates that are more similar to experienced birds (e.g. Coulson 1971). Here, any changes in survival are exactly offset by a change in recruitment or immigration (Fig. 2; Beverton-Holt function; Beverton & Holt 1957). This process can be identified by quantifying the age of recruitment, the proportion of non-breeding individuals in the population and the rates of immigration and emigration. An increase in the age of recruitment following a decrease in resource availability (such as nesting sites) was identified in nine species (Table 34). Similarly, an increase in recruitment from the non-breeding pool of adults in response to an increase in resource availability was identified in five species (Table 34). Finally, an increase in emigration in response to a decrease in resource availability was identified in four species, and an increase in immigration in response to an increase in resource availability was identified in six species. There were also three studies that did not report a relationship between colony size and the proportion of adults in the non-breeding pool (Table 34; Table 35). These studies considered colonies that were increasing at the time of publication. Consequently, the pool of non-breeding adults may have already been exhausted, and thus colony growth was predominantly driven by immigration.

7.2. Potential impacts from offshore wind farm developments

Wind farm developments act in a density-independent manner, i.e. they may affect a proportion of the population regardless of its size. This section offers a brief evaluation of the potential density-dependent processes that a seabird colony may experience in response to offshore wind farm developments.

Reduced survival

In the short term, any elevated rates of mortality may not manifest in the size of the breeding population. This is because any relative increase in resource availability, such as high quality nesting sites (caused by death of one or both of the breeding pair occupying them) will be offset by an increase in productivity, a decrease in the age of recruitment, an increase in the recruitment of individuals from the non-breeding pool or an increase in immigration. However, once these mechanisms have become saturated (i.e. there are no more individuals in the non-breeding pool) a change in the population size is likely to result. In colonies that experience depensatory regulation, this has the potential to be highly destabilising.

Reduced productivity

Seabirds and sea ducks often increase their breeding dispersal in response to failed breeding (e.g. Boulinier *et al* 2008; Dow & Fredga 1983; Perrins & Smith 2000). If wind farm developments decrease rates of productivity (e.g. through displacement of foraging birds, Drewitt & Langston 2006), this may also drive changes in local emigration rates and lead to further reductions in the colony size as birds redistribute. The continued level of disturbance from the offshore development may also prevent colonies from being compensated by elevated levels of immigration or recruitment. This may be especially pertinent to colonies that are within designated Special Protection Areas, where there is a commitment to protect selected colonies.

Conclusions and future directions

This review of density-dependent regulation in seabirds and sea ducks identifies a large number of studies that report compensatory mechanisms to influence rates of productivity, survival, recruitment and dispersal. However, the review highlights that there is not a simple mechanism of density-dependence that can be applied uncritically in all situations. Rather the influence of density-dependence relates to a complex interaction between resource availability, colony size and other local factors, such as predation. The dynamics of a colony should be evaluated if density-dependence is to be realistically incorporated within population models that evaluate the impacts of wind farm developments. The identification of compensation was most prevalent in studies that examined rates of recruitment. This means that per capita growth rates will be higher at low colony densities, and in terms of population modelling, recovery will be quicker if compensatory processes are considered. Thus, in many cases adopting a density-independent modelling framework will overestimate potential impacts. However, in certain species and populations there was also clear evidence that depensatory mechanisms operate on rates of productivity. Here, density-independent models will underestimate potential impacts. Depensatory regulation of productivity rates was reported in almost twice the number of studies that reported compensatory regulation of this demographic parameter, albeit across a similar number of species (Table 35). The ability of compensatory density-dependent mechanisms to support population recovery may largely depend on the demographic rate impacted, and whether the severity of the impact changes over time (e.g. through habituation). Further modelling of compensatory and depensatory functions on different demographic rates is recommended in order to elucidate their influence on the resulting population size.

Table 34. Studies assessing the effect of population density or population size on seabird demography.

Species	Density-dependent process	Demographic process	Effect	Reference
Great cormorant	Compensation	Age of recruitment	Yes	Frederiksen & Bregnballe 2000a
European shag	Compensation	Age of recruitment	Yes	Swann & Ramsay 1983b, cited Swann & Ramsay 1983a
Black-headed gull	Compensation	Age of recruitment	Yes	Lebreton *et al* 1990
Black-legged kittiwake	Compensation	Age of recruitment	Yes	Porter & Coulson 1987
Herring gull	Compensation	Age of recruitment	Yes	Duncan 1981; Chabrzyk & Coulson 1976; Coulson *et al* 1982
Audouin's gull	Compensation	Age of recruitment	Yes	Oro & Pradel 2000
Common guillemot	Compensation	Age of recruitment	Yes	Swann & Ramsay 1983a
Razorbill	Compensation	Age of recruitment	Yes	NERC 1977, cited Swann & Ramsay 1983a
Atlantic puffin	Compensation	Age of recruitment	Yes	Harris 1983; Kress & Nettleship 1988
Great cormorant	Compensation	Emigration	Yes	Frederiksen & Bregnballe 2000a
Common guillemot	Compensation	Emigration	Yes	Crespin *et al* 2006b
Northern gannet	Compensation	Emigration and Immigration	Yes	Murray & Wanless 1997; Moss *et al* 2002
Great skua	Compensation	Emigration and Immigration	Yes	Phillips *et al* 1999b; Meek *et al* 2011
Great cormorant	Compensation	Immigration	Yes	Hénaux *et al* 2007
European shag	Compensation	Immigration	Yes	Aebischer 1995
Herring gull	Compensation	Immigration	Yes	Chabrzyk & Coulson 1976; Duncan 1978; Coulson *et al* 1982
Atlantic puffin	Compensation	Immigration	Yes	Harris 1983
Common eider	Compensation	Incidence of missed breeding	Yes	Coulson 1984

Species	Density-dependent process	Demographic process	Effect	Reference
Common eider	Compensation	Incidence of missed breeding	Yes	Coulson 1984
Goldeneye	Compensation	Incidence of missed breeding	Yes	Eriksson 1979
Northern fulmar	Compensation	Incidence of missed breeding	No	Ollason & Dunnet 1983
Great skua	Compensation	Incidence of missed breeding	Yes	Hamer *et al* 1991
Lesser black-backed gull	Compensation	Incidence of missed breeding	No	Calladine & Harris 1997
Herring gull	Compensation	Incidence of missed breeding	No	Calladine & Harris 1997
Arctic tern	Compensation	Incidence of missed breeding	Yes	Monaghan *et al* 1992
Great northern diver	Compensation	Population growth	Yes	Grear *et al* 2009
Velvet scoter	Compensation	Population growth	Yes	Hartman *et al* 2013
Northern gannet	Compensation	Population growth	Yes	Lewis *et al* 2001
Black legged kittiwake	Compensation	Population growth	Yes	Coulson 1983; Porter & Coulson 1987
Herring gull	Compensation	Population growth	Yes	Chabrzyk & Coulson 1976; Duncan 1978; Coulson *et al* 1982; Raven & Coulson 1997
Great northern diver	Compensation	Productivity	Yes	Evers 2004
Common eider	Compensation	Productivity	Yes	Coulson 1984; 1999; Pöysä & Pöysä 2002
Goldeneye	Compensation	Productivity		Andersson & Eriksson 1982
Northern fulmar	Compensation	Productivity	No	Falk & Møller 1997; Lewis *et al* 2009

Species	Density-dependent process	Demographic process	Effect	Reference
Southern fulmar	Compensation	Productivity	No	Jenouvrier *et al* 2005
Northern gannet	Compensation	Productivity	No	Nelson 1966
Northern gannet	Compensation	Productivity	No	Nelson 1966
European shag	Compensation	Productivity	No	Potts *et al* 1980
Arctic skua	Compensation	Productivity	No	Jones 2003
Great skua	Compensation	Productivity	No	Hamer & Furness 1991
Great skua	Compensation	Productivity	Yes	Jones 2003
Black-legged kittiwake	Compensation	Productivity	No	Coulson 1971
Audouin's gull	Compensation	Productivity	No	Oro *et al* 1999
California gull	Compensation	Productivity	No	Pugesek & Diem 1983
Herring gull	Compensation	Productivity	Yes	Kilpi 1989
Glaucous-winged gull	Compensation	Productivity	Yes	Hunt & Hunt 1976
Great black-backed gull	Compensation	Productivity	Yes	Butler & Trivelpiece 1981
European shag	Compensation	Recruitment from non-breeding population	Yes	Potts *et al* 1980
Great skua	Compensation	Recruitment from non-breeding population	Yes	Klomp & Furness 1992b
Black legged kittiwake	Compensation	Recruitment from non-breeding population	Yes	Porter & Coulson 1987
Audouin's gulls	Compensation	Recruitment from non-breeding population	Yes	Tavecchia *et al* 2007
Common guillemot	Compensation	Recruitment from non-breeding population	Yes	Votier *et al* 2008
Common eider	Compensation	Survival	Yes	Milne 1974
Blue petrel	Compensation	Survival	Yes	Barbraud & Weimerskirch 2003
Great cormorant	Compensation	Survival	Yes	Frederiksen & Bregnballe 2000b

Species	**Density-dependent process**	**Demographic process**	**Effect**	**Reference**
Black-legged kittiwake	Compensation	Survival	Yes	Coulson & Wooller 1976; Coulson 2001
Mediterranean gull	Compensation	Survival	Yes	Marvelde *et al* 2009
Common guillemot	Compensation	Survival	Yes	Crespin 2006b
Atlantic puffin	Compensation	Survival	Yes	Harris & Wanless 1991; Breton *et al* 2006
Atlantic puffin	Compensation	Survival	Yes	Harris & Wanless 1991; Breton *et al* 2006
Arctic skua	Depensation	Productivity	Yes	Phillips *et al* 1998
Black-legged kittiwake	Depensation	Productivity	Yes	Coulson & White 1958; Porter 1990; Falk & Møller 1997; Frederiksen *et al* 2004a
Black-headed gulls	Depensation	Productivity	Yes	Patterson 1965
Sandwich terns	Depensation	Productivity	No	Veen 1997
Common terns	Depensation	Productivity	Yes	Becker 1995; Hernández-Matías *et al* 2003
Common guillemot	Depensation	Productivity	Yes	Birkhead 1977
Common guillemot	Depensation	Productivity	No	Harris & Wanless 1988
Atlantic puffin	Depensation	Productivity	Yes	Harris 1980
Herring gull	Depensation/ Compensation	Productivity	Yes	Parsons 1976
Common tern	Depensation/ Compensation	Productivity	Yes	Becker & Finck 1986; Sudmann 1998

Table 35. Studies assessing the effect of population density or population size on seabird demography. Numbers detail the number of studies identified.

Demographic rate	**Density-dependent mechanism**	**Divers and grebe**	**Sea ducks**	**Procellariiformes**	**Gannet**	**Cormorants**	**Skuas**	**Large gulls**	**Small gulls**	**Terns**	**Auks**	**Total**
Population growth	Compensation	1	1		1			4	2			**9**
	Absent											**0**
Productivity	Depensation						1	1	6	4	2	**14**
	Compensation	1	3				1	3				**8**
	Absent			3	1	1	2	1	2	1	1	**12**
Survival	Compensation		1	1		1			2		3	**8**
	Absent											**0**
Recruitment	Compensation		2			2	1	4	2	1	4	**16**
	Absent			1				2				**3**
Dispersal	Compensation				2	3	2	3			2	**12**
	Absent											**0**

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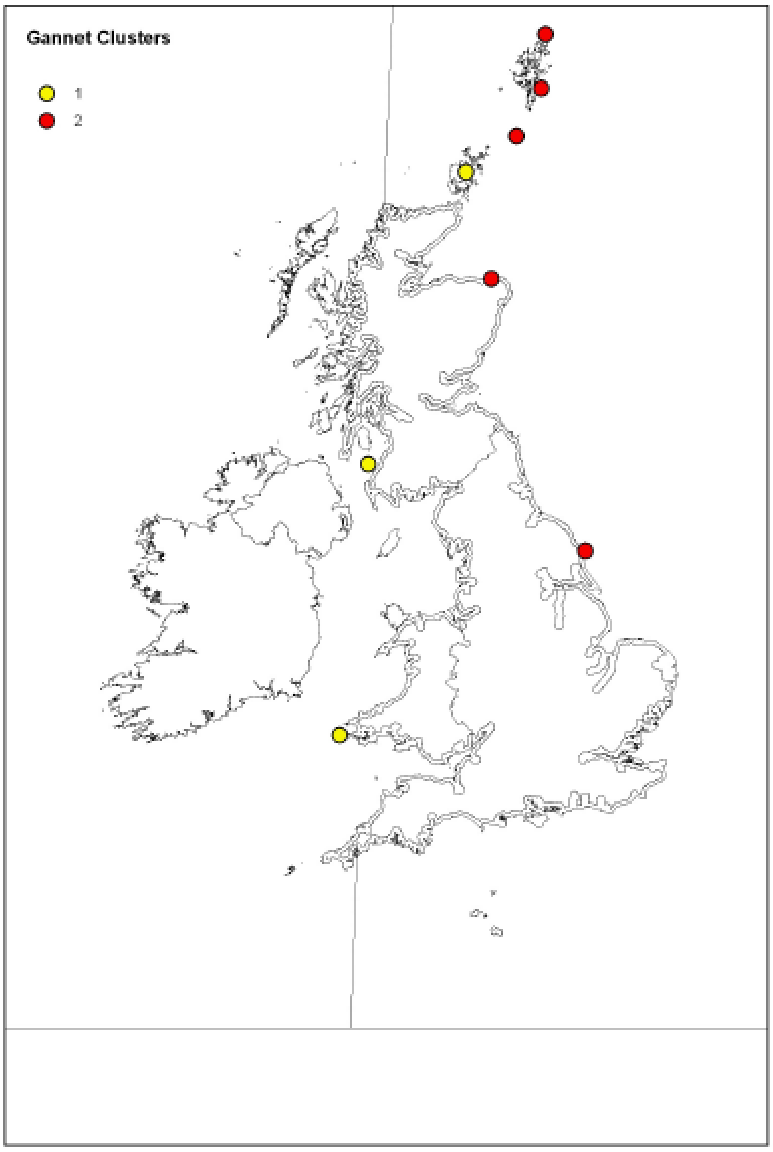
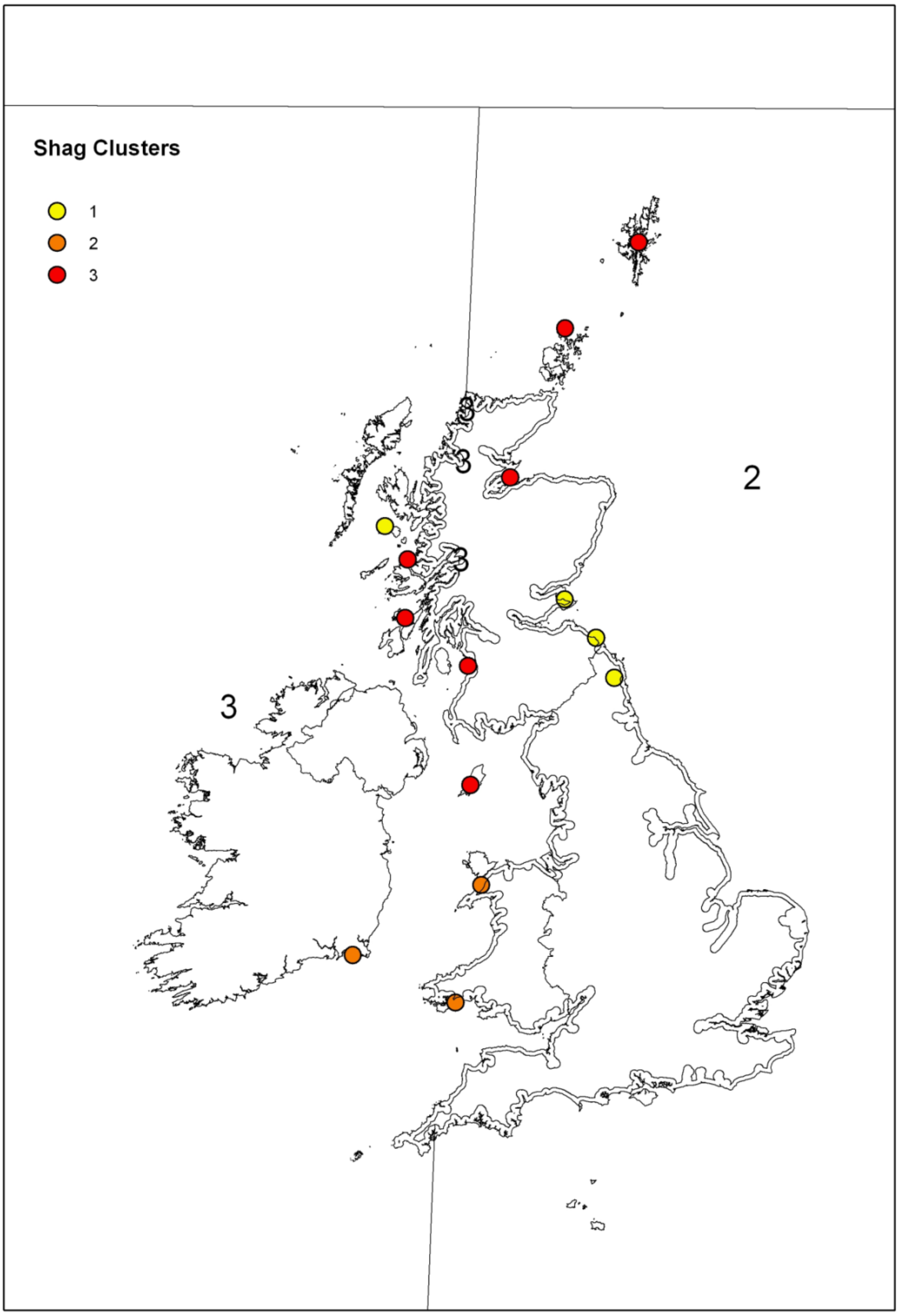
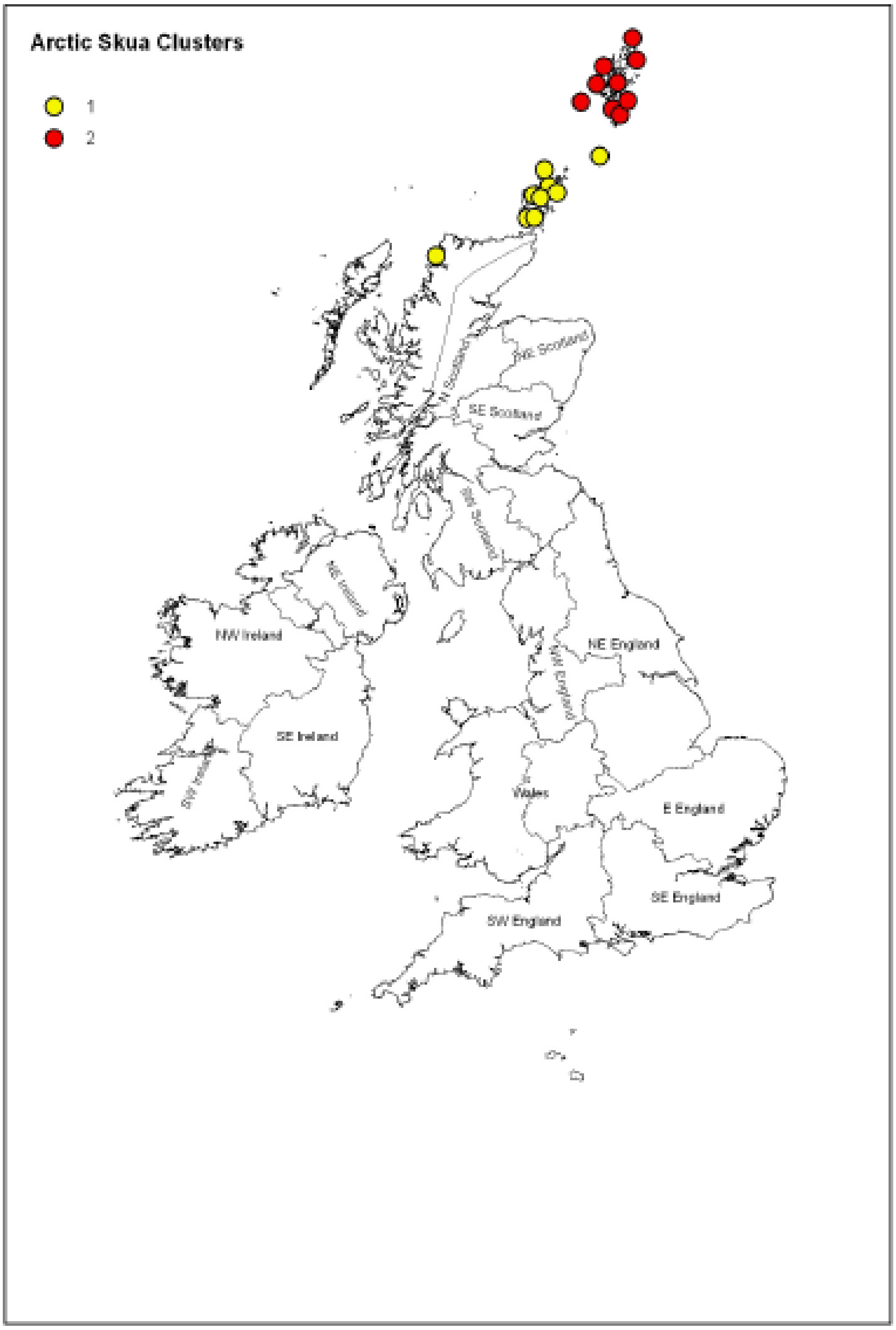
Appendix



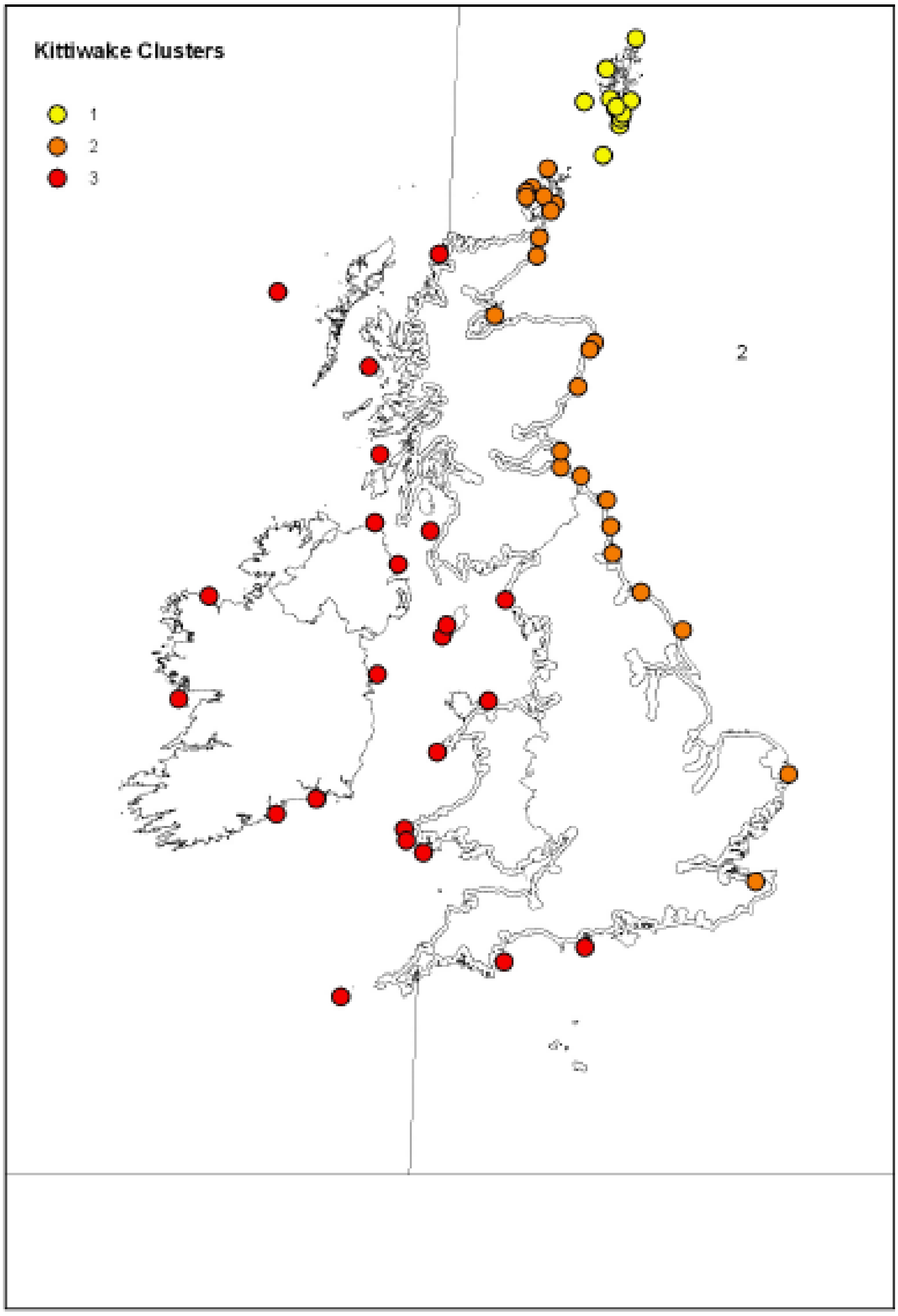
S1. Colony membership of clusters based on analysis of northern gannet breeding success data, overlaid with existing OSPAR monitoring regions (taken from Cook & Robinson 2010). The West Coast cluster is broadly contiguous with the Celtic Sea OSPAR region, with the addition of Orkney, and the East Coast Cluster is broadly contiguous with the Greater North Sea OSPAR region. The West Coast cluster encompasses the North West England and Isle of Man, North West Scotland, Orkney, South West Scotland, Wales, South West Ireland and South East Ireland SMP regions, as well as regional seas 4, 5, 6 and part of 7. The East Coast Cluster encompasses the East England, North East England, South East Scotland, North East England and Shetland SMP regions as well as Regional Seas 1, 2 and part of 7.



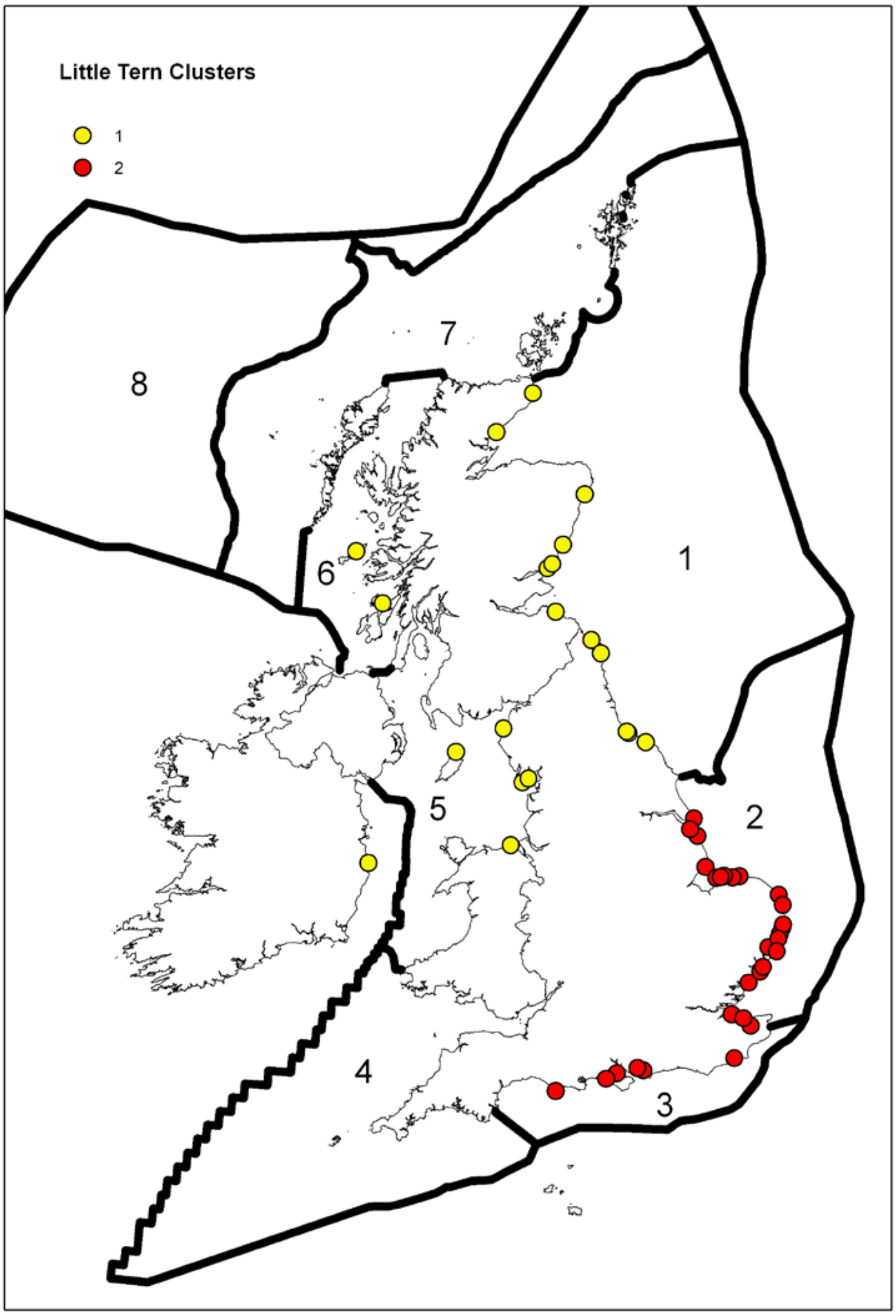
S2. Colony membership of clusters based on analysis of European shag breeding success data, overlaid with existing OSPAR monitoring regions (taken from Cook & Robinson 2010). The East Coast of Scotland and Orkney and the Shetland clusters are within the Greater North Sea OSPAR Region and the West Coast of England and Wales cluster is within the Celtic Sea (Cook & Robinson 2010).



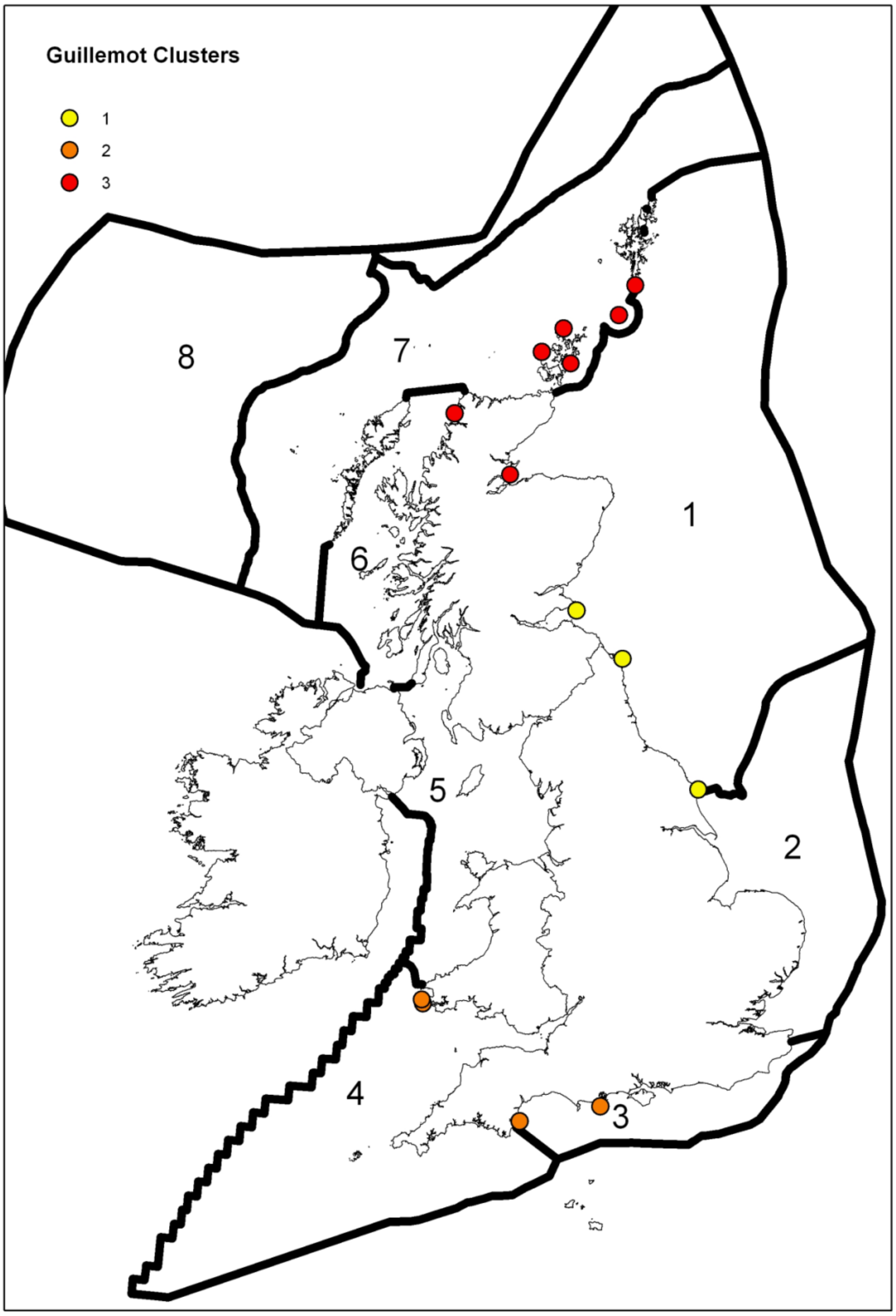
S3. Colony membership of clusters based on analysis Arctic skua breeding success data, overlaid with existing Regional Seas monitoring regions (taken from Cook & Robinson 2010). The Shetland Cluster is within the Greater North Sea OSPAR region and Regional Seas 7. It is contiguous with the Shetland SMP region. The Orkney and North Scotland cluster is split between the Celtic Sea and Greater North Sea OSPAR regions and Regional Seas 6 and 7. It encompasses the North West Scotland and Orkney SMP regions.



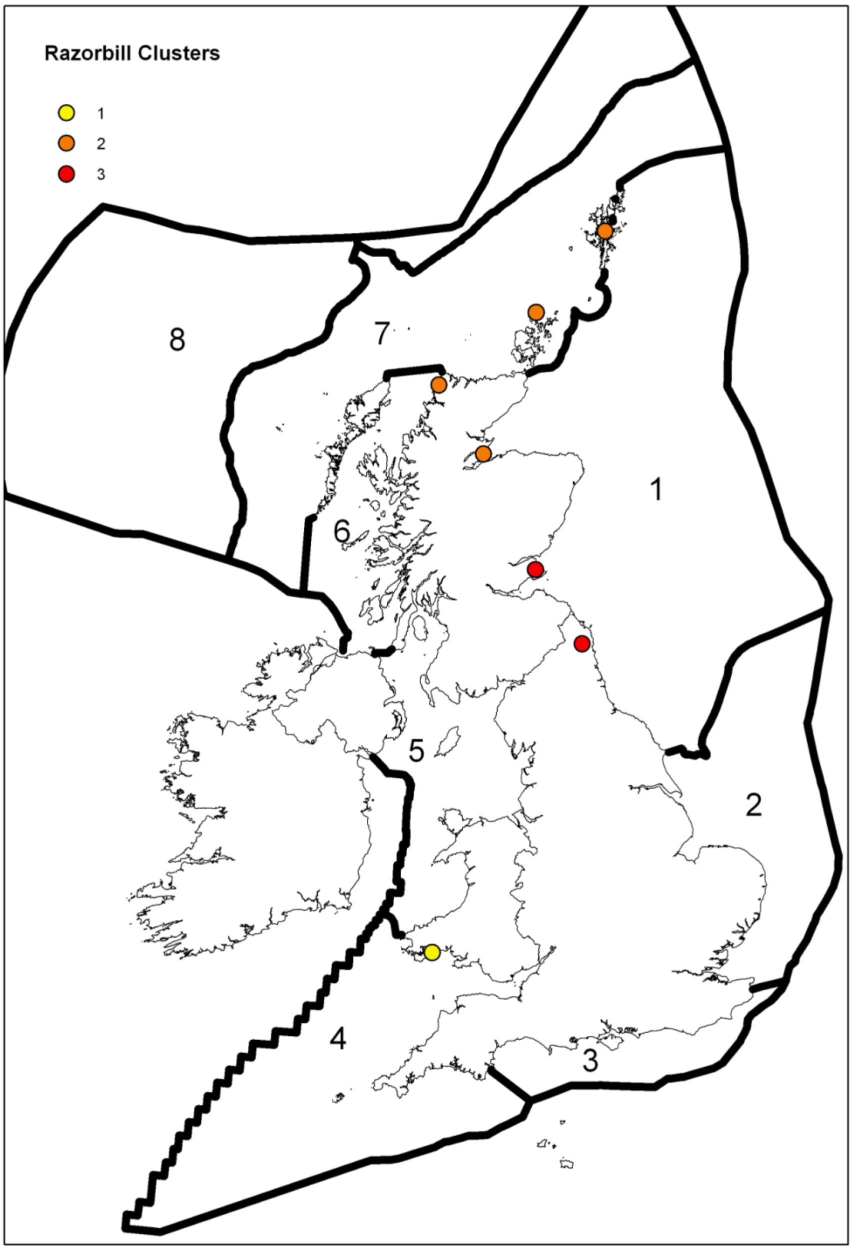
S4. Colony membership of clusters based on analysis of black-legged kittiwake breeding success data, overlaid with existing OSPAR monitoring regions (taken from Cook & Robinson 2010). The Shetland population falls within regional sea 7 and the Shetland SMP region. The Eastern population encompasses regional seas 1, 2 and 3 and parts of regional seas 4 and 7 as well as the South East England, East England, North East England and North East Scotland SMP regions and parts of the North Scotland and South West England SMP Regions. The Western population encompasses regional seas 5 and 6 and parts of regional seas 4 and 7 as well as the Wales, South East Ireland, North West England, South West Scotland and North West Scotland SMP regions and parts of the South West England, North Scotland and Orkney SMP regions.



S5. Colony membership of clusters based on analysis of little tern breeding success data, overlaid with existing OSPAR monitoring regions (taken from Cook & Robinson 2010). The Southern population encompasses the South West England, South East England, East England and North East England SMP (SMP) regions and Regional Seas 2, 3 and 4, all of which are experiencing declines in breeding success. The Northern population encompasses the North Scotland, North East Scotland, North West England, South East Scotland, South West Scotland, Wales and South East Ireland SMP regions and Regional Seas 1, 5, and 6, in which breeding success remains relatively stable.



S6. Colony membership of clusters based on analysis common guillemot breeding success data, overlaid with existing Regional Seas monitoring regions (taken from Cook & Robinson 2010). The North of Scotland group encompasses regional seas 6 and 7 and part of regional sea 1 and the North Scotland, North West Scotland, Orkney and Shetland SMP regions. The East coast group encompasses regional sea 2 and part of regional sea 1 and the South East Scotland and North East England SMP regions. The South West group encompasses regional seas 3, 4 and 5 and the South West England and Wales SMP regions.



S7. Colony membership of clusters based on analysis razorbill breeding success data, overlaid with existing regional seas monitoring regions. The North Scotland, Orkney and Shetland cluster is split between the Greater North Sea and Celtic Sea OSPAR regions. The East Coast of Scotland cluster is within the Greater North Sea OSPAR region and the South Wales cluster is within the Celtic Sea OSPAR region. The North Scotland, Orkney and Shetland cluster is split between regional seas 1, 6 and 7 and between the North West Scotland, North Scotland, North East Scotland, Orkney and Shetland SMP regions. The East Coast of Scotland cluster is within regional sea 1 and split between the South East Scotland and North East England SMP regions. The South Wales cluster is within regional sea 4 and the Wales SMP region.

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M Jitlal, S Burthe, S Freeman and F Daunt

marine scotland

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Testing and validating metrics of change produced by Population Viability Analysis (PVA)

Mark Jitlal, Sarah Burthe, Stephen Freeman and Francis Daunt

Centre for Ecology & Hydrology
NATURAL ENVIRONMENT RESEARCH COUNCIL

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Testing and Validating Metrics of Change Produced by Population Viability Analysis (PVA)
Ref: CR/2014/16

Final report to Marine Scotland Science
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Executive Summary

- The aim of this research project was to review the use of Population Viability Analysis (PVA) metrics in the context of assessing the effect of offshore renewable developments on seabirds and to test PVA metric sensitivity to mis-specification of input parameters. The most useful metrics in this context are those that are least sensitive to such mis-specification, enabling more robust assessment of offshore renewable effects.
- Recent work has tested PVA metric sensitivity using a simulation approach. To complement these findings, the objective in this project was to test metric sensitivity using real-world data. This approach is useful where one wishes to understand a specific region where real data are available, or where one wishes to address generic questions with real data. If the same metrics show low sensitivity in models of real world data as in simulation models, then this would provide re-assurance that these metrics are the most appropriate for use in assessments.
- Five study species were selected: black-legged kittiwake *Rissa tridactyla*; common guillemot *Uria aalge*; razorbill *Alca torda*; herring gull *Larus argentatus* and European shag *Phalacrocorax aristolelis*. Of these, the first four were considered in population modelling in the Forth/Tay region in a previous Marine Scotland Science project (Freeman *et al.* 2014). Similar models have, in the interim, also been fitted for shags in this region so this species was also considered. The SPAs considered in this report were Buchan Ness to Collieston Coast SPA, Fowlsheugh SPA, Forth Islands SPA and St Abb’s Head to Fastcastle SPA.
- Data on abundance, survival and productivity were collated from a variety of sources. Regular or sporadic counts were available from all sites, based on whole colony or plot counts. Productivity was available from all four SPAs for kittiwakes, and for European shags at two SPAs, otherwise data on demographic rates was limited to the Isle of May in the Forth Islands SPA.

- All models were fitted using a Bayesian approach in the software R/WinBUGS. Model fitting was in 'state-space' form, which allows for 'observation error' and environmental stochasticity simultaneously within the same model. Models forecasted the population size for each species at each SPA, for 25 years from 2016 to 2041. Adult survival was set to decline by one of a range of specified rates equating to offshore renewable effects, namely 0% (i.e. no change), 0.5%, 1%, 2% and 3%. Annual productivity was set to decline by 0%, 1%, 2%, 3% and 5%.
- Previous work has indicated that ratio PVA metrics are less sensitive than probabilistic PVA metrics. Accordingly, we tested the sensitivity of six PVA metrics, comprising two ratio metrics (median of the ratio of impacted to un-impacted annual growth rate; median of the ratio of impacted to un-impacted population size); two metrics related to the ratio metrics (median difference in impacted and un-impacted annual growth rates; median difference between impacted and un-impacted population size) and two probabilistic metrics (probability of a population decline exceeding 10%, 25% or 50%; centile for un-impacted population which matches the 50th centile for the impacted population).
- Sensitivity of the six PVA metrics was assessed in relation to mis-specification of input parameters. We considered adult mortality (the complement of survival, since survival is high in seabirds and % increases are limited by the constraint of lying below a survival rate of 1) and productivity to differ from those of the baseline by: -30%, -20%, -10%, 10%, 20% and 30%. We then assessed sensitivities in relation to population status, combining data from all species/SPAs for which we achieved model convergence. Finally, we assessed PVA sensitivities in relation to scenarios of change resulting from the renewables development (i.e. the effect size).
- The state-space modelling approach proved extremely powerful in forecasting population sizes, in particular where censuses were regular. Even in cases where censuses were sporadic, the models generally performed well, though for three species/SPA populations the models would not converge successfully.
- The two ratio metrics were least sensitive to mis-specification in input parameters. They performed well in populations of different status, and under different scenarios of change. The two difference metrics were not readily interpretable, but proved useful when growth rates or population size estimates were small. The probabilistic metrics were more sensitive to mis-specification to input parameters than the ratio PVA metrics. The 'probability of a population decline' metric has been widely used in assessments but proved highly sensitive to mis-specification. The metric representing the

centile from the un-impacted population size equal to the 50th centile of the impacted population size at the end of the wind farm showed moderately low sensitivity to mis-specification of survival and productivity. It performed considerably better than the other probabilistic metric with markedly lower sensitivity to mis-specification, population status and renewables effect size. However, it was more sensitive than ratio metrics, and in some cases showed unstable sensitivity which was less apparent in ratio PVA metrics.

- We recommend that those undertaking assessments consider the relative performance of different metrics with respect to sensitivity to mis-specification of input parameters. Of the two ratio and two probabilistic metrics, the ratio metric ‘median of the ratio of impacted to un-impacted annual growth rate’ was least sensitive, followed by the ratio metric ‘median of the ratio of impacted to un-impacted population size’ and then the probabilistic metric ‘centile for un-impacted population which matches the 50th centile for the impacted population’. If these are used in assessments in future, we recommend that interpretation should factor in their relative sensitivities. Furthermore, a priority for future research would be to analyse the probabilistic metric using simulations, to assess whether the same results are found as in this study. The probabilistic PVA metric ‘probability of a population decline’ was much more sensitive than the other three and is not recommended for use in this context. Finally, we recommend that the two PVA metrics related to the ratio metrics (median difference in impacted and un-impacted annual growth rates; median difference between impacted and un-impacted population size) are used since they are estimable when ratios are being calculated and are useful in some circumstances.

1. Introduction

1.1 Policy Context

The Scottish Government has set a target of 100% of Scottish demand for electricity to be met by renewable sources by 2020. The Scottish Government has a duty to ensure that offshore renewable developments are achieved in a sustainable manner. Scottish Ministers have consented offshore renewable energy sites under Section 36 of the Electricity Act 1989. A licensing process was followed that included the examination of Environmental Statements (ES) which consider the potential impacts and mitigation strategies of the proposed developments.

Offshore renewable developments have the potential to impact on seabird populations that are protected by the EU Birds Directive [2009/147/EC], notably from collisions with turbine blades and through displacement from important habitat (Drewitt & Langston 2006; Larsen & Guillemette 2007; Masden *et al*. 2010; Grecian *et al*. 2010, Langton *et al*. 2011, Scottish Government 2011). Other factors of concern are barrier effects to the movement of migrating or commuting birds, disturbance during construction and operation, toxic and non-toxic contamination and negative effects of developments on the distribution and abundance of prey. Set against these, positive effects may be apparent, in particular if developments result in downstream changes to the physical environment that increase biomass of lower tropic levels (Inger *et al*. 2009). Further, they may act as Fish Aggregating Devices (FADs) creating foraging opportunities for seabirds (Inger *et al*. 2009), though attracting seabirds may increase their vulnerability to other effects such as collision and noise (Scottish Government 2011). Species differ in the sensitivity to disturbance, with auks of intermediate vulnerability and gulls and terns of low vulnerability (Garthe & Hüppop 2004; Langston 2010; Furness *et al*. 2013). These potential effects are predicted to be particularly important for breeding seabirds that, as central place foragers, are constrained to obtain food within a certain distance from the breeding colony (Daunt *et al*. 2002; Enstipp *et al.* 2006).

To aid the future development of offshore renewables, Marine Scotland have developed draft Sectoral Marine Plans for offshore Wind, Wave and Tidal Energy (Scottish Government 2013b) that have involved identifying the available resources and key constraints at a national and regional level, then applying social, economic and environmental assessments to inform the development of plan options. These plans have been subject to a Sustainability Appraisal and public consultation exercise (Scottish Government 2013e) and are underpinned by detailed technical assessments including a Strategic Environmental Assessment (SEA; Scottish

Government 2013d), Habitats Regulations Appraisal (HRA; Scottish Government 2013a) and Socio-economic Assessment (Scottish Government 2013c).

The above analyses have synthesised the potential sensitivities of internationally important seabird populations in Scotland and recognised areas of uncertainty associated with these effects. Therefore, in order to evaluate potential interactions between offshore renewables and marine wildlife in future, Marine Scotland believes that further marine science is required to continue to reduce uncertainty and apply the appropriate level of precaution.

Population Viability Analysis (PVA) provides a robust framework that uses demographic rates to forecast future population levels, either under baseline conditions or under scenarios of change resulting from, for example, an offshore development (Maclean *et al.* 2007; Freeman *et al*. 2014). A sensitivity analysis of PVA metrics to variation in demographic parameters would enable regulators and their advisers to assess the utility of each of these metrics in determining whether a predicted effect is unacceptably large. Demonstrating the validity of these metrics would also ensure that PVA outputs are presented and interpreted in the most suitable way. The outcomes could then be fed back into designing future monitoring requirements. Furthermore, the outputs could inform the establishment of thresholds of acceptable change by regulators, although such an approach has been heavily criticised (Green *et al*. 2016). Finally, they could improve assessments of risk and uncertainty with respect to population viability in environmental assessments and help to ensure that the level of precaution applied is appropriate.

1.2 Project Objectives

An important component of consenting of proposed offshore renewable developments is an assessment of the population consequences on seabirds. Population Viability Analysis (PVA) provides a robust framework that uses assumed or estimated demographic rates (principally survival and productivity) in a mathematical model to forecast future population levels of a wild animal population, either under currently prevailing circumstances or as a consequence of some perturbation to the system (Maclean *et al*. 2007; Freeman *et al*. 2014). Stochastic PVA models are run many times selecting from a distribution of input parameters, resulting in outputs representing the mean, confidence intervals and all quantiles including the 50% (median).

The range of PVA metrics which have the potential to describe the magnitude of a predicted effect on a population include population size by some target date, change in size or growth rate between pairs of consecutive years, trend in population size, counterfactual/ratio of population size or growth rate, probabilities of population decline to below a specific level or a specific percentage of the starting population size, excess probabilities of population decline to below a specific level or a specific percentage of the starting population size, population level predicted to be exceeded with predefined probability (e.g. ‘as likely as not’, Mastrandrea *et al*. 2010) and posterior probabilities (or quantiles derived from them) for any of the above.

This PVA framework allows the sensitivity of these metrics to changes in demographic parameters, notably due to estimation error, to be estimated. This is important as all demographic parameters are estimated with uncertainty, and population change and PVA metrics are disproportionately affected by changes in the magnitude of each. Accordingly, the aim of this project is to review the range of metrics available in PVA analysis and evaluate the sensitivity of these metrics in the context of decision making frameworks.

The report will first review the literature regarding the range of metrics available for use by PVA analysis in the context of renewable assessment frameworks of seabirds. It will then examine the relative sensitivity of a subset of these metrics to mis-specification of input parameters (adult survival and productivity) using PVAs developed on protected seabird populations at SPAs in the Forth/Tay region. It will also assess the impact of mis-specification in the context of population status and effect size of offshore renewable development. Finally, the project will make recommendations on the usefulness and application of the range of metrics within an assessment framework, and make recommendations to inform future assessments that use PVA analysis based on the conclusions of the study.

2. Literature Review

2.1 Introduction

Population Viability Analysis (PVA) uses life-history or population growth rate data to parameterise a mathematical population model to estimate population size and extinction risk of a species into the future (Norton 1995; Beissinger & Westphal 1998; Boyce 2001). Specifically, PVAs have been used for several purposes including predicting the future size of an animal population, estimating the probability of a population going extinct over time, evaluating management strategies most likely to maximise population persistence or exploring how different assumptions consequently alter the viability of small populations (see (Coulson *et al.* 2001)). PVAs have been widely used in conservation biology and wildlife management, aided by the development of intuitive, widely available and user-friendly software packages, particularly to forecast risks of extinction for species of conservation concern (Ludwig 1999). PVAs are a valuable tool because they facilitate the predictive modelling of animal populations under alternative environmental, management or harvesting scenarios and hence can be used to evaluate the effectiveness or consequences of different management decisions. Thus, PVAs can be considered to be a type of risk assessment of the long-term viability of animal populations.

A wide range of models can be considered to be PVAs (Ralls *et al*. 2002). However, in its most common form, PVA utilises life-history parameters (for example growth rates, juvenile and adult mortality, adult fecundity rates etc.) for individuals in a population projection matrix to estimate population size into the future (Boyce 1992). Models can either be deterministic (demographic rates such as survival and reproduction are constant or are determined in a predictable manner) or stochastic (vital rates vary unpredictably over time). Stochastic PVA models, can include demographic stochasticity (e.g. variation between individuals that affects whether a bird survives a given year) and environmental stochasticity (environmental change that would affect all individuals in a group), and hence the variability in the parameters is important, not just the mean values (Maclean, Frederiksen & Rehfisch 2007). PVAs have been developed for a wide range of species from different taxa, including plants (Maschinski *et al.* 2006), invertebrates (for example, sea-urchins (Pfister & Bradbury 1996) and insects (Bauer *et al.* 2013)), amphibians (Pickett *et al.* 2016), reptiles (Enneson & Litzgus 2009), fish (Sweka & Wainwright 2014), birds (Wootton & Bell 2014) and mammals (Pertoldi *et al.* 2013). Although difficult to assess due to the term "PVA" or "Population Viability Analysis" not commonly being included as a keyword, birds appear to be the taxonomic group where PVAs have most commonly been applied. A crude search of Web of Science including the search terms "PVA AND Population Viability Analysis" plus the group (e.g. "mammal") returned 25 citations for plants; 15 for fish; three for reptiles; 38 for birds and 20 for mammals. PVAs have been extensively used in conservation and management with studies focusing on a broad range of topics including: investigating risk of extinction and population viability in small populations (Grayson *et al.* 2014); assessing the impact of different harvesting levels (York *et al.* 2016), predicting population sizes after reintroductions and enhancements (Halsey *et al.* 2015), assessing impacts of threats such as habitat loss (Naveda-Rodriguez *et al.* 2016), climate change (Marrero-Gomez *et al.* 2007) and disease (Haydon, Laurenson & Sillero-Zubiri 2002), assessing effectiveness of alternative management strategies (Ferreras *et al.* 2001); establishing conservation status and strategies (Bevacqua *et al.* 2015); establishing the effectiveness of conservation strategies under a fixed budget (Duca *et al.* 2009); and evaluating which demographic parameters population growth is sensitive to in order to inform management (Mortensen & Reed 2016). As a crude indication, a search in Web of Science found the most published references for the search term "*PVA and management*" (320 references), followed by "*PVA and conservation*" (247), "*PVA and population size*" (167), with few references for "*PVA and renewable energy*" (7) or "*PVA and wind farm(s)*" (2, both on terrestrial wind farms; but note that the majority of studies on PVAs and wind farms are undertaken as part of the planning process e.g. Habitats Regulation Assessments (in Scotland, the law in England and Wales calls them Assessments) and are not published in peer-reviewed journals but within so called "grey- literature").

The outputs of PVAs consist of a predicted population trajectory through time. A suite of metrics have been used to predict the changes in the population of the focal species, both for conservation purposes and as a result of a particular threat or management scheme. Note that the term "metric" is not widely used outside the sphere of PVAs for seabirds and wind farms, where it has broadly been defined (Cook & Robinson 2016a, 2016b) as any value or rule upon which a decision about whether or not a population level effect associated with the impacts of an offshore wind farm is deemed acceptable. We consider the metric to be a value or unit of measurement, and not a rule, and hence cannot be used as an effective search term. A review of the model outputs from general literature in the last five years found that many studies simply reported estimated population sizes or population growth rate for particular time periods (Lopez-Lopez, Sara & Di Vittorio 2012; Wootton & Bell 2014; Naveda-Rodriguez *et al.* 2016). A commonly reported metric was that of quasi-extinction or extinction thresholds, whereby a probability is given for a population declining below a particular threshold (e.g. 10%) after a certain time (e.g. 10 years) or the predicted time to extinction (Blakesley *et al.* 2010; Alemayehu

2013; Hu, Jiang & Mallon 2013; Beissinger 2014; Robinson *et al.* 2016). The difference in extinction probability under different scenarios was reported when comparing management regimes e.g. management Scenario 1 resulted in an X% higher extinction probability than Scenario 2 (Bazzano *et al.* 2014). Susceptibility to quasi-extinction (SQE) has been used to assess whether or not a population is at risk of declining to a specified level (quasi-extinction threshold), a metric which supposedly integrates both parameter uncertainty and stochasticity. This method uses parametric bootstrapping to determine 95% confidence limits of quasi-extinction and then the SQE is defined as the proportion of the bootstrap that indicates a high probability of quasi-extinction (set arbitrarily as >= 0.9 in this paper; Snover and Heppell (2009)).

There are a number of sources of uncertainty that are incorporated within stochastic PVA models (Boyce 1992). There are two main components of uncertainty in time series of demographic variables or population counts: observation and process uncertainty (also called observation and process error or variation). Observation uncertainty (or sampling uncertainty) describes noise in the data that arises due to imprecise or biased empirical data collection methods, for example detection difficulties due to terrain, weather conditions or observer experience and human error. Process uncertainty describes noise that is related to the real variation in the parameter and comprises the real drivers of population fluctuations that are of interest (Bakker *et al.* 2009; Ahrestani, Hebblewhite & Post 2013). Methods for incorporating uncertainty are continuing to advance, including methods for separating out parameter uncertainty and process variation e.g. Heard *et al.* (2013). Therefore, the results of such PVAs are probabilistic, for example risks, probabilities or likelihoods of population decline or extinction. Sensitivity analysis, which determines the amount of change in the model results in response to changes in model parameters, is an important component of PVAs (Saltelli & Annoni 2010; Aiello-Lammens & Akçakaya 2016). Sensitivity analysis can be used to prioritise and inform empirical data collection by establishing the importance of parameters with imperfect knowledge and parameters where improved precision would enhance model predictions. Sensitivity analysis also facilitates understanding and identification of life-history parameters that are highly influential on population size and future viability in order to inform and prioritise conservation or management strategies. Sensitivity analysis is achieved by perturbing the life-history parameters either via a local (one at a time) or global sensitivity analysis (see Mccarthy, Burgman & Ferson (1995); Wisdom, Mills & Doak (2000); Cross & Beissinger (2001); Naujokaitis-Lewis *et al.* (2009); Aiello-Lammens & Akçakaya (2016) for details). Global sensitivity analysis is considered superior to local, because varying local analysis fails to account for the influence of interactions between parameters, but has rarely been applied in part due to computational difficulties and difficulties in quantifying interactions between parameters (Naujokaitis-Lewis *et al.* (2009); Coutts and Yokomizo (2014); but see Aiello-Lammens & Akçakaya (2016)).

Despite the wide application of PVAs to inform and make predictions including the impacts of management or developments, there have been a number of criticisms of their use and how well models can be used to inform management decisions, including how estimates of uncertainty are utilised (Coulson *et al.* 2001; Ellner *et al.* 2002; Reed *et al.* 2002; McCarthy, Andelman & Possingham 2003; Green *et al.* 2016). The quality of the life-history data used to parameterise models may determine how effectively PVAs are able to predict population changes, and for model predictions will only be valid at predicting extinction if the distribution of life-history parameters between individuals and years is stationary in the future (Coulson *et al.* 2001). There is a need to determine and understand how accurately PVAs can predict population size change but the predictions from PVAs are rarely tested against empirical data in the future to establish how well models performed. Criticism has been levied about how the model results can be difficult to understand, assess and interpret by stakeholders (Knight *et al.* 2008; Pe'er *et al.* 2013). Due to uncertainty and variability amongst the input parameters for the PVA models and hence uncertainty associated with the final metrics produced, decision makers may lack confidence in and may misinterpret model predictions (Addison *et al.* 2013; Green *et al.* 2016). Thus, it is critically important that steps are made to solve these challenges where possible (Masden *et al.* 2015; Green *et al.* 2016), since PVAs remain one of the most widely used tools for evaluating the impacts of anthropogenic developments, wildlife management or conservation strategies on focal populations.

2.2 Seabird PVAs and Marine Renewable Developments

One application of PVAs is as a tool to understand the likely impacts of offshore wind farms on seabird populations. The development of offshore wind farms has the potential to be an important anthropogenic intervention into marine habitats. The UK supports nationally and internationally important breeding and wintering populations of seabirds and the UK government has legal obligations to evaluate the effects such developments may have on such populations. The development of offshore wind farms may negatively impact seabird populations by increased mortality associated with direct collisions with turbines, by displacement of birds from suitable foraging areas; and by impeding movements of commuting or migrating birds (Garthe & Huppop 2004; Drewitt & Langston 2006; Everaert & Stienen 2007; Masden *et al.* 2009; Furness, Wade & Masden 2013; Searle *et al.* 2014; Cleasby *et al.* 2015; Vanermen *et al.* 2015; Busch & Garthe 2016). In the UK, a wide number of reports have used PVAs to assess the impact of wind farm developments on seabird populations and to inform the consenting process for approval of these wind farm developments (see Table 1 for examples). It should be noted that details of PVAs for evaluating the impacts of wind farms are largely available through so called “grey literature” (reports and assessments) rather than ISI published papers. PVAs have aimed to either compare the predicted population trajectory into the future with the wind farm development to that without the development, or to quantify the risk that the development poses by establishing probability of future population declines. Both deterministic and stochastic PVA models have been used for evaluating the impacts of wind farms and it has been argued that deterministic models are a more “honest” approach where there is significant uncertainty around demographic parameters because the presented confidence limits from stochastic models imply an unjustified level of precision in the underlying data (WWT 2012). However, stochastic models are more conservative (Lande, Engen & Sæther 2003) and deterministic models do not produce a distribution of results and hence cannot employ probabilistic metrics. A number of different metrics from the PVAs, for example the increase in the probability of a population decreasing by a fixed amount over time, have been used to provide assessments of the impact of wind farms on seabird populations. Metrics have been criticised for being sensitive to uncertainties both in the life-history parameters used to build the models and in the size of the impact of wind farms on the population (Masden *et al.* 2015; Green *et al.* 2016). Uncertainty in the demographic rates used to parameterise models can lead to uncertainty in whether the predicted magnitude of the impact (e.g. increased mortality or reduced productivity) will lead to an adverse effect on the focal population size (Masden *et al.* 2015). Uncertainty in the size of the impact of the wind farms on the population arises due to lack of empirical data on collision risk, displacement or barrier effects on seabird populations. Thus, there is concern that the metrics may not enable accurate predictions and good understanding of the impacts of offshore wind farms on seabird populations (Cook & Robinson 2016a; Green *et al.* 2016). This uncertainty has therefore led to a precautionary approach to assessments (see Thompson *et al.* (2013) for details).

A broad range of metrics have been derived from PVA population models in order to assess the population level effects of wind farm development on seabird populations (Cook & Robinson 2016a). Cook & Robinson (2016a, 2016b) identified 11 metrics that had been derived from population models as part of HRA undertaken for offshore wind farms that were within the planning process. These metrics were summarised from reports from 27 proposed sites at which the population level impacts of offshore wind farms on seabirds had been considered during assessment: Aberdeen Offshore Wind Farm, Beatrice, Burbo Bank Extension, Docking Shoal, Dogger Bank Creyke Beck A, Dogger Bank Creyke Beck B, Dogger Bank Teesside A, Dogger Bank Teesside B, Dudgeon, East Anglia One, Fife Wind Energy Park, Galloper, Hornsea Project One, Inch Cape, London Array Phase II, MORL (MacColl, Stevenson, Telford), Navitus Bay, Neart na Gaoithe, Race Bank, Rampion, Seagreen Alpha, Seagreen Bravo, Triton Knoll 3, Walney I & Walney Extension. The metrics derived from PVAs were split into two broad categories: i) probabilistic approaches (e.g. the probability of the population declining); or ii) ratio approaches (e.g. the ratio of the population size in the presence and absence of the wind farm). Cook & Robinson (2017) builds on this work, but for a reduced set of metrics from the reports, focusing on two PVA metrics (declines in probability difference for both growth rate and population size, equivalent to Metrics 4 and 7 in Table 2; and counterfactual of impacted and un-impacted populations for both growth rate and population size, equivalent to Metrics 2 and 3 in Table 2) and one rule (Acceptable Biological Change) derived from a PVA metric (Metric 15 in Table 2).

2.3 Review Aims

This review builds on the recent report from Cook & Robinson (2016b), which reviewed 11 metrics derived from population models used as part of the HRA undertaken for assessing the impacts of offshore wind farms on seabird populations, by considering a further range of published reports that did not form part of HRAs (see Table 1).

The purpose of our review was to:

1. Provide details of the metrics produced by PVAs;
2. To summarise any evaluations of how sensitive the metrics were to variation in the input parameters in order to recommend which metrics would be useful to pursue further.

In total we review 15 metrics, of which 11 were previously identified in the Cook & Robinson report (2016b). The four additional metrics that we identified were the difference in population growth rate, the difference in population size, the odds ratio of a decline and the centile for un-impacted population which matches the 50th centile for the impacted population (see No’s.s 12-15 in Table 2). It should be noted that for stochastic models comparing impacted and un-impacted scenarios, metrics are derived using a “matched runs” approach (WWT 2012; Green *et al*. 2016). Stochasticity is applied to the population, but the same survival and productivity rates are incorporated for both the impacted and un-impacted populations at each time step prior to any impact from an offshore wind farm being applied.

Table 1

Additional reports reviewed for PVA modelling metrics which were recommended by the project steering group and were not included in the Cook & Robinson reviews (2016a and 2016b). N.B. population growth rate is defined as being the mean rate of growth across the period of interest (ratio of the population in year i+1 to that in year i; also known as the population multiplication rate).

Reference	Species considered	Metrics used	Equivalent metric No. and description if already included in Cook & Robinson 2016b (Table 2 in this report). Metrics in bold are not included.
Mackenzie, A. & Perrow, M.R. (2009) Population viability analysis of the north Norfolk Sandwich tern *Sterna sandvicensis* population. Report for Centrica Renewable Energy ltd and AMEC Power & Process. Mackenzie, A. & Perrow, M.R. (2011) Population viability analysis of the north Norfolk Sandwich tern *Sterna sandvicensis* population. Report for Centrica Renewable Energy ltd and AMEC Power & Process JNCC & NE (2012) Defining the level of additional mortality that the North Norfolk Coast SPA Sandwich tern population can sustain. JNCC & NE.	• Sandwich tern	• Probability of population decline: the probability of the simulated population falling below thresholds compared to the starting population • Change in probability of decline: the difference in probability of decline between impacted and un-impacted populations (also known as the Counterfactual of the probability of population decline; CPD)	• No. 7: Probability of a 10, 25 or 50% population decline • No 8: Change in probability of a 10, 25 or 50% population decline
Trinder, M. (2014) Flamborough and Filey Coast pSPA Seabird PVA Final Report: Appendix N to the response submitted for deadline V. Report for SMart Wind.	• Gannet • Kittiwake • Guillemot • Razorbill • Puffin	• Population growth rate • Predicted change in population growth rate i.e. the reduction in growth rate between un-impacted and impacted populations • Probability of population decline • Change in probability of population decline	• No. 1: Population growth rate • **Not included in Cook & Robinson** but similar to No 2: Ratio of median impacted to un-impacted growth rate • No 7: Probability of a 10, 25 or 50% population decline • No 8. Change in probability of a 10, 25 or 50% decline • No 7: Probability of a 10, 25 or 50% population decline (but considered in the final year)

Reference	Species considered	Metrics used	Equivalent metric No. and description if already included in Cook & Robinson 2016b (Table 2 in this report). Metrics in bold are not included.
		• Probability the population size in the final year for the impacted population will be less than a range of percentages of the un-impacted population size • Change in the probability of the population size in the final year for the impacted population will be less than a range of percentages of the un-impacted population size	• No. 8: Change in probability of a 10, 25 or 50% decline (but considered in the final year)
Trinder, M. (2015) Flamborough and Filey Coast pSPA Seabird PVA Report: Appendix M to the response submitted for deadline IIA. Report for SMart Wind.	• Gannet • Kittiwake • Guillemot • Razorbill • Puffin	• Predicted change in population growth rate i.e. the reduction in growth rate between un-impacted and impacted populations • Ratio of the impacted to un-impacted population size (Counterfactual of population size) at 5 year intervals up to 25 years	• **Not included in Cook Robinson** but similar to No 2: Ratio of median impacted to un-impacted growth rate • No. 3: Ratio of the impacted to un-impacted population size
Inch Cape Offshore Limited (2011) Inch Cape Offshore Wind Farm Environmental Statement: Appendix 15B Population Viability Analysis.	• Kittiwake • Guillemot • Razorbill • Puffin	• Change in probability of a population decline	• No. 8: Change in probability of a 10, 25 or 50% decline
Freeman, S., Searle, K., Bogdanova, M., Wanless, S. & Daunt, F. (2014) Population dynamics of Forth and Tay breeding seabirds: review of available models and modelling of key breeding populations. Final Report to Marine Scotland Science.	• Kittiwake • Guillemot • Razorbill • Puffin • Herring gull	• Probabilities of population decline to threshold percentages (25, 50, 75 and 100%) below the baseline in 2015 • Excess probabilities of population decline compared to that predicted by baseline in 2015 for threshold percentages (25, 50, 75 and 100%) i.e. probability of a decrease in the impacted population minus that for the un-impacted population	• No. 7: Probability of a 10, 25 or 50% population decline • No 8: Change in probability of a 10, 25 or 50% population decline

Reference	**Species considered**	**Metrics used**	**Equivalent metric No. and description if already included in Cook & Robinson 2016b (Table 2 in this report). Metrics in bold are not included.**
Moray Offshore Renewables Ltd (2013) Environmental Statement: Ornithology population viability analysis outputs and review.	• Gannet • Kittiwake • Guillemot • Razorbill • Puffin • Fulmar	• Probabilities of the population dropping below threshold percentages (quasi-extinction) of the baseline population size during the lifespan of the project (25 years or 25 years plus 10 year recovery) • Change in probabilities of the population dropping below threshold percentages (quasi-extinction) of the baseline population size during the lifespan of the project (25 years or 25 years plus 10 year recovery)	• No. 7: Probability of a 10, 25 or 50% population decline • No. 8: Change in probability of a 10, 25 or 50% population decline

Table 2

Description of metrics used to assess population responses to impacts of offshore wind farms. For each metric an indication is given of the scale over which the metric operates and a description of the metric. This table is adapted from Table 1 in Cook & Robinson 2016b and includes an additional four metrics (two based on our additional review of the reports listed in Table 1 and two requested to be included by Marine Scotland Science; additional metrics are numbers 12-15).

No.	Ratio or probabilistic	Can be used to distinguish wind farm effects?	Metric	Scale and meaning (N.B. the scale of 0-1 generally only applies if the impact of the wind farm is negative relative to the un-impacted scenario)	Description	Included in Cook & Robinson 2016b
1	**Neither**	**No**	**Population growth rate**	• Value of 1 indicates a stable population • <1 indicates a declining population • >1 indicates an increasing population	Calculation of population growth rate (calculated as mean rate over the study period; Final population size/Initial population size)$^{1/Nyears}$) in the presence of the wind farm enables evaluation of whether the population will remain stable, increase or decrease through the life time of the project.	Yes
2	**Ratio**	**Yes**	**Ratio of median impacted to un-impacted growth rate (counterfactual of population growth rate)**	• Scale from 0 – 1 • Value of 1 indicates the impacted population growth rate is the same as the un-impacted growth rate (no population-level consequence) • Values close to 0 indicate a large proportional difference between the impacted and un-impacted population growth rates (a strong population-level consequence)	Considering only the growth rate of a population (as in No. 1 above) in the presence of an offshore wind farm enables an assessment of whether the population will remain stable, increase or decrease over time, but it does not make it possible to quantify the impact of the wind farm on that growth rate. However, this is possible if the growth rate of the population in the presence of a wind farm is compared to that expected in the absence of a wind farm. This ratio is also known as the **COUNTERFACTUAL OF POPULATION GROWTH RATE**	Yes
3	**Ratio**	**Yes**	**Ratio of impacted to un-impacted population size (counterfactual of population size)**	• Scale from 0 – 1 • Value of 1 indicates the impacted population size is the same as the un-impacted size (no population-level consequence) • Values close to 0 indicate a large proportional difference between the impacted and un-impacted population size (a strong population-level consequence)	PVA models can be used to estimate population size through time both with and without the offshore wind farm. Comparing the ratio of these two population sizes gives a statistic that can be used to assess the population level impact of the offshore wind farm. Cook and Robinson state that the ratio could be derived either from a simple deterministic model or taken from the mean or median values simulated using a more complex stochastic model. We advocate that the ratio should be obtained from the median of x simulations of matched pairs; or in a Bayesian context the median will come from the posterior distribution of	Yes

No.	Ratio or probabilistic	Can be used to distinguish wind farm effects?	Metric	Scale and meaning (N.B. the scale of 0-1 generally only applies if the impact of the wind farm is negative relative to the un-impacted scenario)	Description	Included in Cook & Robinson 2016b
					the ratios. The ratio of population sizes could be estimated either at a fixed point in time, for example at the end of a project, or at a series of intervals throughout the life time of a project. This ratio is also known as the **COUNTERFACTUAL OF POPULATION SIZE (CPS).** For example,CPS_{25} = Predicted population size at 25 years (with wind farm)/ predicted population size at 25 years (no wind farm)	
4	**Probabilistic**	**No**	**Probability that growth rate <1, 0.95, 0.8**	- Scale from 0 – 1 - 0 indicates that none of the simulations from a stochastic model result in a growth rate <1 (decreasing population) - 1 indicates that all of the simulations from a stochastic model result in a growth rate <1	Calculated from a stochastic model based on the proportion of simulations where the population declines (has a growth rate <1). The probability of a population declining is typically assessed over the lifetime of the project, but other time scales could be selected. The metric could consider the probability of the growth rate being below other values (e.g. 0.95 or 0.8) which could be selected with reference to the status of the population concerned. Referred to as the Decline Probability Difference (DPDλ) in Cook & Robinson (2017)	Yes
5	**Probabilistic**	**Yes**	**Change in probability that growth rate <1, 0.95, 0.8 (linked to No. 4)**	- Scale from 0 – 1 - 0 indicates that there is no likely change in the probability of the growth rate being <1 between impacted and un-impacted populations (no population-level consequence) - values approaching 1 indicate there is an almost certainly change in the probability of the growth rate being <1 between the impacted and un-impacted populations (i.e. a population-level consequence)	Quantifying the probability of a population decline in the presence of an offshore wind farm may not be meaningful if simulations show that the population may already be at risk of declining in the absence of a wind farm, for example if >50% of simulations have a growth rate <1. To assess the population level impact of a wind farm it is necessary in this case to determine how much greater the probability of a decline is in the presence of an offshore wind farm than in the absence of an offshore wind farm. This can be done either at a single fixed point in time, or at intervals throughout the life time of the project.	Yes
6	**Probabilistic**	**No**	**Probability that population is below initial size at any point in time**	- Scale from 0 – 1 - 0 indicates that none of the simulations from a stochastic model result in a population below its initial size at any point in time - 1 indicates that all of the simulations	After an initial impact, environmental stochasticity and density dependence may mean a population is able to recover throughout the life time of a project. This recovery would mean that over 25 years the final population size may not be smaller than starting population size.	Yes

No.	Ratio or probabilistic	Can be used to distinguish wind farm effects?	Metric	Scale and meaning (N.B. the scale of 0-1 generally only applies if the impact of the wind farm is negative relative to the un-impacted scenario)	Description	Included in Cook & Robinson 2016b
				from a stochastic model result in a population below its initial size at any point in time		
7	**Probabilistic**	**No**	**Probability of a 10, 25 or 50% population decline**	- Scale from 0 – 1 - 0 indicates that none of the simulations from a stochastic model show the impacted population declining by a given magnitude (no population-level consequence) - 1 indicates that all simulations show the impacted population declining by at least the given magnitude - The probability thresholds are also known as quasi-extinction or pseudo-extinction thresholds	A metric to assess the population level impact of a development could be derived by estimating the proportion of simulations for a population in the presence of a wind farm in which a decline of a given magnitude was recorded. Referred to as the Decline Probability Difference (DPDn) in Cook & Robinson (2017)	Yes

No.	Ratio or probabilistic	Can be used to distinguish wind farm effects?	Metric	Scale and meaning (N.B. the scale of 0-1 generally only applies if the impact of the wind farm is negative relative to the un-impacted scenario)	Description	Included in Cook & Robinson 2016b
8	**Probabilistic**	**Yes**	**Change in probability of a 10, 25 or 50% decline (Linked to No. 7; also known as Counterfactual of the probability of population decline)**	• Scale from 0 – 1 • 0 indicates that there is no likely change in the probability of the population decreasing by a given magnitude between the impacted and un-impacted populations (no population-level consequence) • Values approaching 1 indicate there is a large change in the probability of the population decreasing by a given magnitude between the impacted and un-impacted populations (a population-level consequence)	Seabird populations are already declining at many UK colonies (JNCC 2013). Hence, the presence of a wind farm may not substantially increase the probability of the population size at these colonies being <1, if all simulations from the baseline scenario already have a population size less than the starting population size. However, the presence of the wind farm may cause a further reduction in population size. It may, therefore, be more meaningful to consider the change in probability of population size decreasing by a given magnitude, for example a X% increase in the probability of a Y% decline. Also referred to as the Counterfactual of the probability of population decline (CPD), for example the $CPD_{25,10}$ is the difference in the probability of a decline from the starting population size of 10% occurring 25 years after the wind farm construction between impacted and un-impacted populations. CPD can be calculated relative to the change from the starting population after a set time, or relative to the median population. Risk to the population concerned based on the changes in probability can be assessed using IPCC based likelihoods (see Mastrandrea *et al*. 2010). Such likelihoods simply convert the probabilities of the population dropping below the starting population into more accessible language for stakeholders according to boundaries	Yes
9	**Probabilistic**	**Yes**	**Probability of a population being a given magnitude below the median size predicted in the**	• Scale from 0 – 1 • 0 indicates that none of the simulations from a stochastic model show the impacted population size being a given magnitude below the un-impacted population size (no population-level consequence)	The metric to assess the population level impacts of a wind farm may be derived by estimating a median size for a population in the absence of an offshore wind farm and calculating the proportion of simulations for a population in the presence of a wind farm which were either below this median population size, or a given magnitude below this median population size.	Yes

No.	Ratio or probabilistic	Can be used to distinguish wind farm effects?	Metric	Scale and meaning (N.B. the scale of 0-1 generally only applies if the impact of the wind farm is negative relative to the un-impacted scenario)	Description	Included in Cook & Robinson 2016b
			absence of an impact	• 1 indicates that all simulations show the impacted population is a given magnitude below the un-impacted population size (population level consequence)		
10	**Probabilistic**	**Yes**	**Probability that impacted population growth rate is 2.5, 5 or 10% less than un-impacted growth rate**	• Scale from 0 – 1 • 0 indicates that none of the simulations from a stochastic model show the impacted population growth rate being a given magnitude below the un-impacted population growth rate (no population-level consequence) • 1 indicates that all simulations show the impacted population growth rate is a given magnitude below the un-impacted population growth rate (population level consequence)	With growth rates simulated from stochastic models, it may be desirable to estimate a mean or median value for the un-impacted population and calculate the proportion of simulations in which the growth rate of the impacted population is lower, or a given percentage lower, than this value. This approach has the advantage of allowing a probabilistic forecast of the impact of the offshore wind farm on a population, e.g. there is a 50% chance that the wind farm will reduce the population growth rate by 10%.	Yes
11	**Probabilistic**	**Yes**	**Overlap of Impacted and Un-impacted Populations**	• Scale from 0 – 1 • 0 indicates that none of the simulated population sizes after 25 years from the stochastic model of the impacted population overlap with the simulated population sizes after 25 years from the un-impacted population • 1 indicates that all of the simulated population sizes after 25 years from the stochastic model of the impacted population overlap with the simulated population sizes after 25 years from the un-impacted population	Using stochastic models, the population size at a fixed point in time (i.e. at the end of a project lifetime e.g. 25 years) may be expressed as a distribution. In these circumstances, it may be desirable to compare the distributions of the impacted and un-impacted populations. Where there is greater overlap between the two populations, impacts may be deemed less significant.	Yes
12	**Closely related to ratio approaches**	**Yes**	**Difference in population growth rate i.e. the reduction in growth rate between un-**	• Similar to No. 2 (Ratio of median impacted to un-impacted growth rate) but absolute not ratio values (one growth rate subtracted from the other) • The magnitude of the value relates to the magnitude of the difference	Considering only the growth rate of a population (as in No. 1) in the presence of an offshore wind farm enables an assessment of whether the population will remain stable, increase or decrease over time, but it does not make it possible to quantify the impact of the wind farm on that growth rate. However, as with No. 2, this is possible if the growth rate of the population in	No; closely related to No. 2

No.	Ratio or probabilistic	Can be used to distinguish wind farm effects?	Metric	Scale and meaning (N.B. the scale of 0-1 generally only applies if the impact of the wind farm is negative relative to the un-impacted scenario)	Description	Included in Cook & Robinson 2016b
			impacted and impacted populations	between the two growth rates	the presence of a wind farm is compared to that expected in the absence of a wind farm.	
13	**Closely related to ratio approaches**	**Yes**	**Difference in population size i.e. the reduction in population size between un-impacted and impacted populations**	- Similar to No. 3 (Ratio of impacted to un-impacted population size) but absolute not ratio values (one population size subtracted from the other) - The magnitude of the value relates to the magnitude of the difference between the two population sizes	PVA models can be used to estimate population size through time both with and without the offshore wind farm. Comparing these two population sizes by looking at the difference between them enables assessment of the population level impact of the offshore wind farm. As with No 3, the metric of population sizes could be estimated either at a fixed point in time, for example at the end of a project, or at a series of intervals throughout the life time of a project.	No; closely related to No. 3
14	**Probabilistic**	**Yes**	**Odds Ratio of a threshold population decline comparing impacted to un-impacted populations**	- An odds ratio of 1 implies that the presence of the wind farm has no effect on the probability of an event (e.g. a threshold population decline) - An odds ratio >1 implies that the wind farm leads to an increase in the probability of the event	Odds ratios are a way of quantifying the odds of an event happening and provide an additional way of reporting the impacts of a wind farm on seabird populations. However, we did not find any instances where odds ratios were used as metrics for PVAs associated with wind farms in the literature examined in Table 1. The odds ratio essentially provides a summary of the difference between the probabilities for impacts and un-impacted populations so is an alternative way of quantifying the difference between the raw probabilities. For example: - If a decline of 50% in the population (N.B. the level of the decline is not actually relevant to the calculation of the odds ratio) has been estimated to have a probability of 0.2 in the absence of a wind farm, but 0.5 when the wind farm is present - then the odds ratio for the effect of the wind farm is: (0.5 / (1 - 0.5)) / (0.2 / (1 - 0.2)) = 4 - the wind farm has the effect of multiplying the odds of the event (a 50% decline) by four.	No; closely related to No. 8

No.	Ratio or probabilistic	Can be used to distinguish wind farm effects?	Metric	Scale and meaning (N.B. the scale of 0-1 generally only applies if the impact of the wind farm is negative relative to the un-impacted scenario)	Description	Included in Cook & Robinson 2016b
15	**Probabilistic**	**Yes**	**Centile for un-impacted population which matches the 50th centile for the impacted population**	• Related to No. 11 • Values between 0 and 100	This metric is the centile for the un-impacted population which matches the 50th centile of the impacted population. The centile values are taken from the distributions of the impacted and un-impacted populations. The metric from which Acceptable Biological Change (Marine Scotland 2015) is derived.	No; closely related to No. 11

2.4 Sensitivity of PVA Metrics

The second aim of our review was to **summarise any evaluations of how sensitive the metrics were to variation in the input parameters in order to recommend which metrics would be useful to pursue further.** Metrics have been criticised as being sensitive to uncertainties in the demographic parameters used in the modelling process and in the magnitude of the impact predicted on populations (Green *et al.* 2016). In order to evaluate this, Cook & Robinson (2016b) conducted analyses to quantify how sensitive the conclusions drawn from each model were to uncertainty in the demographic parameters used in the population models, the structure of the population models used to derive the metrics and the magnitude of the impact considered. Cook & Robinson (2017) built on this sensitivity analysis by comparing model sensitivity for the counterfactual metrics (No's.s 2 and 3 in Table 2) between models run using a matched runs approach and those without (i.e. where base demographic rates within a stochastic population model vary between un-impacted and impacted populations).

Overall, Cook & Robinson evaluated the metrics according to whether the metric responses were **clear** (the metric shows a noticeable change in response to impacts of increasing magnitude) and **consistent** (the shape of the relationship between the metric and the magnitude of the impact was linear). A clear response would make it easier to distinguish between population level changes associated with differing magnitudes of the impact. Thus, if metrics are not clear then it may be difficult to distinguish impacts arising as a result of the wind farm from natural variation in the population. The shape and consistency of the response are also important because if the response is consistent then it is easier to understand and predict the relationship between the metric and the population level impacts and to understand the consequences of under- or over-estimating the magnitude of impacts. Curved relationships between metrics and the magnitude of the impact are more difficult to interpret than linear relationships because the effects on the population will depend on the magnitude of the impact and hence conclusions are more vulnerable to mis-specification of model parameters. Cook & Robinson concluded that none of the 11 metrics they considered showed both a clear and consistent response to impacts of increasing magnitude, and that none of the probabilistic approaches gave responses that were clear or consistent. Of the 11 metrics, population growth rate, ratio of impacted to un-impacted population growth rate and ratio of impacted to un-impacted population size were the most promising (see Cook & Robinson 2016b; Cook & Robinson 2017). Population growth rate and ratio of impacted to un-impacted population growth rate were promising because of a consistent linear relationship with the magnitude of the impact. However, due to overlap in the confidence limits for these metrics and the range over which they operate, distinguishing population level effects with and without the wind farm would be difficult unless the magnitude of the impact was very large. The ratio of impacted to un-impacted population size was promising because it was the only metric that showed a clear response to the range of impacts considered in the analysis.

Cook & Robinson specifically tested sensitivity to the following:

1. **Population trend**: whether the metric was sensitive to the population trend prior to wind farm construction increasing, decreasing or being stable.
2. **Mis-specification of the demographic parameters:** whether the metrics are sensitive to changes in the demographic parameters (i.e. a large change in the metric arises from a small change in the demographic parameter; for:

    i. Adult survival;
    ii. Immature survival;
    iii. Chick survival;
    iv. Productivity.

3. **Density dependence:** whether the metric is sensitive to inclusion of density dependence on productivity and breeding adult survival in the models.
4. **The form of density dependence:** whether the metric is sensitive to the form of density dependence in the models i.e. how quickly the adult survival rate changes as the population approaches or moves away from the carrying capacity (rather than whether this is compensatory i.e. population growth rate will reduce with increasing density or depensatory i.e. population growth rates will reduce with decreasing density).
5. **Whether stochastic or deterministic:** whether the metric is sensitive to the inclusion of stochasticity (i.e. is modelled from input parameters over a range of values rather than a fixed value).

The most promising metrics for use in assessing the population level effects of wind farms on seabirds were considered to be the ratio of impacted to un-impacted population growth rate (No. 2 in Table 2) and the ratio of impacted to un-impacted population size (No. 3 in Table 2). Cook & Robinson (2017) recommended that stochastic models using a matched run approach are used because this is likely to reflect the most precautionary approach. The median values of the decision criteria predicted for the counterfactual metrics (Metrics 2 and 3) were greater when a matched run approach was used than when models were run without (see Cook & Robinson 2017). See Table 3 for a full summary of sensitivity of all metrics to the five criteria listed above and a summary of how clear and consistent the metrics were. Table 4 summarises the main strengths and weaknesses of each metric and how the metric should be used and interpreted if being used to assess the impacts of wind farms.

Table 3

Sensitivity of metrics used to determine the impacts of offshore wind farms on seabird populations to variation in the input parameters (adapted from Table 5 in Cook & Robinson (2016b)). Shading indicates how well each metric performs: light grey indicates good, dark grey moderate and black poor performance. The two main criteria (highlighted with a thick black line) are whether there was a clear and consistent relationship between the magnitude of the effect and the metric. N.B. probabilistic metrics cannot be calculated from deterministic models, so the comparison between stochastic and deterministic models is not applicable. No’s.12-14 from Table 2 were not included as these were not included in the sensitivity analysis from Cook & Robinson (2016b).

No.	Metric	**Clear** • **Green**: clear difference between metrics for impacts of increasing magnitude • **Amber**: metric varies over a very narrow range • **Red**: the metric reaches an asymptote with impacts of increasing magnitude	**Consistent** • **Green**: linear relationship • **Amber**: non-linear curved relationship • **Red**: stepped relationship	**Insensitive to population trend** • **Green**: identical regardless of population trend • **Amber**: <10% change in metric in relation to population trend • **Red**: >10% change in metric in relation to population trend	**Insensitive to adult survival** • **Green**: sensitivity to misspecification <1% for 10% impacts on survival or productivity • **Amber**: sensitivity to misspecification <5% for 10% impacts on survival or productivity • **Red**: sensitivity to misspecification >5% for 10% impacts on survival or productivity	**Insensitive to immature survival** • **Green**: sensitivity to misspecification <1% for 10% impacts on survival or productivity • **Amber**: sensitivity to misspecification <5% for 10% impacts on survival or productivity • **Red**: sensitivity to misspecification >5% for 10% impacts on survival or productivity	**Insensitive to chick survival** • **Green**: sensitivity to misspecification <1% for 10% impacts on survival or productivity • **Amber**: sensitivity to misspecification <5% for 10% impacts on survival or productivity • **Red**: sensitivity to misspecification >5% for 10% impacts on survival or productivity	**Insensitive to productivity** • **Green**: sensitivity to misspecification <1% for 10% impacts on survival or productivity • **Amber**: sensitivity to misspecification <5% for 10% impacts on survival or productivity • **Red**: sensitivity to misspecification >5% for 10% impacts on survival or productivity	**Insensitive to incorporation of density dependence** • **Green**: median values the same with or without density dependence • **Amber**: <10% change with density dependence vs independent • **Red**: >10% change with density dependence vs independent	**Insensitive to the form of density dependence incorporated** • **Green**: straight line regardless of shape of density dependence or max value of pructivity/adult survival • **Amber**: wavy line • **Red**: clear relationship with density dependence	**Insensitive to stochastic/deterministic model** • **Green**: median values same for both model types • **Amber**: <10% change between models • **Red**: >10% change between models
1	Population growth rate	▓	░	▓	░	░	░	░	▓	░	▓
2	Ratio of median impacted to un-impacted growth rate	▓	░	▓	░	░	░	░	▓	░	▓
3	Ratio of impacted to un-impacted population size after 25 years	░	▓	▓	▓	░	░	░	█	█	▓
4	Probability that growth rate <1	█	▓	█	▓	▓	░	░	█	▓	
5	Change in probability that growth rate <1	█	▓	█	█	▓	▓	▓	█	▓	
6	Probability that population is below initial size at any point in time	█	▓	█	▓	░	░	░	▓	░	
7	Probability of a 25% population decline	█	▓	█	▓	▓	▓	▓	█	█	
8	Change in probability of a 25% decline	█	▓	█	█	▓	▓	▓	█	█	
9	Probability of a population being 50% below un-impacted population	█	▓	█	▓	▓	▓	▓	█	█	
10	Probability that impacted population growth rate is 2.5% less than un-impacted growth rate	█	▓	█	▓	▓	▓	▓	█	█	
11	Overlap of Impacted and Un-impacted Populations	█	▓	█	█	█	█	█	█	▓	

Table 4

Overview of the strengths and weaknesses of the different metrics and information on how the metric should be used to assess the impacts of wind farms. Table adapted from Table 6 in Cook & Robinson (2016b) with the addition of numbers 12 and 13 which were not included in the sensitivity analysis from Cook & Robinson (2016b). We have not included metrics 14 or 15 since sensitivity of these metrics to input parameter specification has not been assessed, so it is not possible to synthesise their strengths and weaknesses.

No.	Metric	Strengths	Weaknesses	How to use and interpret the metrics
1	**Population growth rate**	• Easy to interpret • Consistent relationship between metric and magnitude of impact: easier to make predictions about likely impacts • Relatively insensitive to misspecification of the input parameters	• On its own can't be used to assess wind farm impacts due to lack of comparison with un-impacted population • Variability around the estimates mean it can be difficult to distinguish between variation in the baseline population growth rate and the impacts from the wind farm	• Not a meaningful metric on its own- need to compare the population growth rate of the un-impacted population with that of the impacted population in order to understand then population level effect associated with a wind farm • Lack of a significant difference between impacted and un-impacted populations does not necessarily mean that there would be no population level consequences of the wind farm (due to overlapping confidence intervals)
2	**Ratio of median impacted to un-impacted growth rate**	• Consistent relationship between metric and magnitude of impact: easier to make predictions about likely impacts • Insensitive to misspecification of the input parameters and relatively insensitive to uncertainty in parameter estimates • Insensitive to population trend: metric reflects impact of wind farm and not population status	• Metric varies over a limited range, with the overlapping confidence limits this makes it hard to determine likely population level effects from different magnitudes of effect • Hard to assess effects of the wind farm in a population context due to this limited range	• Metric can be used regardless of population status or trend • Metric should be presented as a median value with 95% confidence limits • Thresholds for determining a wind farm impact are subjective but could be set in reference to the status or trend of the population • Models should be run with a matched run approach
3	**Ratio of impacted to un-impacted population size**	• Easy to interpret in context of a population effect • Clear relationship between metric and magnitude of impact: easier to make	• Sensitive to population declines • More sensitive to misspecification of the demographic parameters than population growth rate or ratio of impacted to un-	• Metric can be used for stable or increasing populations on its own • May be useful context for the ratio of impacted to un-impacted population

No.	Metric	Strengths	Weaknesses	How to use and interpret the metrics
	after 25 years	predictions about likely impacts • Relatively insensitive to uncertainty in the demographic parameters	impacted population growth rate	growth rate regardless of trend • Metric should be presented as a median value with 95% confidence limits • Thresholds for determining a wind farm impact are subjective but could be set in reference to the status or trend of the population • Models should be run with a matched run approach
4	**Probability that growth rate <1**	• Easy to understand, intuitive	• On its own can't be used to assess wind farm impacts due to lack of comparison with un-impacted population • Sensitive to misspecification of adult survival rate • Sensitive to population trends: if population is stable/declining then metric only varies over limited range and so it is difficult to identify population level effects associated with different impacts • True variation in parameters and that based upon observation error are usually not distinguished • Measures are sensitive to any change in conditions in the future	• Not a meaningful metric on its own- need to compare the population growth rate of the un-impacted population with that of the impacted population in order to understand then population level effect associated with a wind farm • Can only be used when the population was increasing prior to the wind farm construction • Requires robust measures of site-specific adult survival • Thresholds for determining a wind farm impact are subjective but could be set in reference to the status or trend of the population
5	**Change in probability that growth rate <1**	• Easy to understand, intuitive: metric quantifies the change in probability of a population declining as a result of a wind farm	• Sensitive to population trend • Sensitive to misspecification of demographic parameters • True variation in parameters and that based upon observation error are usually not distinguished • Measures are sensitive to any change in	• Should not be used when the populations were declining prior to wind farm construction where the change in probability of growth rate is already close to 1 • Requires robust, site specific data on demographic parameters

No.	Metric	Strengths	Weaknesses	How to use and interpret the metrics
			conditions in the future	• Thresholds for determining a wind farm impact are subjective but could be set in reference to the status or trend of the population
6	**Probability that population is below initial size at any point in time**	• Accounts for the fact that populations may recover over the lifetime of the wind farm	• On its own can't be used to assess wind farm impacts due to lack of comparison with un-impacted population • Sensitive to population trends prior to wind farm construction • Sensitive to misspecification of the demographic parameters • True variation in parameters and that based upon observation error are usually not distinguished • Measures are sensitive to any change in conditions in the future	• Not a meaningful metric on its own - need to compare the population growth rate of the un-impacted population with that of the impacted population in order to understand then population level effect associated with a wind farm • Can only be used when the population was increasing prior to the wind farm construction • Requires robust measures of site-specific adult survival • Thresholds for determining a wind farm impact are subjective but could be set in reference to the status or trend of the population
7	**Probability of a 25% population decline**	• Easy to understand • Can be related to established conservation assessments (e.g. (Eaton *et al.* 2015)	• On its own can't be used to assess wind farm impacts due to lack of comparison with un-impacted population • Sensitive to population trends prior to wind farm construction • Sensitive to misspecification of the demographic parameters • True variation in parameters and that based upon observation error are usually not distinguished • Measures are sensitive to any change in conditions in the future	• Not a meaningful metric on its own - need to compare the population growth rate of the un-impacted population with that of the impacted population in order to understand then population level effect associated with a wind farm • Can only be used when the population was increasing prior to the wind farm construction • Requires robust measures of site-specific adult survival • Thresholds for determining a wind farm impact are subjective but could be set in reference to the status or trend of the population

No.	Metric	Strengths	Weaknesses	How to use and interpret the metrics
8	**Change in probability of a 25% decline**	• Easy to understand, intuitive: metric quantifies the change in probability of a population declining by 25% as a result of a wind farm	• Sensitive to population trends prior to wind farm construction • Sensitive to misspecification of the demographic parameters • True variation in parameters and that based upon observation error are usually not distinguished • Measures are sensitive to any change in conditions in the future	• Should not be used when the populations were declining prior to wind farm construction where the change in probability of growth rate is already close to 1 • Requires robust, site specific data on demographic parameters
9	**Probability of a population being 25% below un-impacted population**	• Easy to understand, intuitive comparison of impacted and un-impacted populations • Can be related to established conservation assessments (e.g. (Eaton *et al.* 2015)	• Some sensitivity to population trends prior to wind farm construction • Sensitive to misspecification of the demographic parameters • True variation in parameters and that based upon observation error are usually not distinguished • Measures are sensitive to any change in conditions in the future	• The 25% threshold is subjective and may not be appropriate. Consideration needs to be given to whether to whether alternative thresholds may be more appropriate considering the status and importance of the focal population • Requires robust, site specific data on demographic parameters • Sensitivity to the form and inclusion of density dependence means that models with density dependence should only be used where there is robust evidence for it occurring within the population
10	**Probability that impacted population growth rate is 2.5% less than un-impacted growth rate**	• Relates the impacted population growth rate to that of the un-impacted population	• Difficult to understand in a population context • May be statistically difficult to detect a 2.5% difference in growth rate. Could use higher levels of change but more severe impacts would be required to detect them • Sensitive to population trends prior to wind farm construction • Sensitive to misspecification of the demographic parameters • True variation in parameters and that based upon observation error are usually	• Should not be used when the populations were declining prior to wind farm construction where the change in probability of growth rate is already close to 1 • Requires robust, site specific data on demographic parameters • Sensitivity to the form and inclusion of density dependence means that models with density dependence should only be used where there is robust evidence for it occurring within the population

No.	Metric	Strengths	Weaknesses	How to use and interpret the metrics
			not distinguished • Measures are sensitive to any change in conditions in the future	
11	**Overlap of Impacted and Un-impacted Populations**	• Straightforward comparison that looks at how similar the model outputs are for impacted and un-impacted populations	• Sensitive to population trends prior to wind farm construction • Sensitive to misspecification of the demographic parameters • Sensitive to estimates of uncertainty surrounding the demographic parameters • Value can depend on the number of simulations used in the modelling to obtain the metric • True variation in parameters and that based upon observation error are usually not distinguished • Measures are sensitive to any change in conditions in the future	• Sensitive to population trends means the metric should only be used where there is good understanding of the status of the focal population • Requires robust, site specific data on demographic parameters and the uncertainty surrounding them • Sensitivity to the form and inclusion of density dependence means that models with density dependence should only be used where there is robust evidence for it occurring within the population • Needs careful analysis to ensure enough simulations are used in the models
12	**Difference in population growth rate i.e. the reduction in growth rate between un-impacted and impacted populations**	• consistent relationship between metric and magnitude of impact: easier to make predictions about likely impacts • Insensitive to misspecification of the input parameters and relatively insensitive to uncertainty in parameter estimates • Insensitive to population trend: metric reflects impact of wind farm and not population status	• Metric varies over a limited range, with the overlapping confidence limits this makes it hard to determine likely population level effects from different magnitudes of effect • Hard to assess effects of the wind farm in a population context due to this limited range • Provides absolute values of difference between population growth rate rather than ratios and may need to be interpreted also in the context of No. 2	• Metric can be used regardless of population status or trend • Metric should be presented as a median value with 95% confidence limits • Thresholds for determining a wind farm impact are subjective but could be set in reference to the status or trend of the population
13	**Difference in population size i.e. the reduction in size between un-impacted and**	• consistent relationship between metric and magnitude of impact: easier to make predictions about likely impacts • Insensitive to misspecification of the input parameters and relatively insensitive to	• Provides absolute values of difference between populations rather than ratios and may need to be interpreted also in the context of No. 3	• Metric can be used regardless of population status or trend • Metric should be presented as a median value with 95% confidence limits • Thresholds for determining a wind farm

No.	Metric	Strengths	Weaknesses	How to use and interpret the metrics
	impacted populations	uncertainty in parameter estimates - Insensitive to population trend: metric reflects impact of wind farm and not population status		impact are subjective but could be set in reference to the status or trend of the population

2.5 Criticisms of PVA Metrics in Assessing Wind Farm Impacts

A number of criticisms have been levied against using the metrics derived from PVAs to assess the impact of wind farms (Cook & Robinson 2016a; Green *et al.* 2016). The main criticisms (some of which e.g. No. 1 are equally applicable to broader modelling contexts) were as follows:

1. Lack of empirical data to provide robust estimates and associated confidence limits of collision, barrier and displacement effects on seabirds.
2. Due to this lack of robust estimates of impact levels, probabilistic methods for assessing the risk of population impacts from wind farms are not scientifically robust or defensible - this includes metrics from PVAs that estimate e.g. the difference in probability of a decline between impacted and un-impacted populations.
3. Thresholds are subjective and it should not be claimed that these have been set based on scientific evidence.

Green *et al.* 2016 makes a number of recommendations for providing a scientifically robust and defensible method of assessing population-level impacts of wind farms on seabirds. In the context of PVA modelling the ratio of the expected population size with the wind farm to that without it (No. 3 in Table 2; also termed the so-called Counterfactual of Population size (CPS)) is recommended as a robust metric because this metric is relatively insensitive to uncertainties about demographic rates because they apply to both impacted and un-impacted scenarios. Cook & Robinson (2016b) also advocate the use of this metric, which in conjunction with the ratio of population growth rate (No. 2 in Table 2), is considered to score well in the assessments of sensitivity in Table 3. However, it should be noted that the ratio of impacted to un-impacted population size was sensitive to incorporation and the form of density dependence (see Table 3). Uncertainty can be incorporated, as in Cook & Robinson 2016b, if metrics are derived from a stochastic model or across a range of impact levels. Bayesian approaches, such as those utilised by Freeman *et al.* (2014) and a potential method for conducting Global Sensitivity Analysis developed by Aiello-Lammens & Akçakaya (2016) show promise in being able to separate out the uncertainty associated with input parameter values used in the modelling with that from scenarios of impact on a population (for example different levels of collision mortality or displacement risk), and thus have potential to help address the criticisms levied by Green *et al.* (2016). It has been highlighted that the strength of PVAs lies not in predicting absolute values of viability or costs of management but rather in evaluating the relative effects of different management scenarios (Perkins, Vickery & Shriver 2008). Green *et al.* (2016) is highly critical of interpreting effects based on arbitrary boundaries, which includes probabilistic approaches including probabilities and changes in probabilities of population declines below quasi-extinction thresholds (No. 7 and No. 8 in Table 2), and interpretation of such boundaries advocated for species conservation using IPCC based approaches detailed in Mastrandrea *et al.* (2011) where, for example, an effect is considered to be 'moderate-high' if there is a > 5 % increase in the likelihood of a 20 % population reduction.

2.5.1 Density Dependence

Green *et al.* (2016) also recommends that PVAs should be constructed using density-independent matrix models because such models would be more precautionary in their assessments of population impacts than models including density dependence (as compensatory density dependence, widely assumed to be the most common form, would tend to reduce the impact on population size). However, density-dependent processes may be depensatory, thus slowing the rate of population growth at lower population densities rather than at high densities. Establishing whether compensatory or depensatory density-dependent processes are occurring for species that are the focus of PVAs for wind farms is important: if depensatory processes are operating and are ignored in PVAs then a population decline arising from a wind farm could have larger consequences on the population than are predicted by the models, accelerating population decline and delaying population recovery. Recent work has identified depensation occurring due to increased anti-predator vigilance or colonial defence decreasing rates of productivity in smaller populations in eight species of seabird and seaduck, including species that have been the focus of PVAs for wind farms (Arctic skua, kittiwake, black-headed gull, sandwich tern, common tern, guillemot, puffin and herring gull; Horswill & Robinson 2015; Horswill *et al.* 2016). Indeed, depensation was reported almost twice as often as compensation as a mechanism regulating productivity rates and the authors highlight that this positive feedback mechanism on population size has the potential to be highly destabilising. However, density-dependent effects can vary significantly between colonies in relation to local conditions. Cook & Robinson (2016b) concluded from their sensitivity analyses that density dependent processes operating on the population would mitigate any impacts arising from the wind farm and hence that assuming no density dependence is present is likely to be the most precautionary approach unless depensatory density dependence is known to be operating. Furthermore, Cook & Robinson (2016a) recommend that density-dependence could be incorporated within models where careful consideration has deemed this appropriate, but that density independent models are likely to represent a more precautionary approach in many cases.

2.5.2 Consideration of the Time-Span used to Assess Impacts

Consideration needs to be given to the time-span over which metrics are used to determine whether the wind farm is likely to have an impact on seabird populations, for example whether the assessment is made at time increments from the construction period of the wind farm or at the end of the wind farm operating period e.g. 25 years. The time period selected needs to consider that there will be increasing uncertainty for both impacted and un-impacted scenarios with extrapolation in to the future and hence increased risk of false conclusions on the predicted magnitude of population level effects, but conversely short time windows do not reflect the duration of the lifespan of the wind farm licence (typically 25 years).

2.6 Knowledge Gaps

Cook & Robinson (2016b) adopted a conventional PVA approach whereby they assumed values for demographic parameters (specifically survival, varying between ages, and productivity) and projected simulated population predictions forward in time from a specified starting point (typically at an ‘equilibrium’ age-structure). No data were directly used, so no models were fitted and the results could be assumed valid for any species with demography approximately similar to that adopted in the simulations. With such an approach, since values appropriate for a given species will often be unknown with accuracy, a range of values tend to be considered, and this is the approach the BTO adopted. The advantage of this approach is that since no data fitting is required, there is a considerable reduction in computational demands. The second advantage is that it is possible to model a range of seabird life history strategies. As such, one can construct an analysis that is potentially relevant to all species and regions. However, this approach is less desirable where one wishes to understand a specific region where real data are available, or where one wishes to address generic questions with real data. One example of the latter is the need for a generic solution to the common situation where there are non-local empirical data that are relevant to the focal colony which itself lacks data. Another feature of these models is that the confidence intervals can be unrealistically narrow. A further consideration is that although the Cook & Robinson (2016b) sensitivity analysis undertook a comprehensive assessment of metric sensitivity using simulation approaches, a key knowledge gap is that metric sensitivity has not been comprehensively examined using real data. A project that focussed on this would be complementary to the work undertaken by the BTO. If the same metrics show low sensitivity in models of real world data as in simulation models, then this would provide re-assurance that these metrics are the most promising. Furthermore, such an approach would enable generic questions to be addressed with real data. One example which is very common with UK seabird populations is where data are absent from the focal colony but available from an adjacent colony, thereby offering a natural, informative prior. We would recommend that such approaches are undertaken so that sensitivity of metrics can be tested using real-world data.

2.7 Recommendations from Literature Review

- The two metrics that have been recommended for use in establishing the impact of a wind farm on seabird populations are the **Ratio of median impacted to un-impacted growth rate** and the **ratio of impacted to un-impacted population size (also known as counterfactual of population size).**
- The two metrics of the **difference in population growth rate between impacted and un-impacted populations** and the **difference in population size** should also be considered as these may be more useful if the growth rates or population size estimates being compared are small (ratios may be misleading in this context).
- Metrics should be obtained from stochastic models using a matched run approach because this is likely to reflect the most precautionary approach.
- Should probabilistic metrics be used, based on the rationale that they have been widely used in the past within published conservation science literature, and may still be used extensively in the future, it should be acknowledged that these have received criticism in Green *et al.* (2016) and Cook & Robinson (2016b).
- Density dependence should only be included where there is evidence that this may be occurring in the population of interest, otherwise use of density-independent models, or a range of density dependent structures, is advised.
- Global Sensitivity Analysis approaches detailed in Aiello-Lammens & Akçakaya (2016) and Bayesian approaches utilised by Freeman *et al.* (2014) to separate model outcome uncertainty that arises due to uncertainty in the parameter estimates used to build the models from the uncertainty in the effects of the management action (in this case wind farms) should be considered.

3. Population Modelling: Methods

3.1 Modelling Approach

A key early decision by the Steering Group was to agree which population modelling approach to use. Conventionally, PVA have been applied by assuming values for demographic parameters (specifically survival, varying between ages, and productivity) and projecting simulated population predictions forward in time from a specified starting point (typically at an ‘equilibrium’ age-structure). No data are directly used, so no models are fitted and the results can be assumed valid for any species with demography approximately similar to that adopted in the simulations. In practice, since values appropriate for a given species will rarely be known with much accuracy, a range of values tend to be considered. The advantage of this approach is that since no data fitting is required, there is a considerable reduction in computational demands. The second advantage is that it is possible to model a range of seabird life history strategies. As such, one can construct an analysis that is potentially relevant to all species and regions. This approach is less desirable where one wishes to understand a specific region where real data are available, or where one wishes to address generic questions with real data. One example of the latter is the need for a generic solution to the common situation where there are non-local empirical data that are relevant to the focal colony which itself lacks data (see next section). Another feature of these models is that the confidence intervals can be unrealistically narrow.

In the previous population modelling contract CEH undertook for Marine Scotland Science, we fitted state-space models using Bayesian techniques via WinBUGS to data from four SPAs for five species in the Forth/Tay region (Freeman *et al.* 2014). Here, no parameter values were specified beforehand; all were estimated from the data prior to projecting the population predictions forwards to beyond the period of the data. In these models, the population is assumed to change stochastically (the ‘state process’) and the counts to be equal in expectation to the population level (or part of it), subject also to sampling variability (the ‘observation process’). Using this method, sampling co-variances of parameter estimates are naturally accommodated. In Freeman *et al.* (2014), demographic parameters were assumed to vary about a mean value, with a specified variance, where these were estimated from models applied at sites with more substantial data (generally the Isle of May). While the need for defining parameter configurations *a priori* are reduced in such models, the results are dependent upon the data used (precision, for example, will depend in part upon the likely representativeness of the data from the well-studied colony). One advantage of this approach is in the case where there is interest in specific colonies/study areas, thereby providing a rationale for fitting the model to real data. Of the various methods that can be used to fit models to data, we consider this approach to be the most robust because of greater realism in the estimating of credibility intervals, in particular due to the partitioning of observation and process error, in cases where there are empirical data (counts and/or demography data) or informative priors (see Freeman *et al.* 2014 for a discussion of this). A second advantage of this approach is in addressing generic questions with real-world data. One example has been addressed above that we think is particularly relevant in this context, where data are absent from the focal colony but available from an adjacent colony, thereby offering a natural, informative prior. However, considerable thought is required before adopting this approach since information from another colony cannot automatically be assumed to apply elsewhere, to other species and/or regions, and any assumptions should be clearly specified. Two more advantages arise from this approach within the specific context of this project: a) Cook & Robinson (2016) have undertaken a comprehensive sensitivity analysis of PVA metrics using simulations in a traditional framework, so there would be a benefit in testing the performance of the same suite of metrics in an empirical analysis, with confidence gained if the same metrics perform well using both approaches; b) there is continuity with the previous report (Freeman *et al.* 2014). The main disadvantage of this approach is the analytical and computational demands. Furthermore, if there is no interest in specific colonies/regions, or if the generic questions that can be addressed using real-world data, then a simulation approach is the logical way forward.

The Steering Group decided that there was such interest, and that it would be complementary to the recent work by Cook & Robinson (2016), so this was the method that was undertaken. Further, the decision was to focus on the three main issues emerging from past work and stakeholder interest: sensitivity in a range of PVA metrics including a comparison of ratio and probabilistic types, effect of population status on sensitivity, and effect of renewables effect size on sensitivity. Finally, it was agreed following consideration of the literature that density dependence would not be included in the models (see literature review).

3.2 Modelling Methods

3.2.1 Input Data

Five study species were selected: black-legged kittiwake, common guillemot, razorbill, herring gull and European shag. Of these kittiwake, guillemot, razorbill and herring gull were considered in Freeman *et al.* (2014). As similar models have, in the interim, also been fitted for shags we also consider this extra species. We accumulated data sets on abundance, survival and productivity from four SPAs (Buchan Ness to Collieston Coast SPA; Fowsheugh SPA; Forth Islands SPA; St Abb's Head to Fastcastle SPA).

New data were added up to 2016 where available (Freeman *et al.* 2014 modelled data up to 2012). Data include colony counts, in full if possible but often such data are available only in a limited number of years, or else have been made only in smaller parts of the main colony (i.e. plots). Demography is estimated from ringing data (survival) or nest record data (productivity per nest/pair). Such data have long been gathered by CEH at the Isle of May in the Forth Islands SPA, but are often missing elsewhere in the region. Data availability and sources for the species considered are given in Tables 5 and 6, respectively.

Counts and demographic parameter estimates can be found in Appendix 1.

Table 5

Data availability for each species at each SPA. Regular census means annual or near-annual. Sporadic census is less regular – typically every four to seven years. Sources: [a]Seabirds Monitoring Programme online database; [b]Vicky Anderson/Edward Grace, RSPB, pers comm; [c]Roddy Mavor, JNCC pers comm.; [d]Harris *et al.* 2009, 2013; [e]Frederiksen *et al.* 2004 updated; [f]Lahoz-Monfort *et al.* 2011, 2014; [g]Newell *et al.* 2012; [h]Lahoz-Monfort *et al.* 2013; [i]BTO ringing and recovery data, purchased for Freeman *et al.* 2014

Species	**SPA**	**Counts**	**Survival (Adult birds)**	**Productivity**
Kittiwake	Forth Islands	Regular census[a]	Regular survey[e]	Regular census[a,g]
	St Abb's Head	Regular census[a]	No	Regular census[a]
	Fowlsheugh	Sporadic census[a]	No	Regular census[a]
	Buchan Ness	Sporadic census[a]	No	Regular census[a]
Guillemot	Forth Islands	Regular census[a]	Regular survey[f]	Regular census[a,g,h]
	St. Abb's Head	Sporadic census [a] Regular sub-plot survey[a]	No	No
	Fowlsheugh	Sporadic census[a] Regular sub-plot survey[b]	No	No
	Buchan Ness	Sporadic census[a] Sporadic sub-plot survey[c]	No	No
Razorbill	Forth Islands	Regular census[a]	Regular survey[f]	Regular census[a,g,h]
	St Abb's Head	Sporadic census[a] Regular sub-plot survey[a]	No	No
	Fowlsheugh	Sporadic census[a] Regular sub-plot survey[b]	No	No
Herring gull	Forth Islands	Regular census[a]	Historical survey[i]	Regular census[a]
	St Abb's Head	Regular census[a]	No	No
Shag	Forth Islands	Regular census[a]	Regular survey[a]	Regular census[a]
	St Abb's Head	Regular census[a]	No	Regular census[a]
	Buchan Ness	Sporadic census[a]	No	No

Table 6

Data source for each species at each SPA.

Species	SPA	Counts	Adult survival	Productivity
Kittiwake	Forth Islands	Forth Islands	Isle of May	Isle of May
	St Abb’s Head	St Abb’s Head	Isle of May	St Abb’s Head
	Fowlsheugh	Fowlsheugh	Isle of May	Fowlsheugh
	Buchan Ness	Buchan Ness	Isle of May	Buchan Ness
Guillemot	Forth Islands	Forth Islands	Isle of May	Isle of May
	St. Abb’s Head	St. Abb’s Head	Isle of May	Isle of May
	Fowlsheugh	Fowlsheugh	Isle of May	Isle of May
	Buchan Ness	Buchan Ness	Isle of May	Isle of May
Razorbill	Forth Islands	Forth Islands	Isle of May	Isle of May
	St Abb’s Head	St Abb’s Head	Isle of May	Isle of May
	Fowlsheugh	Fowlsheugh	Isle of May	Isle of May
Herring gull	Forth Islands	Forth Islands	Isle of May	Isle of May
	St Abb’s Head	St Abb’s Head	Isle of May	Isle of May
Shag	Forth Islands	Forth Islands	Isle of May	Isle of May
	St Abb’s Head	St Abb’s Head	Isle of May	St Abb’s Head
	Buchan Ness	Buchan Ness	Isle of May	Isle of May

3.2.2 Population Models

The models adopted for these data are as described in Freeman *et al.* (2014) and we provide only a brief overview here. A state-space model for the annual counts was adopted, with the expected number of breeding pairs of a population in year *t* given by N_t, where, for a species such as shag that begins breeding at age three is:

$$N_t = Nr_t + Na_t$$

$$Nr_t \sim Poisson\left(N_{t-3}\left(\frac{f_{t-3}}{2} \times \varphi_{j,t-3}\varphi_{j,t-2}\varphi_{j,t-1}\right)\right)$$

$$Na_t \sim Binomial\left(N_{t-1}, \varphi_{a,t-1}\right)$$

Where Nr_t and Na_t are respectively the numbers of new recruits, and survivors of the previous breeding population, in year *t*. The model is straightforwardly amended to accommodate those species that do not begin breeding until aged five or six. Juvenile survival probabilities $\varphi_{j,t}$ are assumed to take a constant value φ_j , unknown but estimable from the data; those for adults $\varphi_{a,t}$ are assumed normally distributed mean values and variance estimated from a set of ringing data at the Isle of May. Completing the model the annual numbers of chicks per pair f_t are estimated with means and variance from nest record data gathered at the site in question, where available, or using the data from the Isle of May where site-specific productivity data are unavailable. Due to the paucity of kittiwake counts at Fowlsheugh and Buchan Ness to Collieston Coast, these were modelled simultaneously in a single (multivariate) state-space model, with a common juvenile survival rate. As in Freeman *et al.* (2014) there were problems modelling the Kittiwakes at the Forth Islands SPA; this was due to low counts in 1994, which subsequently recovered for a few years, and so the 1994 counts were omitted from the data that informed the state space model.

Models were fitted using Bayesian techniques using the software JAGS (Plummer 2013). As in Freeman *et al.* (2014), multiple projections for 25 future years (2016 to 2041) of wind farm impact under various scenarios (given below) are made by repeatedly sampling from the distributions above, effectively generating posterior distributions for the abundance in future years. Using the model above, we thus produce ‘baseline’ predictions, under the assumption that prevailing conditions apply in future years. We then produced a series of alternative ‘impacted’ population trajectories assuming that adult survival, productivity or both were negatively affected by some ‘perturbation’, equating to the effect of an offshore wind farm. This enables a comparison of future predictions following perturbation with those under the ‘status quo' assumptions, known as the baseline. In consultation with the Steering Group, adult survival was set to decline by one of a range of specified rates, namely 0% (i.e. no change), 0.5%, 1%, 2% and 3%. Declines in annual productivity were set to 0%, 1%, 2%, 3% and 5%. Finally, combined effects of survival and productivity were set to, respectively, 0%/0%, 1%/1%, 2%/2%, 3%/3% and 0.5%/5%. Note that these are percentage point changes, as requested by the Steering Group, which differs from the approach taken in Cook & Robinson (2016b) where percentage changes were investigated. In all models, an additional five years were projected with no change in survival or productivity, representing a post-wind farm decommissioning period.

3.3 PVA Metric Sensitivity

The above modelling framework allowed us to examine the population changes under various levels of impact upon the demographic parameters, given that these take the values of the model. It is, of course, plausible that the average values of adult survival and productivity experienced by the populations may differ from those implied by the demographic data used, especially where these are 'borrowed' from adjacent sites for those without such data of their own (for survival, this is all sites apart from the Forth Islands; even there, all ringing data are from a single study at the Isle of May). Therefore, we also repeated the entire procedure with demographic parameters "mis-specified" to varying degrees. Specifically, we considered median adult mortality (the complement of survival, since survival is generally high in seabirds and percentage increases are greatly limited by the constraint of lying below a survival rate of one) and productivity to differ from those of the baseline by, in turn -30%, -20%, -10%, 10%, 20% and 30%. The consequences of uncertain adoption of demographic parameters could then be examined by plotting a suite of PVA metrics against this rate of mis-specification, under a range of renewables effect sizes.

The Steering Group, having considered the findings of the literature review, requested that we examine the sensitivity of five PVA metrics, and Marine Scotland Science requested that we include a sixth metric (PVA F):

1) Median of the ratio of impacted to un-impacted (=baseline) annual growth rate (PVA A; Metric No. 2 in Table 2).
2) Median of the ratio of impacted to un-impacted population size after 25 years (PVA B; Metric No. 3 in Table 2).
3) Median difference in impacted and un-impacted annual growth rates (PVA C; Metric No. 12 in Table 2.)

4) Median difference between impacted and un-impacted population size after 25 years (PVA D; Metric No. 13 in Table 2).
5) Probability of a population decline over 25 years exceeding a) 10% b) 25% and c) 50% (PVA E1, E2 and E3 respectively; Metric No. 7 in Table 2).
6) Centile for un-impacted population which matches the 50th centile for the impacted population after 25 years (PVA F; Metric No. 15 in Table 2).

PVAs A and B are ratio metrics, PVAs C and D are metrics related to ratio metrics and PVAs E and F are probabilistic metrics. All of these metrics are readily estimable from the repeated simulations above, with posterior distributions of the ratios/differences arising from a “matched runs” approach, as recommended (WWT 2012; Green *et al.* 2016; Cook & Robinson 2017) i.e. the parameters defining the expected annual counts in each replicate are identical, except insofar as the expected impacted figures are adjusted to reflect the level of the impact. Plotting these metrics against alternative levels of adult survival or productivity used gives a visual assessment of the sensitivity of these metrics to the choice of demographic parameters.

Note that for the models of razorbills at Fowlsheugh, two of the thirteen models exhibited formal warnings via the Brooks-Gelman statistic values regarding convergence for juvenile survival. However, the estimates of the PVA metrics from these models appear to be consistent with the pattern as shown by other species/SPAs and so these are retained in the plots.

However, for three species/SPA combinations there were inherent problems with the “baseline” model (with no mis-specification). This was for shags at Buchan Ness, having a baseline model which “converged”, but not to anything sensible (the observation error was greater than the counts) and for herring gull at both sites, which had problems with the convergence of key parameters, adult survival and juvenile survival. Therefore, we considered these three species/SPAs to be unreliable and did not use them in the assessment of the sensitivity of the PVA metrics.

3.4 Structure of the Results

The Steering Group requested that we examine the sensitivity of these PVA metrics to mis-specification in adult mortality and productivity, and investigate to what extent this sensitivity varied with predicted population status and size of renewables effect. Accordingly, the results section is split into three parts.

First, we provide the full results of population modelling, including retrospective data fitting, population forecasts and PVA sensitivities for one species/SPA population: kittiwakes at Forth Islands. It was considered by the Steering Group necessary to show this comprehensive output for one population only, although models presented were undertaken on all populations. Combining the mis-specifications in adult mortality or productivity with the scenarios of annual decline in adult survival or productivity provides four graphical outputs:

1. Mis-specification in adult mortality with scenarios of renewables-induced change in productivity;
2. Mis-specification in adult mortality with scenarios of change in adult survival;
3. Mis-specification in productivity with scenarios of renewables-induced change in productivity;
4. Mis-specification in productivity with scenarios of change in adult survival.

Second, we present PVA sensitivities in relation to population status, combining data from all species/SPAs for which we achieved model convergence. We estimated the projected population growth rate as follows:

$$\lambda = \left(\frac{Estimated\ median\ total\ population\ in\ 2041}{Estimated\ median\ total\ population\ in\ 2016}\right)^{1/25}$$

Lambda is calculated for the baseline model and takes the values for the various species/SPA combinations shown in Table 7. Populations were classed as increasing (lambda > 1) or decreasing (lambda < 1). Of the four combinations outlined above, we only show results from the analysis of mis-specification in adult mortality with the maximum scenario of change in adult survival (3%), to maximise clarity.

Third, we present PVA sensitivities in relation to scenarios of change resulting from the renewables development (i.e. the effect size). Of the four combinations outlined above, we only show results from the analysis of mis-specification in adult mortality with scenarios of change in adult survival.

Table 7

Projected population growth rates over the period 2016-2041 for Species/SPA populations.

Species/SPA population	**Lambda**
Kittiwakes:	
Forth Islands	0.964
St Abb’s Head	0.937
Fowlsheugh	0.969
Buchan Ness to Collieston Coast	0.967
Guillemots:	
Forth Islands	1.012
St Abb’s Head	1.018
Fowlsheugh	0.997
Buchan Ness to Collieston Coast	1.022
Razorbills:	
Forth Islands	1.023
St Abb’s Head	0.991
Fowlsheugh	1.040
Shags:	
Forth Islands	1.004
St Abb’s Head	0.980

4. Population Modelling: Results

4.1 Population Modelling and PVA Sensitivity in Forth Islands Kittiwakes

The data available for Forth Island kittiwakes, the population for which we present the full set of outputs, ranges from 1984 to 2016. The annual variation in the median adult survival and productivity as well as the posterior distribution of juvenile survival and the observation error are given in Figure 1. The latter two parameters approximate a normal distribution, with a mean juvenile survival of 0.685. The model suggests that Kittiwakes at the Forth SPA have declined from an initial abundance of just over 10,000 to about 4,000 in 2016. Future projections indicate further declines (Figures 2a-c), though note the wide credible intervals, broadening as time passes, as uncertainty increases in these estimates.

For the sensitivity analysis, the median population size after 25 projected years (2041) was estimated under a range of mis-specifications in adult mortality or productivity and scenarios of annual decline in adult survival or productivity (Figure 3). The estimated population size when adult survival or productivity does not change and there is no mis-specification in the Bayesian model results in an estimate of approximately 1,300 birds. As expected, population size under all effect size scenarios declines with increasing mortality and increases with increasing productivity (Figure 3). These relationships are non-linear, and different scenarios of annual decline diverge as the overall effect of mis-specification strengthens, because percentage point changes in mis-specification have a relative, not absolute effect on population size.

The outputs of PVA metric sensitivity can be found in Figures 4a-h for PVA A, B, C, D, E1, E2, E3 and F, respectively (see Section 3.3 of the methods for a definition of each metric). We estimated the PVA metrics using seven model runs for changes in adult mortality (-30% to +30% at 10% increments) and seven runs for productivity (-30% to +30% at 10% increments). The model run of no change in adult mortality or productivity is shared by both, hence a total of thirteen models were run.

The ratio of impacted to un-impacted annual growth rate (PVA A; Figure 4a) was very close to one for the full range of scenarios and, matching theory and past evidence using simulations, was insensitive to mis-specification in demographic parameters. One possibility for the low sensitivity of PVA A is the scale of values, with all values being close to one, and, therefore, sensitivity potentially appearing low in a visual assessment even in cases where it is not. However, we show that this is not the case in Appendix 2, where we consider a 25 year growth rate, whereby lines deviate markedly from one and low sensitivity is still apparent.

Estimates for the ratio of impacted to un-impacted population size after 25 years (PVA B; Figure 4b) showed a range of values with respect to scenarios of change in productivity and, in particular, mortality, but it was also insensitive to mis-specification in demographic parameters. The PVA metric representing the difference in impacted and un-impacted growth rates (PVA C; Figure 4c) was also comparatively insensitive. In contrast, the PVA metric representing the difference in impacted and un-impacted population size (PVA D; Figure 4d) was considerably more sensitive, and showed non-linear patterns of change which were dependent on the effect size scenario, associated with the relationship between absolute and relative changes in population size (as with Figure 3).

As regards the probabilistic metrics, the metric presenting the probability of a population decline over 25 years exceeding 10%, 25% and 50% (PVAs E1, E2 and E3; Figure 4e, f and g respectively) showed high sensitivity to mis-specification both in mortality and reproduction. Each shows a non-linear pattern of change in line with expectations and past use of these metrics, including the expected variation between PVAs E1, E2 and E3 in relation to the stated exceedance thresholds of 10%, 25% and 50%. In contrast, the metric representing the centile for un-impacted population which matches the 50% centile for the impacted population after 25 years (PVA F; Figure 4h) showed moderately low sensitivity to mis-specification of survival and productivity. It was less sensitive than PVA E with and more sensitive than ratio metrics PVA A and B.

Graphical presentation of sensitivity of PVA metrics for all 13 species/SPA combinations can be found in Appendix 3.

Figure 1: Diagnostics plot from the Bayesian state space model for adult survival, productivity, juvenile survival and observation error.

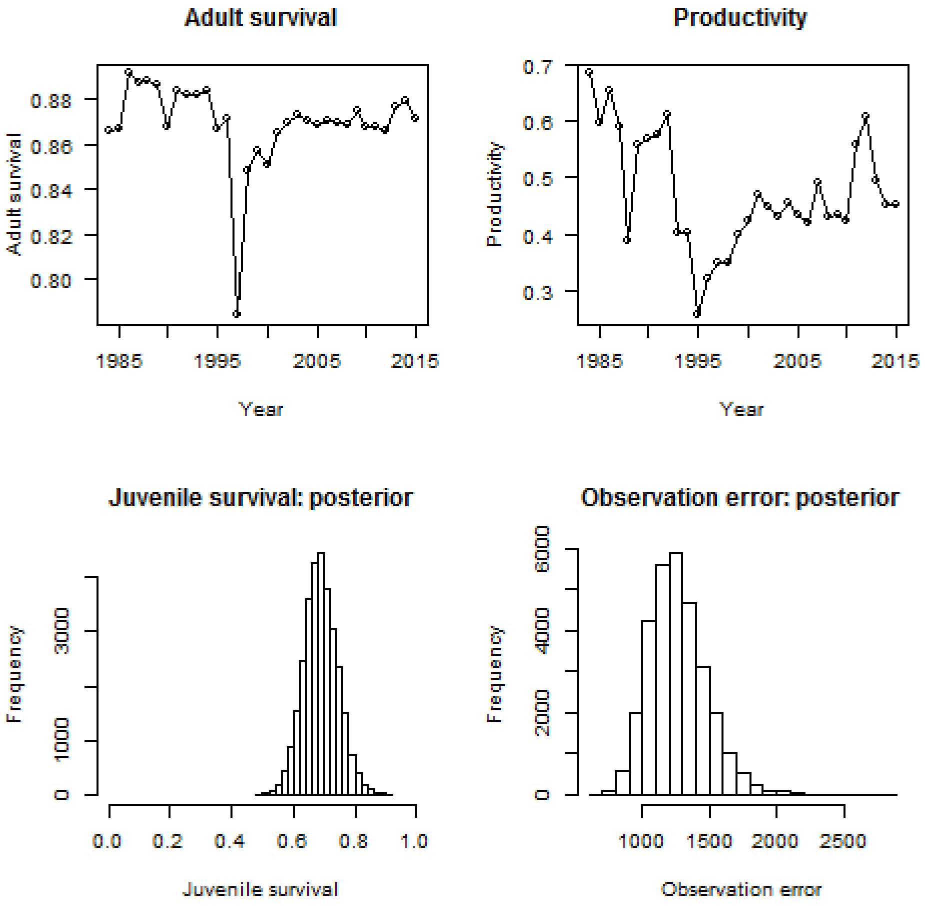


Figure 2a: Estimated total abundance from 1984 to 2016, with an additional 25 years of projections with various declines in productivity and a final five years of projections with no decline in productivity.

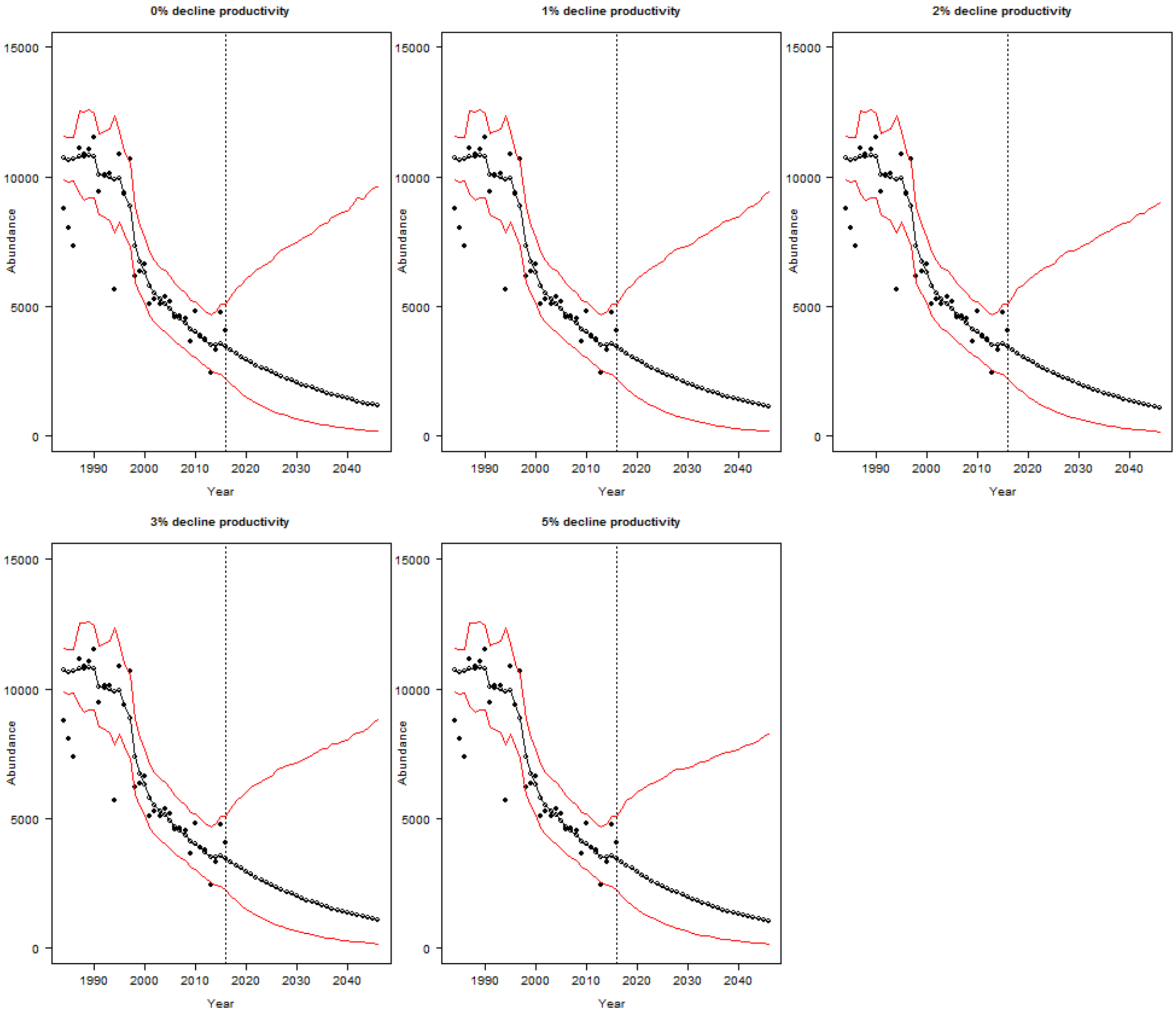


Figure 2b: Estimated total abundance from 1984 to 2016, with an additional 25 years of projections with various declines in adult survival and a final five years of projections with no decline in adult survival.

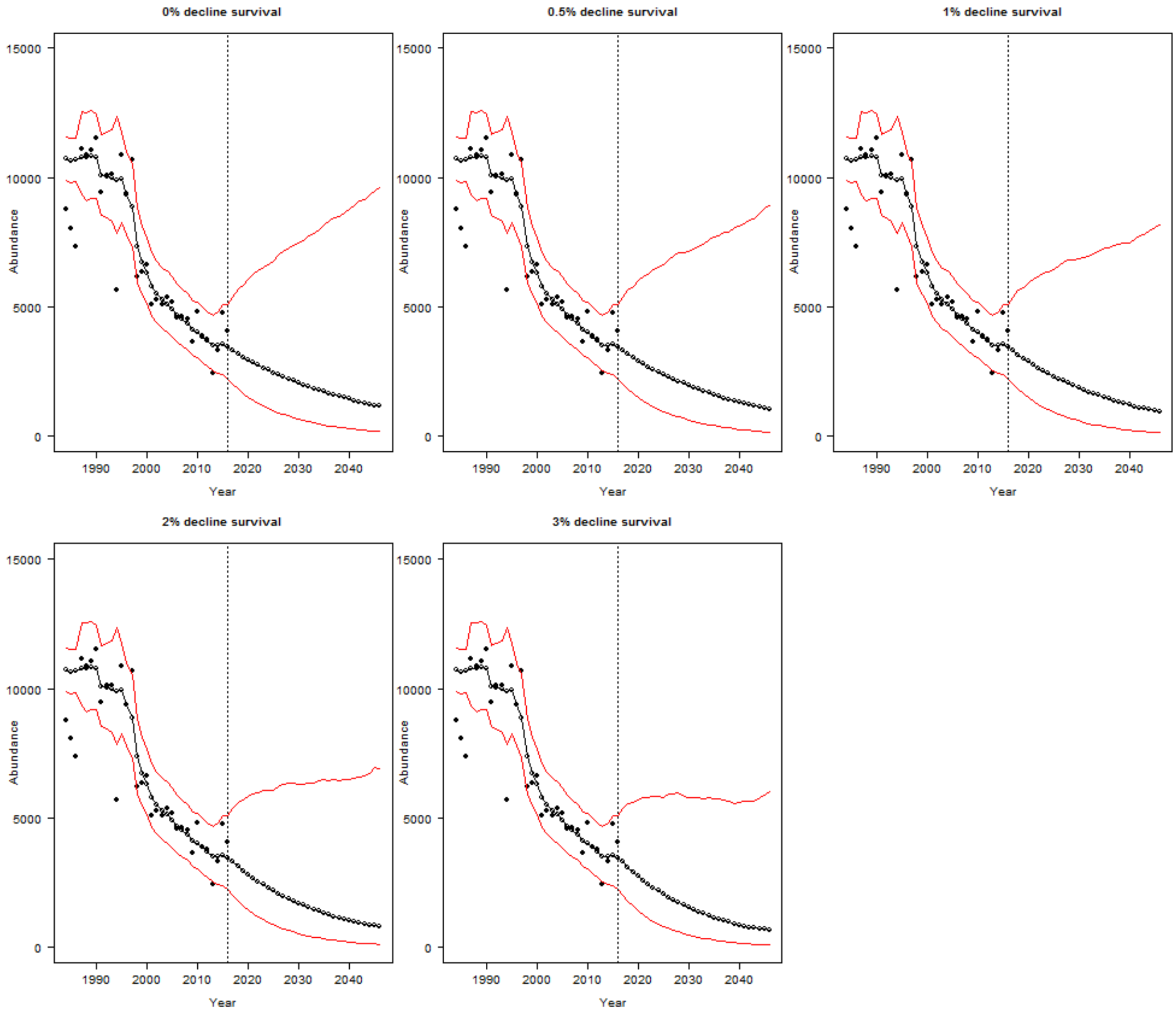


Figure 2c: Estimated total abundance from 1984 to 2016, with an additional 25 years of projections with various declines in both productivity and adult survival and a final five years of projections with no decline in either productivity or adult survival.

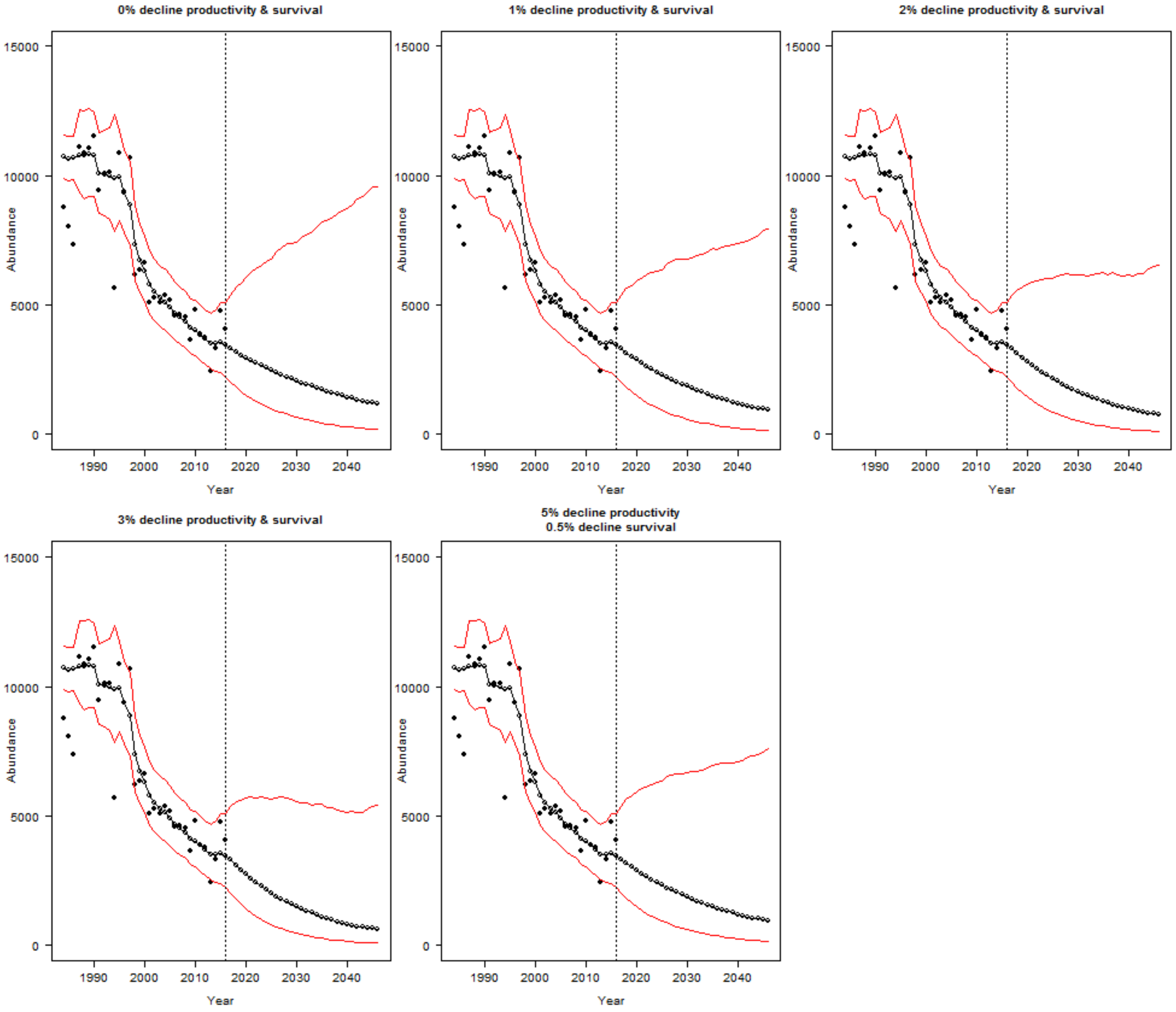


Figure 3: Median impacted population size after 25 years of projections under various scenarios of mis-specification in productivity and adult mortality. Adult mortality mis-specification is illustrated in the upper panels and productivity mis-specification in the lower panels. Mis-specification was varied from -30% to +30% (with 0% representing no mis-specification). The five coloured lines represent the different levels of potential impact on annual productivity (left panels) or annual adult survival (right panels) over the hypothetical 25 year lifetime of the wind farm (2016-2041).

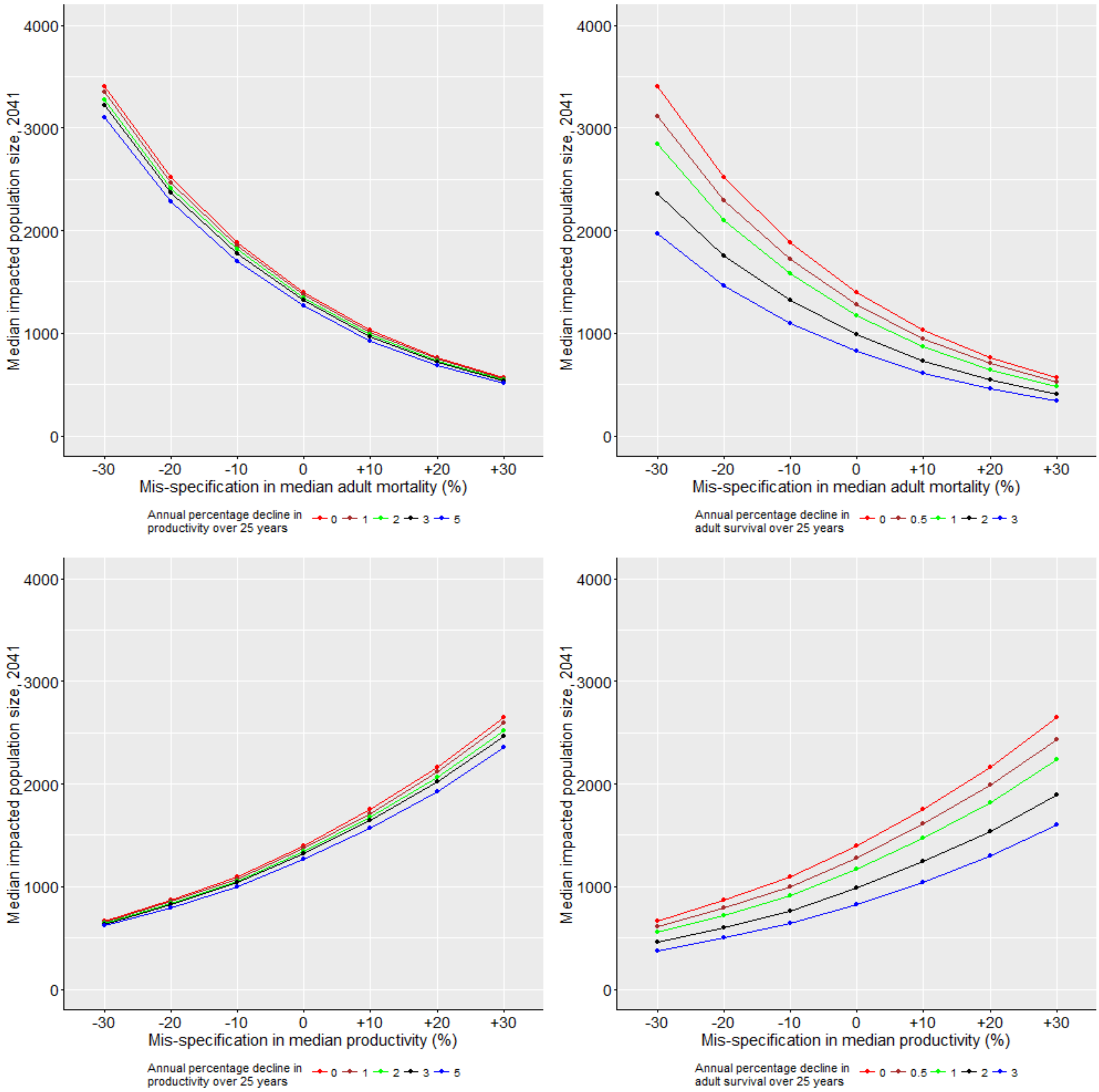


Figure 4a: PVA Metric A – ratio of population growth rate from 2016-2041, comparing impacted population vs. un-impacted population. Adult mortality mis-specification is illustrated in the upper panels and productivity mis-specification in the lower panels. Mis-specification was varied from -30% to +30% (with 0% representing no mis-specification). The five coloured lines represent the different levels of potential impact on annual productivity (left panels) or annual adult survival (right panels) over the hypothetical 25 year lifetime of the wind farm (2016-2041).

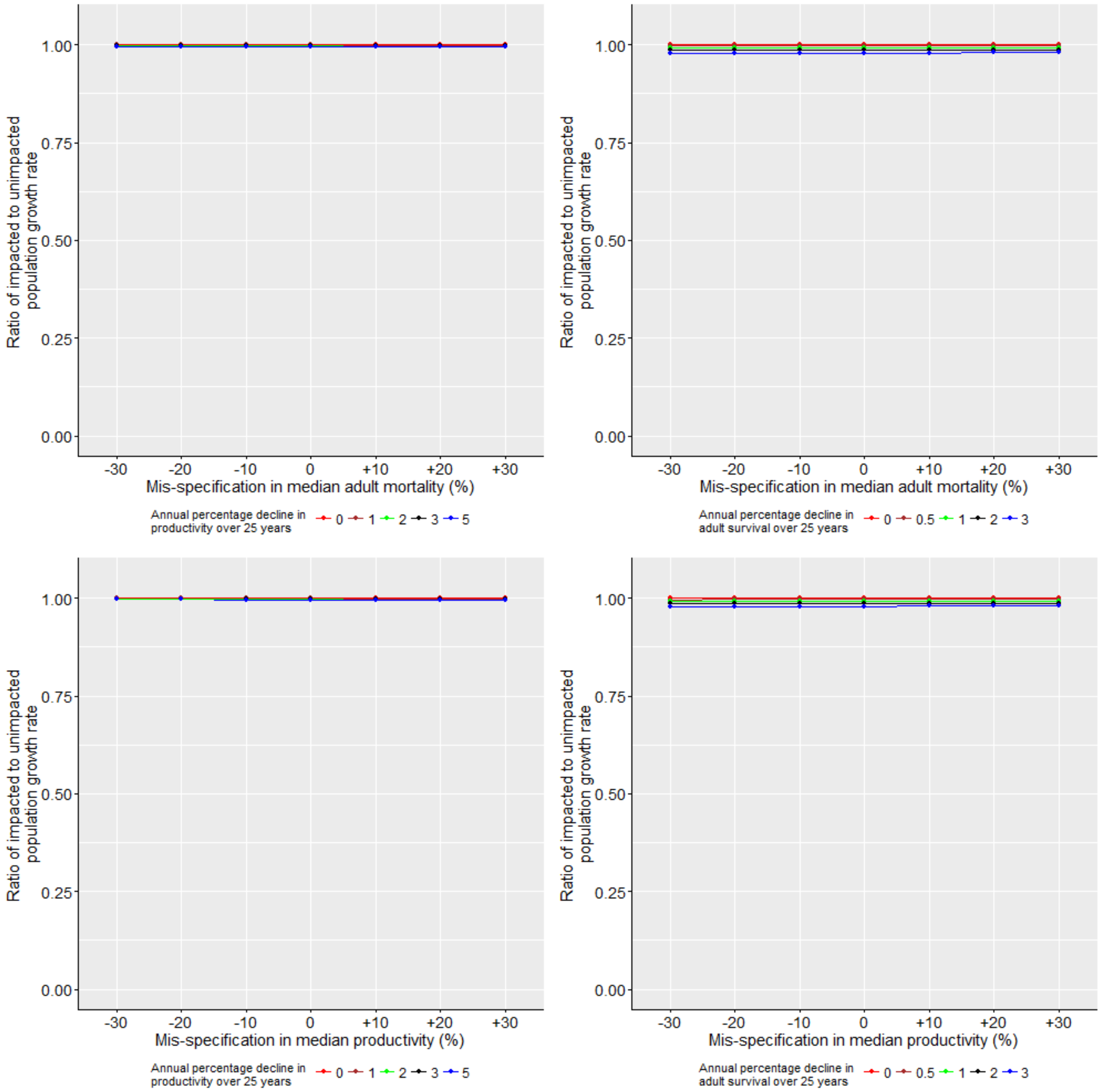


Figure 4b: PVA Metric B – ratio of population size at 2041, comparing impacted population vs. un-impacted population. Adult mortality mis-specification is illustrated in the upper panels and productivity mis-specification in the lower panels. Mis-specification was varied from -30% to +30% (with 0% representing no mis-specification). The five coloured lines represent the different levels of potential impact on annual productivity (left panels) or annual adult survival (right panels) over the hypothetical 25 year lifetime of the wind farm (2016-2041).

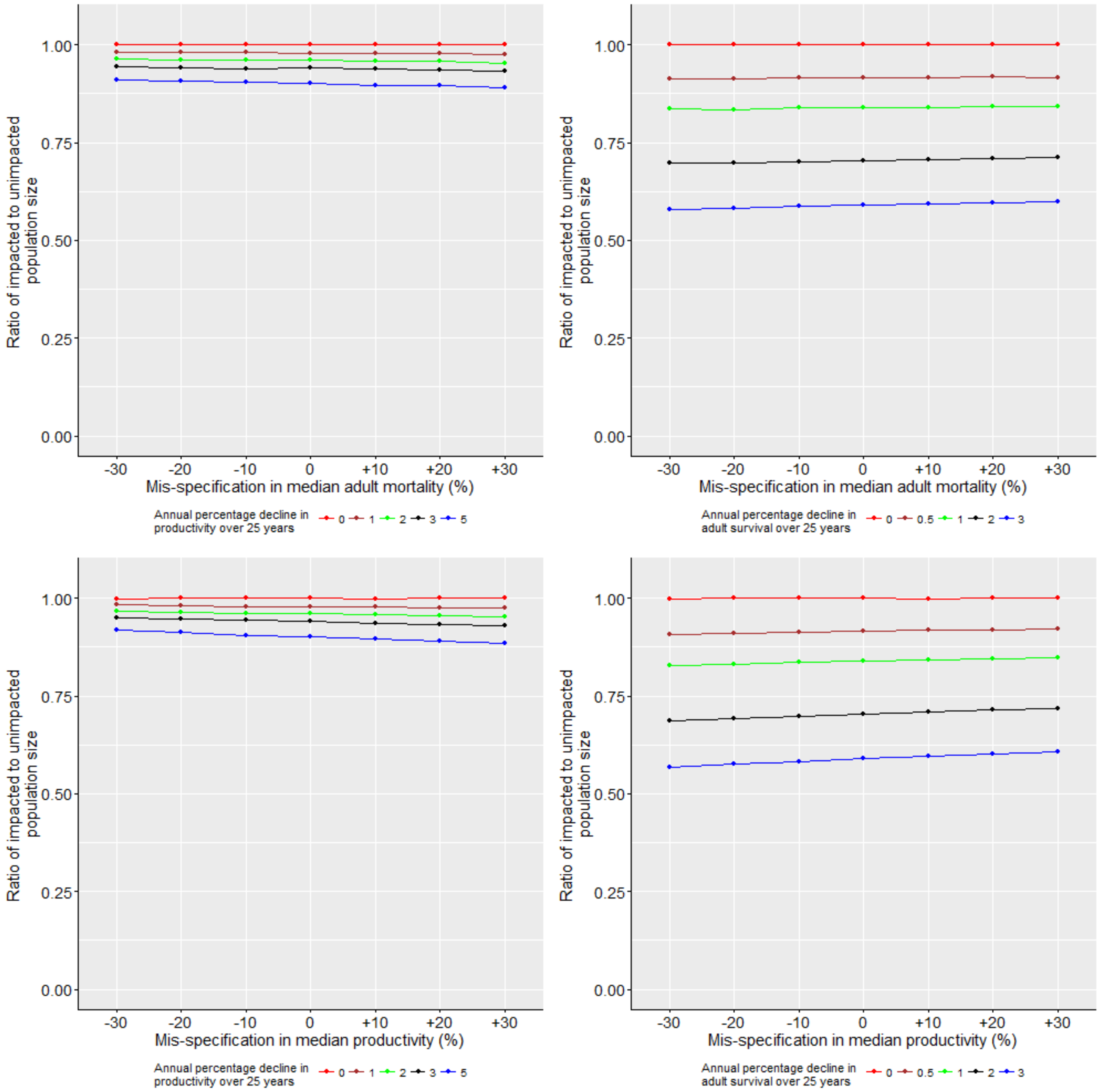


Figure 4c: PVA Metric C – difference in population growth rate from 2016-2041, comparing impacted population vs. un-impacted population. Adult mortality mis-specification is illustrated in the upper panels and productivity mis-specification in the lower panels. Mis-specification was varied from -30% to +30% (with 0% representing no mis-specification). The five coloured lines represent the different levels of potential impact on annual productivity (left panels) or annual adult survival (right panels) over the hypothetical 25 year lifetime of the wind farm (2016-2041).

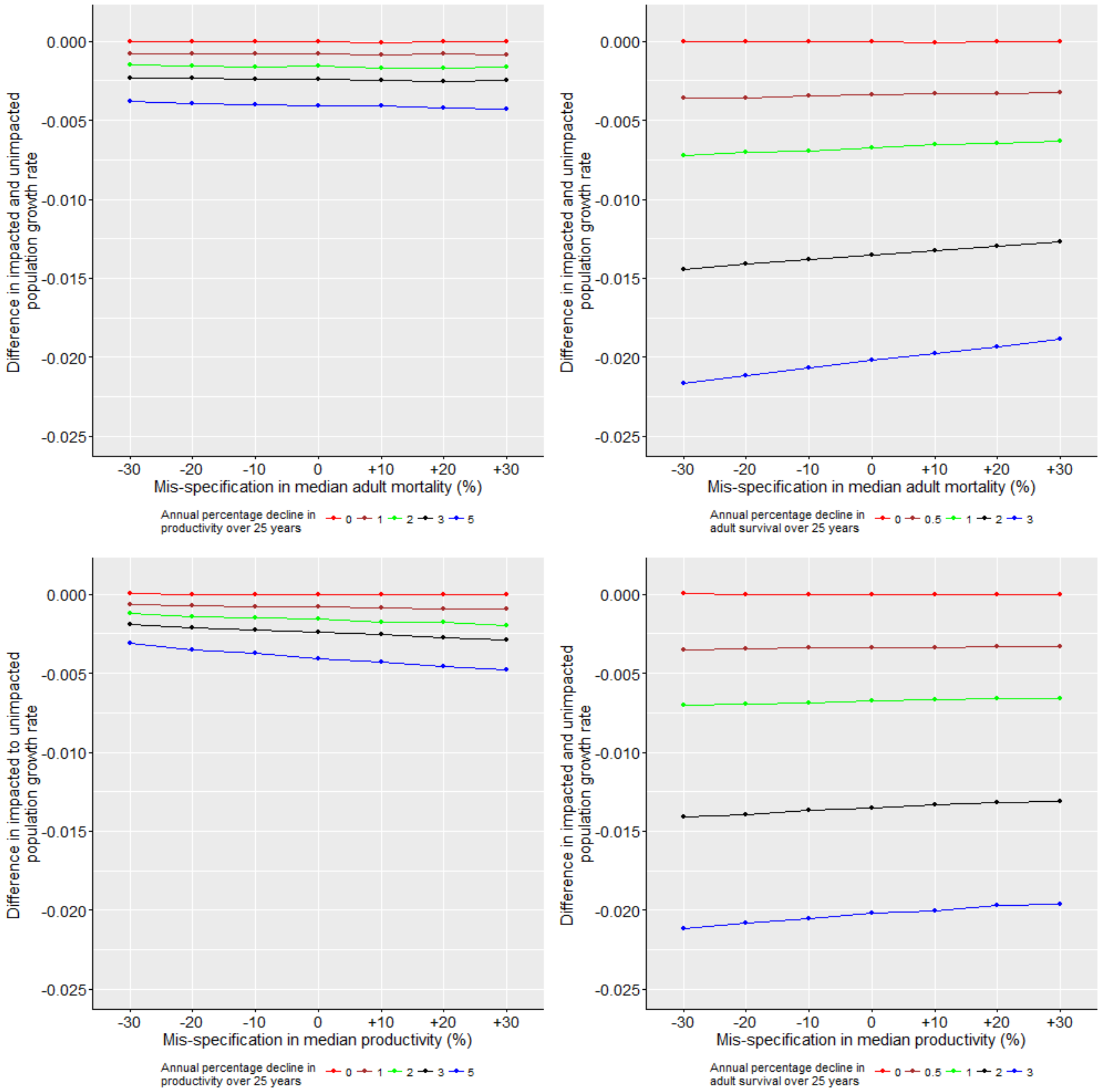


Figure 4d: PVA Metric D – difference in population size at 2041, comparing impacted population vs. un-impacted population. Adult mortality mis-specification is illustrated in the upper panels and productivity mis-specification in the lower panels. Mis-specification was varied from -30% to +30% (with 0% representing no mis-specification). The five coloured lines represent the different levels of potential impact on annual productivity (left panels) or annual adult survival (right panels) over the hypothetical 25 year lifetime of the wind farm (2016-2041).

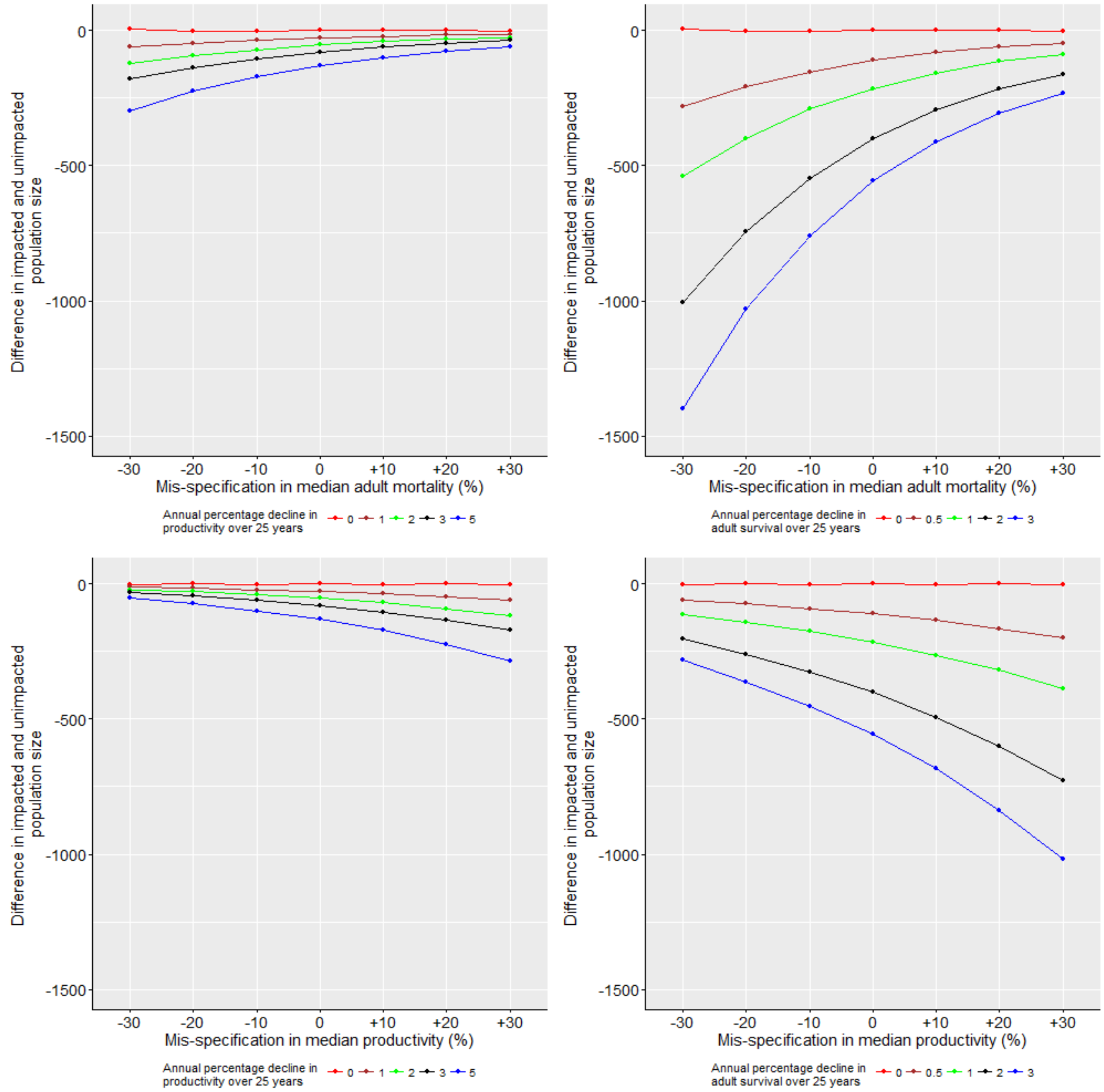


Figure 4e: PVA Metric E1 – probability of population decline greater than 10% from 2016-2041. Adult mortality mis-specification is illustrated in the upper panels and productivity mis-specification in the lower panels. Mis-specification was varied from -30% to +30% (with 0% representing no mis-specification). The five coloured lines represent the different levels of potential impact on annual productivity (left panels) or annual adult survival (right panels) over the hypothetical 25 year lifetime of the wind farm (2016-2041).

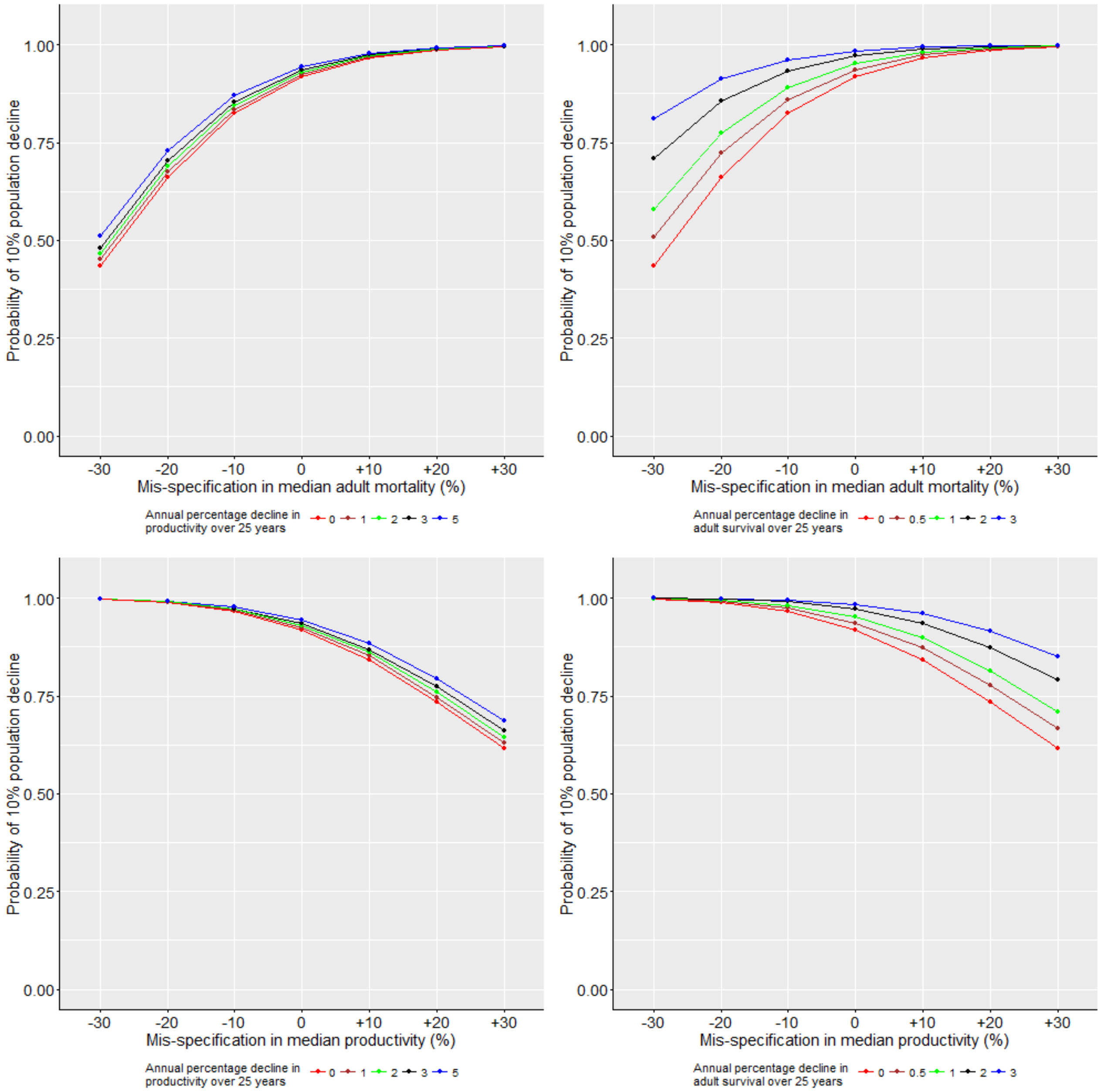


Figure 4f: PVA Metric E2 – probability of population decline greater than 25% from 2016-2041. Adult mortality mis-specification is illustrated in the upper panels and productivity mis-specification in the lower panels. Mis-specification was varied from -30% to +30% (with 0% representing no mis-specification). The five coloured lines represent the different levels of potential impact on annual productivity (left panels) or annual adult survival (right panels) over the hypothetical 25 year lifetime of the wind farm (2016-2041).

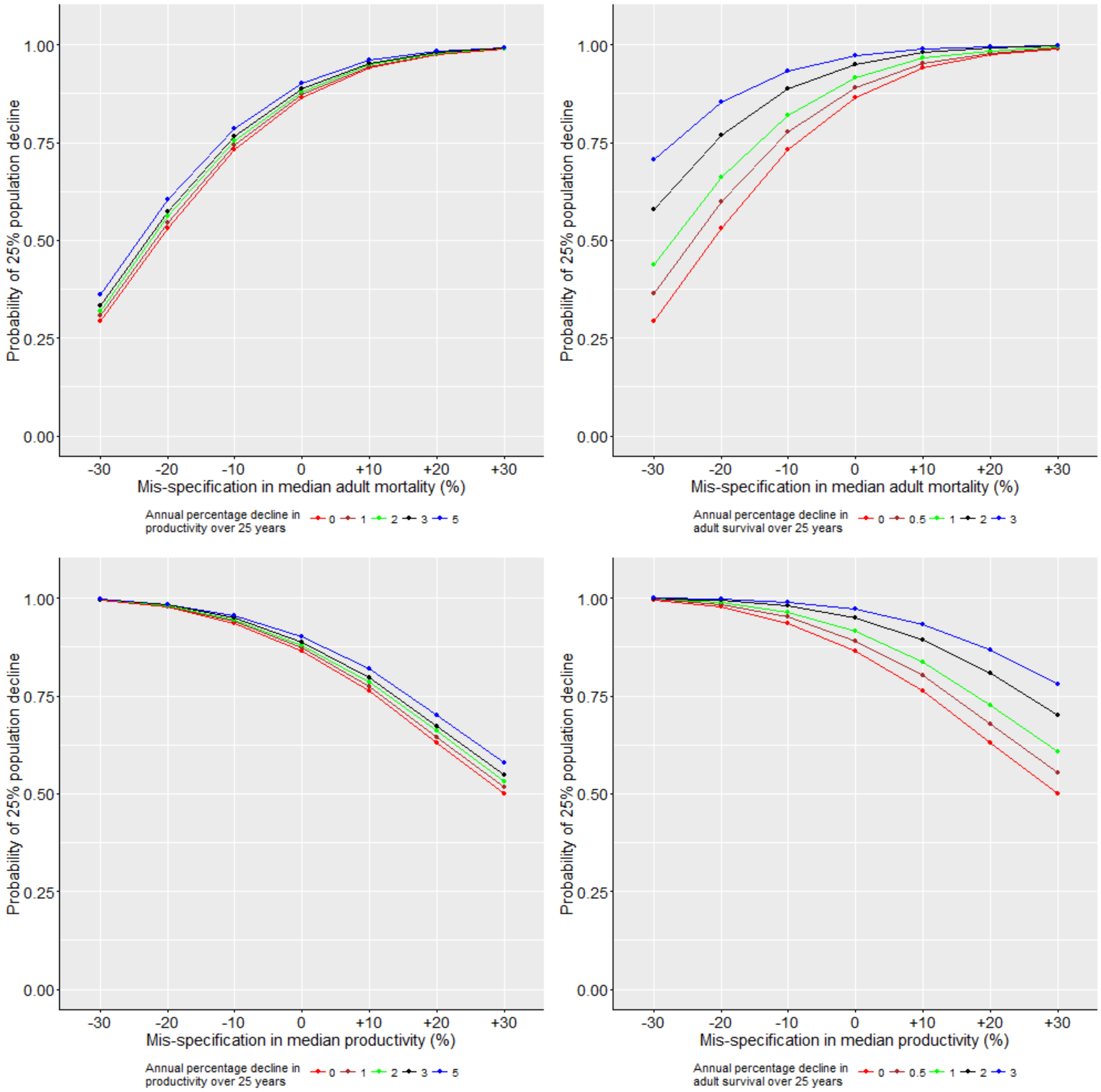


Figure 4g: PVA Metric E3 – probability of population decline greater than 50% from 2016-2041. Adult mortality mis-specification is illustrated in the upper panels and productivity mis-specification in the lower panels. Mis-specification was varied from -30% to +30% (with 0% representing no mis-specification). The five coloured lines represent the different levels of potential impact on annual productivity (left panels) or annual adult survival (right panels) over the hypothetical 25 year lifetime of the wind farm (2017-2041).

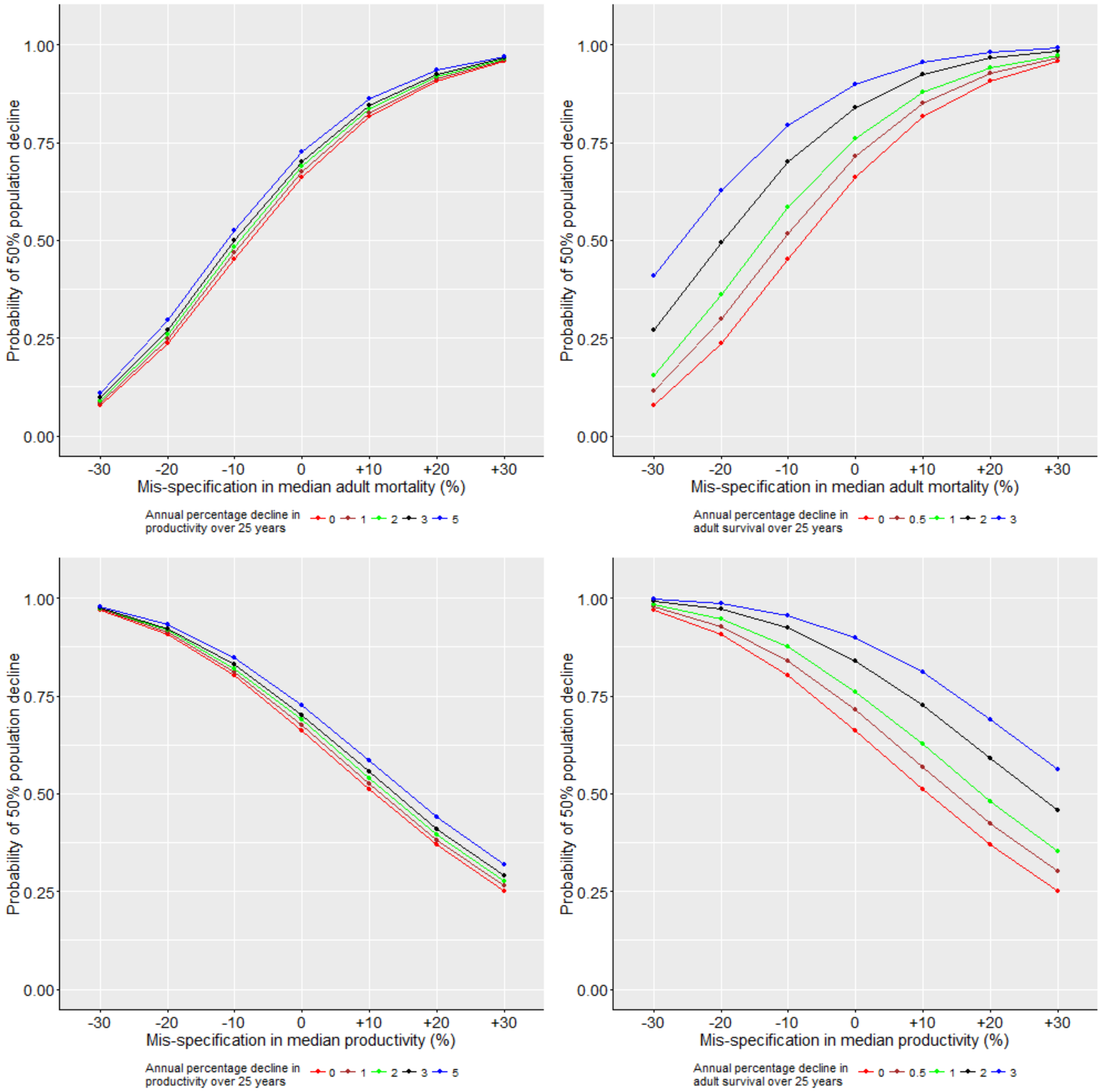
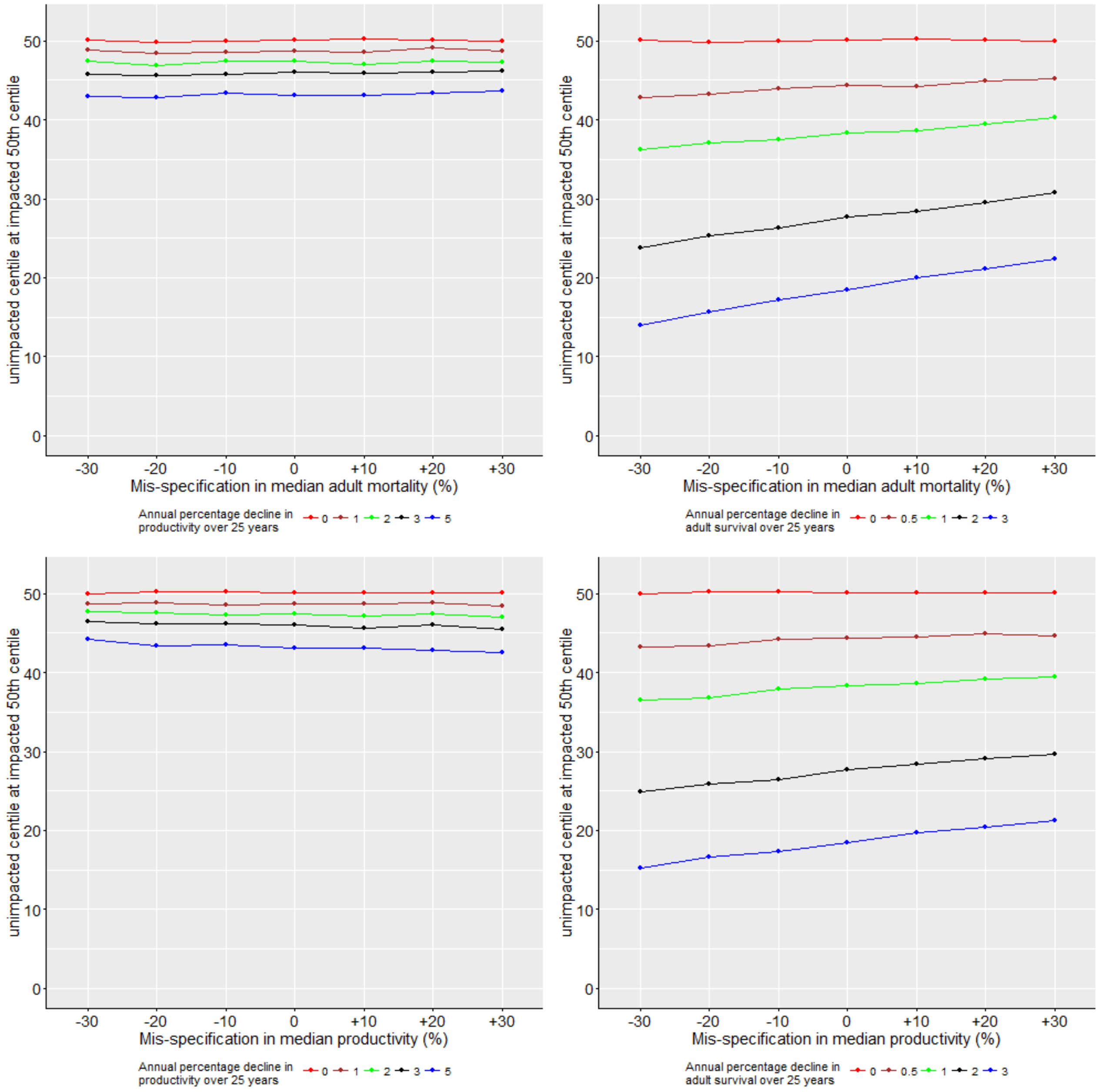


Figure 4h: PVA Metric F – centile from un-impacted population size equal to the 50th centile of the impacted population size, at 2041. Adult mortality mis-specification is illustrated in the upper panels and productivity mis-specification in the lower panels. Mis-specification was varied from -30% to +30% (with 0% representing no mis-specification). The five coloured lines represent the different levels of potential impact on annual productivity (left panels) or annual adult survival (right panels) over the hypothetical 25 year lifetime of the wind farm (2016-2041).



4.2 PVA Sensitivity in Relation to Population Status and Renewables Effect Size

To examine the effects of population status and renewables effect size, we integrated the results for the 13 SPA species/combinations for which we had good model convergence at the time of writing:

- Kittiwakes: Forth Islands; St Abb’s Head; Fowlsheugh; Buchan Ness to Collieston Coast
- Guillemots: Forth Islands; St Abb’s Head; Fowlsheugh; Buchan Ness to Collieston Coast
- Razorbills: Forth Islands; St Abb’s Head; Fowlsheugh
- Shags: Forth Islands; St Abb’s Head

Six of the thirteen indicated increasing abundance over time. These are guillemots at Forth Islands, St Abb’s Head and Buchan Ness to Collieston Coast, razorbills at Forth Islands and Fowlsheugh and shags at Forth Islands, while the remainder showed a decrease (Table 7), providing a comparatively even balance facilitating this comparison. Results for differences in sensitivity in decreasing and increasing populations can be found in Figures 5a-h for PVA A, B, C, D, E1, E2, E3 and F, respectively. These plots show results from the analysis of mis-specification in adult mortality with the maximum scenario of change in adult survival (3%).

We present PVA sensitivities in relation to scenarios of renewables effect size in Figures 6a-h for PVA A, B, C, D, E1, E2, E3 and F, respectively. Of the four combinations shown in Figures 3 and 4, we only show results from the analysis of mis-specification in adult mortality with scenarios of change in adult survival, with effect sizes of 0.5%, 1%, 2% and 3%.

For PVA A, values approximate one (range 0.977-1) and there was no discernible difference in sensitivity between decreasing and increasing populations or with respect to renewables effect size (Figures 5a and 6a). Note that although annual growth rates are close to one, 25 year growth rates will show a discernible difference. For example, an annual growth rate of 0.977, results in a 25 year growth rate of 0.559.

For PVA B, there was also no discernible difference in sensitivity between decreasing and increasing species (Figure 5b). There was an increase in sensitivity with increasing effect sizes, with slopes flatter at 0.5% effect size compared with 3%

effect size, though the effect was small and the metric can be considered comparatively insensitive to all scenarios of effect size (Figure 6b).

PVAs C and D had higher sensitivity than PVAs A and B overall, but showed a similar response to population status and renewables effect size to PVA B, such that there was no clear difference between decreasing and increasing species in slope (Figure 5c and 5d), and a slight increase in gradient with increasing effect size from 0.5% to 3% (Figure 6c and 6d).

For PVA E, increasing populations showed greater sensitivity to probability of population decline greater than 10% than decreasing populations (Figure 5e), whereas the converse was true for a probability of population decline greater than 50% (Figure 5g). Similar sensitivities were apparent at 25% (Figure 5f). These differences reflect the pattern of probabilities of thresholds of change in population size relative to population status, with mis-specification having a smaller effect on probability of a smaller change in population size (10%) in a decreasing population since probability of this outcome is very high in most circumstances, and a smaller effect on probability of a larger change in population size (50%) in an increasing population (where probability of this outcome is very low in most circumstances). There was no clear difference in sensitivity with respect to renewables effect size, being comparatively high and variable in all scenarios at all three thresholds (Figures 6e-g).

PVA F showed a similar response to PVAs A, B, C and D with respect to population status and effect size. Thus, there was no clear difference in sensitivity between decreasing and increasing species in slope, with sensitivity overall being moderately low, higher than ratio metrics but lower than PVA E (Figure 5h). Sensitivity was also comparatively unaffected by effect sizes (Figure 6h).

Figure 5a: PVA Metric A – ratio of population growth rate from 2016-2041, comparing impacted population vs. un-impacted population, for changing adult mortality and a 3% decrease in adult survival, across decreasing populations and increasing populations.

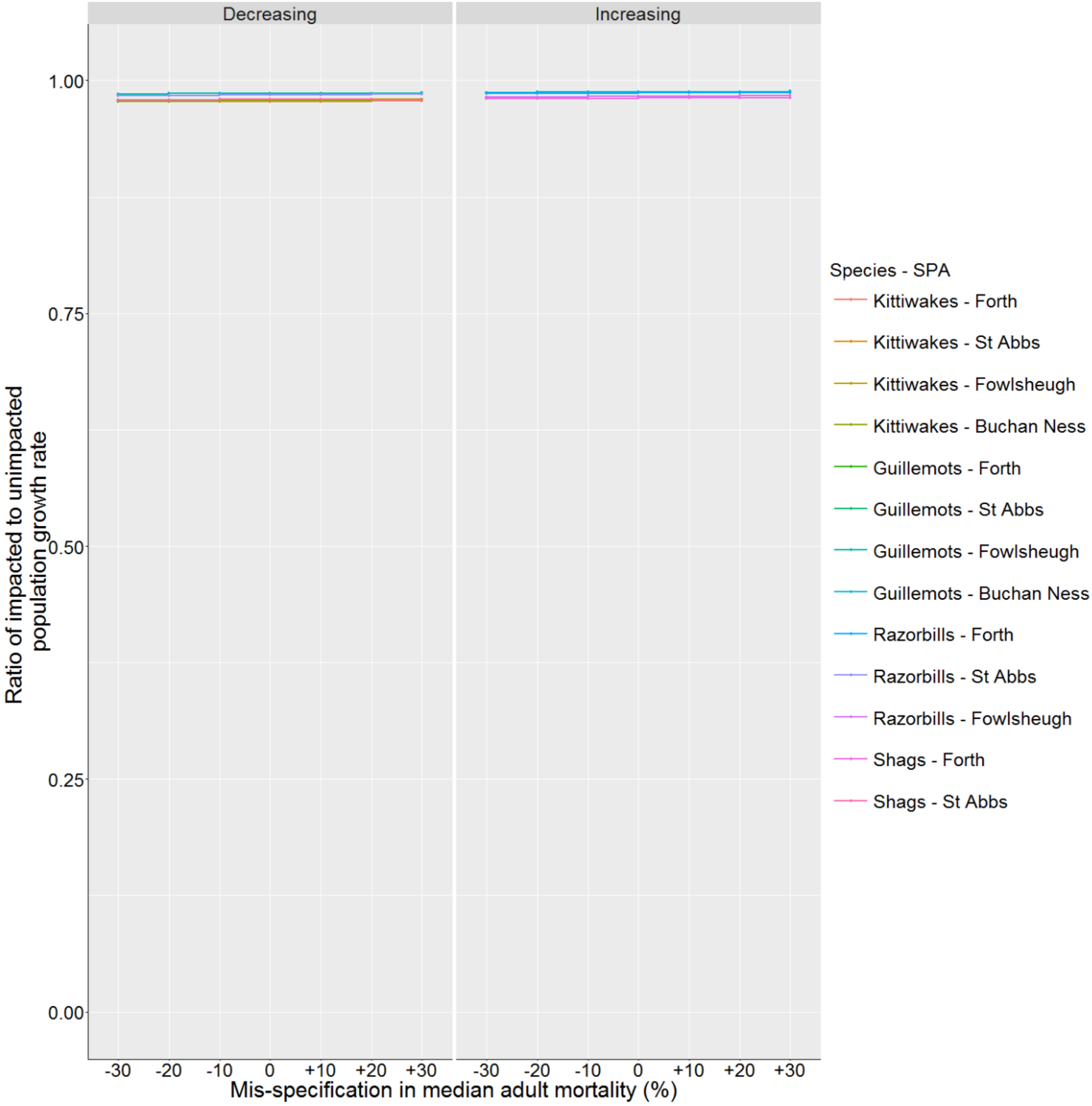


Figure 5b: PVA Metric B – ratio of population size at 2041, comparing impacted population vs. un-impacted population, for changing adult mortality and a 3% decrease in adult survival, across decreasing populations and increasing populations.

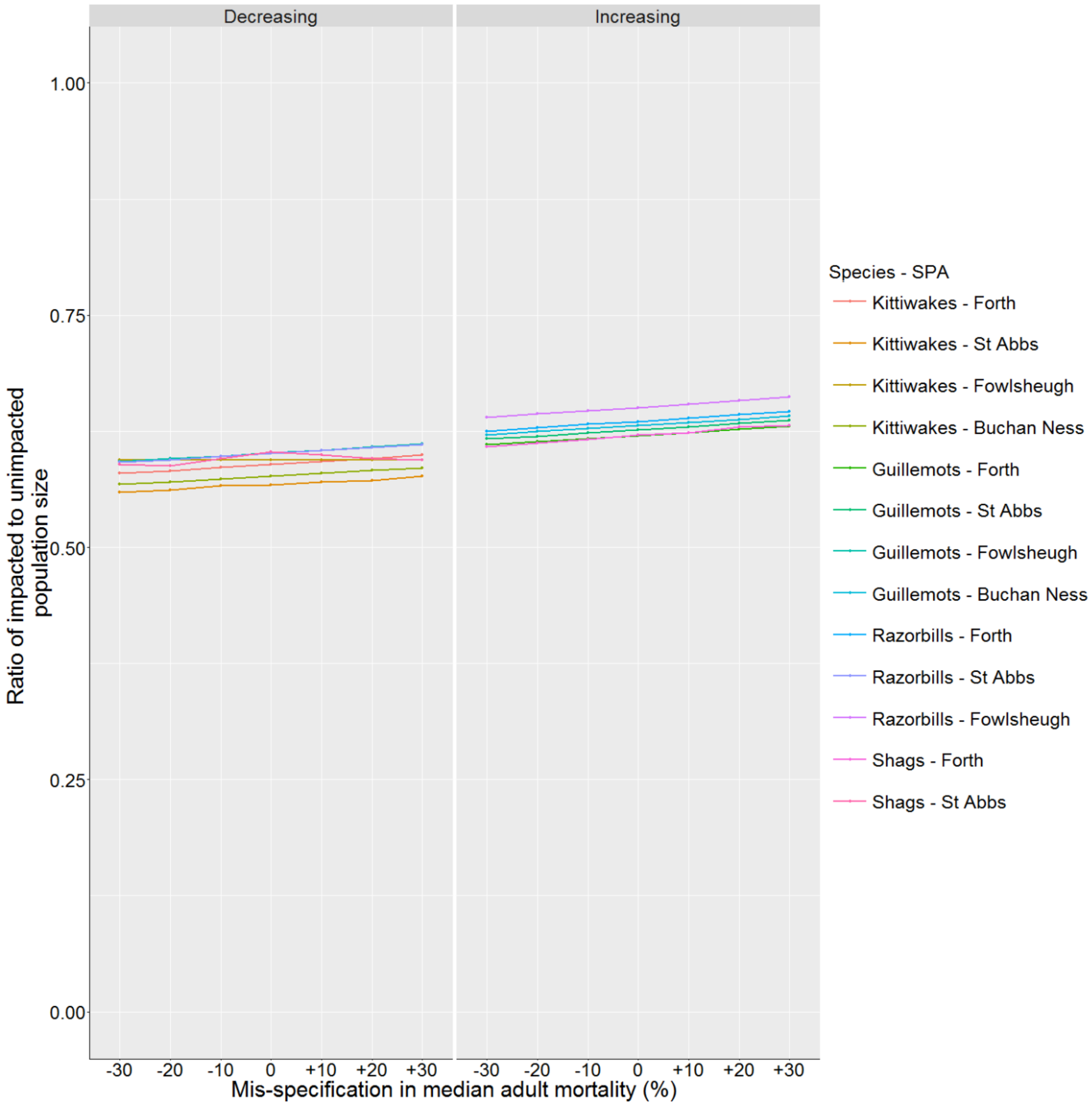


Figure 5c: PVA Metric C – difference in population growth rate from 2016-2041, comparing impacted population vs. un-impacted population, for changing adult mortality and a 3% decrease in adult survival, across decreasing populations and increasing populations.

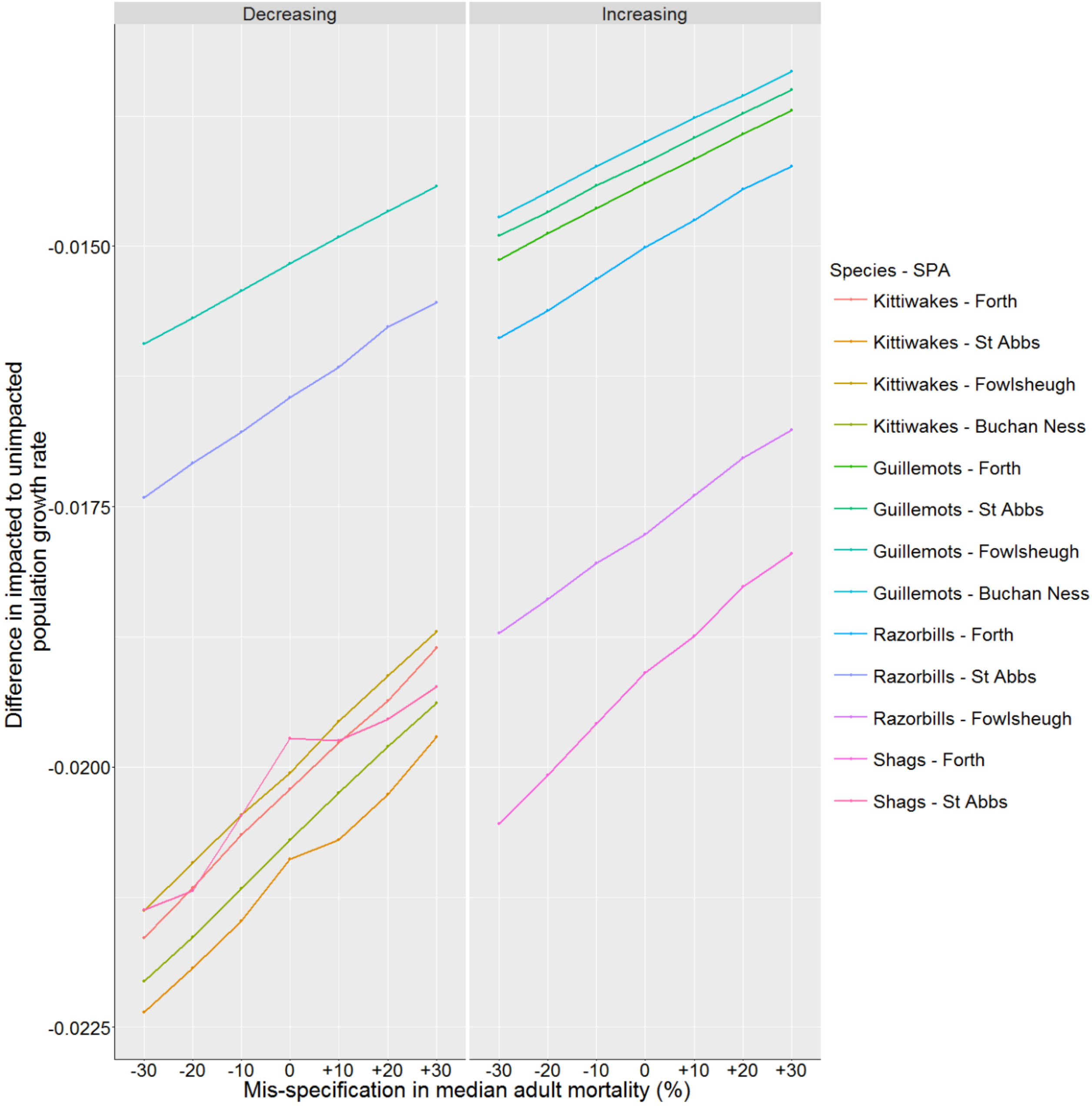


Figure 5d: PVA Metric D – difference in population size at 2041, comparing impacted population vs. un-impacted population, for changing adult mortality and a 3% decrease in adult survival, across decreasing populations and increasing populations.

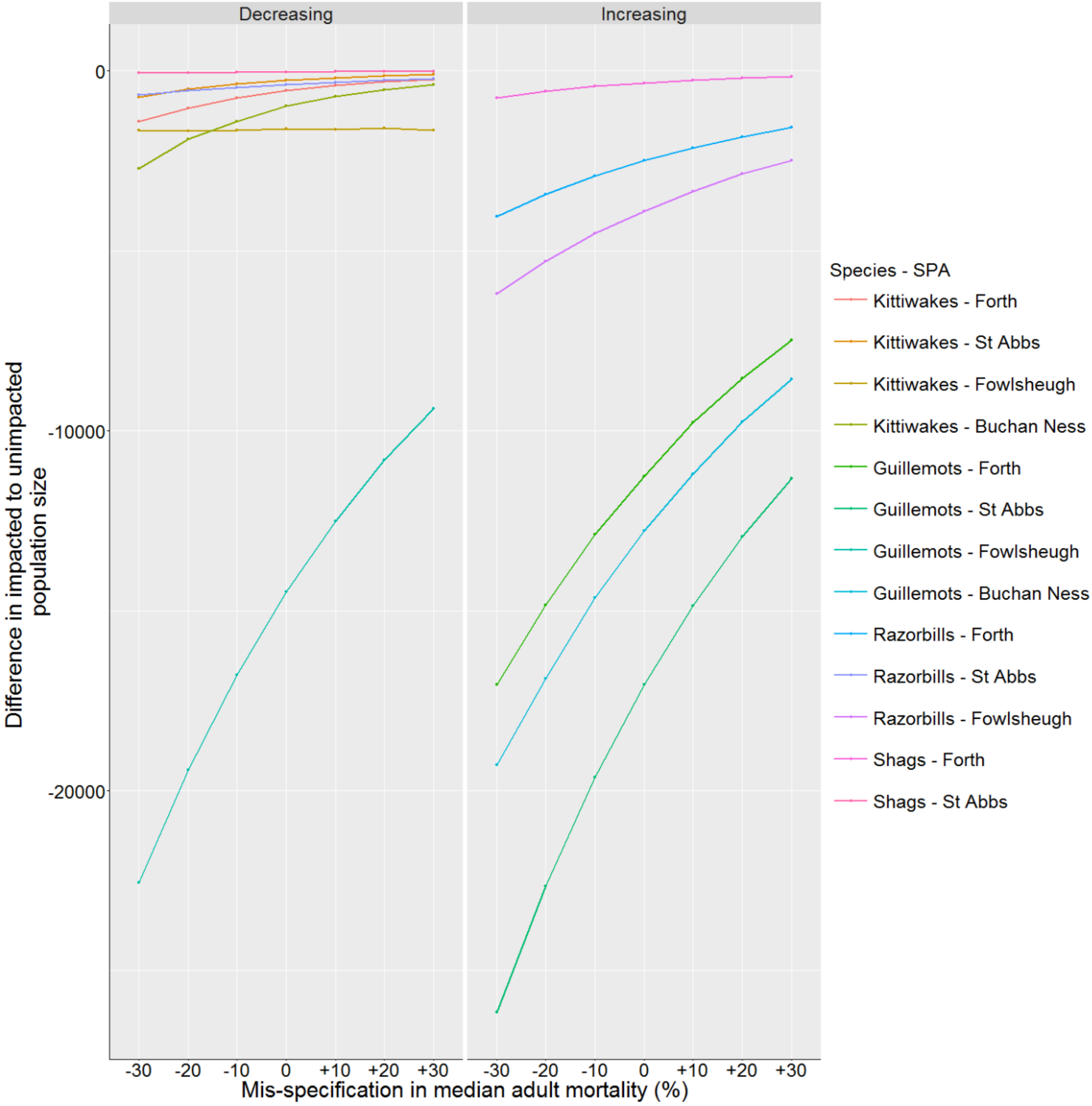


Figure 5e: PVA Metric E1 – probability of population decline greater than 10% from 2016-2041, for changing adult mortality and a 3% decrease in adult survival, across decreasing populations and increasing populations.

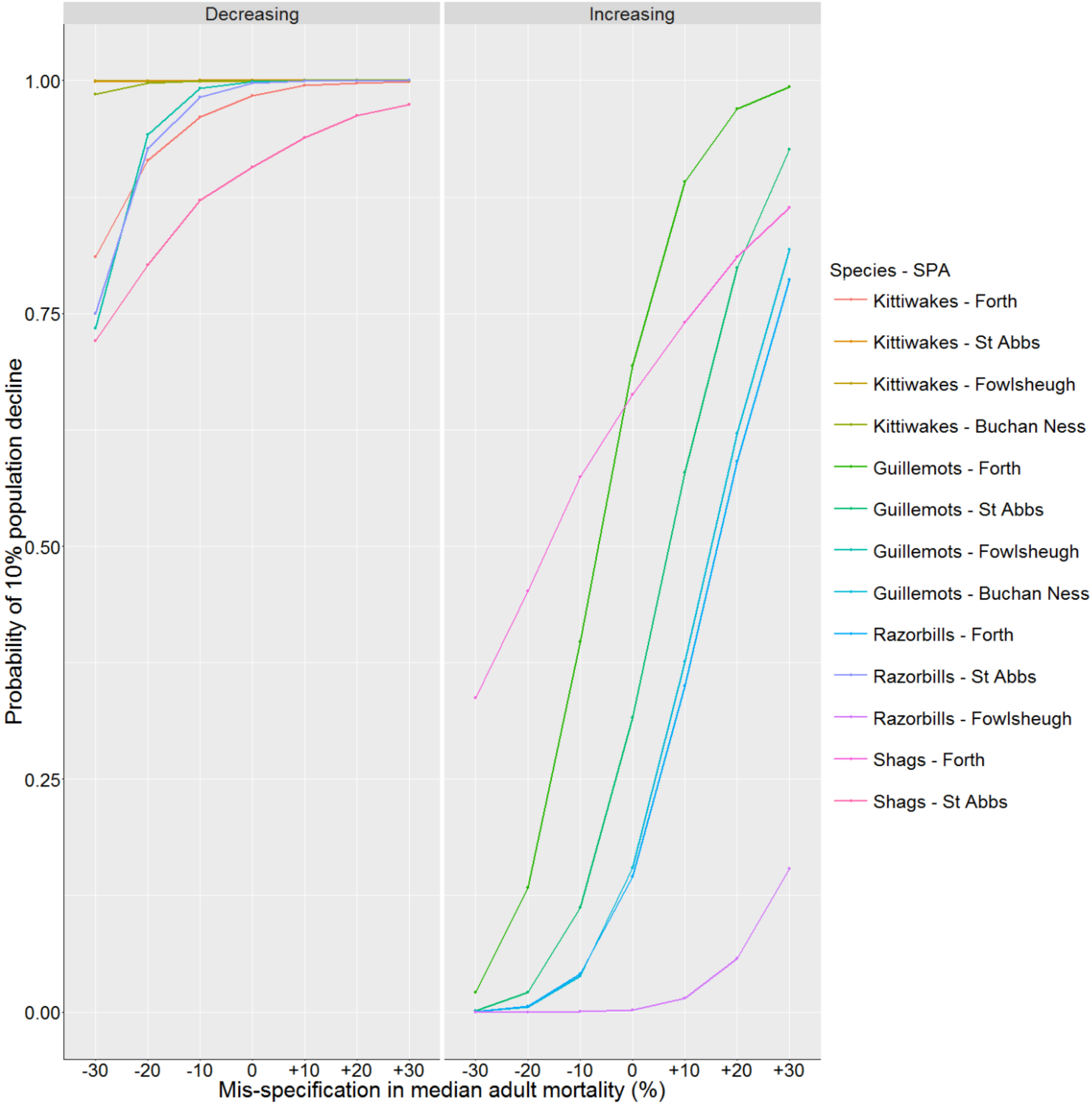


Figure 5f: PVA Metric E2 – probability of population decline greater than 25% from 2016-2041, for changing adult mortality and a 3% decrease in adult survival, across decreasing populations and increasing populations.

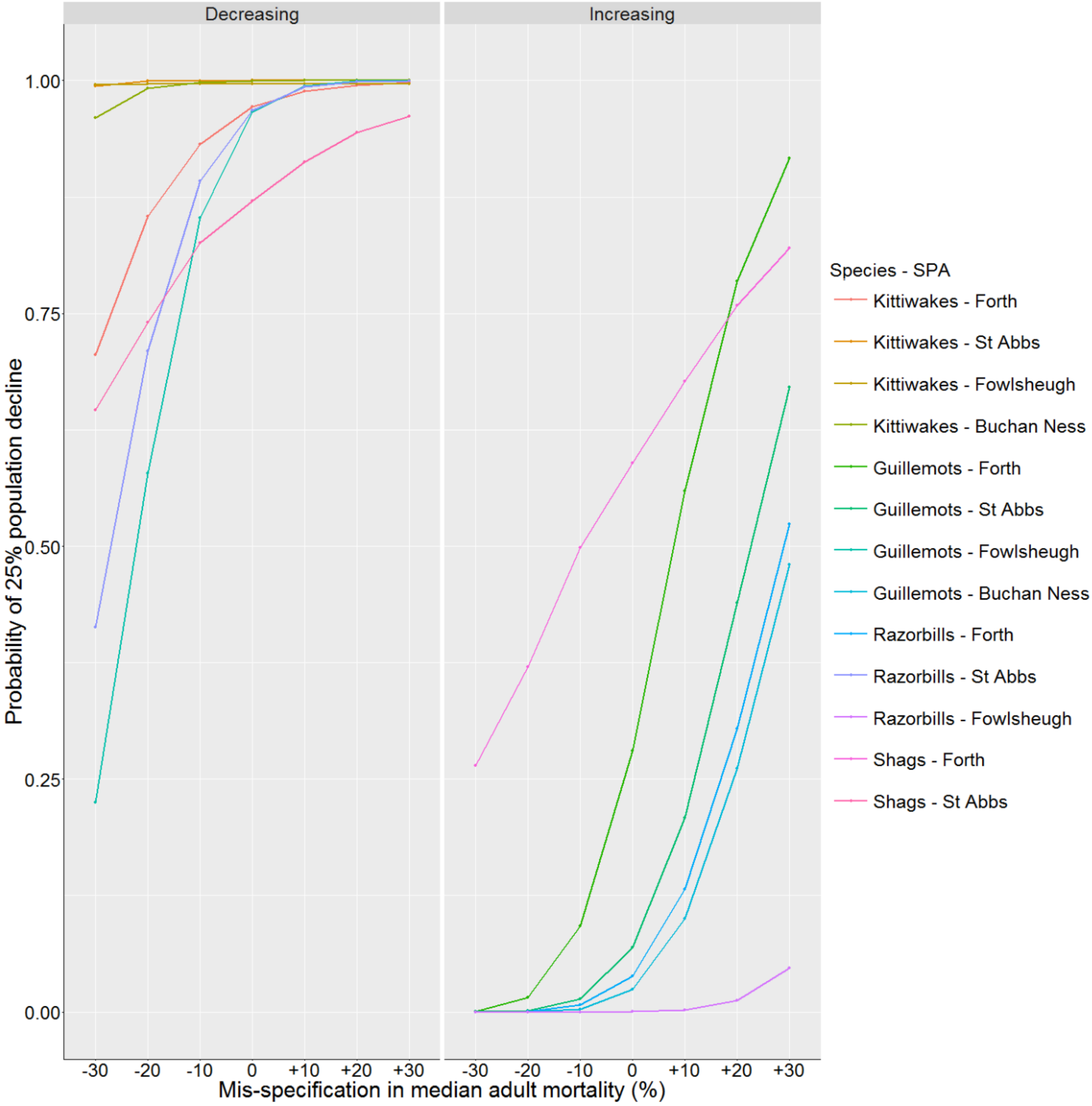


Figure 5g: PVA Metric E3 – probability of population decline greater than 50% from 2016-2041, for changing adult mortality and a 3% decrease in adult survival, across decreasing populations and increasing populations.

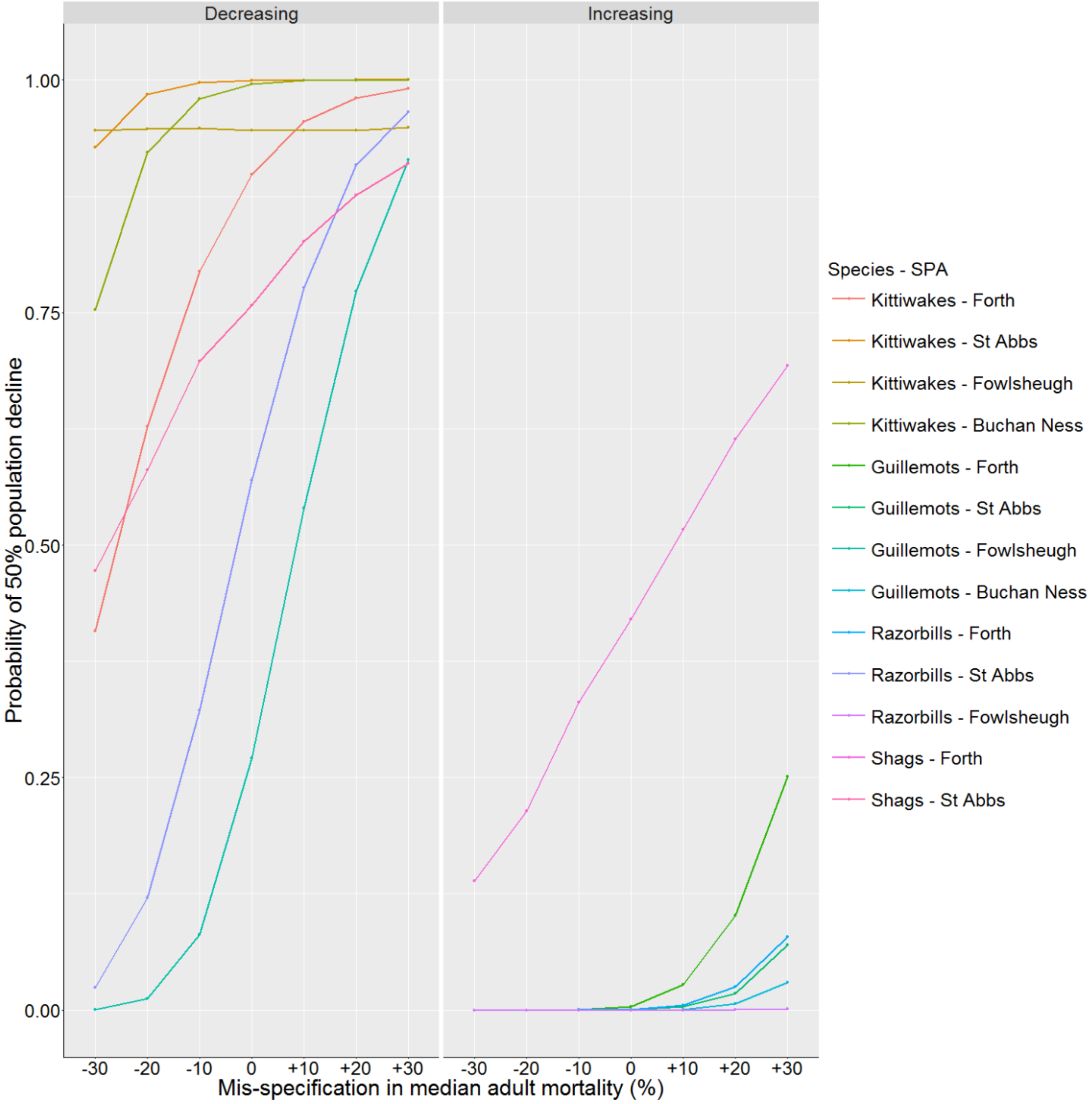


Figure 5h: PVA Metric F – centile from un-impacted population size equal to the 50th centile of the impacted population size, at 2041, for changing adult mortality and a 3% decrease in adult survival, across decreasing populations and increasing populations.

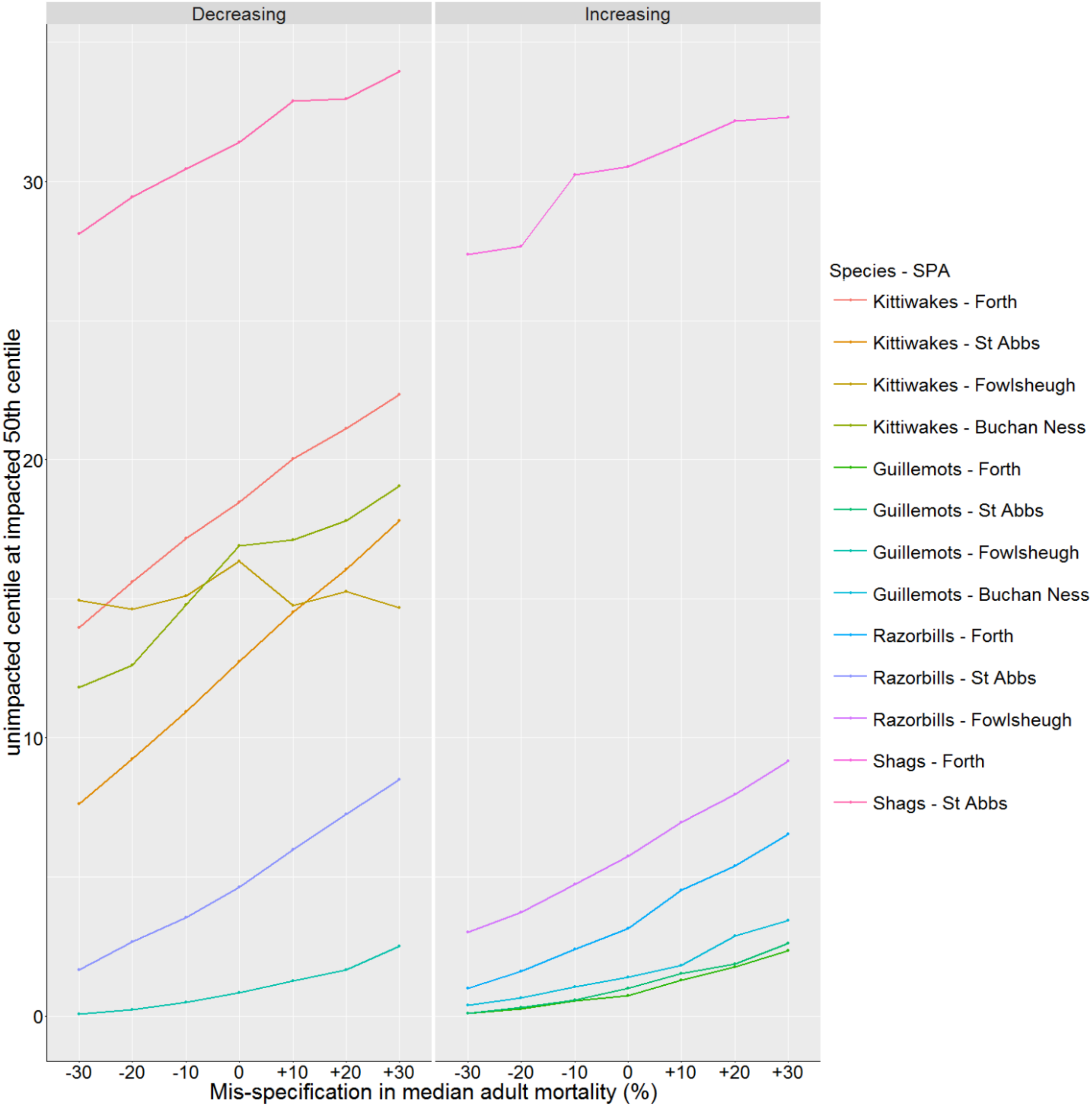


Figure 6a: PVA Metric A – ratio of population growth rate from 2016-2041, comparing impacted population vs. un-impacted population, for changing adult mortality and various decreases in adult survival, across all populations.

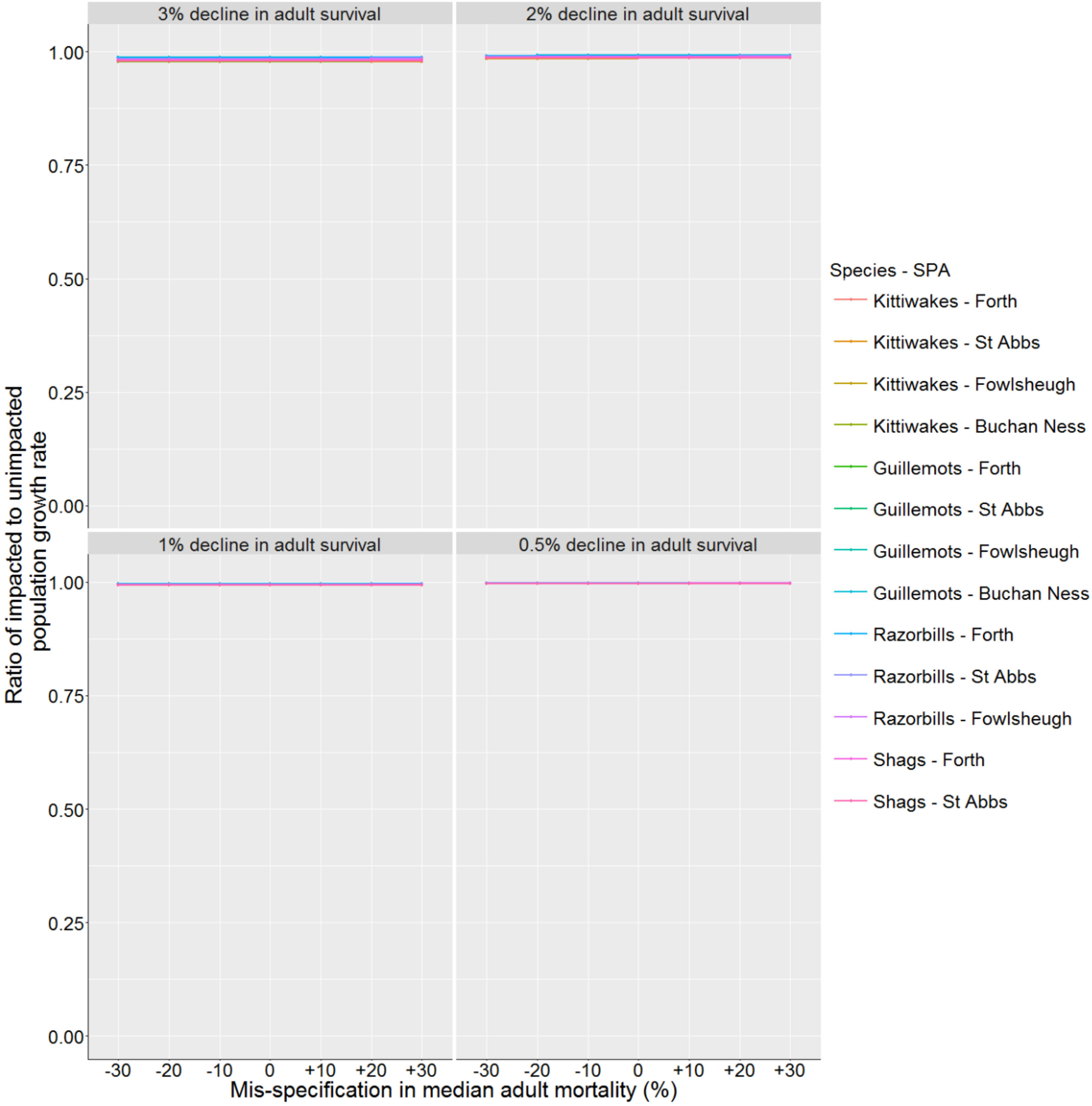


Figure 6b: PVA Metric B – ratio of population size at 2041, comparing impacted population vs. un-impacted population, for changing adult mortality and various decreases in adult survival, across all populations.

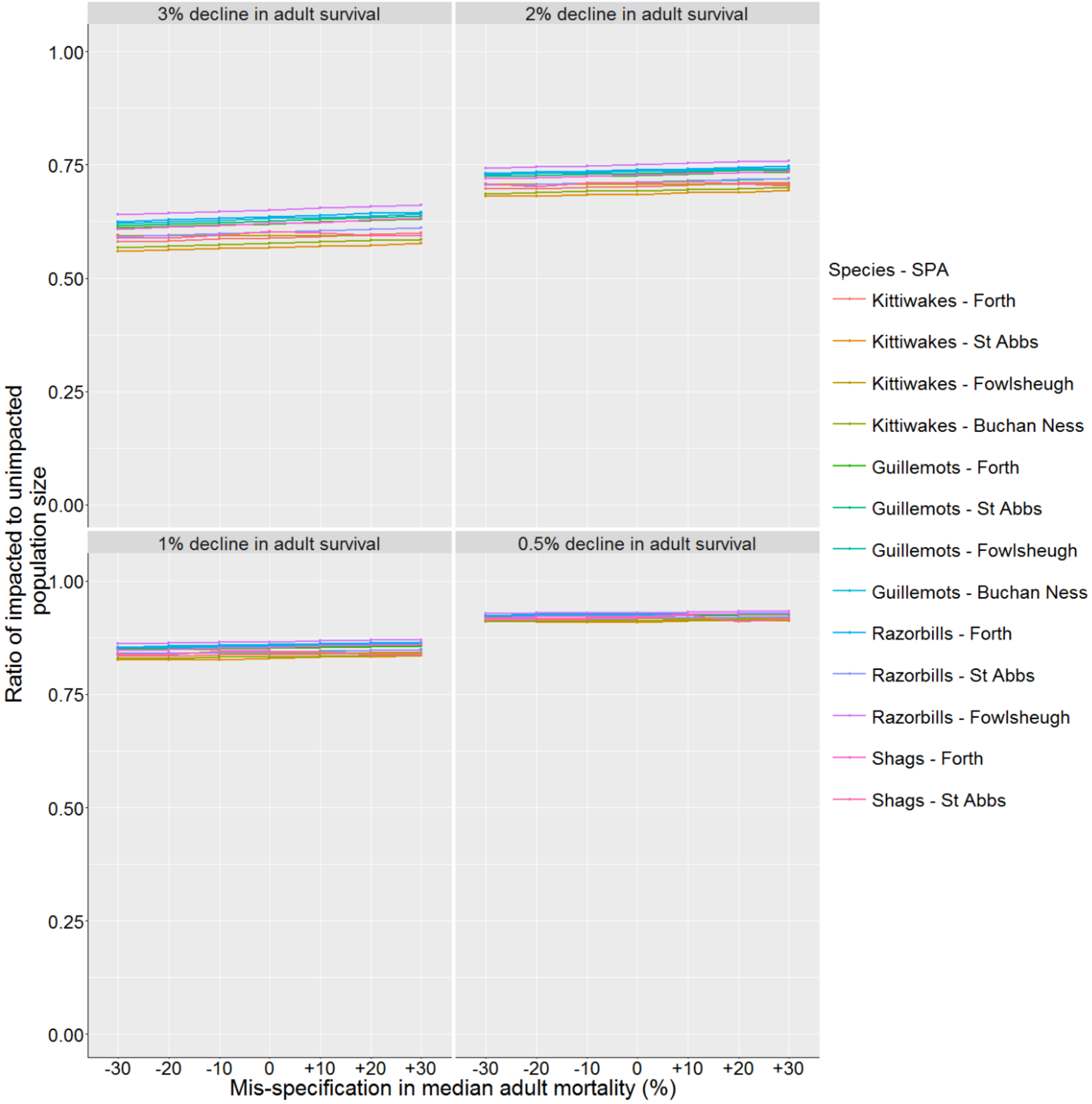


Figure 6c: PVA Metric C – difference in population growth rate from 2016-2041, comparing impacted population vs. un-impacted population, for changing adult mortality and various decreases in adult survival, across all populations.

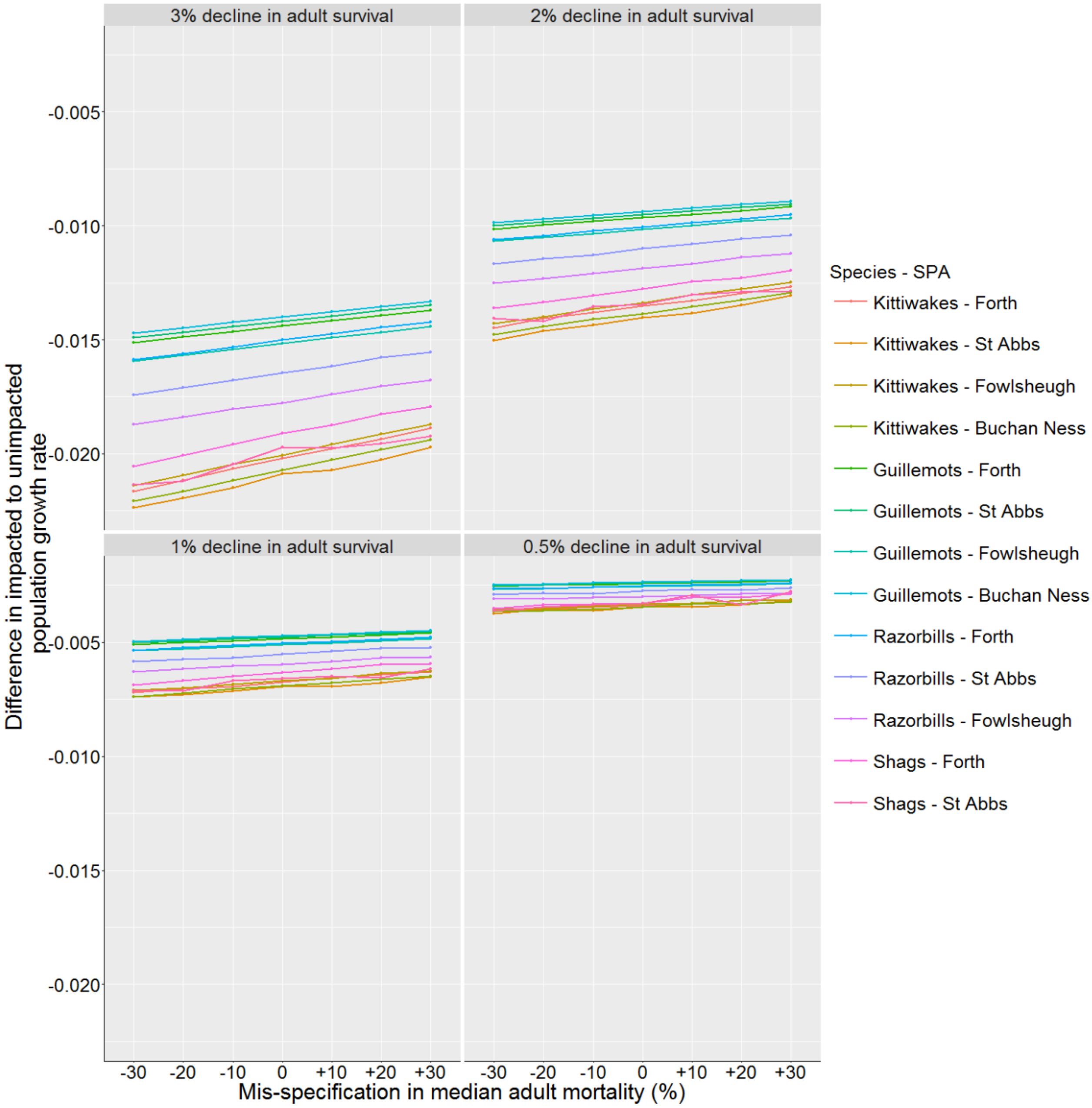


Figure 6d: PVA Metric D – difference in population size at 2041, comparing impacted population vs. un-impacted population, for changing adult mortality and various decreases in adult survival, across all populations.

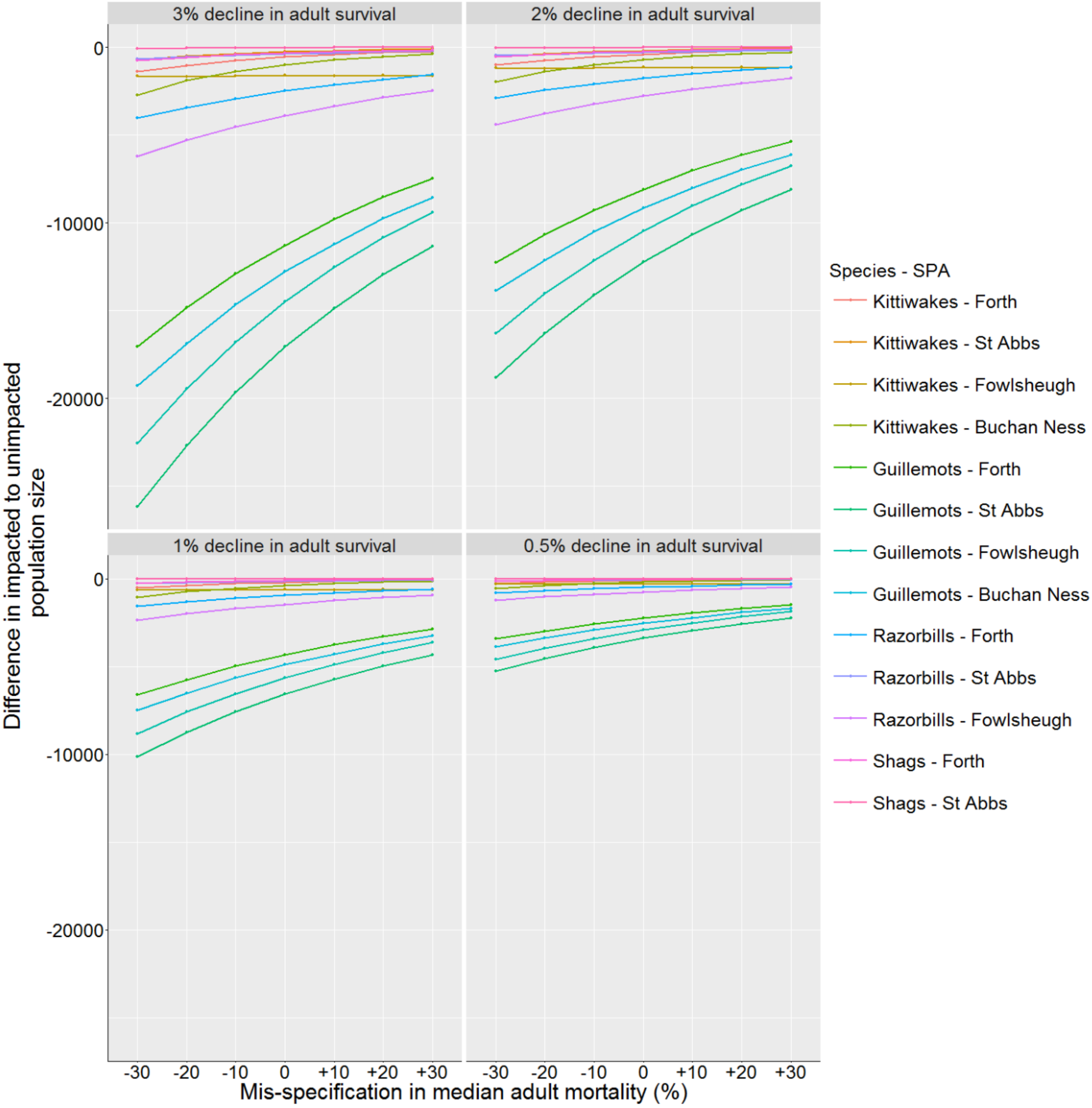


Figure 6e: PVA Metric E1 – probability of population decline greater than 10% from 2016-2041, for changing adult mortality and various decreases in adult survival, across all populations.

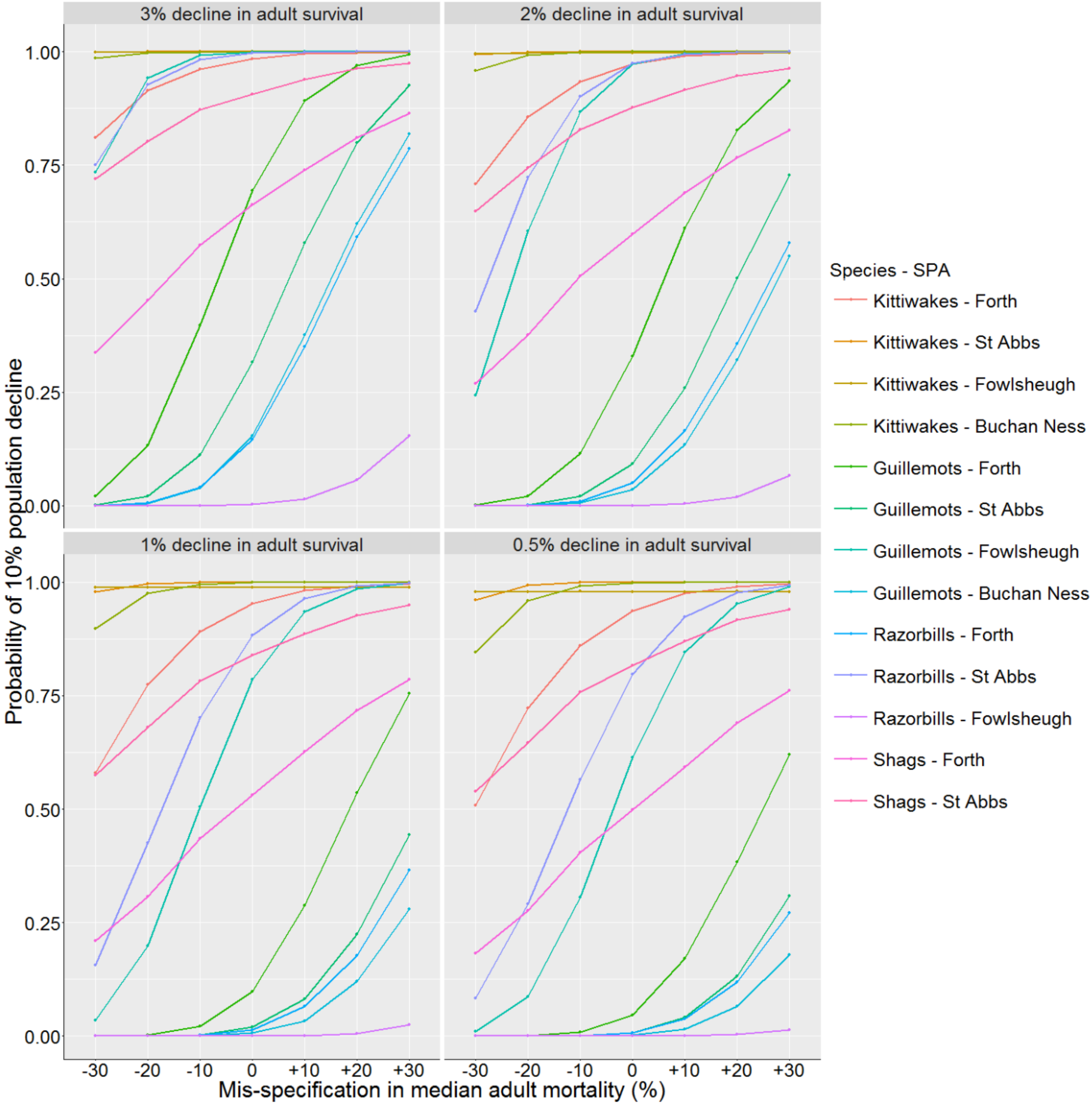


Figure 6f: PVA Metric E2 – probability of population decline greater than 25% from 2016-2041, for changing adult mortality and various decreases in adult survival, across all populations.

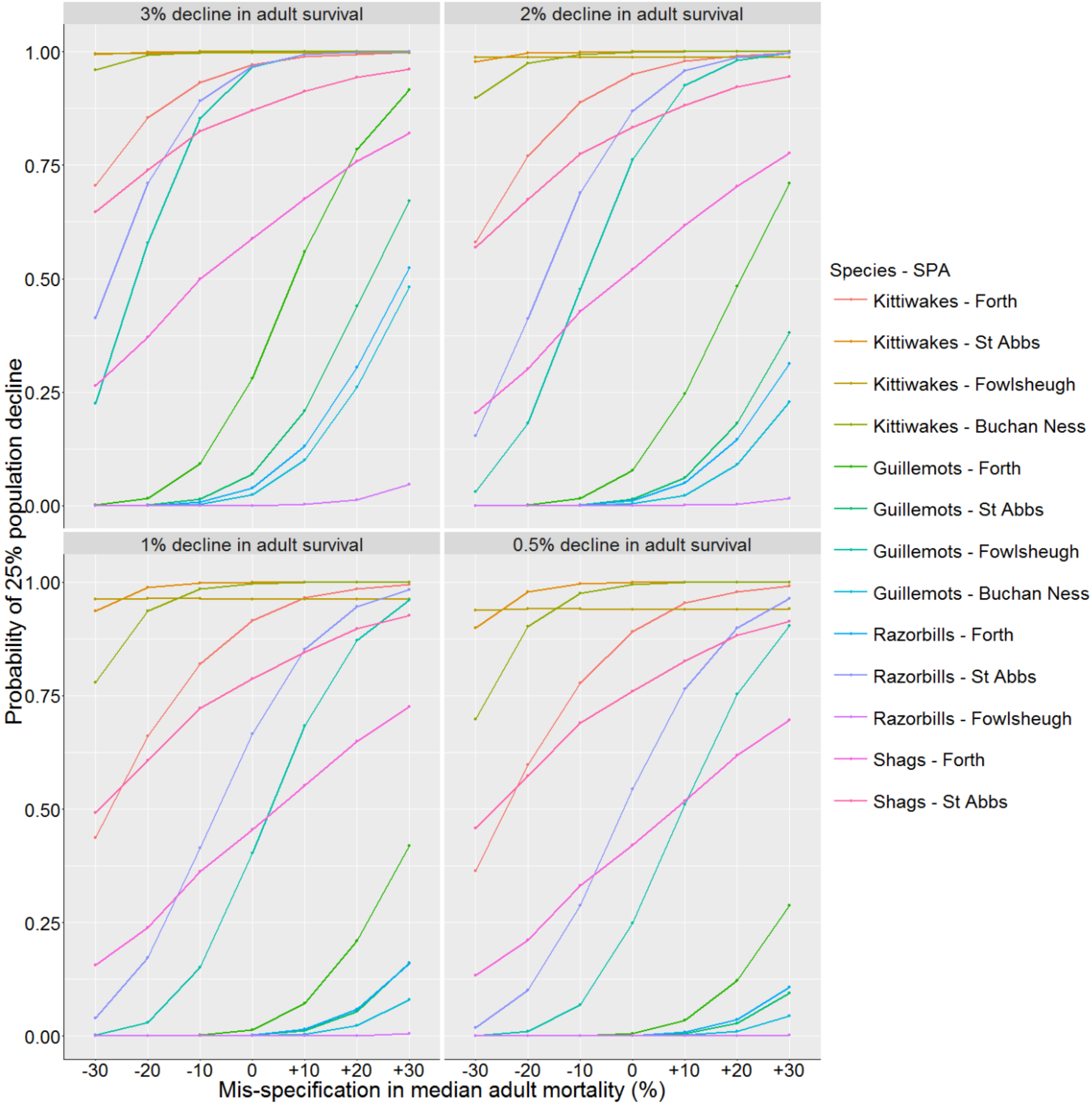


Figure 6g: PVA Metric E3 – probability of population decline greater than 50% from 2016-2041, for changing adult mortality and various decreases in adult survival, across all populations.

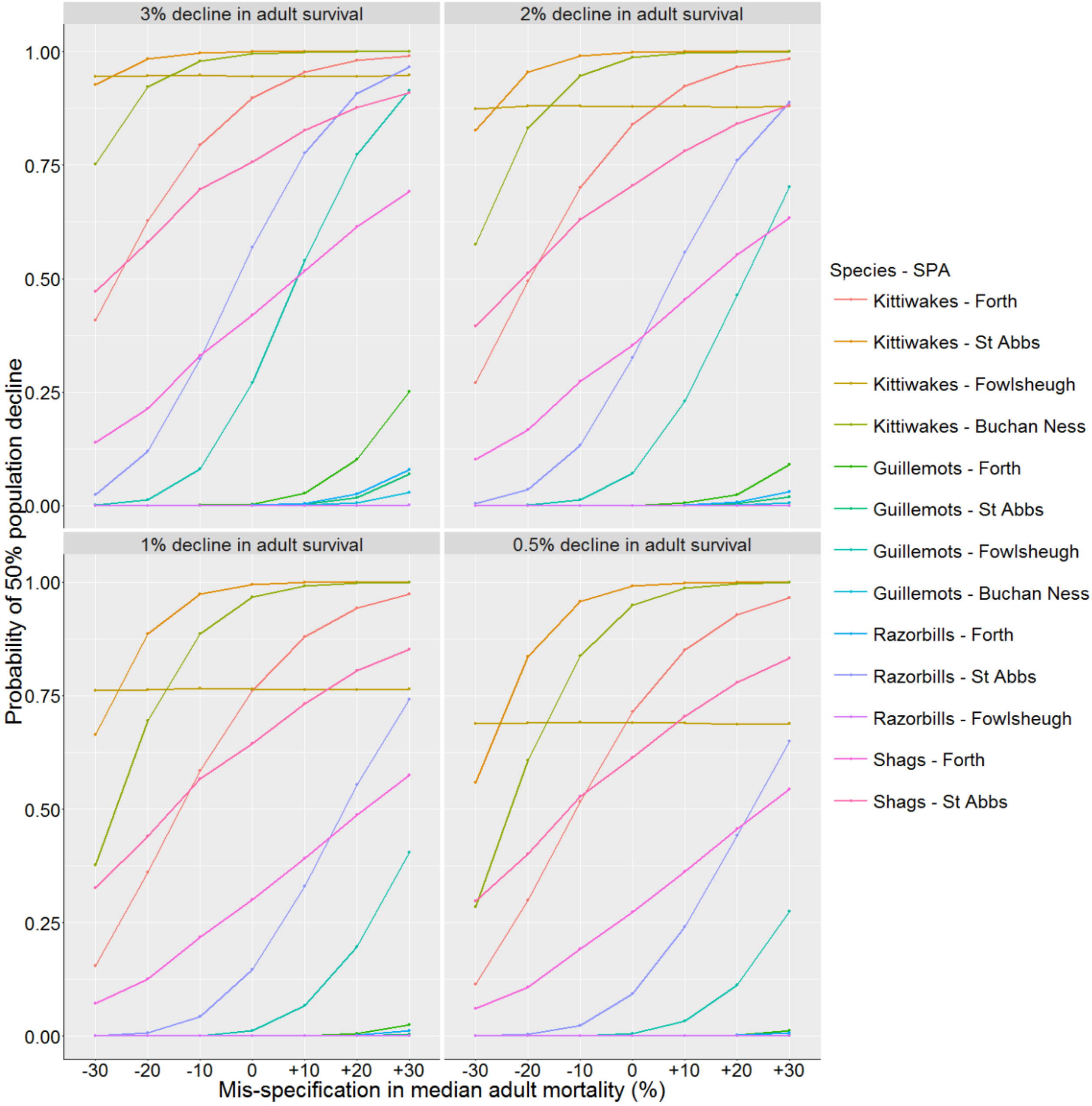
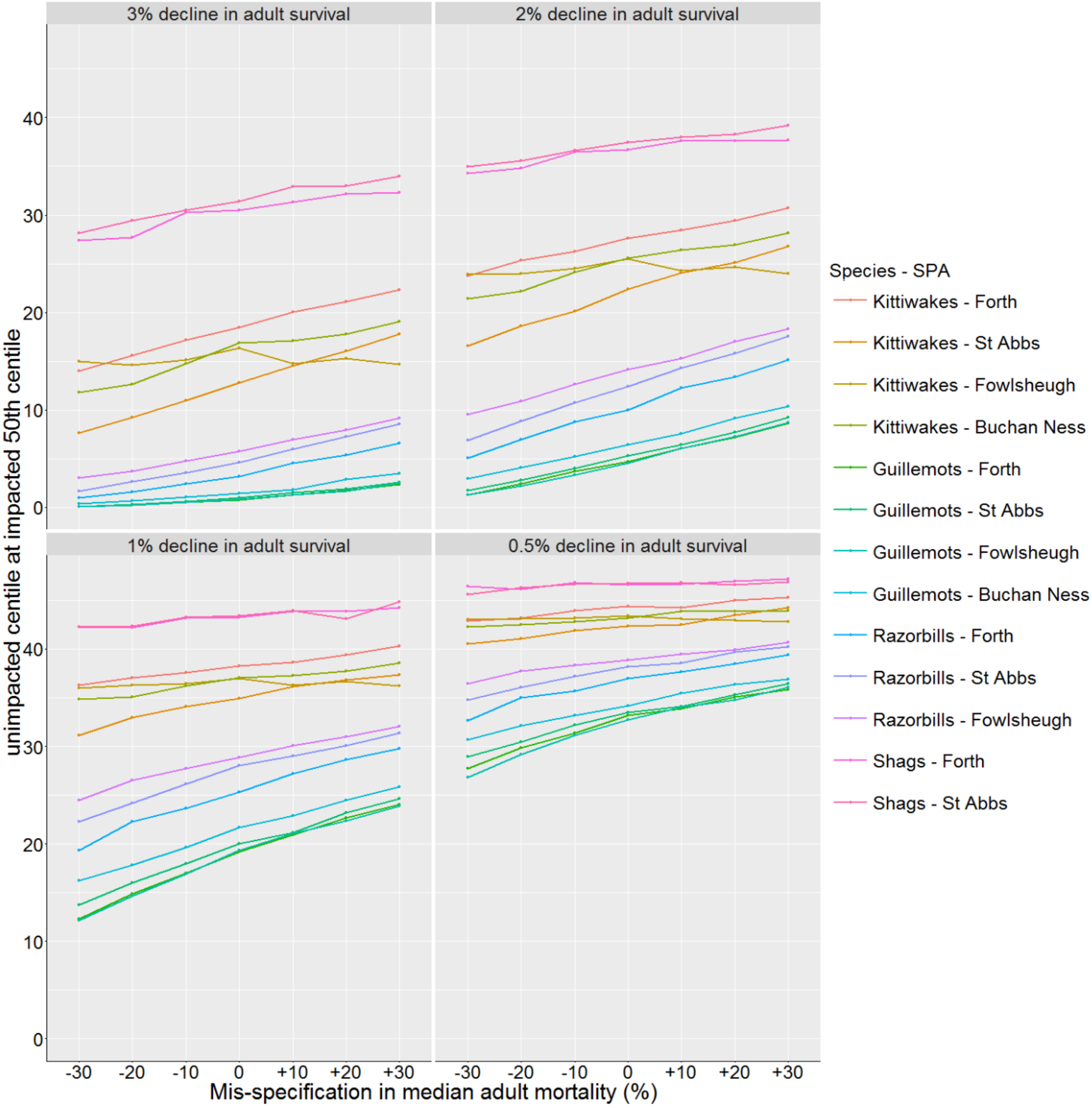


Figure 6h: PVA Metric F – centile from un-impacted population size equal to the 50th centile of the impacted population size, at 2041, for changing adult mortality and various decreases in adult survival, across all populations.



5. Discussion and Recommendations

5.1 PVA Metric Sensitivity

This study represents the most comprehensive assessment of PVA metric sensitivity to mis-specification of demographic rates in relation to population status and perturbation effect sizes in the seabird/marine renewable context using real-world data. Using available data on abundance, survival and productivity in a well-studied region of the UK and Bayesian population modelling approaches, we compared the sensitivity to mis-specification of input demographic parameters of six PVA metrics, comprising two ratio metrics (PVAs A and B), two metrics related to ratio metrics (PVAs C and D) and two probabilistic metrics (PVAs E and F).

By undertaking an analysis of real-world data sets, our work provides a useful complement to recent work on sensitivity of PVA metrics to input parameter uncertainty using simulation modelling of generic seabird species with varying life histories (Cook & Robinson 2016b, 2017). The close accordance in findings provides confidence on choice of PVA metrics that are least sensitive to such mis-specification, and, therefore, most suitable for use in wind farm assessments.

5.2 Recommendations on PVA Metrics

The two ratio metrics performed best among the six metrics considered with respect to sensitivity to mis-specification in input parameters. The ratio of impacted to un-impacted annual growth rate (PVA A) and ratio of impacted to un-impacted population size after 25 years (PVA B) both showed low sensitivity to demographic input mis-specification, in accordance with findings from other studies (Green *et al.* 2016; Cook & Robinson 2016b, 2017), with PVA A performing consistently better than PVA B.

The calculations of difference in impacted and un-impacted annual growth rates (PVA C) and between impacted and un-impacted population size after 25 years (PVA D) were not so readily interpretable but they are useful when growth rates or population size estimates are small.

In keeping with other work, we found that the probability PVA metric (PVA E) was highly sensitive and we, therefore, caution against using it in this context, in accordance with recommendations by other authors (Green *et al.* 2016; Cook & Robinson 2016b, 2017). We were not tasked with testing the sensitivity of counterfactual probabilistic metrics, in particular Metric 8 in Table 2 (“Change in probability of a 10, 25 or 50% decline", also known as "Counterfactual of the probability of population decline", and linked to Metric 7 in Table 2/PVA E in this report), a metric that has been used frequently in assessments, often in association with PVA E. However, a visual examination of the figures presenting PVA metric E shows in almost all cases, a clear divergence between the lines across the range of values of mis-specification, and this change in the difference between values across effect sizes represents sensitivity to mis-specification of demographic rates in the excess probability referred to here. Good examples where this is clear are Figure 4e (all four panels) and Figure 4f (all four panels). It is not clear in all cases – see for example Figure 4g (top left panel). However, overall we can conclude that this counterfactual is comparatively more sensitive to mis-specification than ratio metrics.

Finally, the metric representing the centile from the un-impacted population size equal to the 50th centile of the impacted population size at the end of the wind farm (PVA F) showed moderately low sensitivity to mis-specification of survival and productivity. It performed considerably better than the other probabilistic metric (PVA E - probability of a population decline) with markedly lower sensitivity to mis-specification, population status and renewables effect size. However, it was more sensitive than ratio metrics, and in some cases showed unstable sensitivity which was less apparent in PVA metrics A and B (see Figures 5 a, b and h; Figures 6 a, b and h).

We recommend that those undertaking assessments consider the relative performance of different metrics with respect to sensitivity to mis-specification of input parameters. To summarise, of the two ratio and two probabilistic metrics considered here, the order with respect to sensitivity to mis-specification of input parameters was PVA A; PVA B; PVA F; PVA E. PVA E was much more sensitive than the other three and is not recommended for use in this context. If the first three are used in assessments in future, we recommend that interpretation should factor in their relative sensitivities. We also recommend that PVA metrics (C and D) are used since they are estimable when ratios are being calculated.

Note that we do not make recommendations on appropriate thresholds in relation to the above metrics, which is a societal choice and a matter for regulators.

5.3 Recommendations on PVA Analysis in Assessments of Renewables on Seabirds

We believe that Population Viability Analysis is a robust framework for forecasting future population change of seabirds under baseline conditions and under conditions of varying perturbations on demographic rates caused by renewable developments.

Furthermore, we believe that Bayesian state-space models have considerable potential in Population Viability Analysis using real data. Forecasts are made straightforward by the adoption of this approach, since posterior distributions are naturally generated. Furthermore, these methods do not suffer from the same criticism aimed at traditional methods that confidence intervals are unrealistically narrow. In addition, the study region has some of the most comprehensive demographic data available on seabirds in the UK, collected by CEH at their long term field site on the Isle of May, which has proved extremely valuable in carrying out this work. However, the restricted availability of high quality data left us with no alternative but to use these data on other populations where no such data exist. Despite this, the models of these other populations generally performed well. Exceptions were where population counts were sparse and variable, a particular issue at the Buchan Ness to Collieston Coast SPA.

5.4 Future Research and Monitoring Priorities

A fruitful avenue for future research would be extension to more complex models that incorporate environmental covariates or density dependence. Although there remains a lack of empirical evidence linking environmental covariates and seabird demography (Daunt *et al.* 2017), examples do exist (e.g. Frederiksen *et al.* 2004) and could form the rationale for future modelling including covariates. Evidence for density dependence in UK seabird populations is emerging (Horswill *et al.* 2016) and could be included where there is strong evidence for its occurrence including, crucially, whether the form of density dependence is compensatory or depensatory.

It would also be beneficial to estimate PVA metric sensitivity across a broader range of real world examples, comprising more species with differing life histories than we could consider here. This approach would enable a more comprehensive assessment of ratio and probabilistic metrics. Furthermore, it would be useful to test PVA F using a simulation modelling approach (Cook & Robinson 2016b, 2017) to establish whether a similar sensitivity to mis-specification of input parameters was apparent using that method. Another future priority would be to test sensitivity of different metrics using different population modelling methods: in addition to

Bayesian state-space models, other methods that may be more suited to sparse data could be incorporated, such as age-structured population growth models.

It is encouraging to note the value of plot counts, since these can be maintained on an annual or near- annual basis much more readily than full colony counts. However, we would recommend that full counts continue to be undertaken regularly to ensure that plots continue to be representative. Local data on survival and productivity add significantly to the ability to model populations effectively. However, our study demonstrates that PVA metrics, and their sensitivity to mis-specification, can be estimated where data are absent from the focal colony but available from an alternative, ideally nearby colony, thereby offering a natural, informative model prior. However, considerable thought is required before adopting this approach since information from another colony cannot automatically be assumed to apply elsewhere to other species and/or regions.

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Appendix 1

Input Parameters to the Bayesian State Space Models

This appendix details the input values for the population models.

Input parameters for adult survival and productivity are provided at two scales. In the Bayesian models, they are on the logit or the log scale (Table A1.1). However, these can be somewhat difficult to understand, so we have back transformed those that are on the log scale (productivity for kittiwakes and shags), using the mean and variance on the log scale to estimate the mean and variance of the untransformed productivity, which is log-normally distributed; these estimates can be verified with simulations. The two approaches matched. We, therefore, ran simulations for the parameters on the logit scale and estimate the mean and variance for the remaining untransformed survival and productivity parameters (Table A1.2).

Population counts are provided for all populations that were successfully modelled in this project in Tables A1.3 (kittiwakes), A1.4a and A1.4b (guillemots), A1.5a and A1.5b (razorbills) and A1.6 (shags).

Table A1.1

Input parameters into the Bayesian state space models for kittiwakes, guillemots, razorbills and shags at Forth Island, St Abbs, Buchan Ness and Fowlsheugh SPAs. Note that adult survival is on the logit scale and productivity is on the log scale for kittiwakes and shags, and on the logit scale for guillemots and razorbills (see Table A1.2 for values on the untransformed scale).

Species	**SPA**	**Adult survival: mean (sd)**	**Productivity: mean (sd)**
Kittiwake	Forth Islands	1.875 (0.546)	-0.790 (0.898)
	St Abb's Head	1.875 (0.546)	-0.615 (0.679)
	Fowlsheugh	1.875 (0.546)	-0.313 (0.492)
	Buchan Ness to Collieston Coast	1.875 (0.546)	-0.678 (0.699)
Guillemot	Forth Islands	2.705 (0.634)	1.041 (0.583)
	St Abb's Head	2.705 (0.634)	1.041 (0.583)
	Fowlsheugh	2.705 (0.634)	1.041 (0.583)
	Buchan Ness to Collieston Coast	2.705 (0.634)	1.041 (0.583)
Razorbill	Forth Islands	2.494 (0.685)	0.552 (0.350)
	St Abb's Head	2.494 (0.685)	0.552 (0.350)
	Fowlsheugh	2.494 (0.685)	0.552 (0.350)
Shag	Forth Islands	2.147 (1.215)	-0.052 (0.637)
	St Abb's Head	2.147 (1.215)	0.170 (0.590)

Table A1.2

Input parameters into the Bayesian state space models for kittiwakes, guillemots, razorbills and shags at Forth Island, St Abbs, Buchan Ness and Fowlsheugh SPAs. Note that adult survival and productivity are on the untransformed scale.

Species	**SPA**	**Adult survival: mean (sd)**	**Productivity: mean (sd)**
Kittiwake	Forth Islands	0.855 (0.067)	0.679 (0.755)
	St Abb’s Head	0.855 (0.067)	0.681 (0.521)
	Fowlsheugh	0.855 (0.067)	0.825 (0.432)
	Buchan Ness to Collieston Coast	0.855 (0.067)	0.648 (0.515)
Guillemot	Forth Islands	0.927 (0.045)	0.725 (0.111)
	St Abb’s Head	0.927 (0.045)	0.725 (0.111)
	Fowlsheugh	0.927 (0.045)	0.725 (0.111)
	Buchan Ness to Collieston Coast	0.927 (0.045)	0.725 (0.111)
Razorbill	Forth Islands	0.910 (0.058)	0.631 (0.080)
	St Abb’s Head	0.910 (0.058)	0.631 (0.080)
	Fowlsheugh	0.910 (0.058)	0.631 (0.080)
Shag	Forth Islands	0.847 (0.145)	1.163 (0.823)
	St Abb’s Head	0.847 (0.145)	1.410 (0.909)

Table A1.3

Kittiwake breeding population sizes used in population models for each SPA. Values represent number of breeding pairs.

SPA	**Forth Islands**	**Forth Islands**	**Forth Islands**	**Forth Islands**	**Forth Islands**	**St Abbs to Fast Castle SPA**	**Fowlsheugh SPA**	**Buchan Ness to Collieston Coast SPA**
Site	**Bass Rock**	**Craigleith**	**Fidra**	**Isle of May**	**The Lamb**	**St Abb's Head NNR**	**Fowlsheugh**	**Boddam to Collieston**
1981				6115				
1982								
1983								
1984				6012				
1985				5510				
1986		725	532	4801	167	13940	22051	19498
1987	2400		726	6765	214	15182		
1988		770	610	7638	175	16200		
1989		840	705	7564	250	19066		
1990		850	598	8129	187	17642		
1991			494	6535	106	16183	23522	
1992			489	6916	223	16524	34872	
1993		1028	452	7009	84	15268		
1994		564	330	3751	160	13007		
1995		951	435	7603	210	13670		24957
1996	2142	509	314	6269	143	13437		
1997	3044	714	298	6518	119	13393		
1998			243	4306		8044		
1999	1307	511	225	4196	115	9576	18800	
2000	1000	539	343	4618	132	11077		
2001	670	440	243	3639	117	8028		14091
2002	774	383	315	3666	139	8890		
2003	910	450	273	3335	124	6642		
2004	660	501	217	3876	126	6239		13330
2005	563	492	257	3790	94	7239		
2006	505	444	275	3167	202	6288	11140	
2007	377	508	244	3424	96	6463		12542
2008	323	513	222	3354	110	5298		
2009	425	594	237	2316	82	4616	9454	
2010	440	600	232	3422	133	4744		
2011	313	542	204	2685	140	4688		
2012	395	620	191	2465	95	4314	9388	
2013	270	293	128	1712	47	3403		
2014	324	300	167	2464	84	3625		
2015	441	537	275	3433	99	4209	9655	
2016	325	468	259	2912	101	2779		

Table A1.4a

Guillemot breeding population sizes used in population models for Forth Islands SPA. Values represent number of breeding pairs. Counts of individuals were converted to pairs using k-values from the Isle of May (Harris *et al.* 2015a, updated). Count type WCC = whole colony count.

SPA	**Forth Islands SPA**	**Forth Islands SPA**	**Forth Islands SPA**	**Forth Islands SPA**	**Forth Islands SPA**
Site	**Bass Rock**	**Craigleith**	**Fidra**	**Isle of May**	**The Lamb**
Count type	**WCC**	**WCC**	**WCC**	**WCC**	**WCC**
1981				11250	
1982					
1983				14750	
1984				13000	
1985				13000	
1986		1404	126	13700	1967
1987	1797		53	11680	572
1988		969	88	11223	1604
1989		1181	101	12736	2502
1990		1167	67	12632	1807
1991			134	11440	1631
1992			161	11511	2136
1993		981	143	12418	2287
1994		1400	219	13843	2309
1995		1263	172	15326	1887
1996	1911	1112	153	14500	2163
1997	2682	507	173	17340	2829
1998			207	17384	2063
1999	1890	1333	293	16933	2935
2000	2373	1913	427	17979	1677
2001	2395	2087	448	18442	1431
2002	2452	1291	506	20185	820
2003	2057	1546	434	19519	1449
2004	1966	1549	492	20332	1517
2005	1547	1208	583	18858	1313
2006	2346	1215	333	15578	1268
2007	1030	1058	541	15536	1283
2008	1402	1347	353	15036	2541
2009	2136	1512	439	14143	1842
2010	1329	919	429	15029	1806
2011	1906	1625	316	14955	1944
2012	1328	1371		14100	
2013	1546	1347	372	13349	2224
2014	1759	2498	550	14248	2403
2015	2385	2254	467	15945	2289
2016	1562	1798	325	16132	2150

Table A1.4b

Guillemot breeding population sizes used in population models for St Abbs Head to Fast Castle SPA, Fowsheugh SPA and Buchan Ness to Collieston Coast SPA. Values represent number of breeding pairs. Counts of individuals were converted to pairs using k-values from the Isle of May (Harris *et al.* 2015a, updated). Count type WCC = whole colony count; PC = mean of plot means.

SPA	**St Abbs to Fast Castle SPA**	**St Abbs to Fast Castle SPA**	**Fowlsheugh SPA**	**Fowlsheugh SPA**	**Buchan Ness to Collieston Coast SPA**	**Buchan Ness to Collieston Coast SPA**
Site	**St Abb's Head NNR**	**St Abb's Head NNR**	**Fowlsheugh**	**Fowlsheugh**	**Boddam to Collieston**	**Boddam to Collieston**
Count type	**WCC**	**PC**	**WCC**	**PC**	**WCC**	**PC**
1981						
1982						
1983						
1984		142		198		
1985		119		209		
1986	16443	157	37453	173	9225	
1987	17775	156		208		
1988	18667	143		194		
1989	21394	165		232		
1990	21790	172		206		
1991		174				
1992		167	39381	240		126
1993	20036	180		217		
1994		190		216		
1995		199		237	16602	137
1996		177		244		
1997		240		244		
1998	26254	219		234		148
1999		232	48651	295		
2000		272		234		
2001		248		253	19286	185
2002		318				
2003	29502	264		296		
2004		255		300		202
2005		283		243		
2006		238	39370	216		
2007		270		225	17876	153
2008	29079	252		305		
2009		304	42339	244		
2010		229		240		163
2011		299		285		
2012		232	37277	233		
2013	29828	253		221		158
2014		265		199		
2015		223	40979	236		
2016		236		201		194

Table A1.5a

Razorbill breeding population sizes used in population models for Forth Islands SPA. Values represent number of breeding pairs. Counts of individuals were converted to pairs using k-values from the Isle of May (Harris *et al.* 2015b, updated). Count type WCC = whole colony count. Unrealistic k-values were recorded in 2005 so population counts were excluded.

SPA	**Forth Islands SPA**	**Forth Islands SPA**	**Forth Islands SPA**	**Forth Islands SPA**	**Forth Islands SPA**
Site	**Bass Rock**	**Craigleith**	**Fidra**	**Isle of May**	**The Lamb**
count type	**WCC**	**WCC**	**WCC**	**WCC**	**WCC**
1988		79	120	1903	26
1989		74	91	2075	33
1990		38	48	1508	21
1991		70	79	1425	28
1992		34	53	1909	30
1993		41	44	2052	9
1994		56	62	2227	26
1995		79	59	3108	34
1996	165	64	65	2989	64
1997	138	66	81	2719	19
1998			86	3126	
1999	71	114	147	3429	92
2000	65	157	86	3105	68
2001	111	111	72	3346	78
2002	180	131	111	2844	90
2003	64	117	63	2233	81
2004	128	138	82	2677	85
2005					
2006	169	175	123	2975	62
2007	119	181	128	2735	77
2008	85	147	95	2591	80
2009	70	117	127	2400	70
2010	63	136	123	2557	42
2011	94	185	108	2705	70
2012	106	157	70	3068	66
2013	105	129	109	2879	59
2014	124	110	170	2987	65
2015	144	193	139	3202	46
2016	91	186	122	3570	82

Table A1.5b

Razorbill breeding population sizes used in population models for St Abbs Head to Fast Castle SPA and Fowsheugh SPA. Values represent number of breeding pairs. Counts of individuals were converted to pairs using k-values from the Isle of May (Harris *et al.* 2015b, updated). Count type WCC = whole colony count; PC = mean of plot means.

SPA	**St Abbs to Fast Castle SPA**	**St Abbs to Fast Castle SPA**	**Fowlsheugh SPA**	**Fowlsheugh SPA**
Site	**St Abb's Head**	**St Abb's Head NNR**	**Fowlsheugh**	**Fowlsheugh**
count type	**WCC**	**PC**	**WCC**	**PC**
1988	1343	21		
1989	1398	23		
1990	1072	18		
1991		29		
1992		24	6827	
1993	1187	21		
1994		25		
1995		29		
1996		23		
1997		33		
1998	1793	29		
1999		28	5808	
2000		30		
2001		26		
2002		32		
2003	1595	20		
2004		15		9
2005		29		
2006		20	3341	20
2007		21		19
2008	1262	18		
2009		23	3696	18
2010		18		14
2011		24		
2012		23	4883	21
2013	1269	22		14
2014		20		18
2015		16	5180	22
2016		18		20

Table A1.6

Shag breeding population sizes used in population models for each SPA. Values represent number of breeding pairs.

SPA	Forth Islands SPA	Forth Islands SPA	Forth Islands SPA	Forth Islands SPA	Forth Islands SPA	Forth Islands SPA	St Abbs to Fast Castle SPA	Buchan Ness to Collieston Coast SPA
Site	**Bass Rock**	**Craig-leith**	**Fidra**	**Inch-mickery**	**Isle of May**	**The Lamb**	**St Abb's Head NNR**	**Boddam to Collieston**
1973		164	17		1076	244		
1974		225	27		933	255		
1975	180	214	25		644	233		
1976	213	201	20	8	497	210	187	
1977	201	186	18	12	921	156	193	
1978	202	208	23	14	769	143	134	
1979	188	215	25	14	966	160		
1980	191	198	25	11	1041	143		
1981	154	252	43	14	1163	220		
1982	194	344	59	22	1425		209	
1983	170	356	66	42	1567	283		
1984	193	379	64	22	1639	284		
1985	101	345	55	29	1524	303	268	
1986	75	388	67	24	1310	301	364	440
1987	162	465	64	24	1916		396	
1988	93	435	86	24	1290	250	318	
1989	111	544	124	29	1703	286	366	
1990	121	522	116	28	1386	290	338	
1991		646	242	33	1487	305	463	
1992		665	255	36	1634	318	450	
1993	20	155	88	28	715	65	300	
1994	13	106	73	10	403	36	115	
1995		171	84	20	503	81	173	223
1996	47	159	81	18	512	77	175	
1997	41	180	107	28	502	65	160	
1998			86	25	621		196	
1999	30	131	61	33	259	76	165	
2000	28	208	123	32	541	46	233	
2001	39	237	139	41	734	99	300	415
2002	25	233	186	52	676	102	296	
2003	24	197	254	70	968	124	365	
2004	46	324	272	78	687	111	369	594
2005	18	131	115	52	281	49	131	
2006	36	118	198	57	485	65	162	
2007	28	199	169	57	399	73	132	331
2008	22	133	146	55	427	97	131	
2009	15	200	159	54	465	75	138	
2010	16	207	204	55	492	114	157	
2011	25	281	191	62	540	66	160	
2012	11	258	172	71	648	77	171	
2013	31	117	153	59	322	44	94	363
2014	12	137	162	65	338	49	107	

Appendix 2

Ratio of Impacted to Un-Impacted 25 Year Population Growth Rate

One possibility for the low sensitivity of PVA metric A (median of the ratio of impacted to un-impacted annual growth rate) is the scale of values, with all values being close to one, and, therefore, sensitivity potentially appearing low in a visual assessment even in cases where it is not. However, here we consider a 25 year growth rate, where lines deviate markedly from 1 and sensitivity is more discernible. This analysis shows that low sensitivity is still apparent (Figure A2.1).

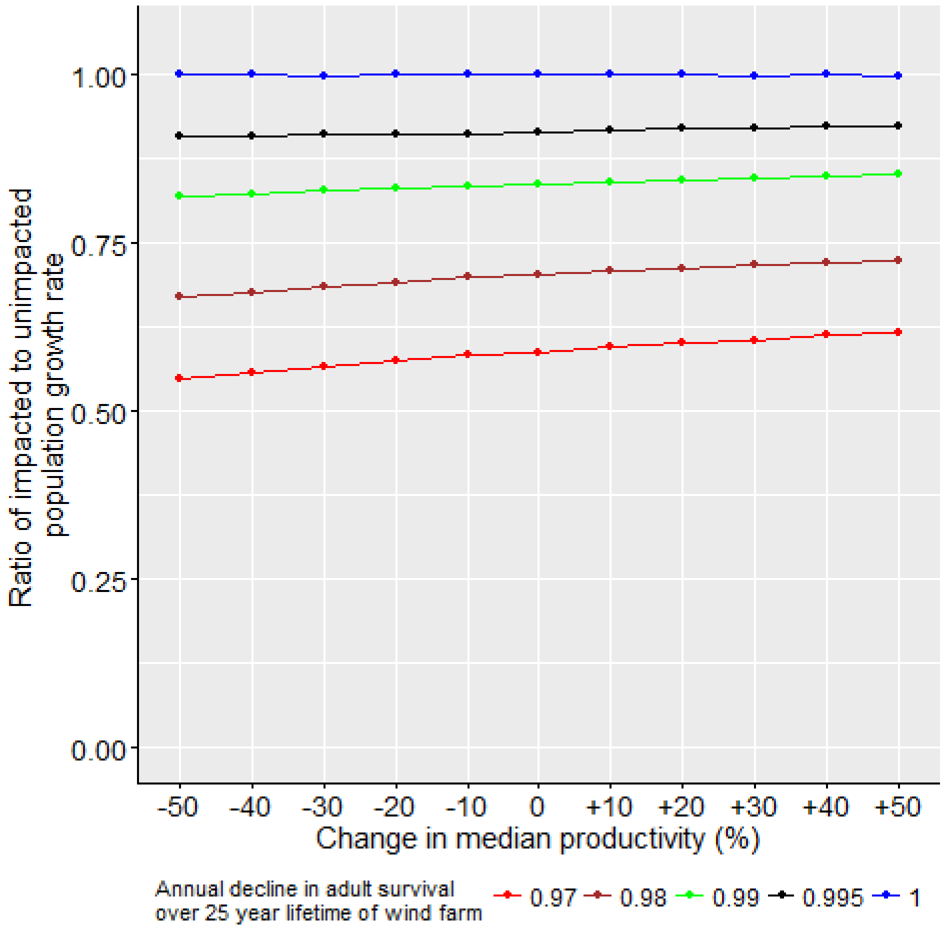


Figure A2.1: PVA Metric A – ratio of 25 year population growth rate, comparing impacted population vs. un-impacted population, showing productivity mis-specification varied from -50% to +50% (with 0% representing no mis-specification) in Forth Islands kittiwakes. The five coloured lines represent the different levels of potential impact on annual adult survival.

Appendix 3

PVA Metric Sensitivity for all Populations

This Appendix presents graphical output of PVA metric sensitivity for the 13 populations considered in this project. For each species, the sequence of figures is as presented in Figure 4 of the main report for Forth Islands kittiwakes. For completeness, we include Forth Islands kittiwakes here.

In all figures, adult mortality mis-specification is illustrated in the upper panels and productivity mis-specification in the lower panels. Mis-specification was varied from -30% to +30% (with 0% representing no mis-specification). The five coloured lines represent the different levels of potential impact on annual productivity (left panels) or annual adult survival (right panels) over the hypothetical 25 year lifetime of the wind farm (2017-2041).

1. **Kittiwakes at Forth Islands SPA**:

Figure A2.1a: PVA Metric A for Forth Kittiwakes – ratio of population growth rate from 2016-2041, comparing impacted population vs. un-impacted population.

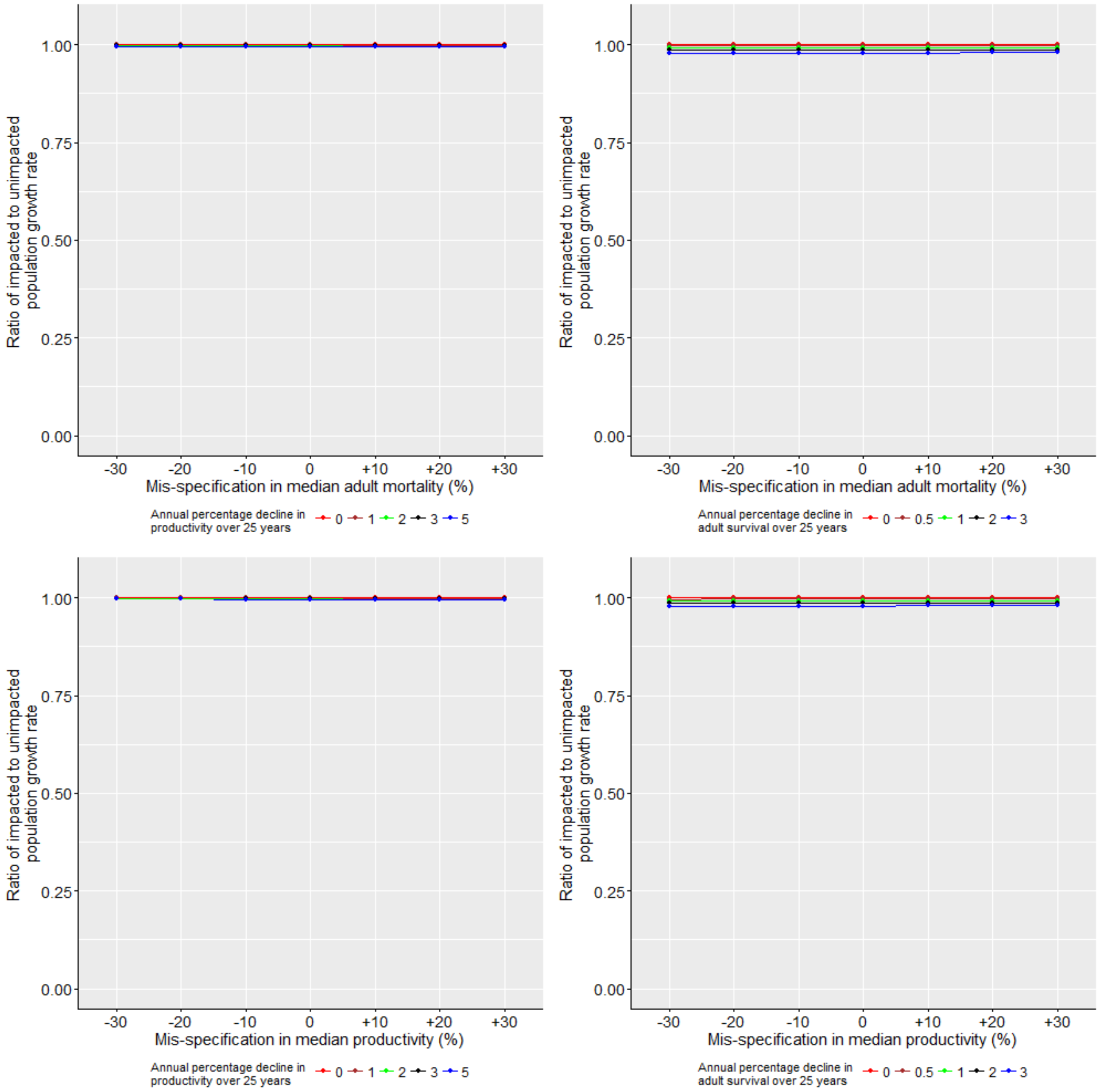


Figure A2.1b: PVA Metric B for Forth Kittiwakes – ratio of population size at 2041, comparing impacted population vs. un-impacted population.

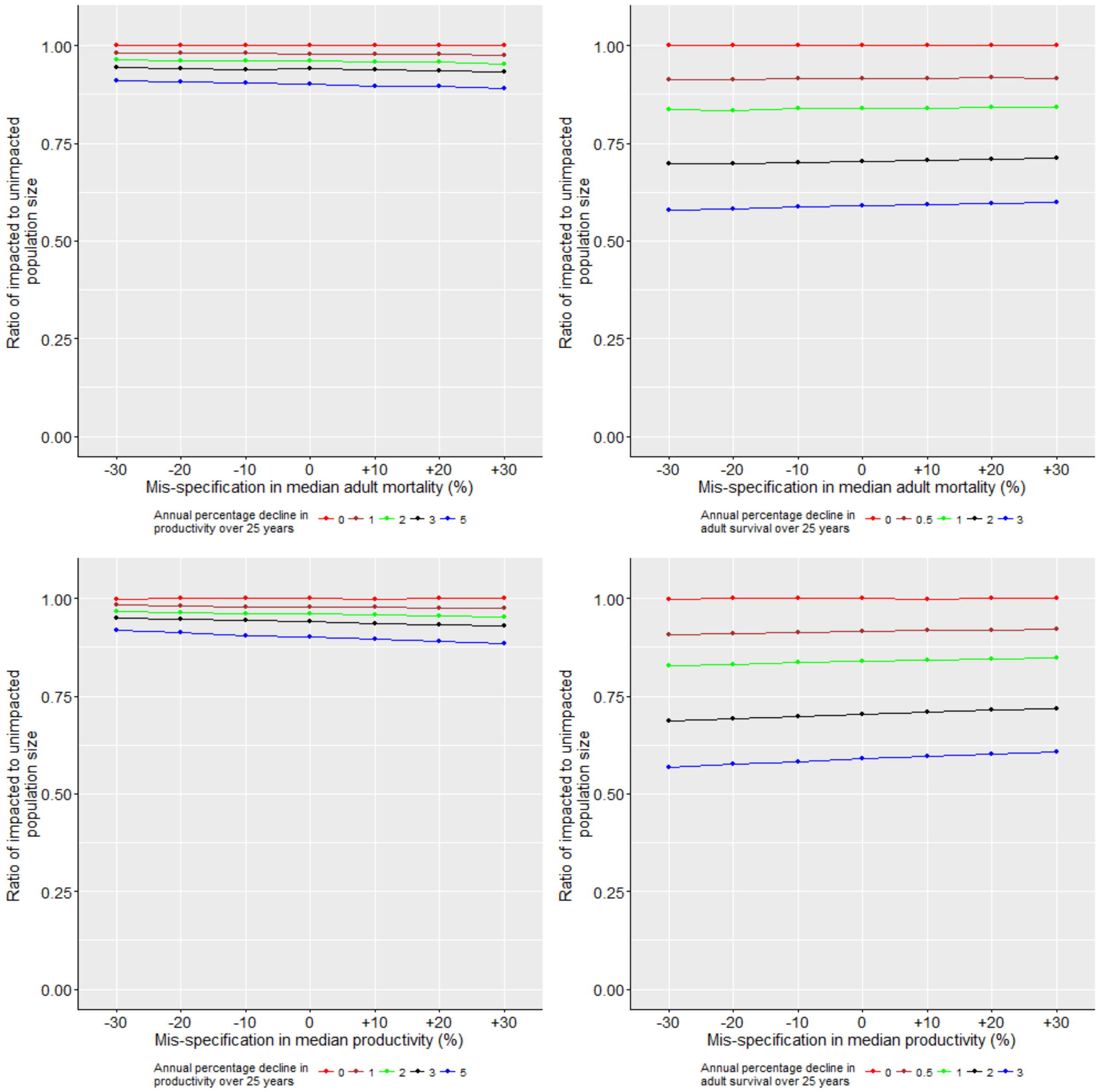


Figure A2.1c: PVA Metric C for Forth Kittiwakes – difference in population growth rate from 2016-2041, comparing impacted population vs. un-impacted population.

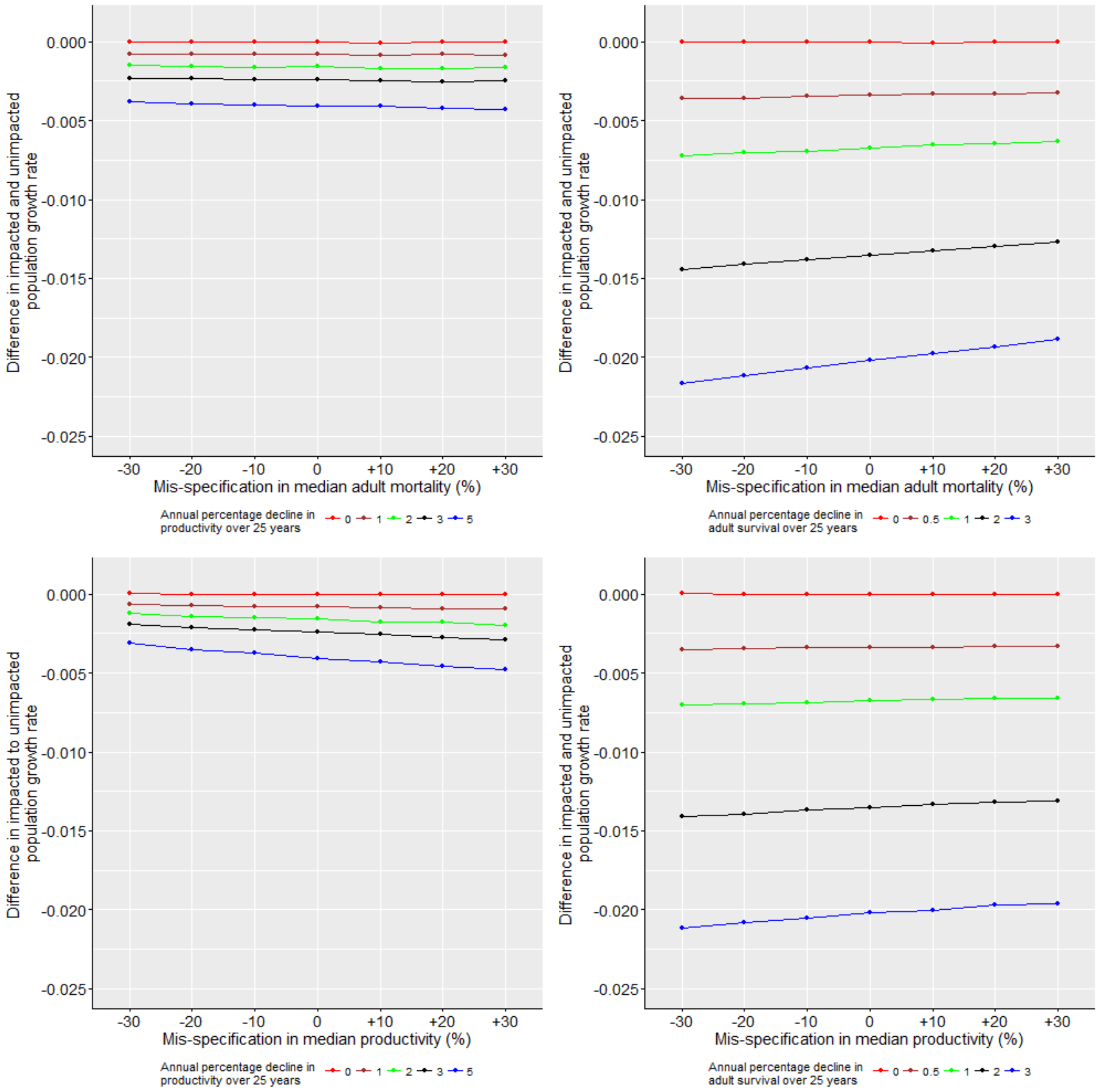


Figure A2.1d: PVA Metric D for Forth Kittiwakes – difference in population size at 2041, comparing impacted population vs. un-impacted population.

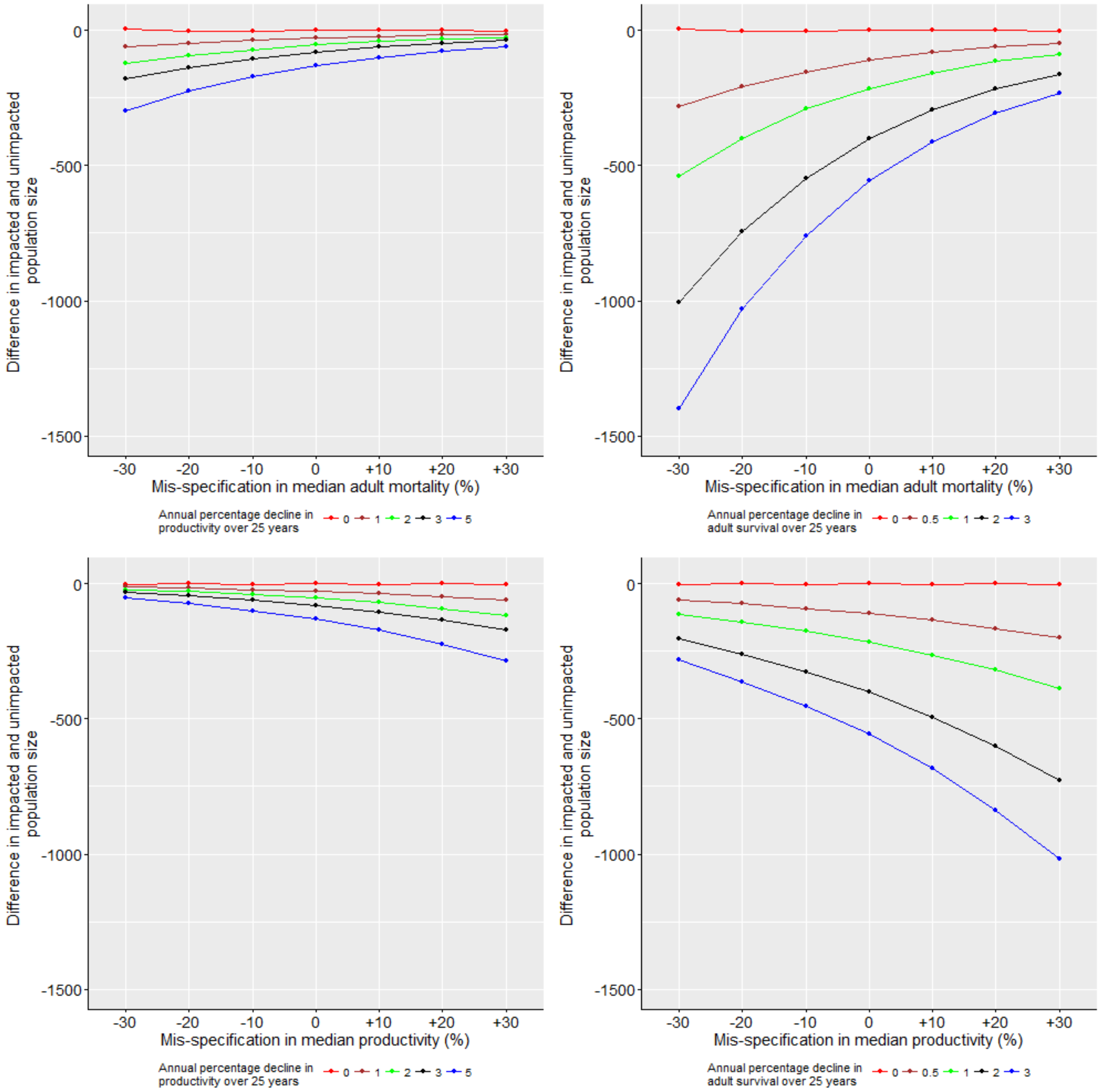


Figure A2.1e: PVA Metric E1 for Forth Kittiwakes – probability of population decline greater than 10% from 2016-2041.

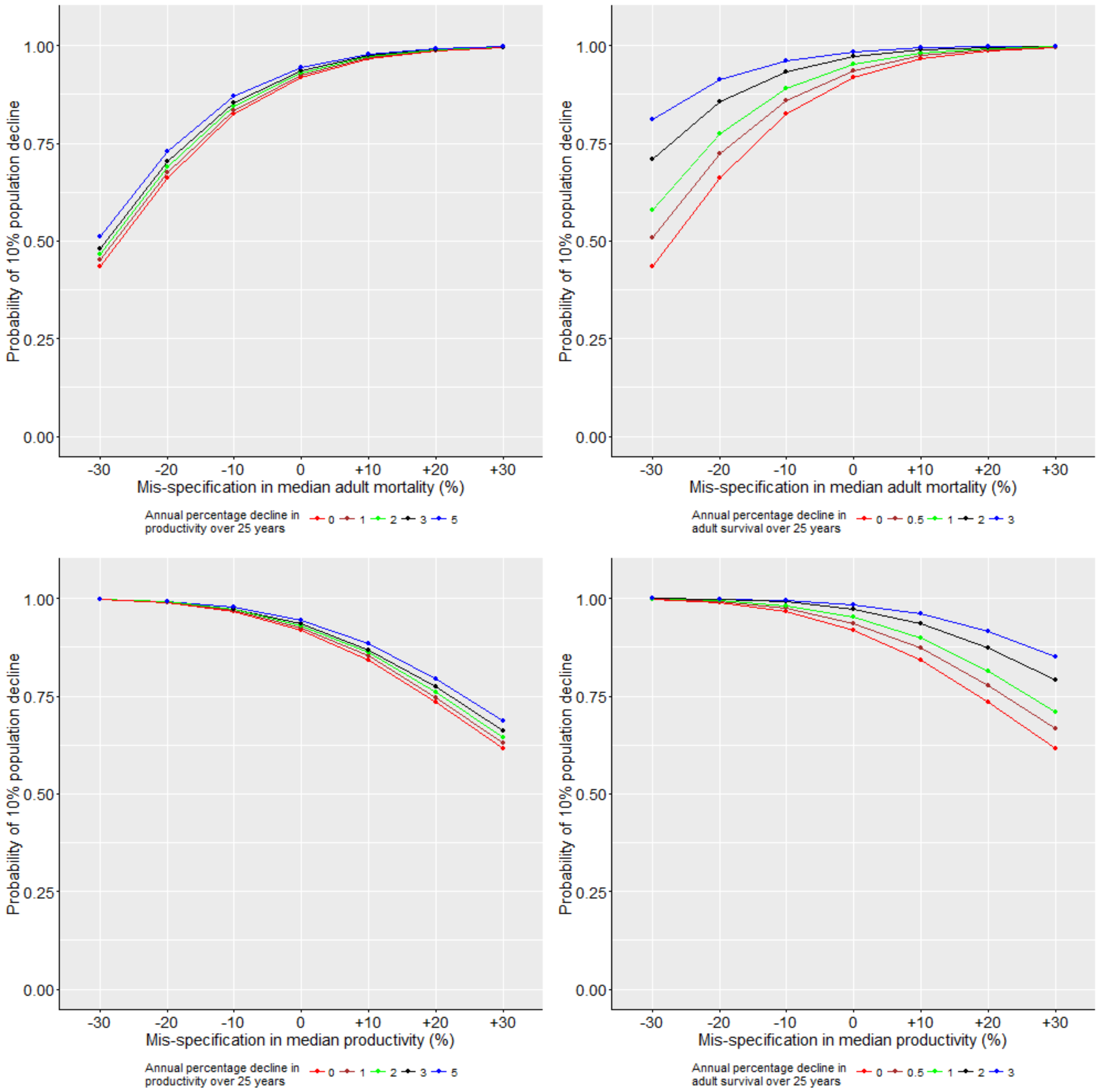


Figure A2.1f: PVA Metric E2 for Forth Kittiwakes – probability of population decline greater than 25% from 2016-2041.

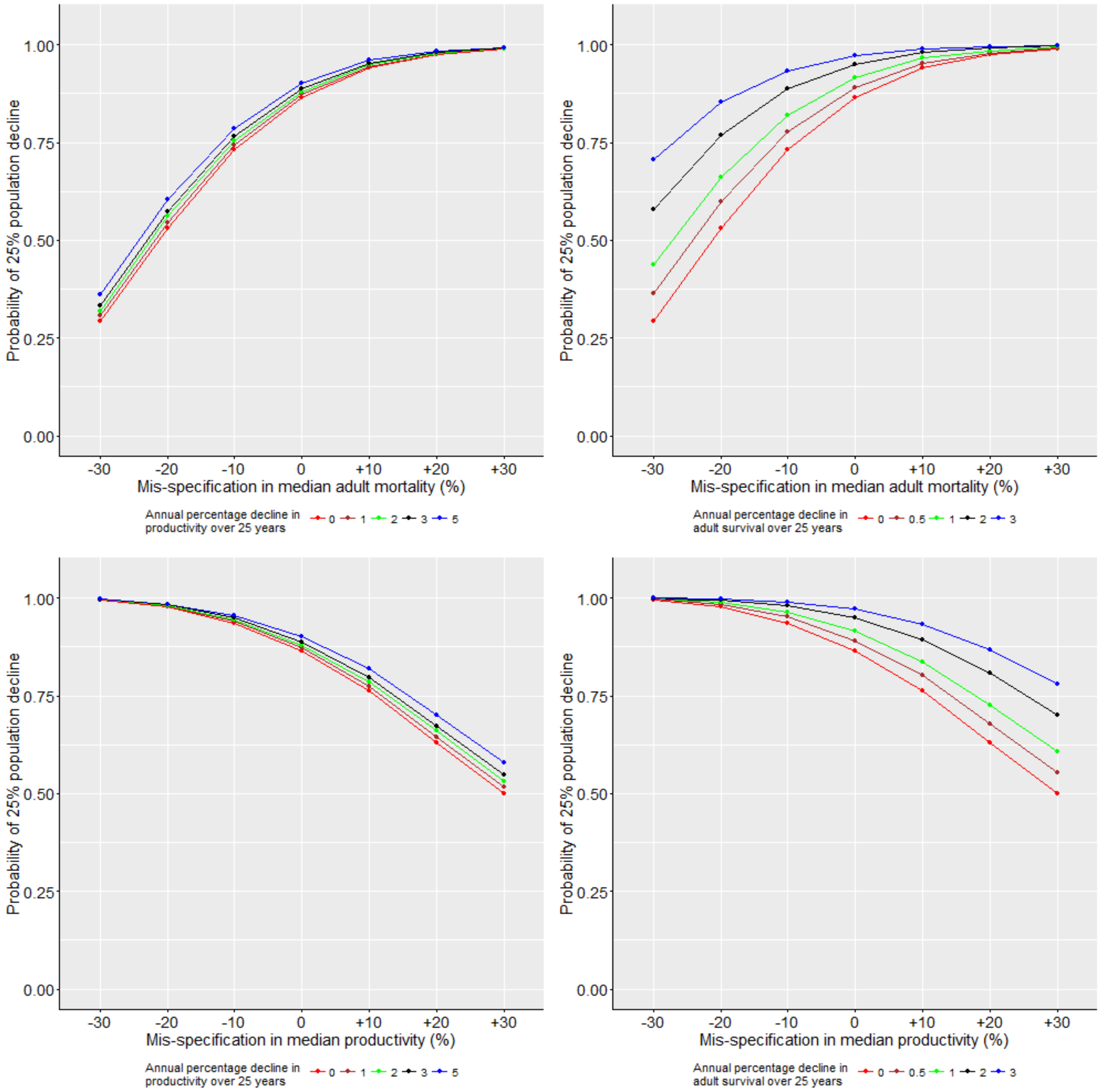


Figure A2.1g: PVA Metric E3 for Forth Kittiwakes – probability of population decline greater than 50% from 2016-2041.

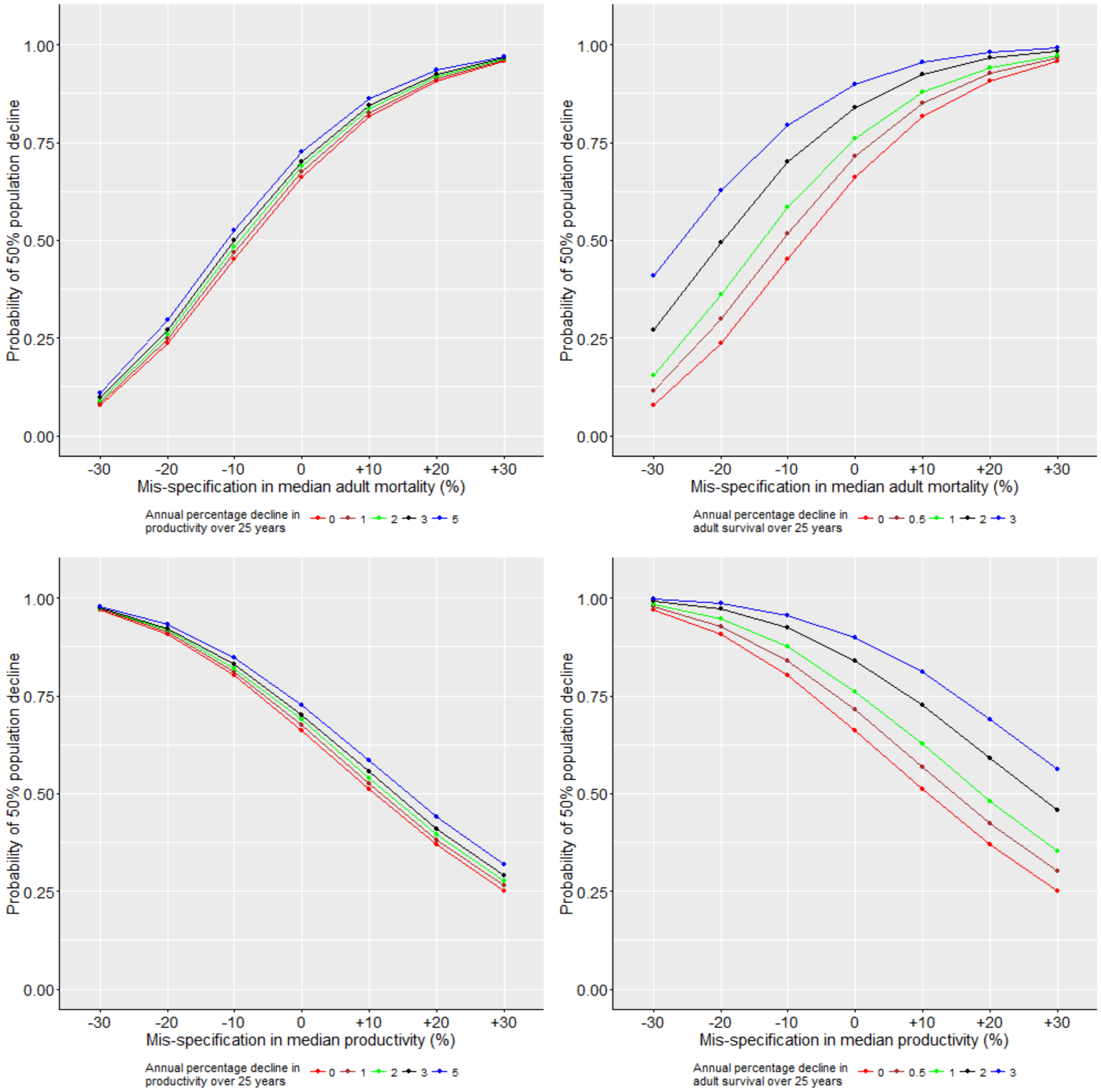
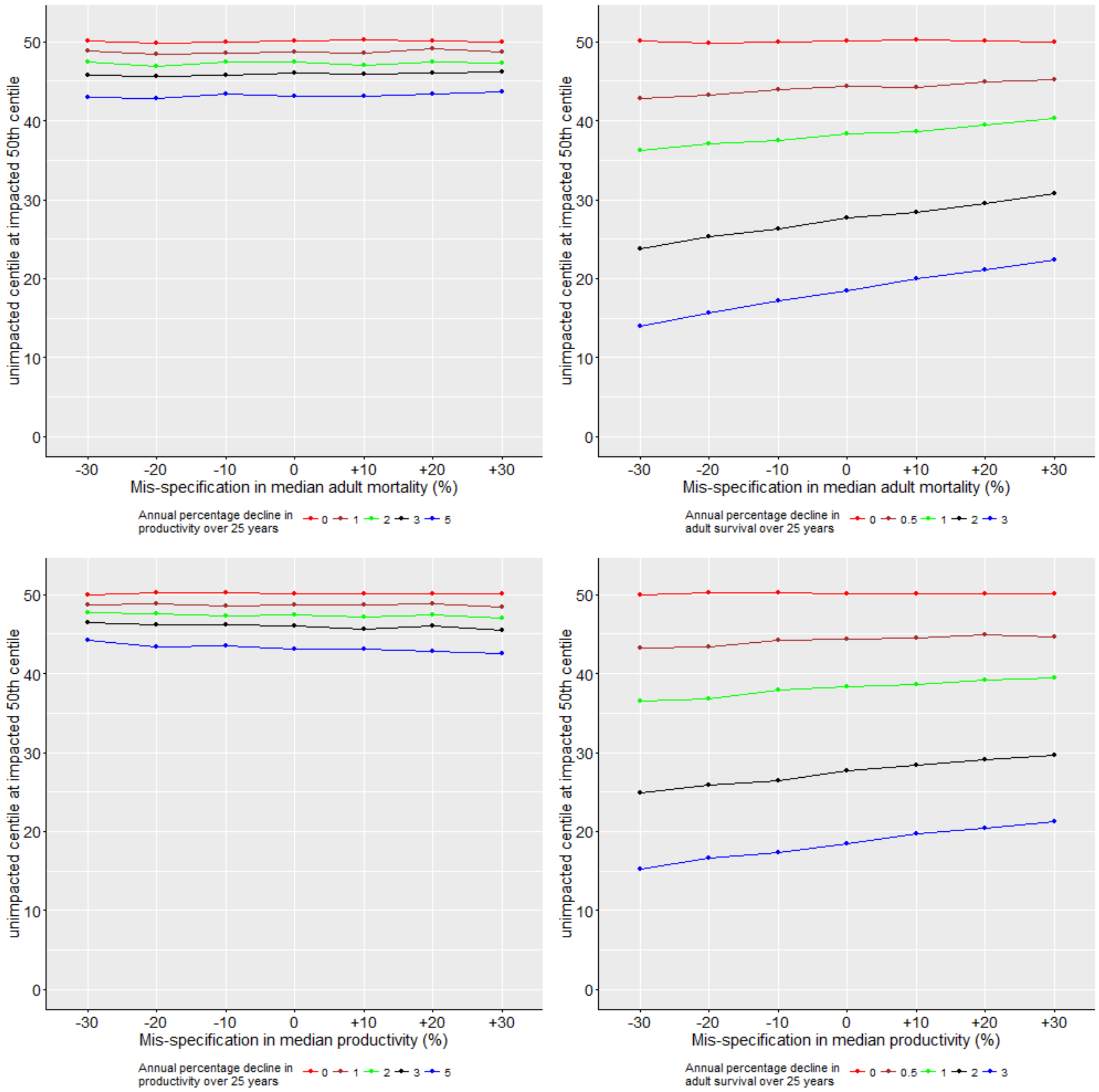


Figure A2.1h: PVA Metric F for Forth Kittiwakes – centile from un-impacted population size equal to the 50th centile of the impacted population size, at 2041.



2. **Kittiwakes at St Abb's Head SPA**:

Figure A2.2a. PVA Metric A for St Abb's Kittiwakes – ratio of population growth rate from 2016-2041, comparing impacted population vs. un-impacted population.

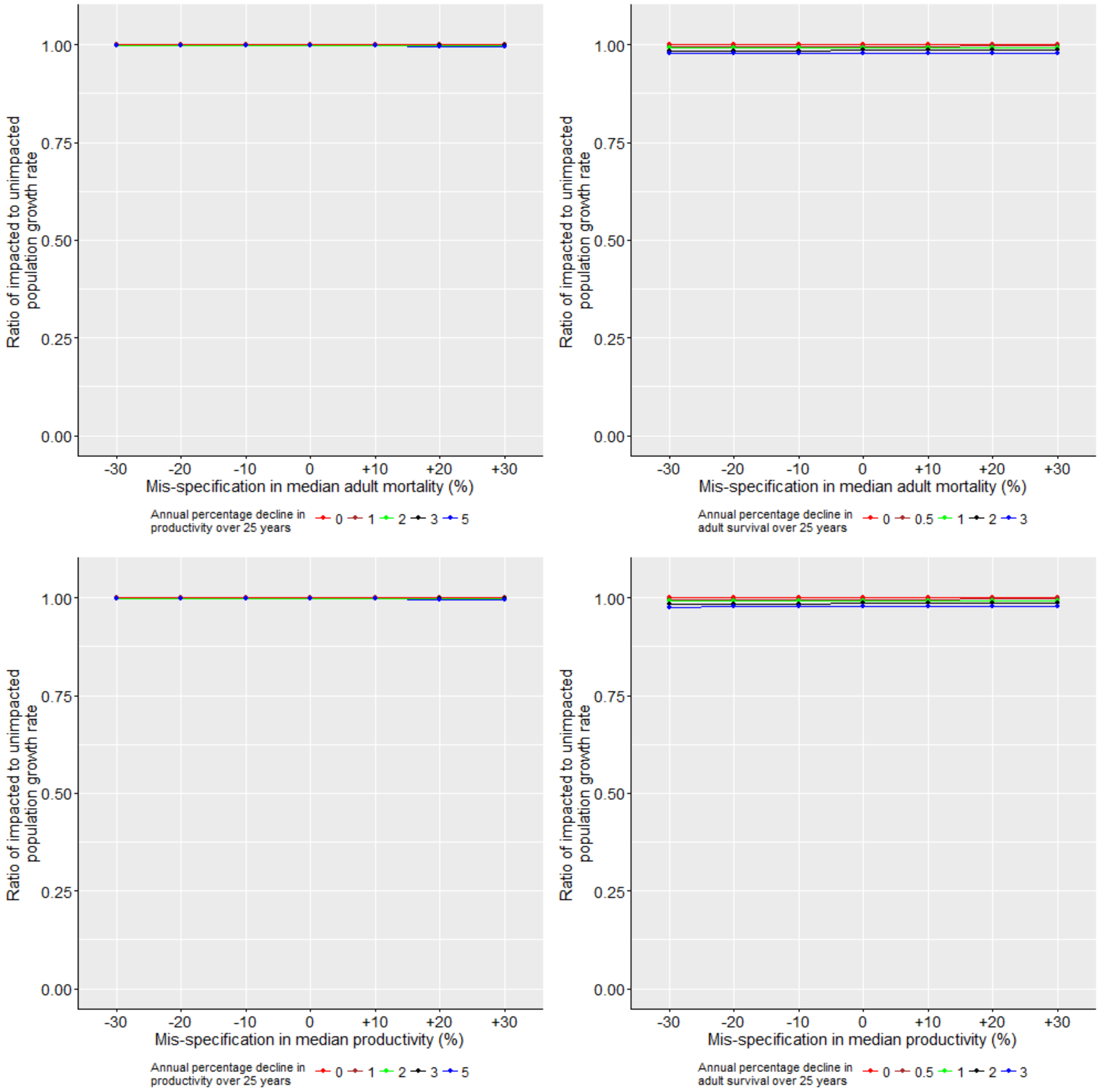


Figure A2.2b. PVA Metric B for St Abb’s Kittiwakes – ratio of population size at 2041, comparing impacted population vs. un-impacted population.

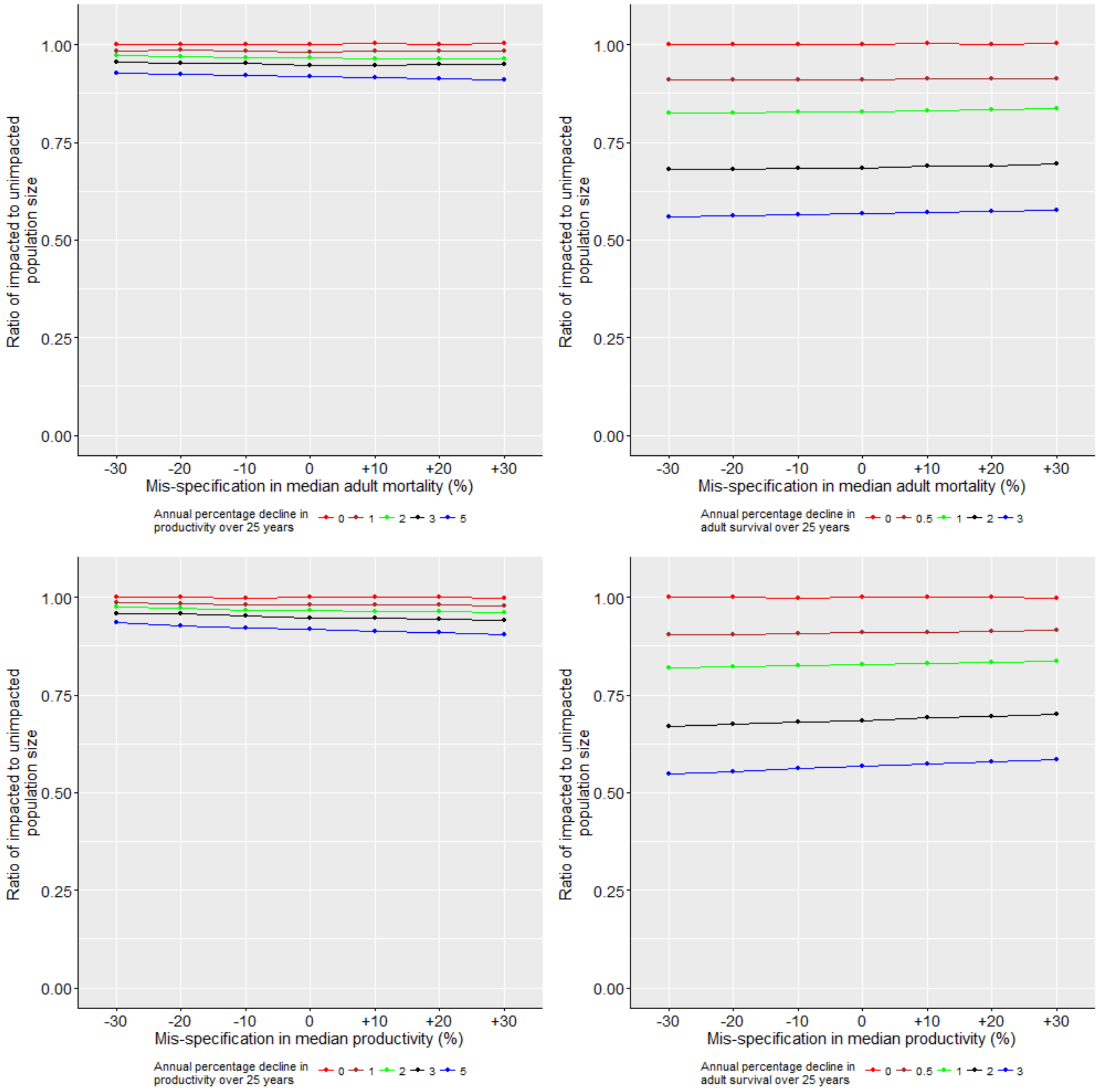


Figure A2.2c. PVA Metric C for St Abb’s Kittiwakes – difference in population growth rate from 2016-2041, comparing impacted population vs. un-impacted population.

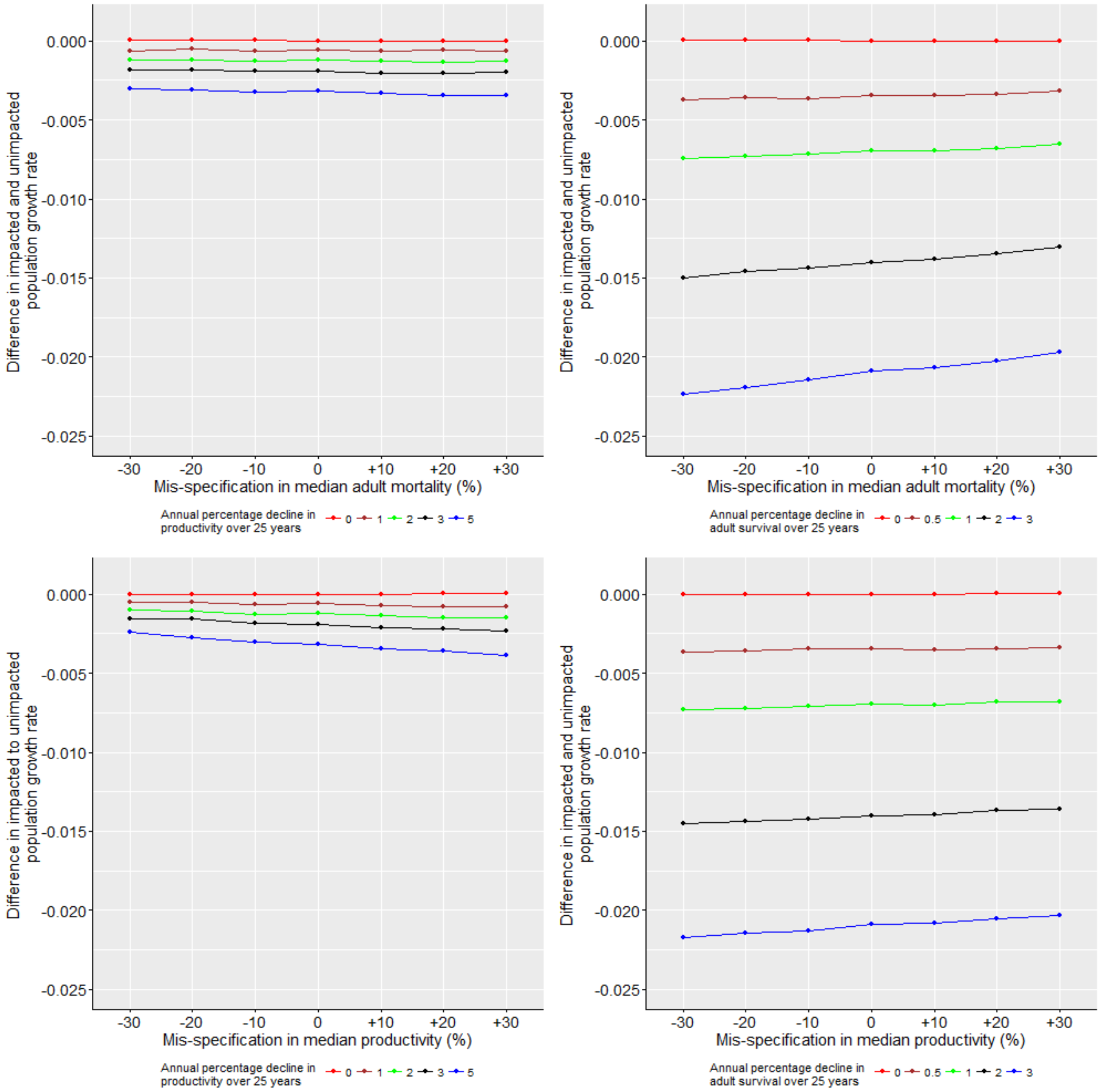


Figure A2.2d. PVA Metric D for St Abb's Kittiwakes – difference in population size at 2041, comparing impacted population vs. un-impacted population.

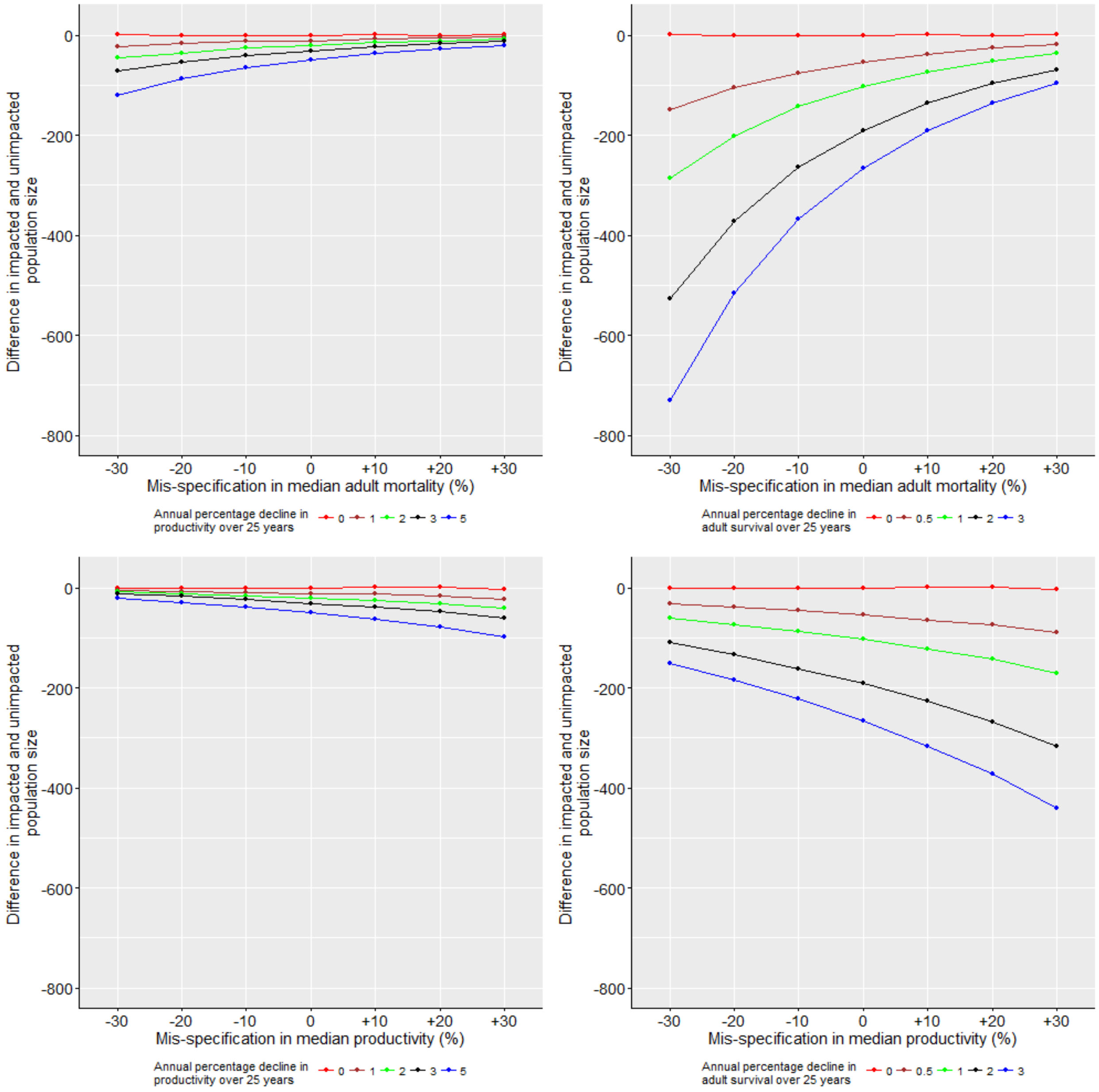


Figure A2.2e. PVA Metric E1 for St Abb’s Kittiwakes – probability of population decline greater than 10% from 2016-2041.

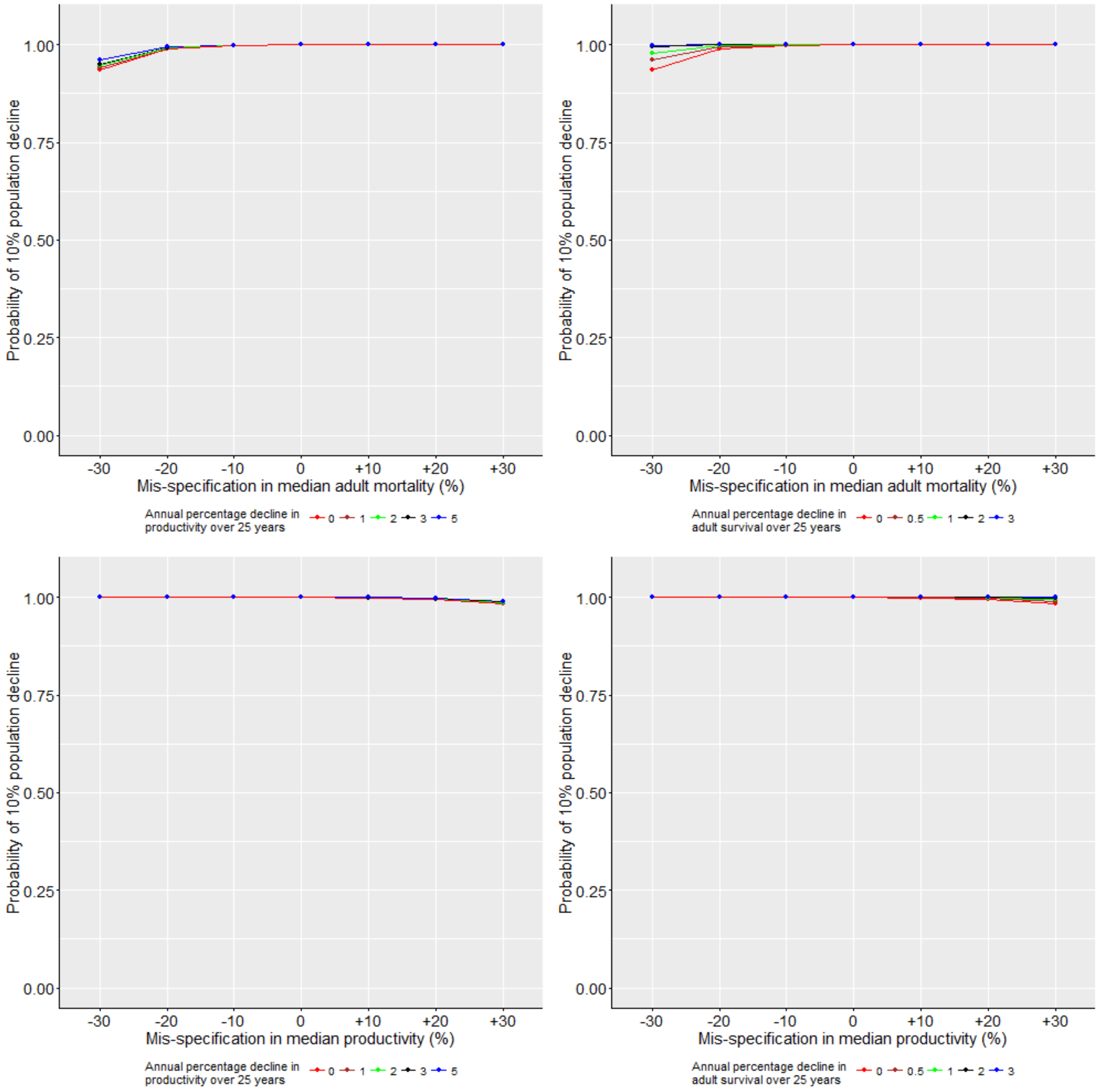


Figure A2.2f. PVA Metric E2 for St Abb’s Kittiwakes – probability of population decline greater than 25% from 2016-2041.

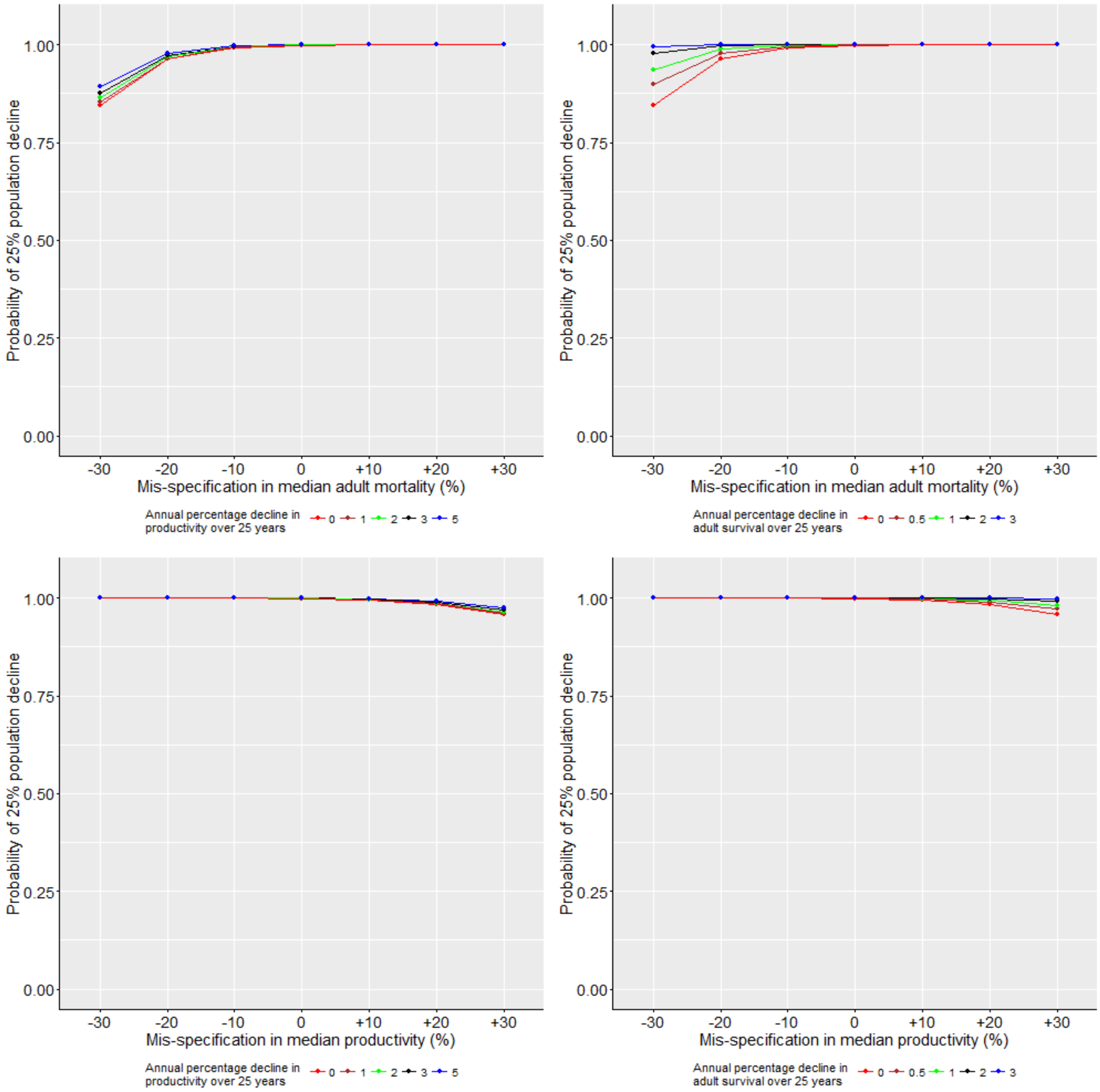


Figure A2.2g. PVA Metric E3 for St Abb's Kittiwakes – probability of population decline greater than 50% from 2016-2041.

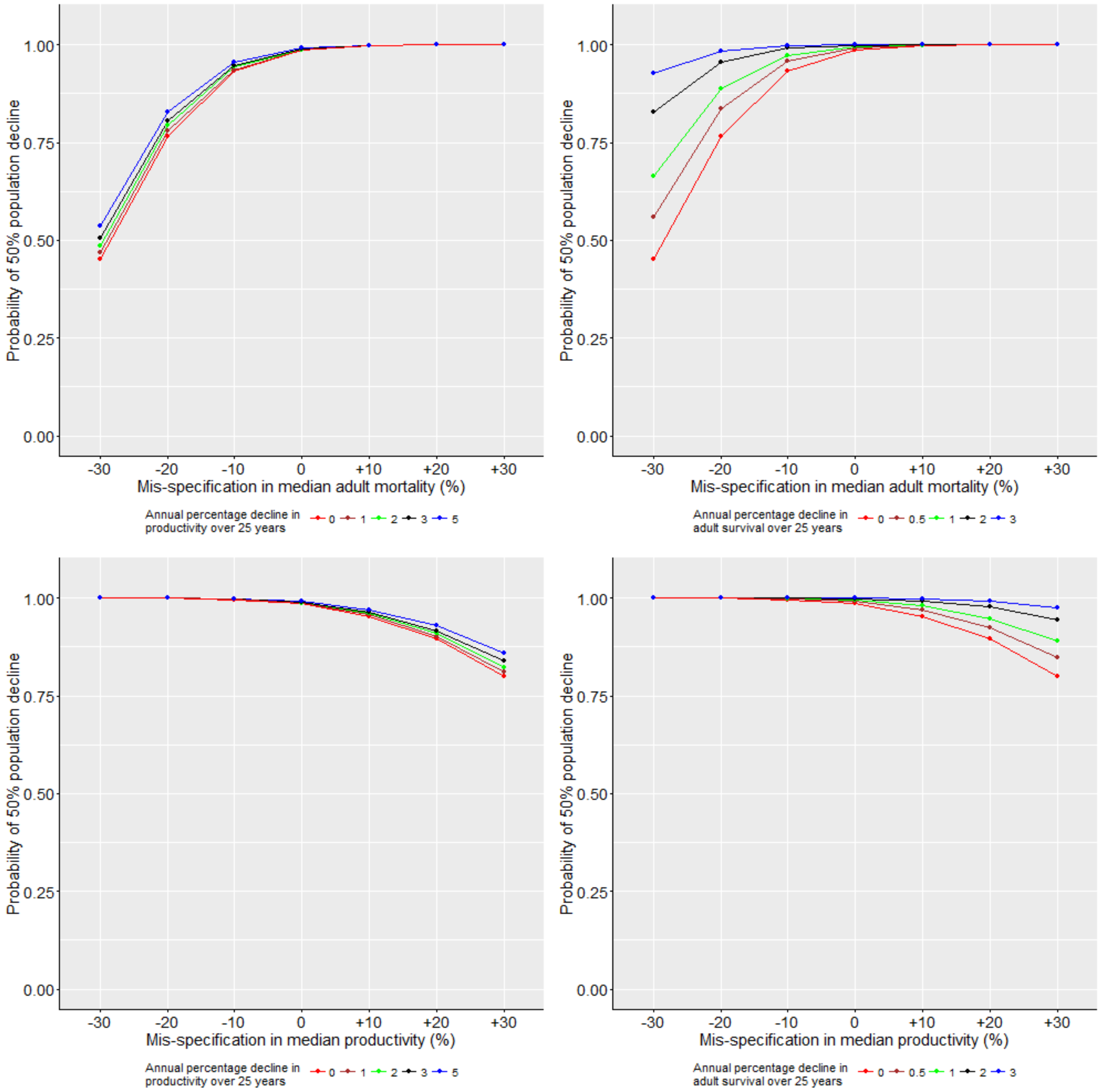
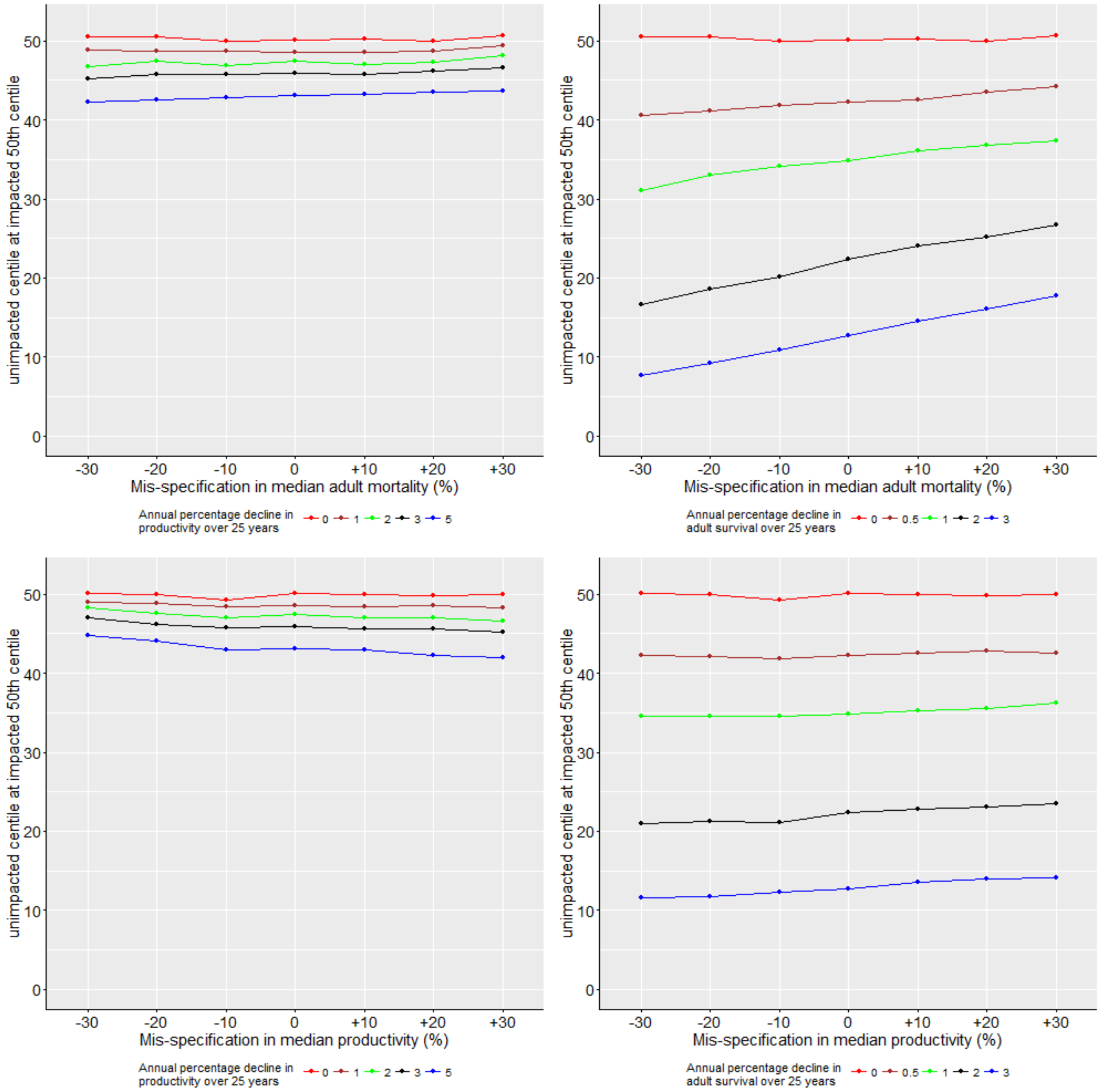


Figure A2.2h. PVA Metric F for St Abb's Kittiwakes – centile from un-impacted population size equal to the 50th centile of the impacted population size, at 2041.



3. **Kittiwakes at Fowlsheugh SPA**:

Figure A2.3a. PVA Metric A for Fowlsheugh Kittiwakes – ratio of population growth rate from 2016-2041, comparing impacted population vs. un-impacted population.

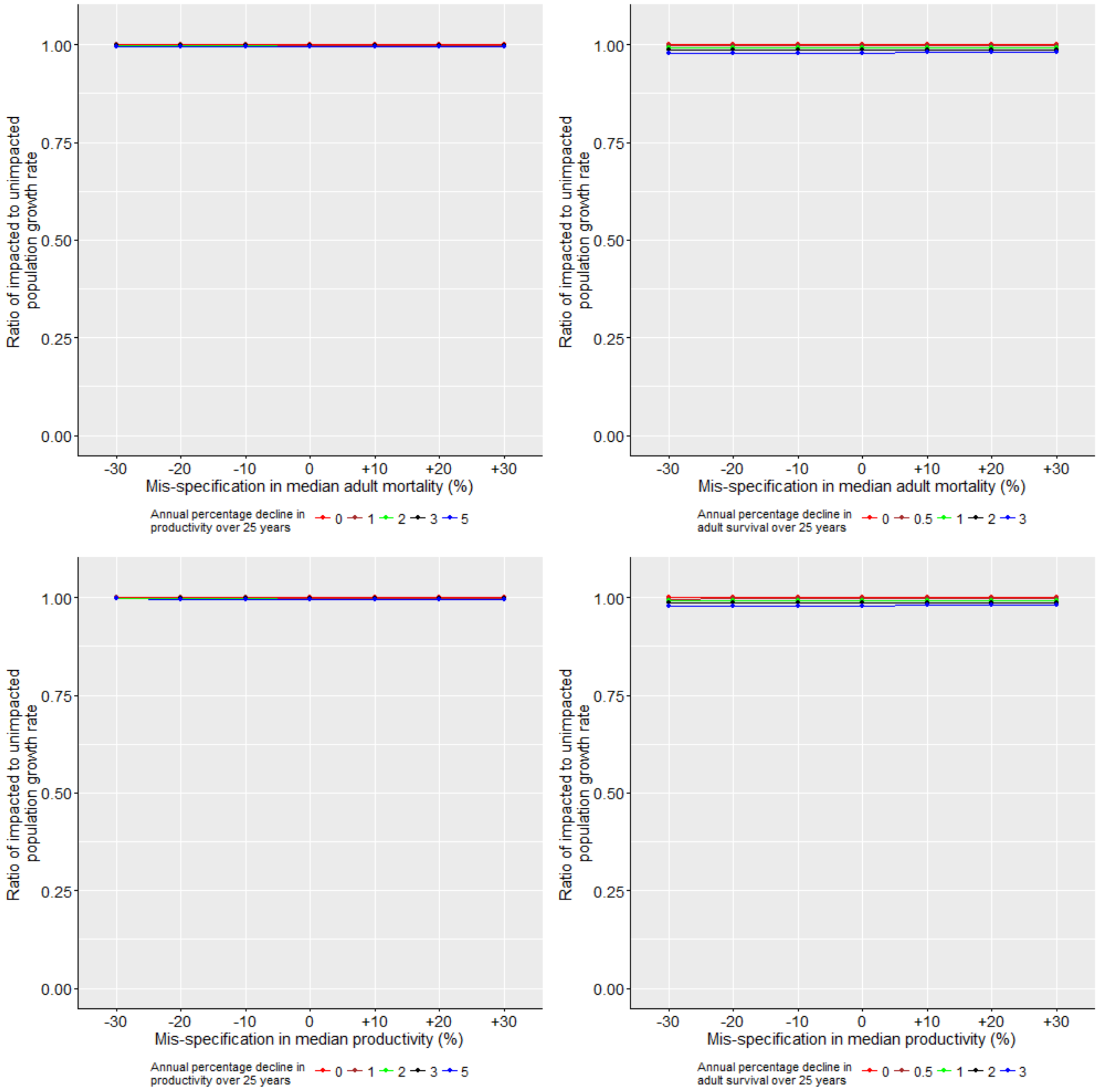


Figure A2.3b. PVA Metric B for Fowlsheugh Kittiwakes – ratio of population size at 2041, comparing impacted population vs. un-impacted population.

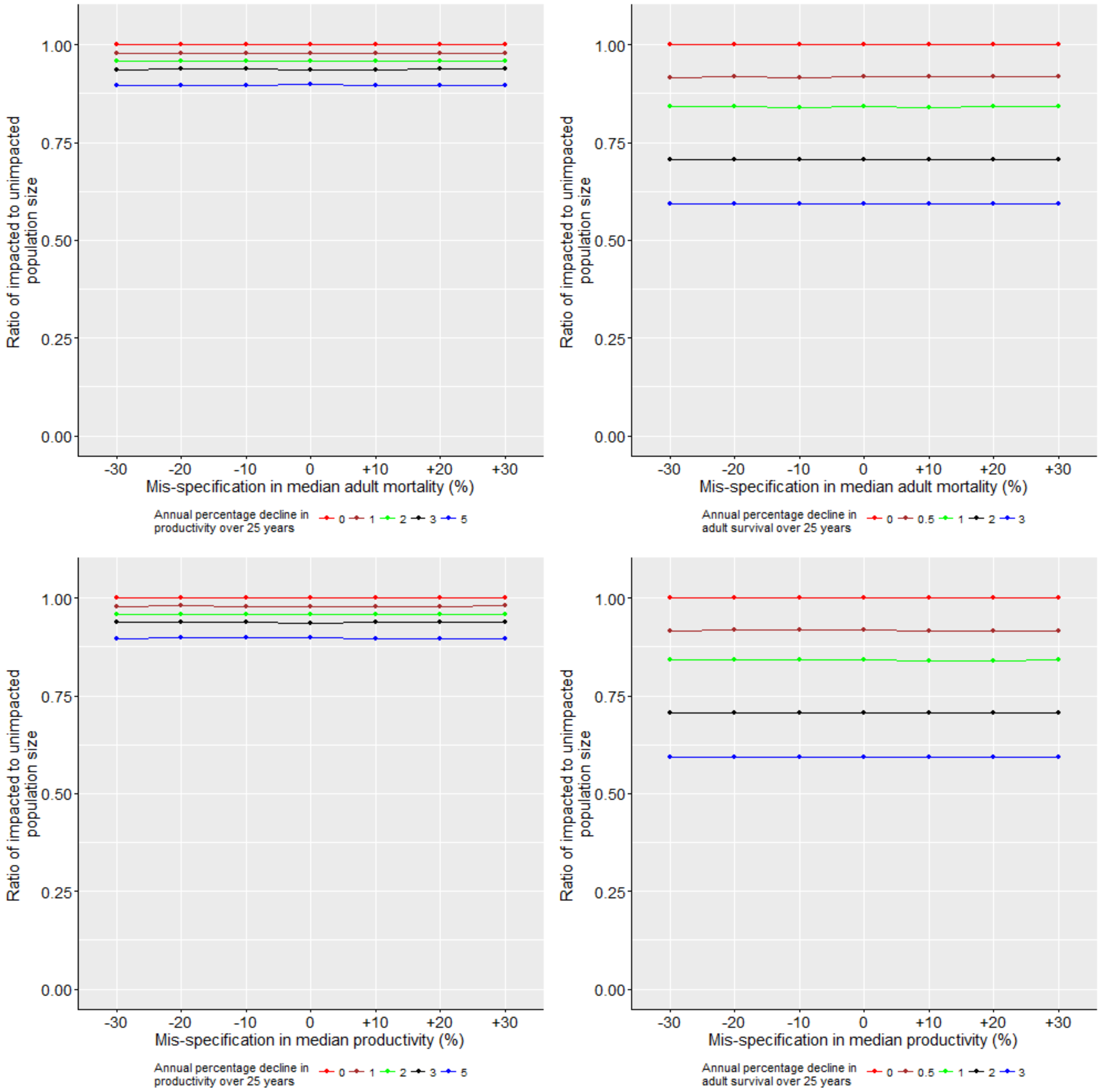


Figure A2.3c. PVA Metric C for Fowlsheugh Kittiwakes – difference in population growth rate from 2016-2041, comparing impacted population vs. un-impacted population.

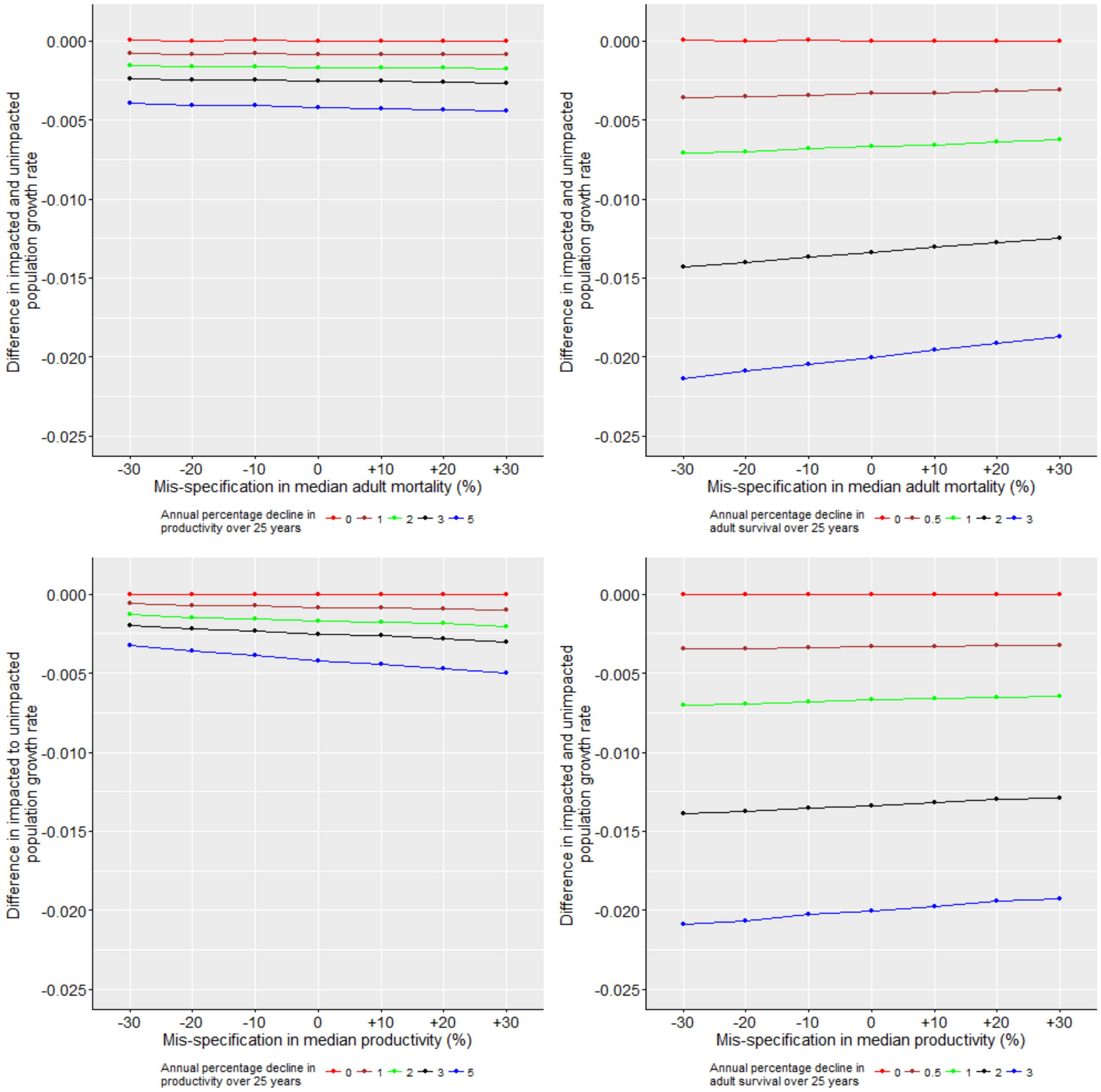


Figure A2.3d. PVA Metric D for Fowlsheugh Kittiwakes – difference in population size at 2041, comparing impacted population vs. un-impacted population.

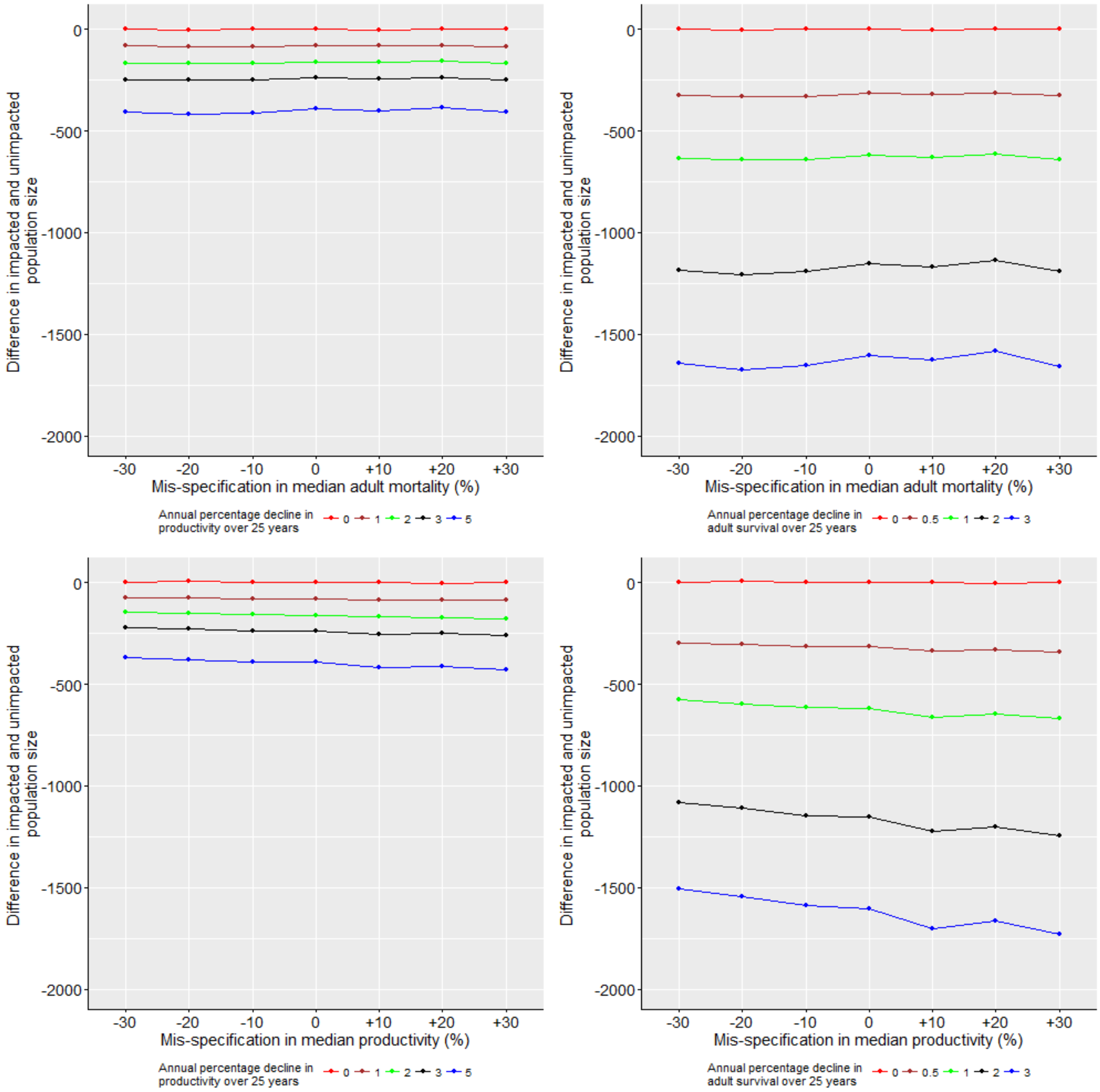


Figure A2.3e. PVA Metric E1 for Fowlsheugh Kittiwakes – probability of population decline greater than 10% from 2016-2041.

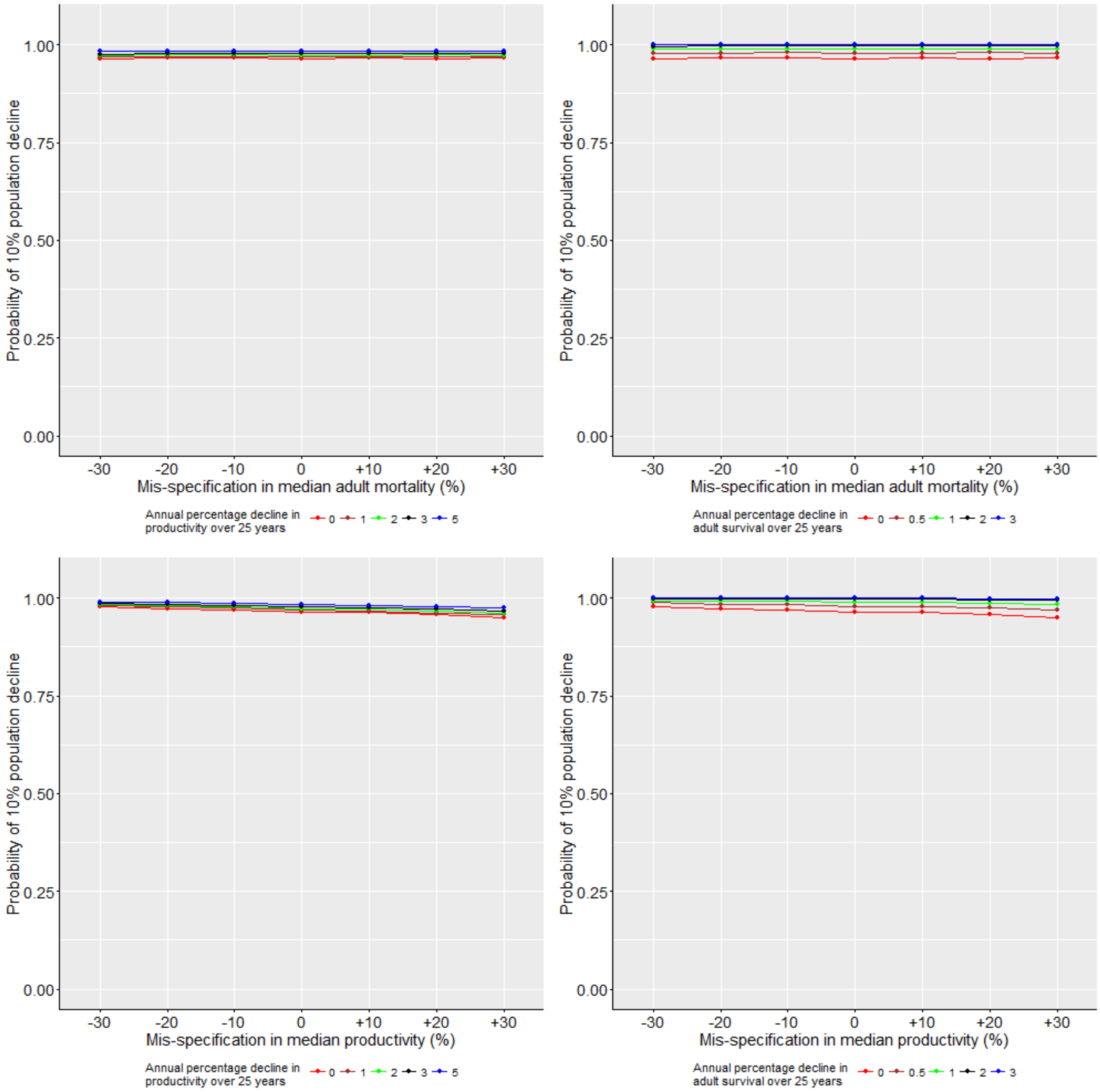


Figure A2.3f. PVA Metric E2 for Fowlsheugh Kittiwakes – probability of population decline greater than 25% from 2016-2041.

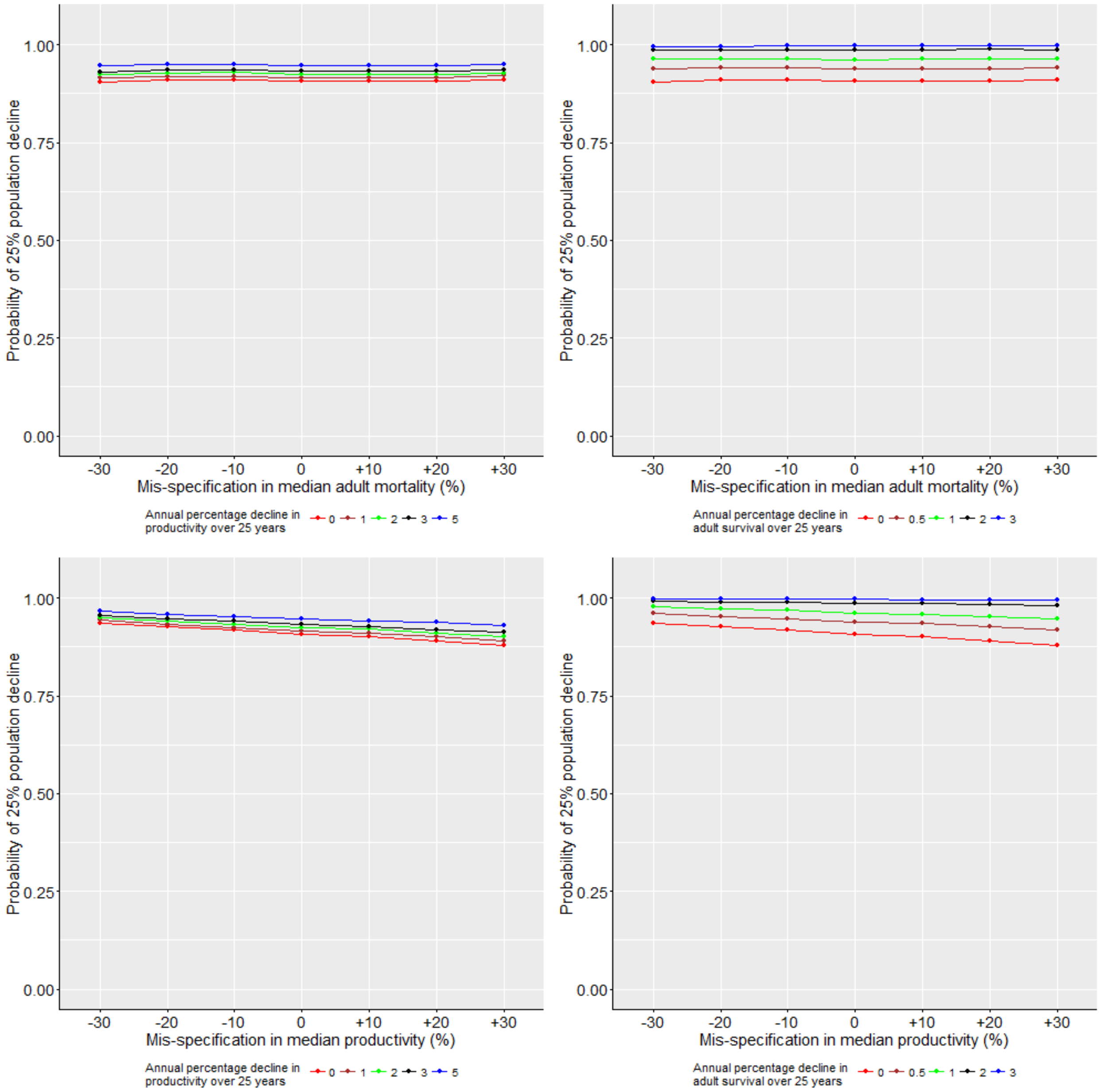


Figure A2.3g. PVA Metric E3 for Fowlsheugh Kittiwakes – probability of population decline greater than 50% from 2016-2041.

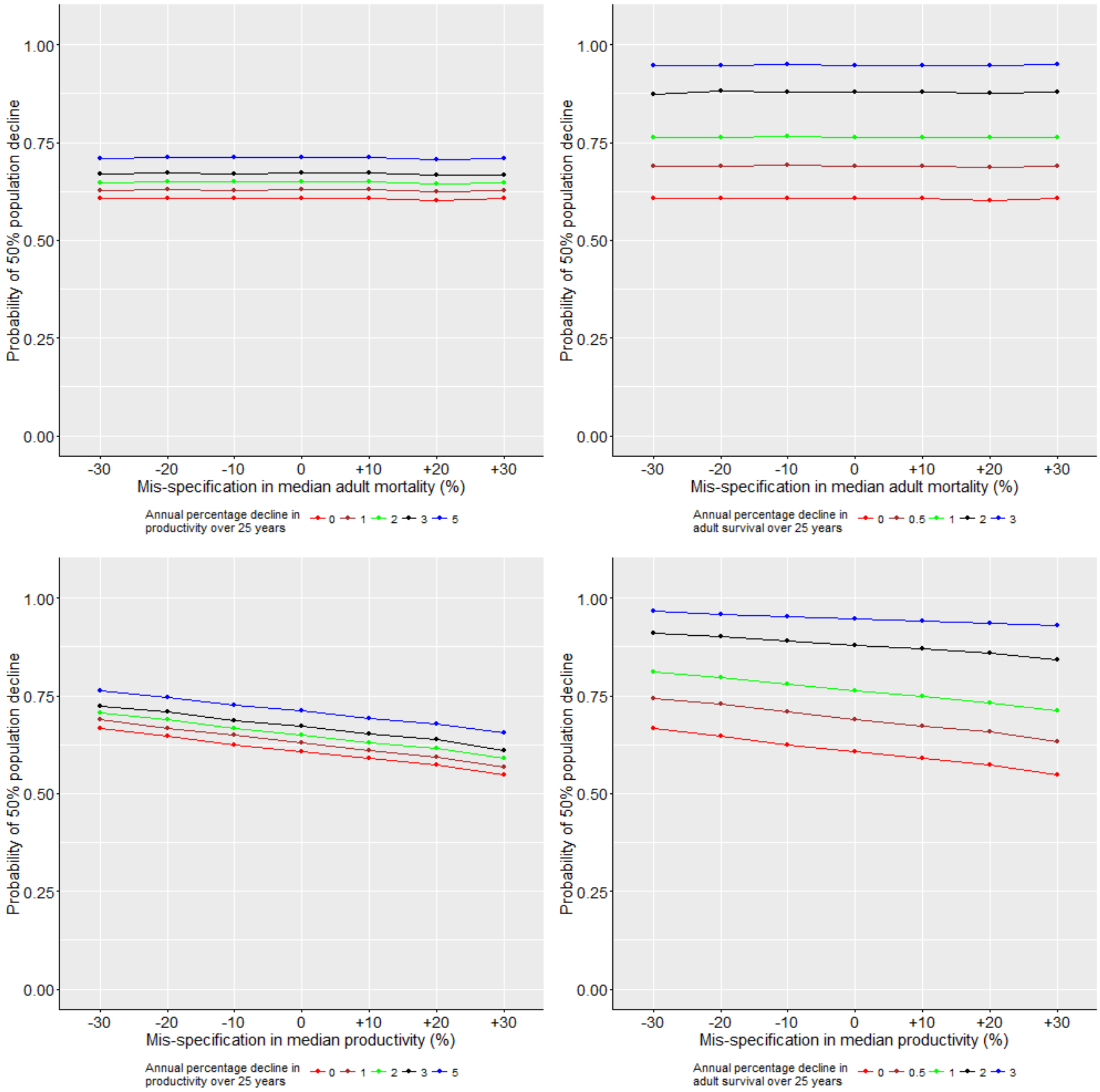
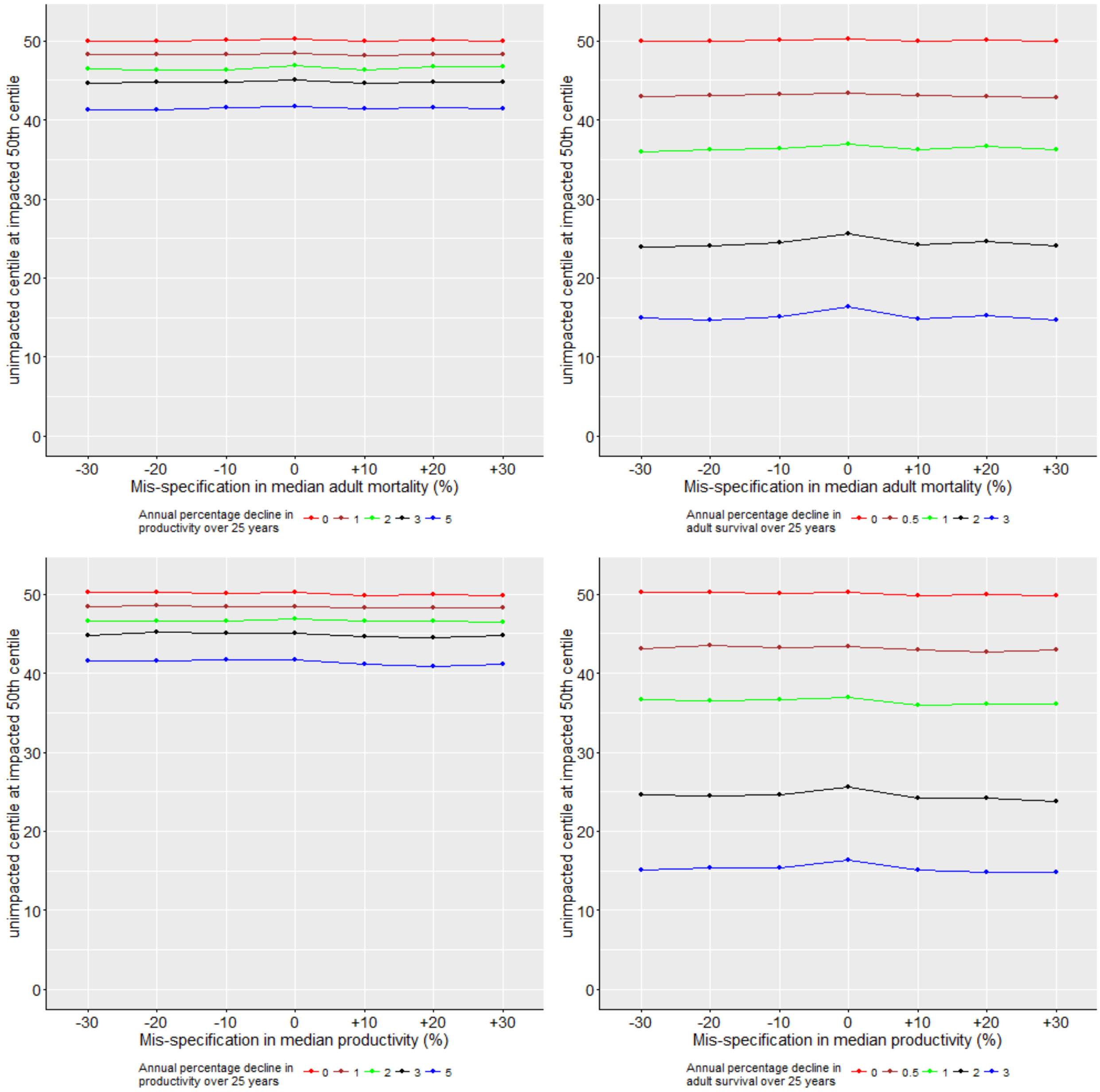


Figure A2.3h. PVA Metric F for Fowlsheugh Kittiwakes – centile from un-impacted population size equal to the 50th centile of the impacted population size, at 2041.



4. **Kittiwakes at Buchan Ness to Collieston Coast SPA**:

Figure A2.4a. PVA Metric A for Buchan Ness Kittiwakes – ratio of population growth rate from 2016-2041, comparing impacted population vs. un-impacted population.

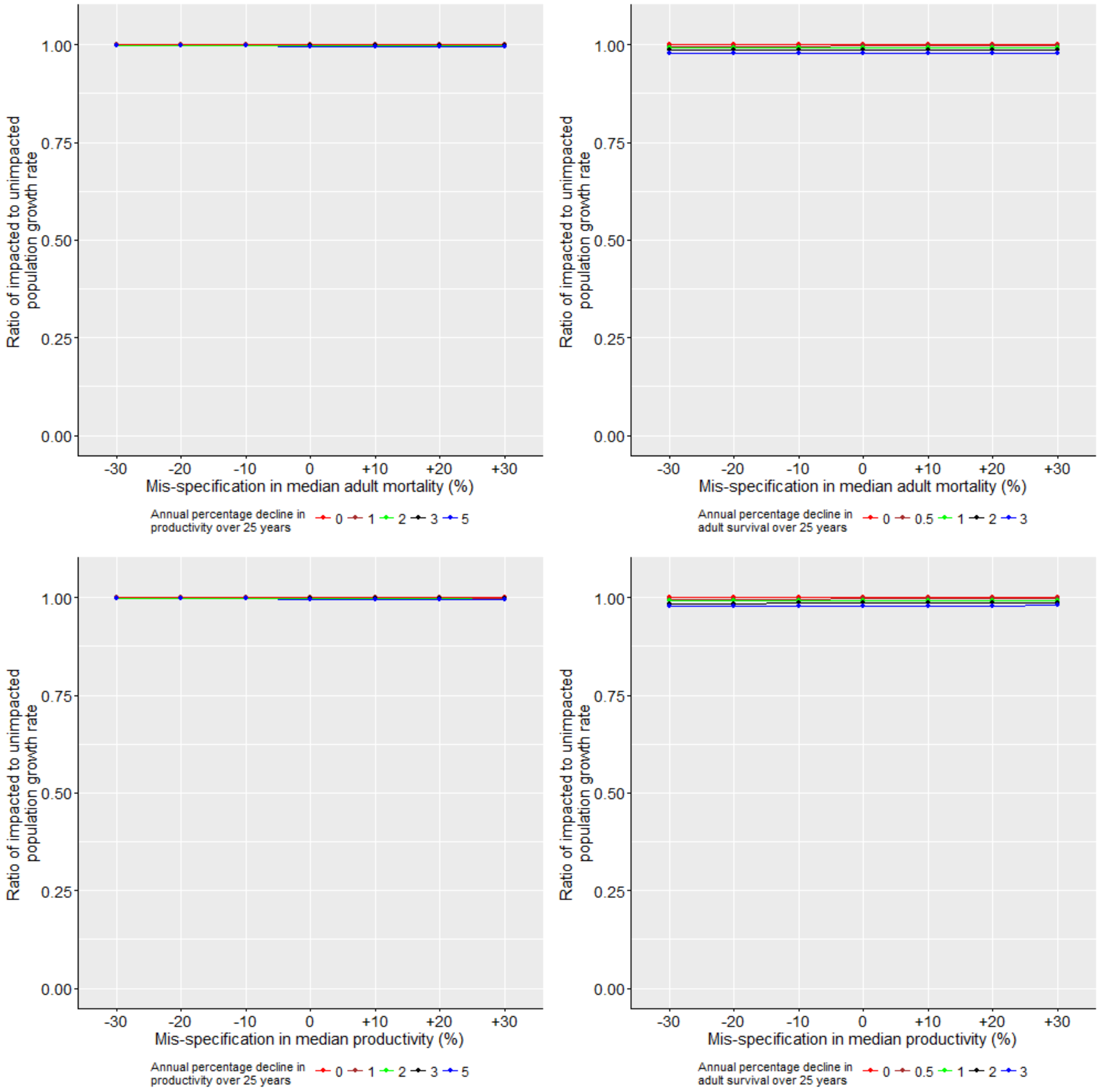


Figure A2.4b. PVA Metric B for Buchan Ness Kittiwakes – ratio of population size at 2041, comparing impacted population vs. un-impacted population.

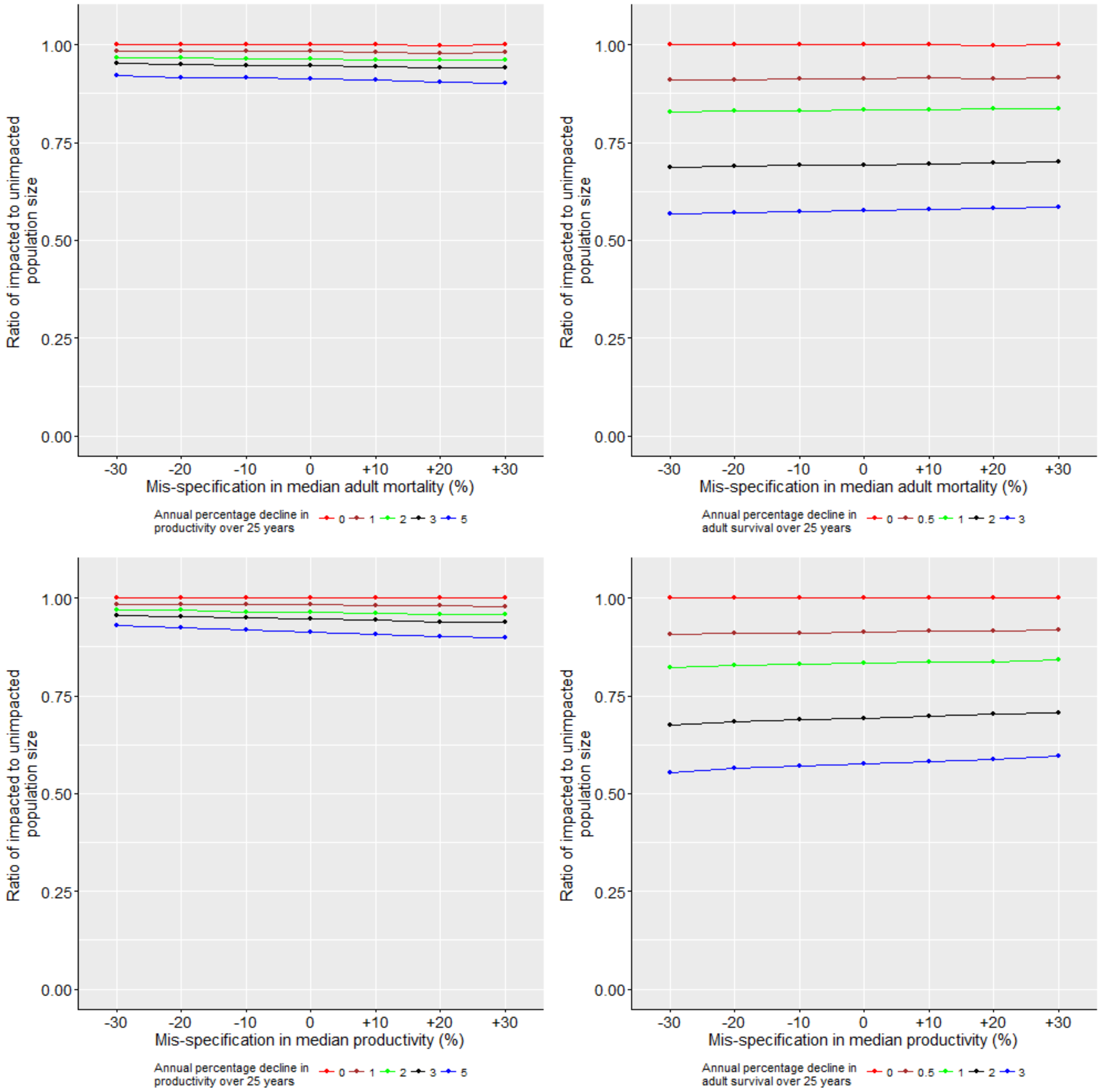


Figure A2.4c. PVA Metric C for Buchan Ness Kittiwakes – difference in population growth rate from 2016-2041, comparing impacted population vs. un-impacted population.

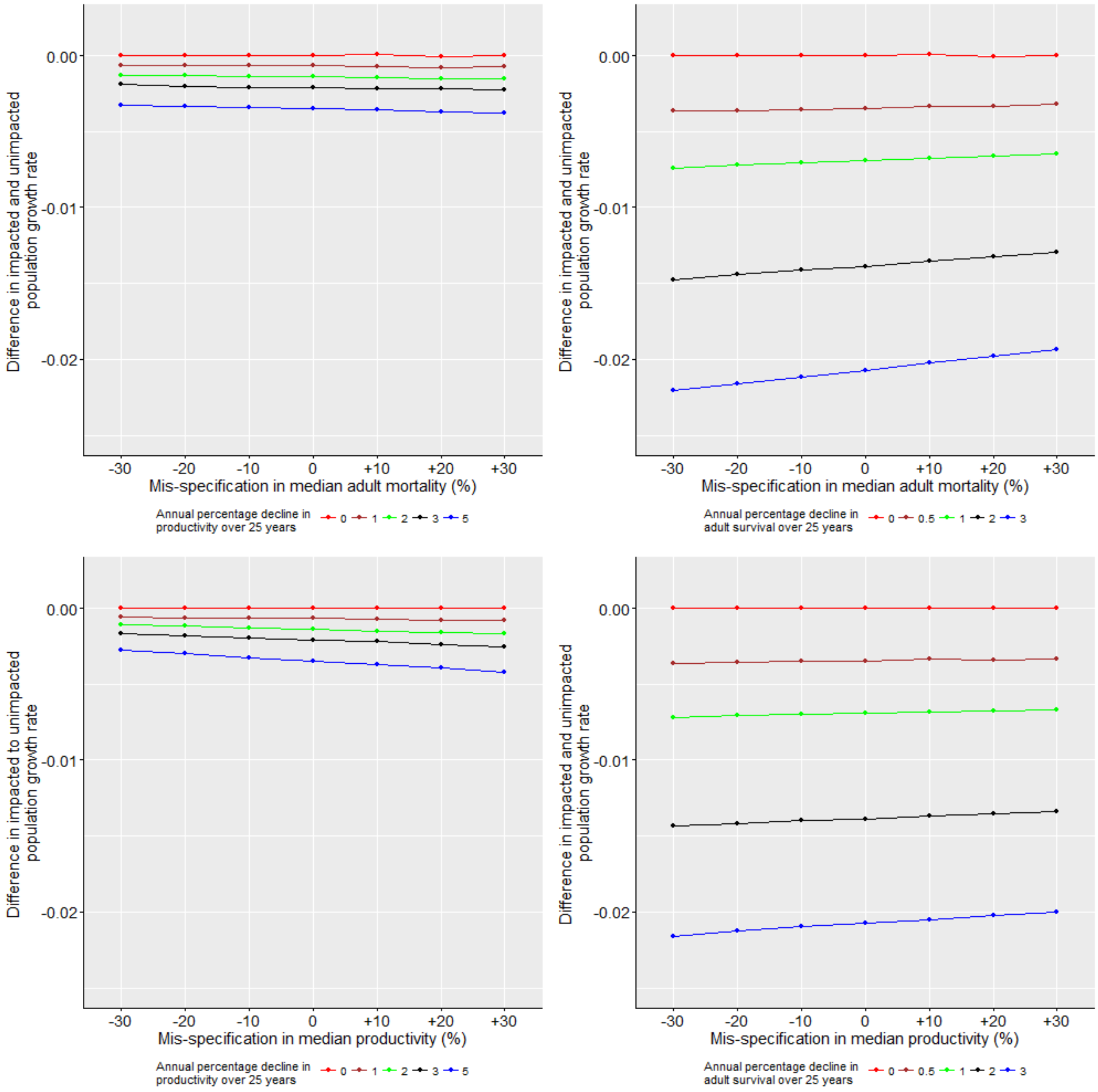


Figure A2.4d. PVA Metric D for Buchan Ness Kittiwakes – difference in population size at 2041, comparing impacted population vs. un-impacted population.

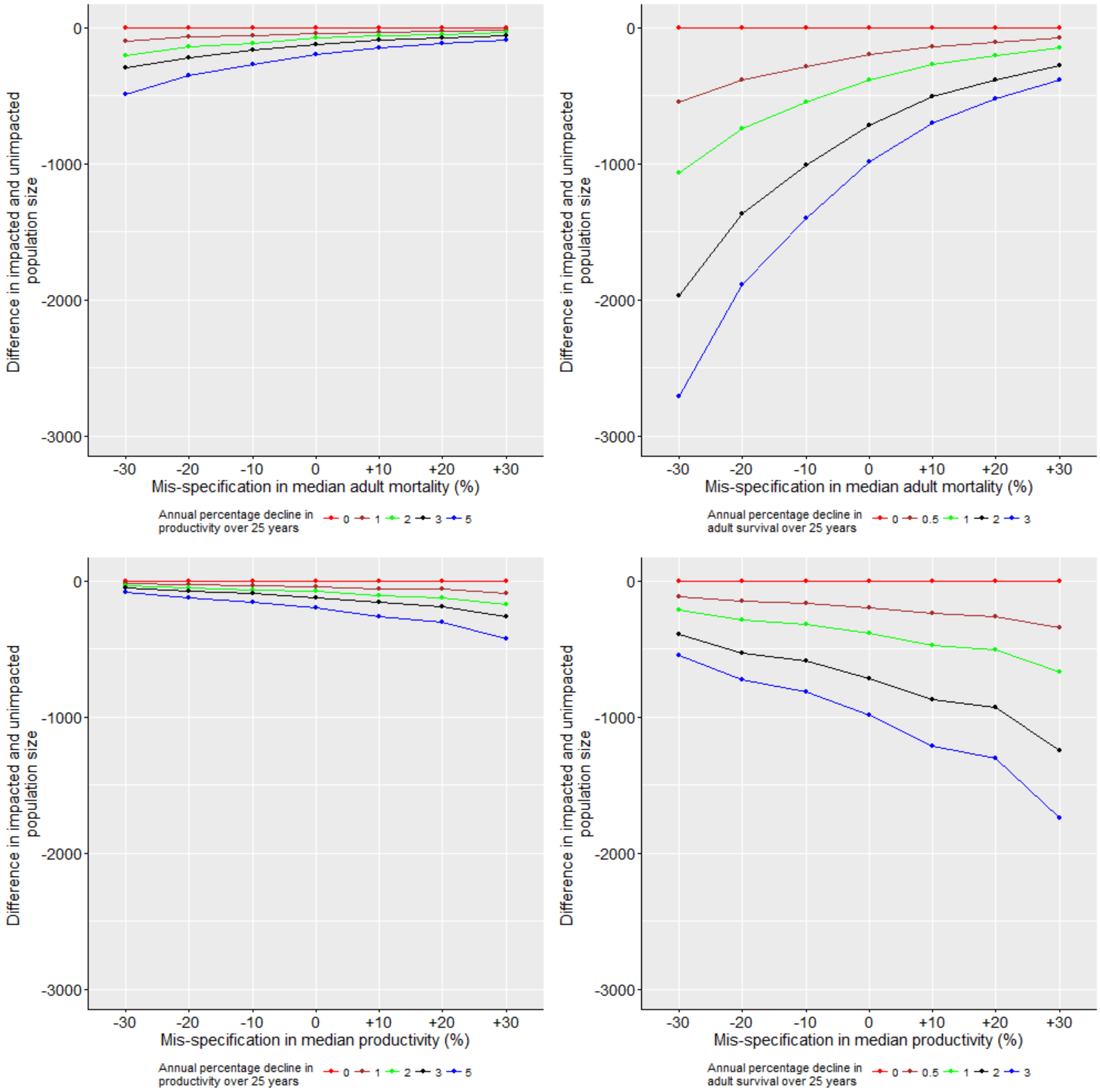


Figure A2.4e. PVA Metric E1 for Buchan Ness Kittiwakes – probability of population decline greater than 10% from 2016-2041.

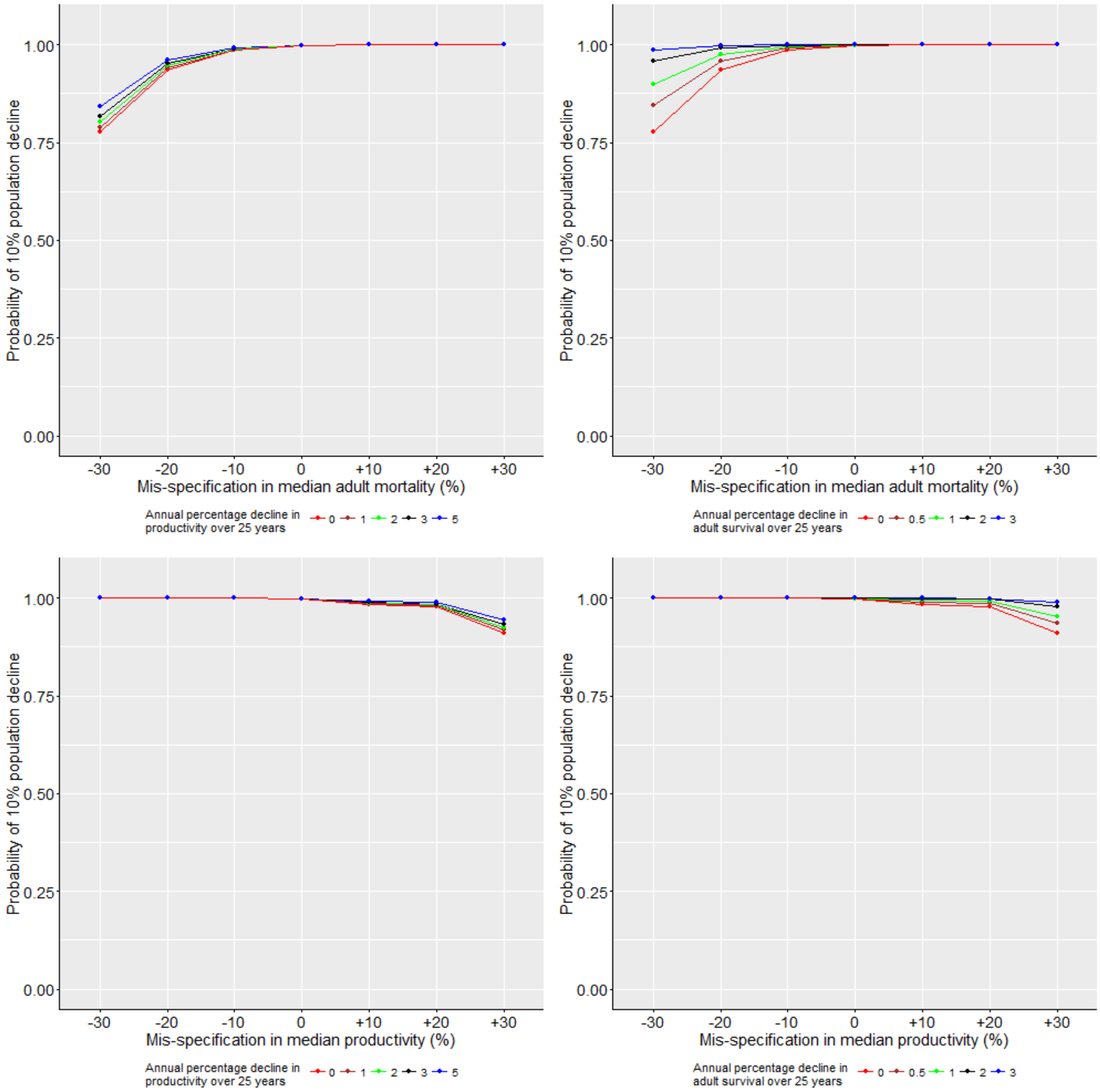


Figure A2.4f. PVA Metric E2 for Buchan Ness Kittiwakes – probability of population decline greater than 25% from 2016-2041.

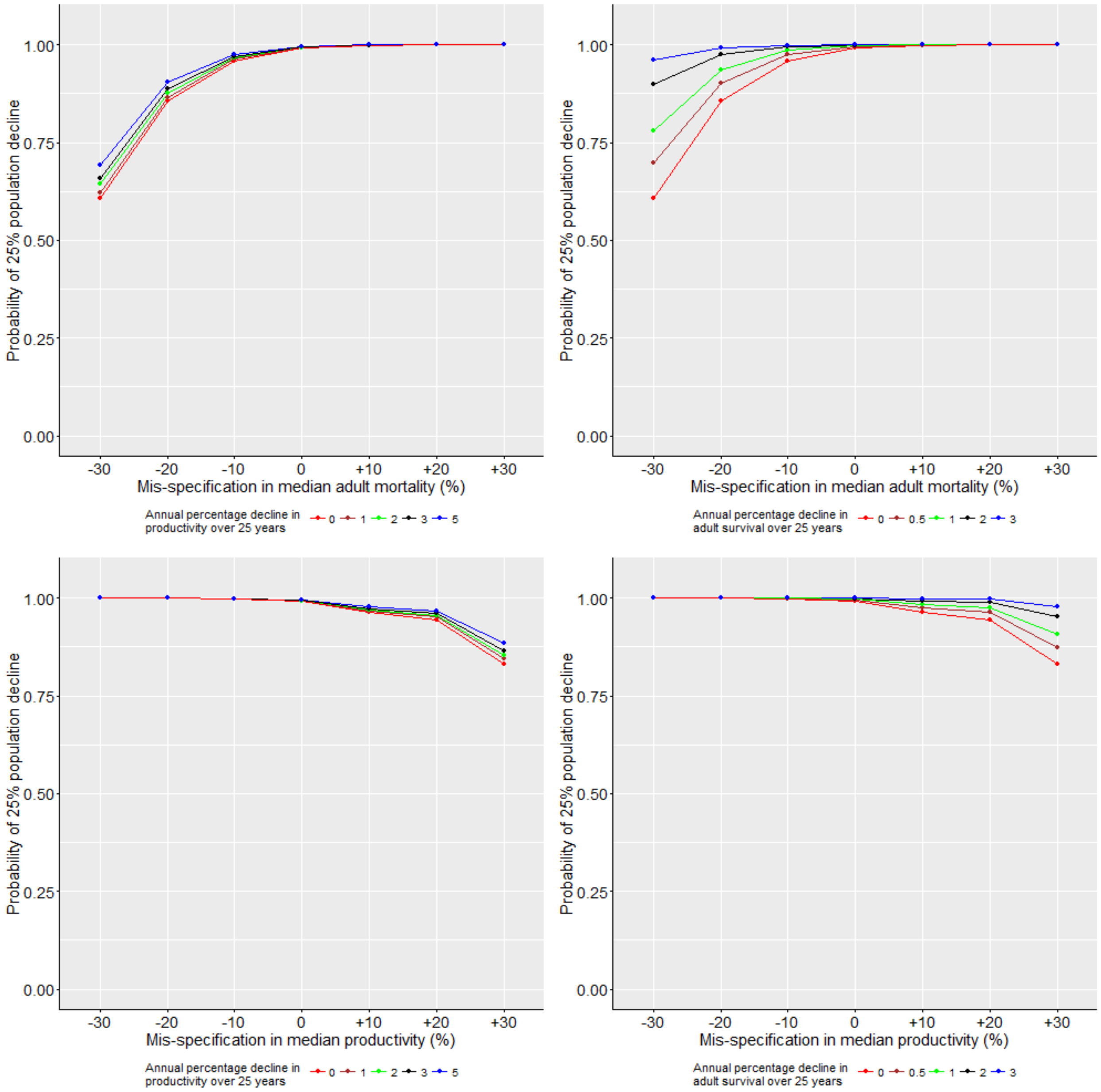


Figure A2.4g. PVA Metric E3 for Buchan Ness Kittiwakes – probability of population decline greater than 50% from 2016-2041.

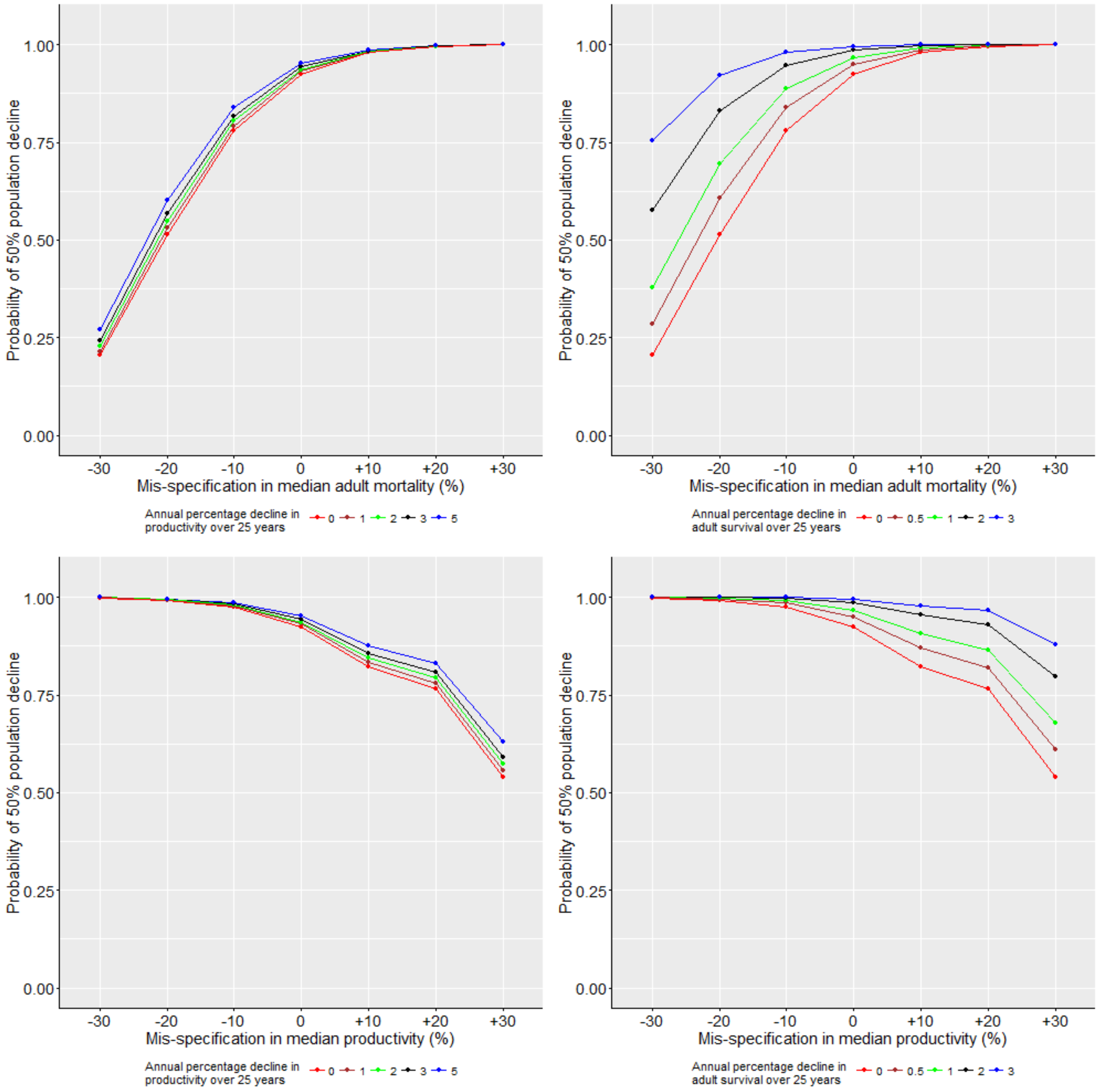
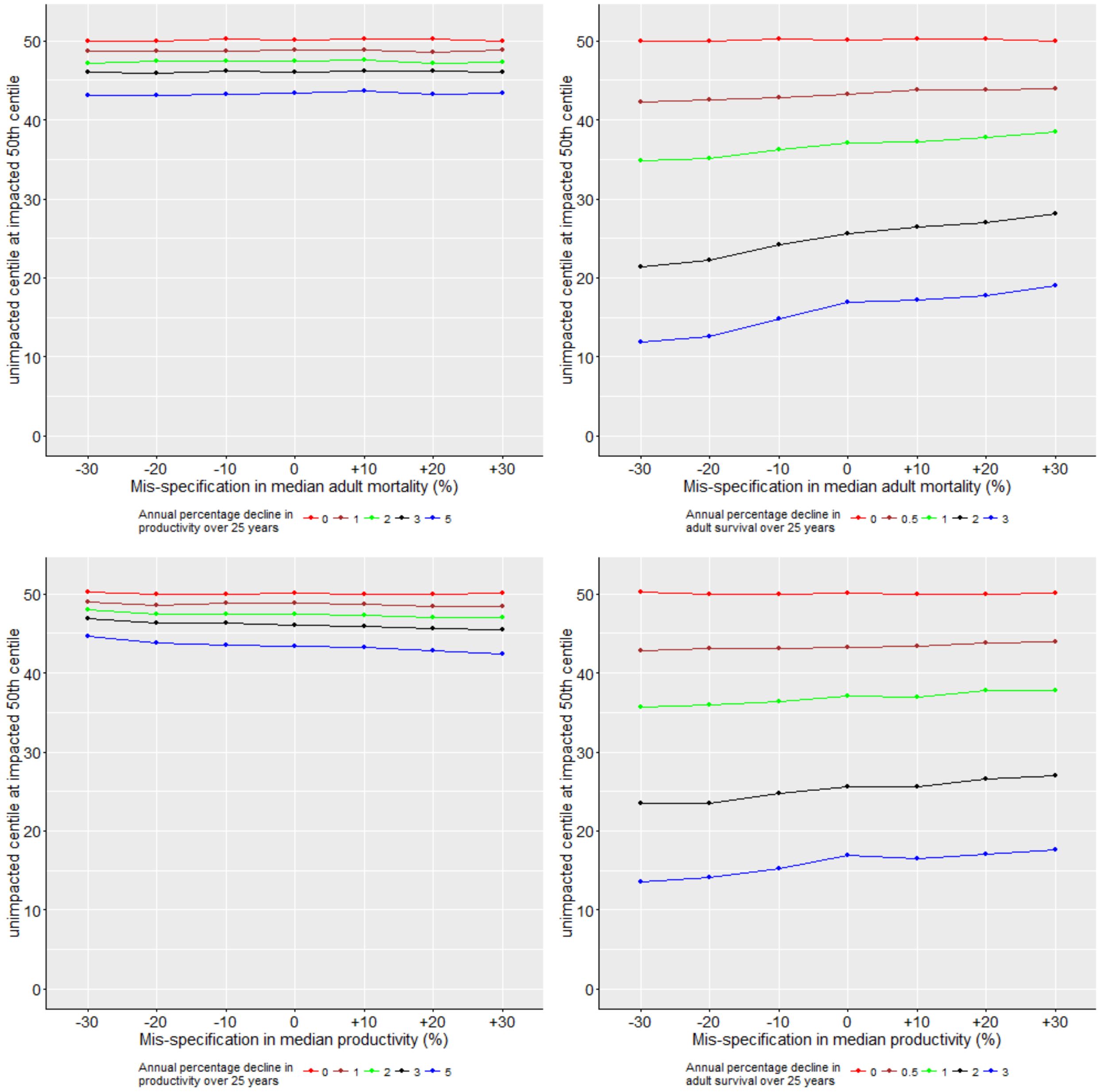


Figure A2.4h. PVA Metric F for Buchan Ness Kittiwakes – centile from un-impacted population size equal to the 50th centile of the impacted population size, at 2041.



5. **Guillemots at Forth Islands SPA**:

Figure A2.5a. PVA Metric A for Forth Guillemots – ratio of population growth rate from 2016-2041, comparing impacted population vs. un-impacted population.

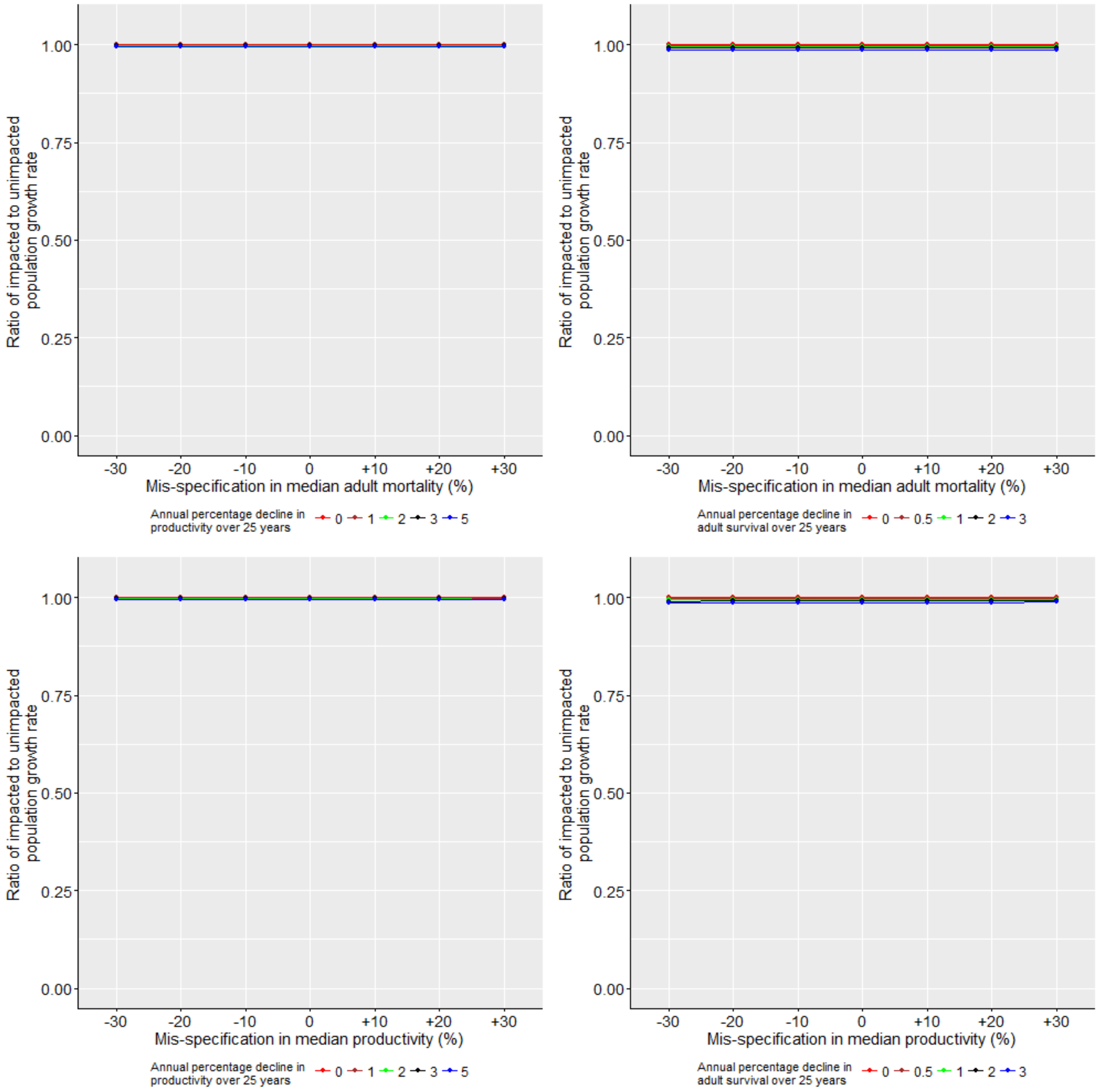


Figure A2.5b. PVA Metric B for Forth Guillemots – ratio of population size at 2041, comparing impacted population vs. un-impacted population.

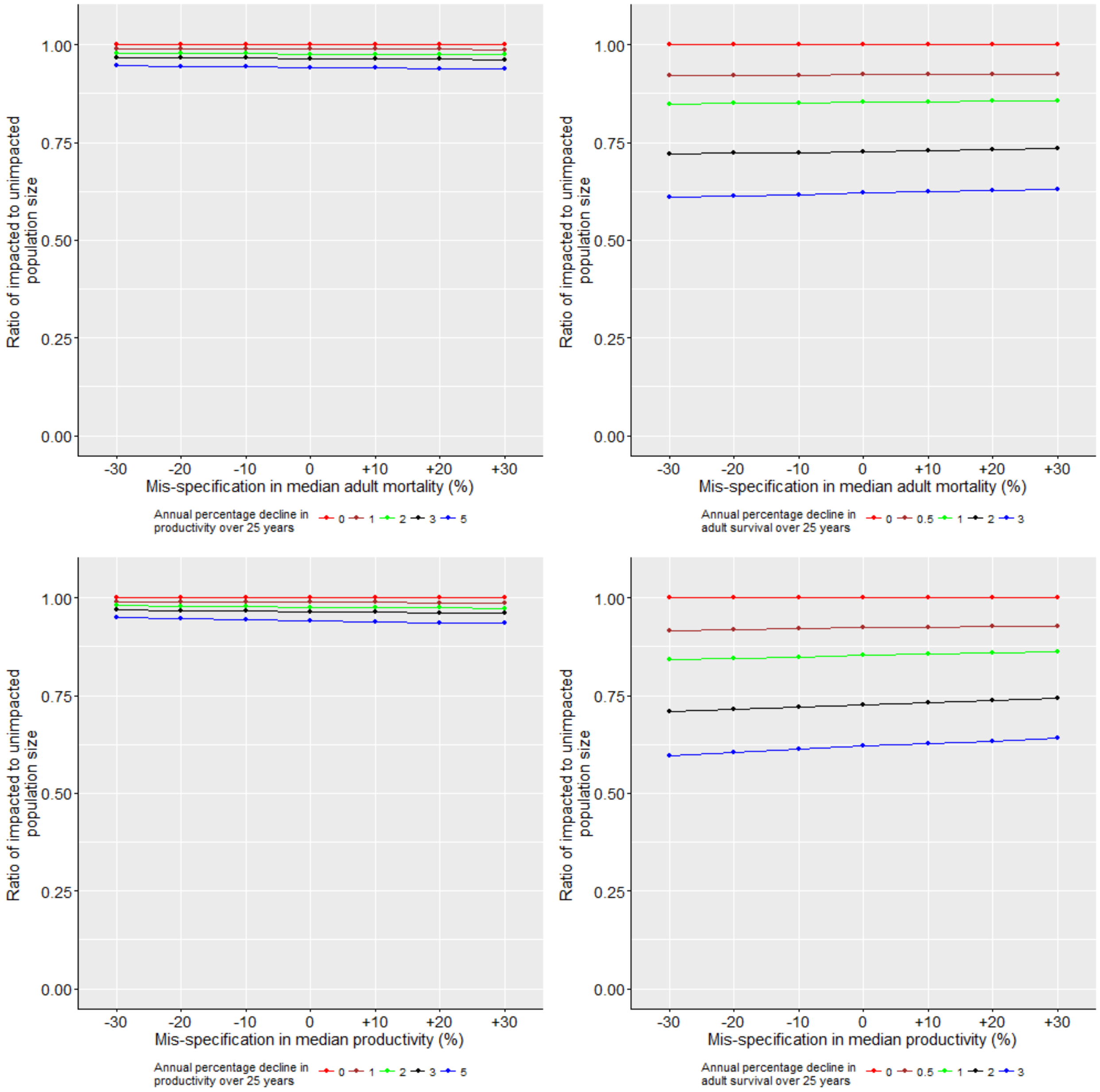


Figure A2.5c. PVA Metric C for Forth Guillemots – difference in population growth rate from 2016-2041, comparing impacted population vs. un-impacted population.

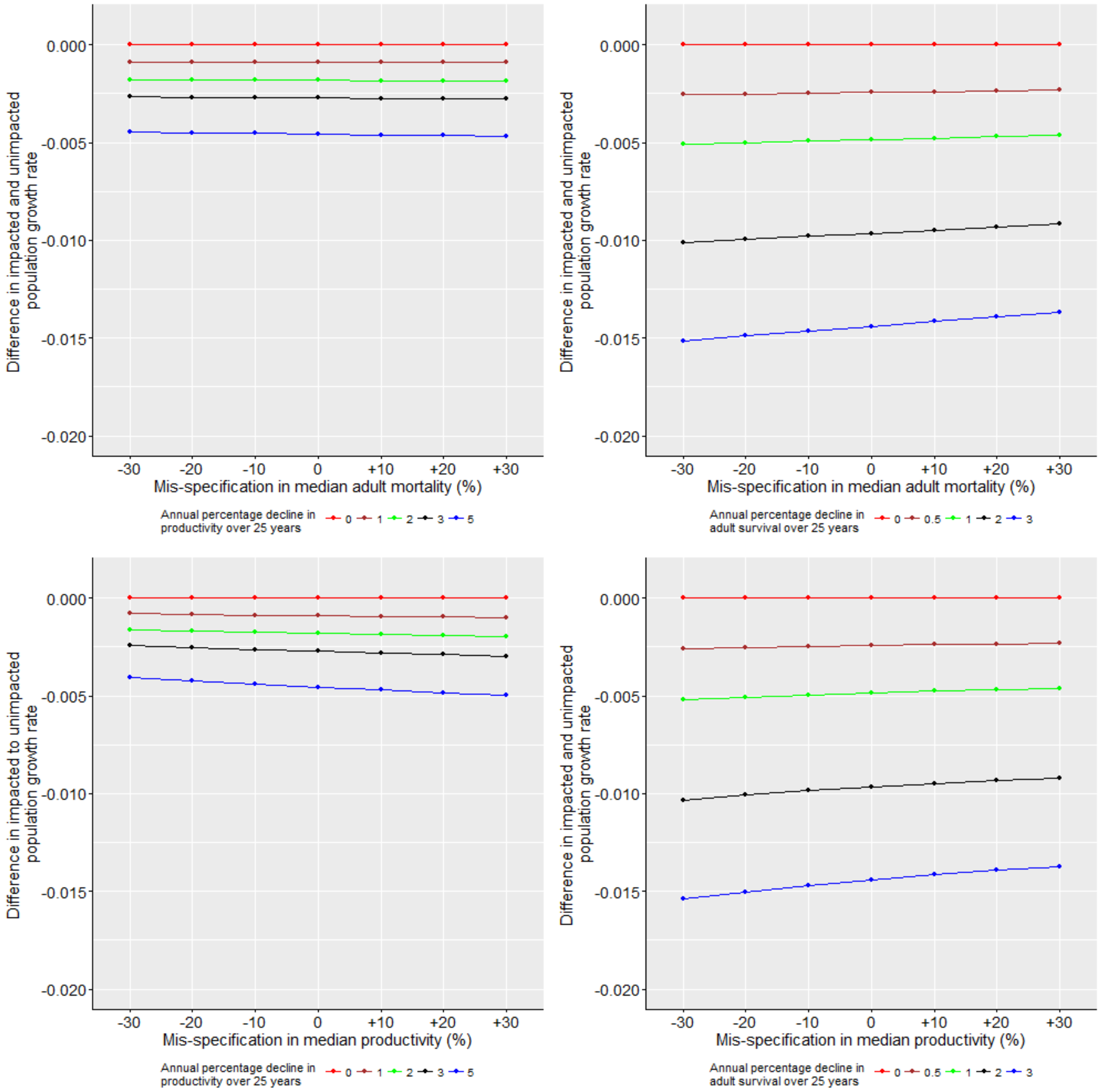


Figure A2.5d. PVA Metric D for Forth Guillemots – difference in population size at 2041, comparing impacted population vs. un-impacted population.

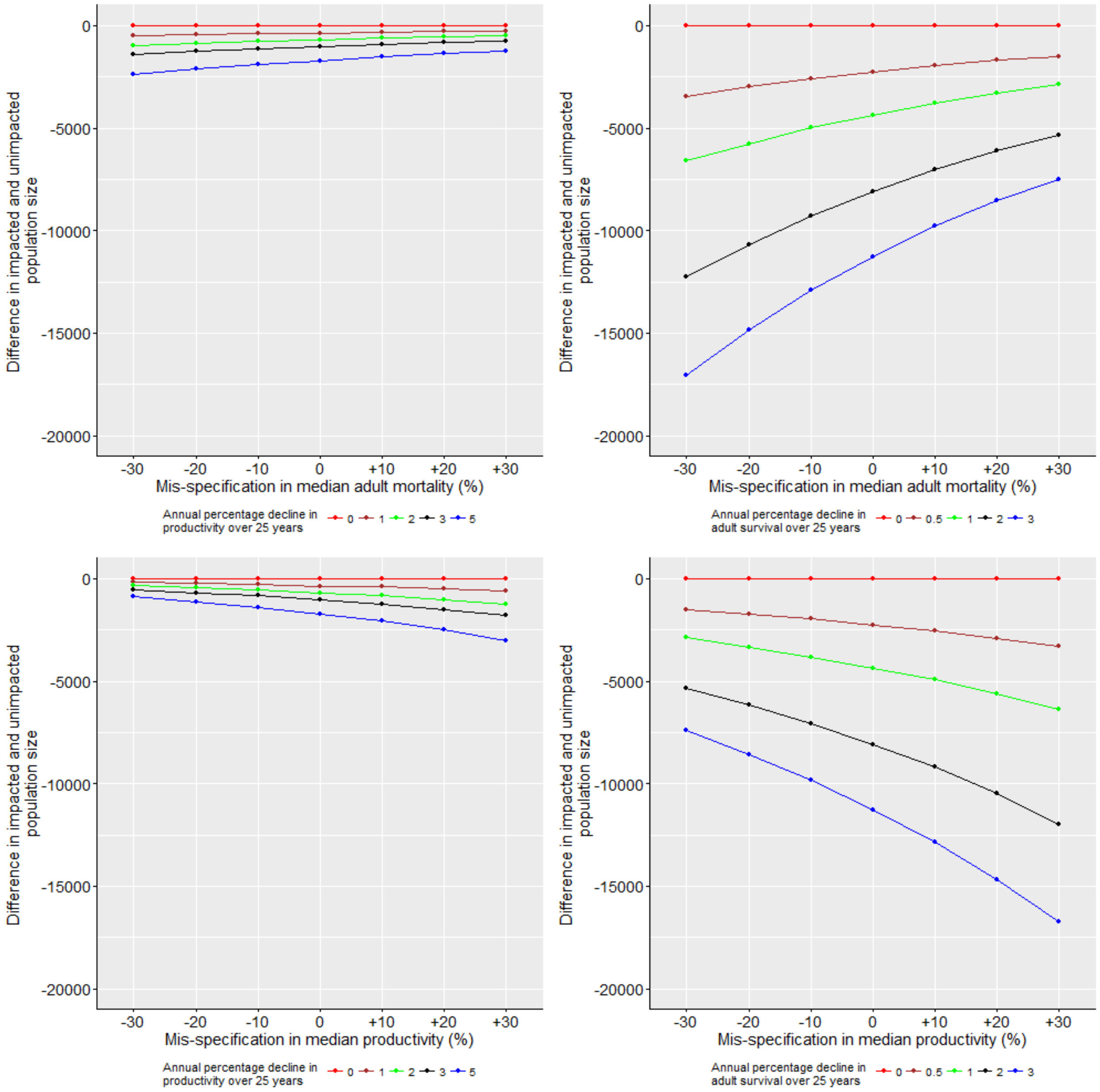


Figure A2.5e. PVA Metric E1 for Forth Guillemots – probability of population decline greater than 10% from 2016-2041.

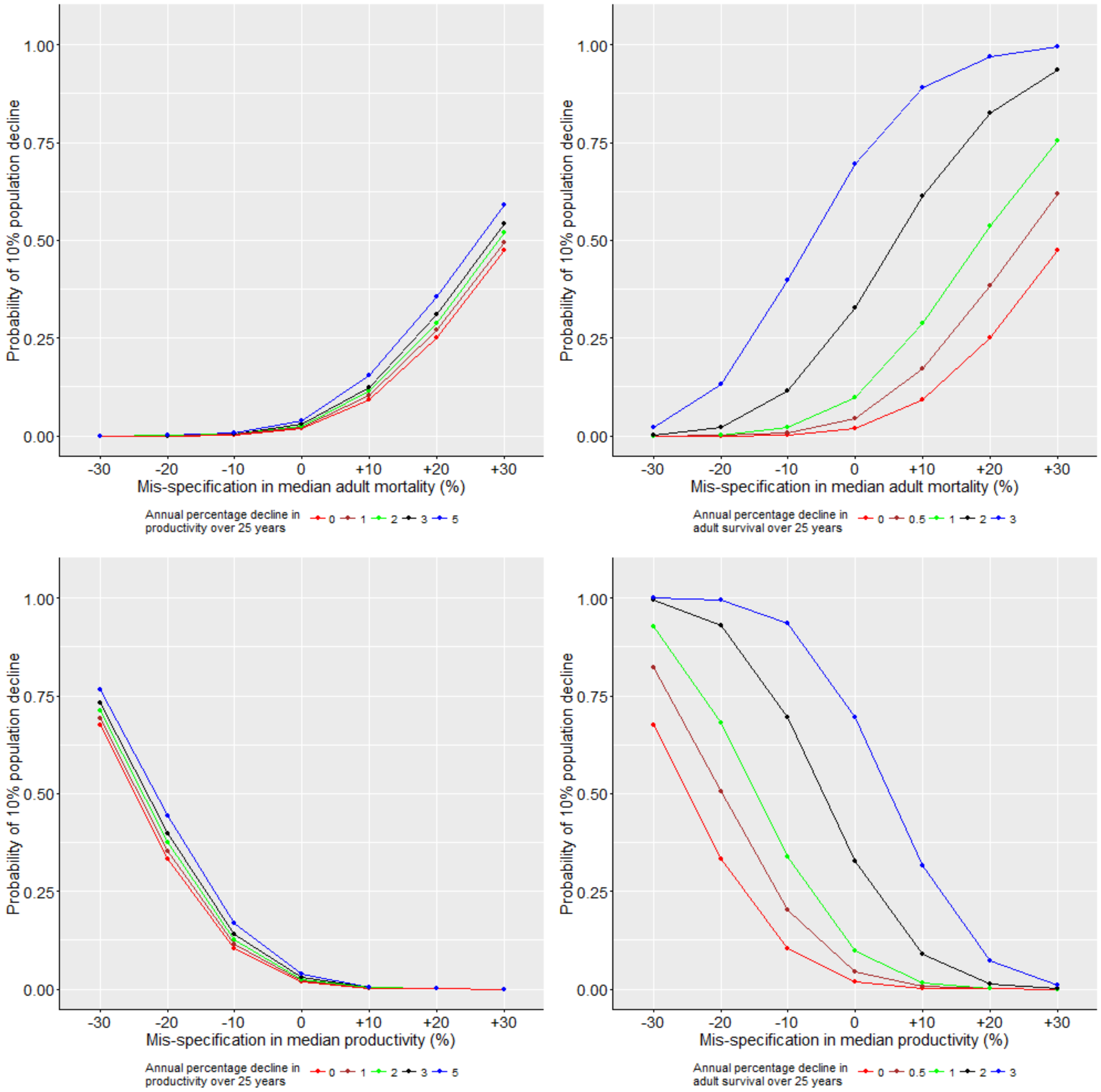


Figure A2.5f. PVA Metric E2 for Forth Guillemots – probability of population decline greater than 25% from 2016-2041.

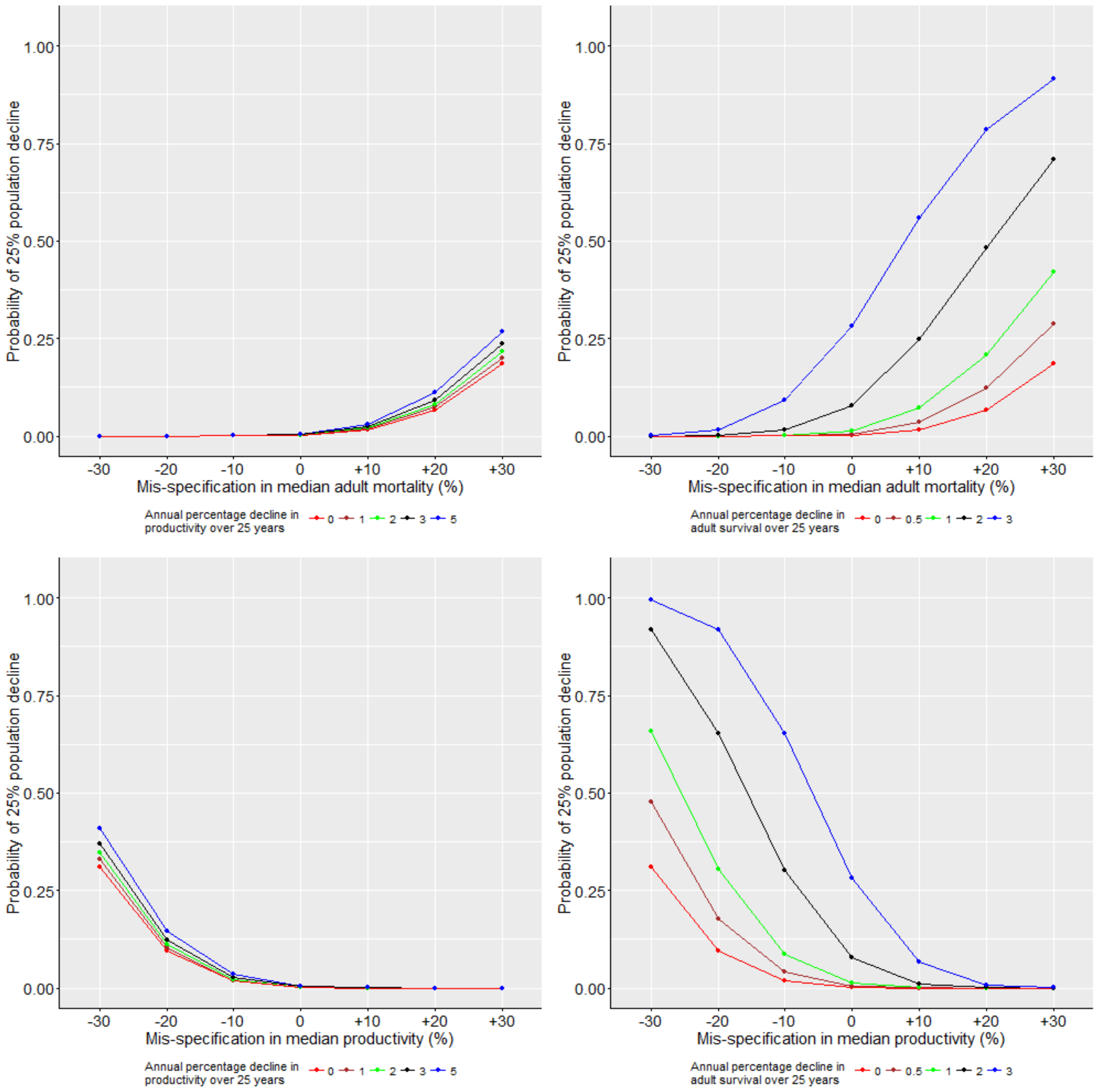


Figure A2.5g. PVA Metric E3 for Forth Guillemots – probability of population decline greater than 50% from 2016-2041.

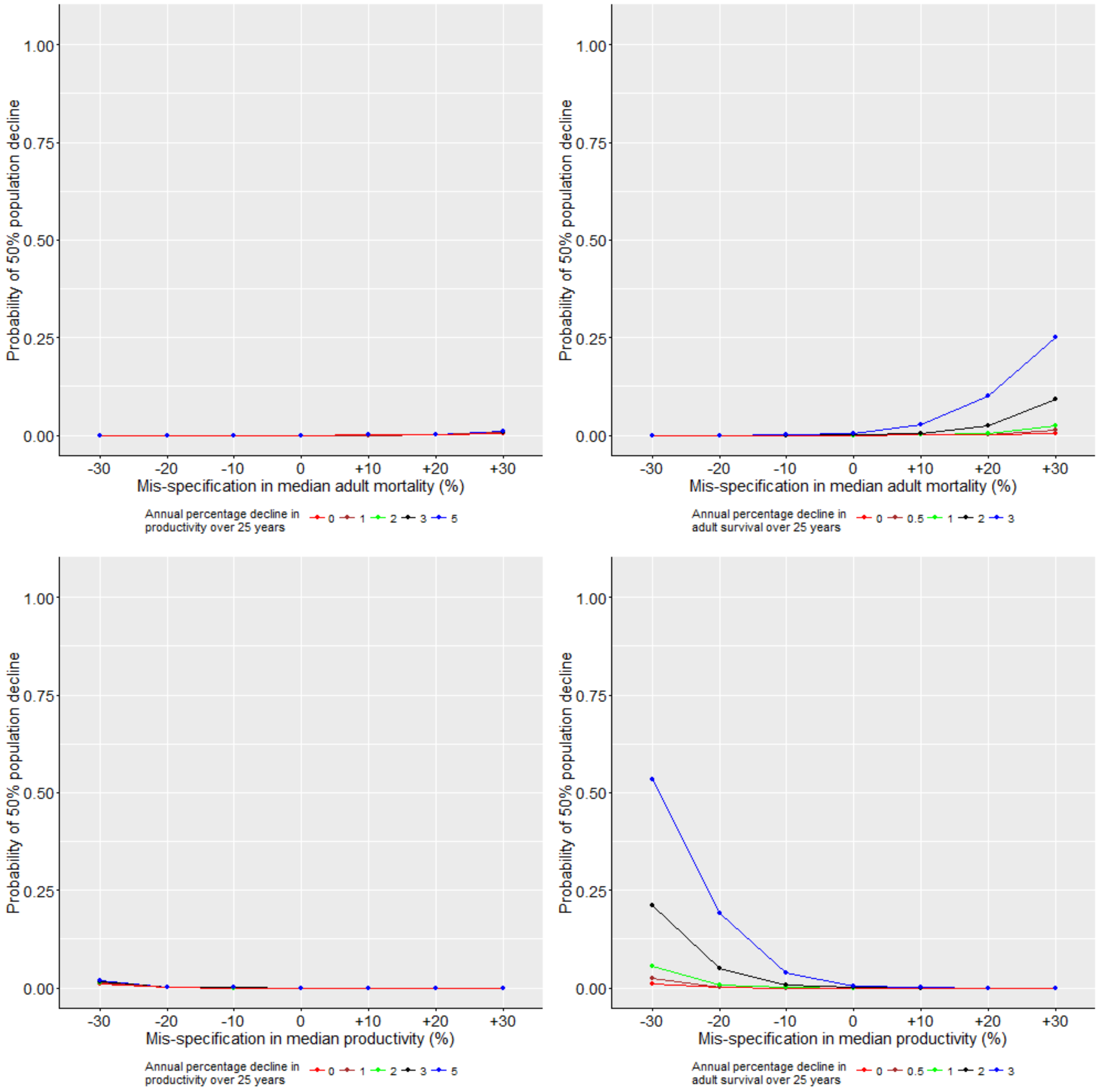
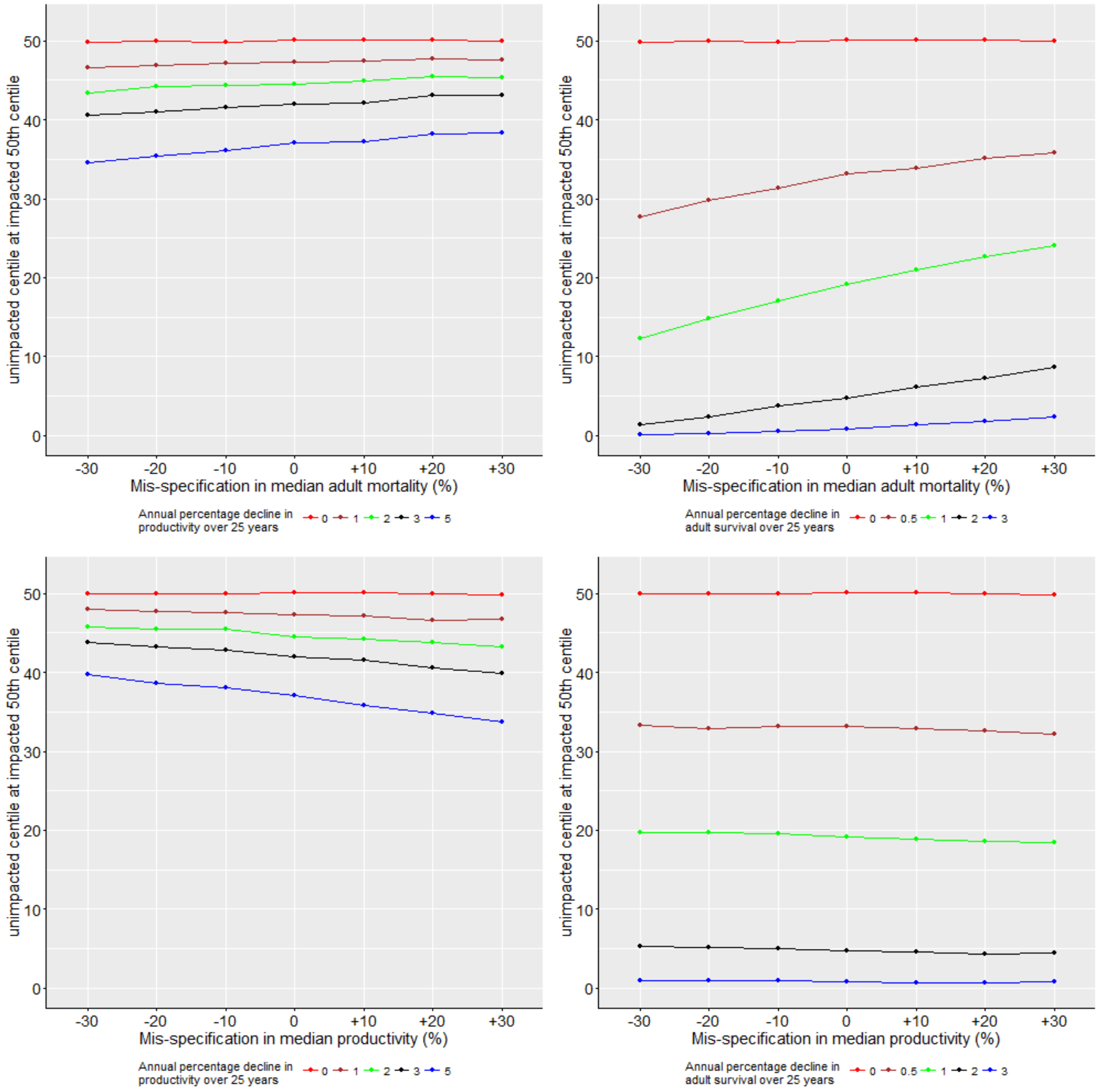


Figure A2.5h. PVA Metric F for Forth Guillemots – centile from un-impacted population size equal to the 50th centile of the impacted population size, at 2041.



6. **Guillemots at St Abb’s Head SPA**:

Figure A2.6a. PVA Metric A for St Abb’s Guillemots – ratio of population growth rate from 2016-2041, comparing impacted population vs. un-impacted population.

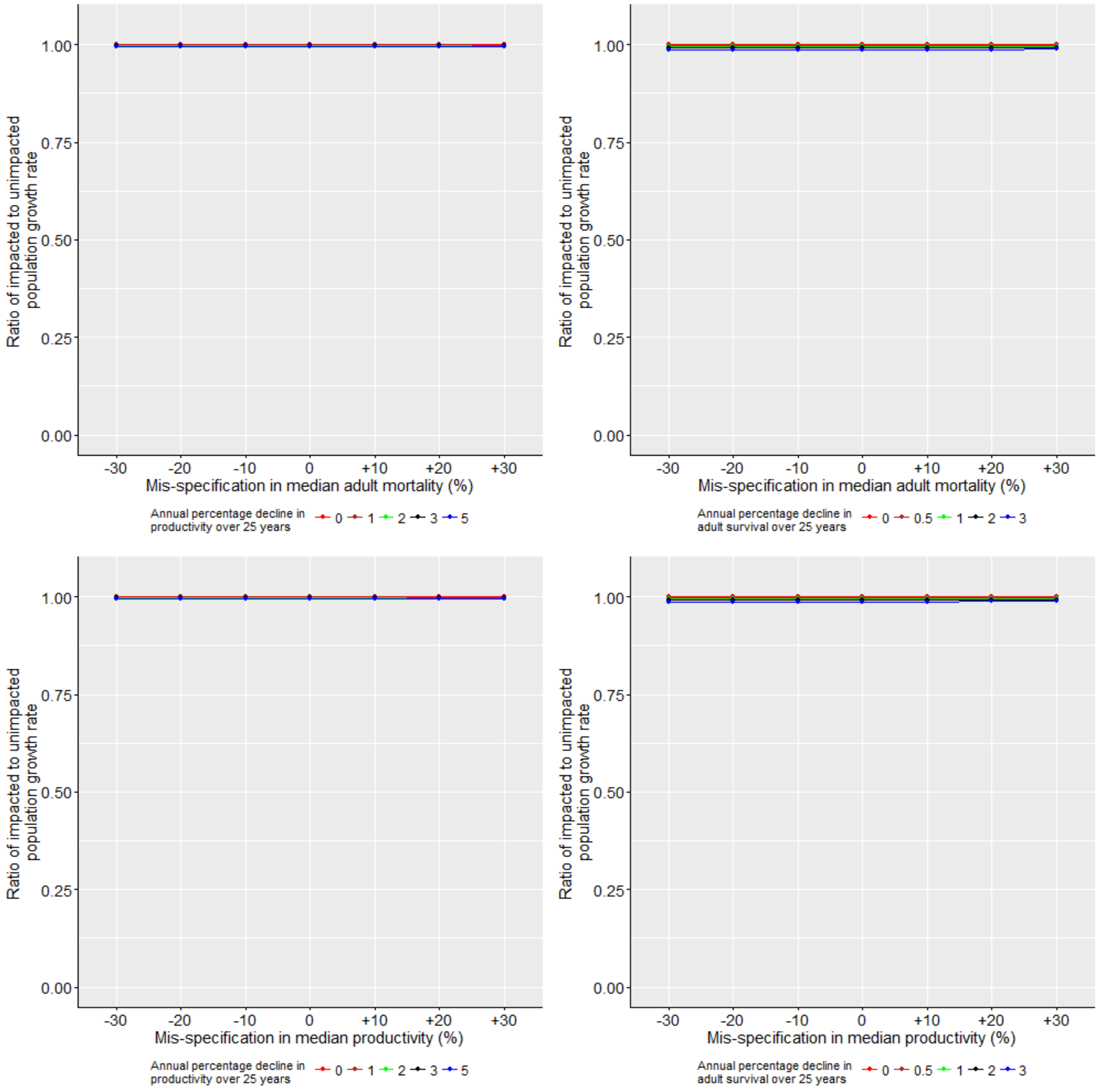


Figure A2.6b. PVA Metric B for St Abb’s Guillemots – ratio of population size at 2041, comparing impacted population vs. un-impacted population.

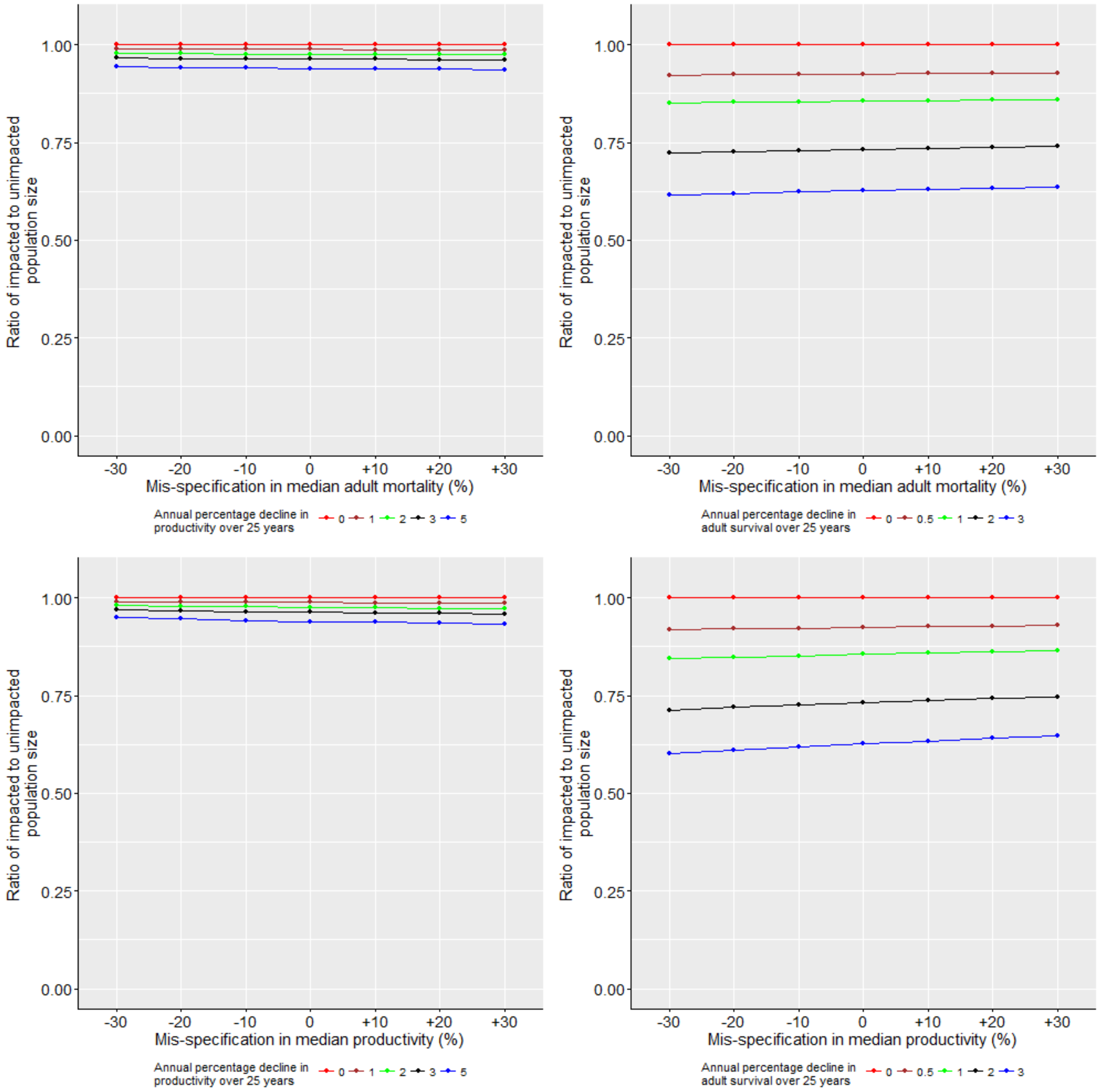


Figure A2.6c. PVA Metric C for St Abb's Guillemots – difference in population growth rate from 2016-2041, comparing impacted population vs. un-impacted population.

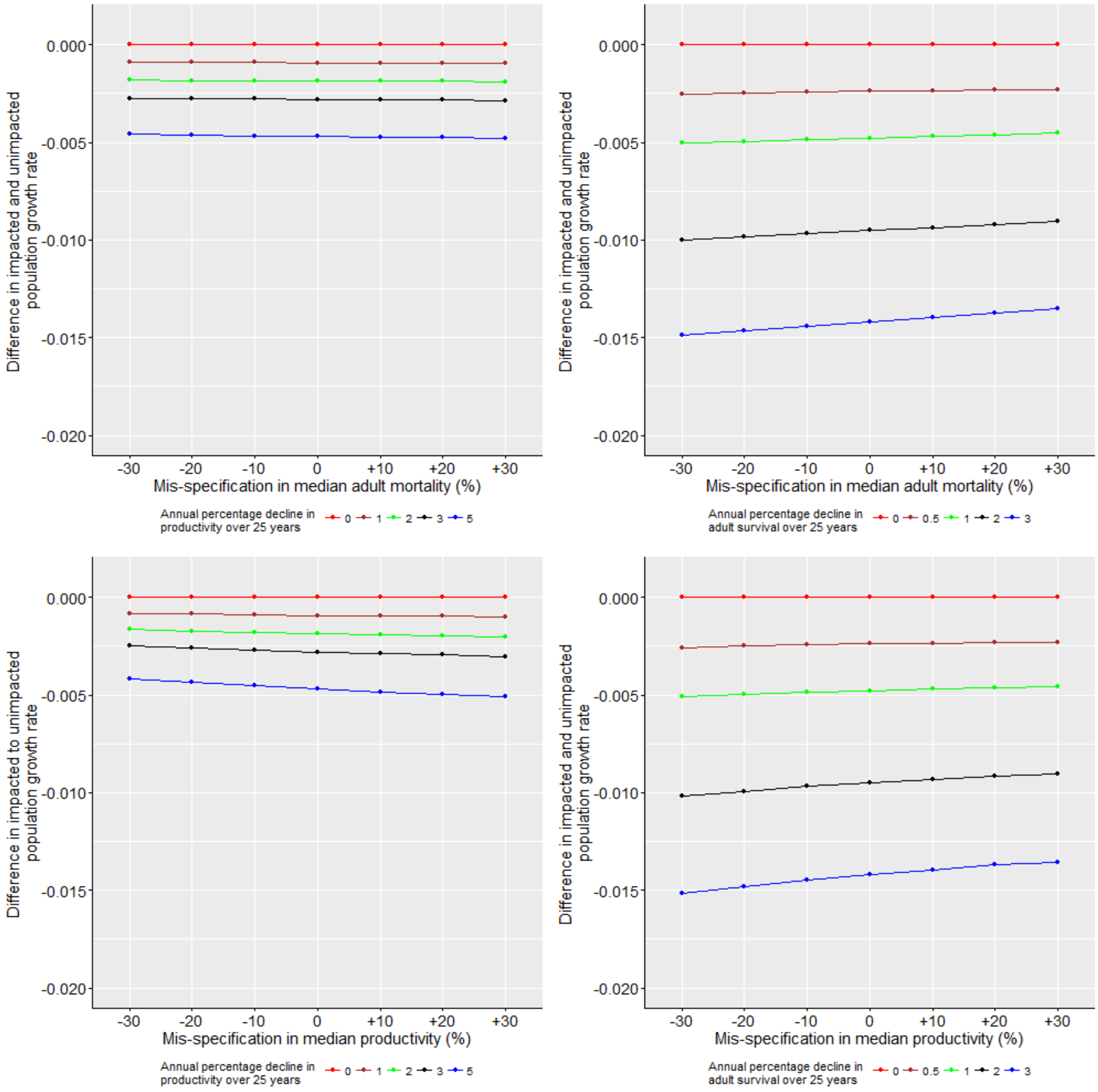


Figure A2.6d. PVA Metric D for St Abb’s Guillemots – difference in population size at 2041, comparing impacted population vs. un-impacted population.

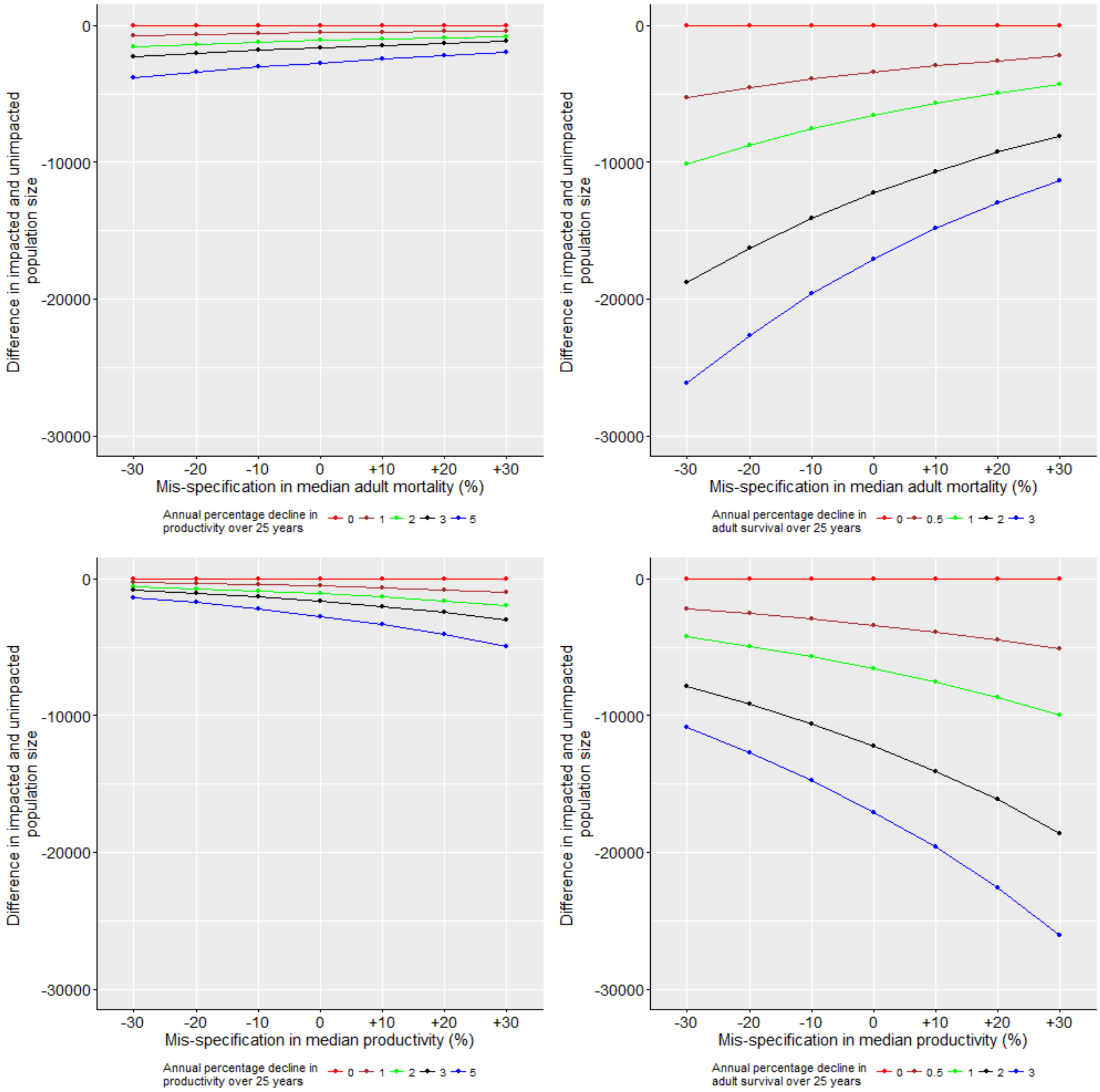


Figure A2.6e. PVA Metric E1 for St Abb’s Guillemots – probability of population decline greater than 10% from 2016-2041.

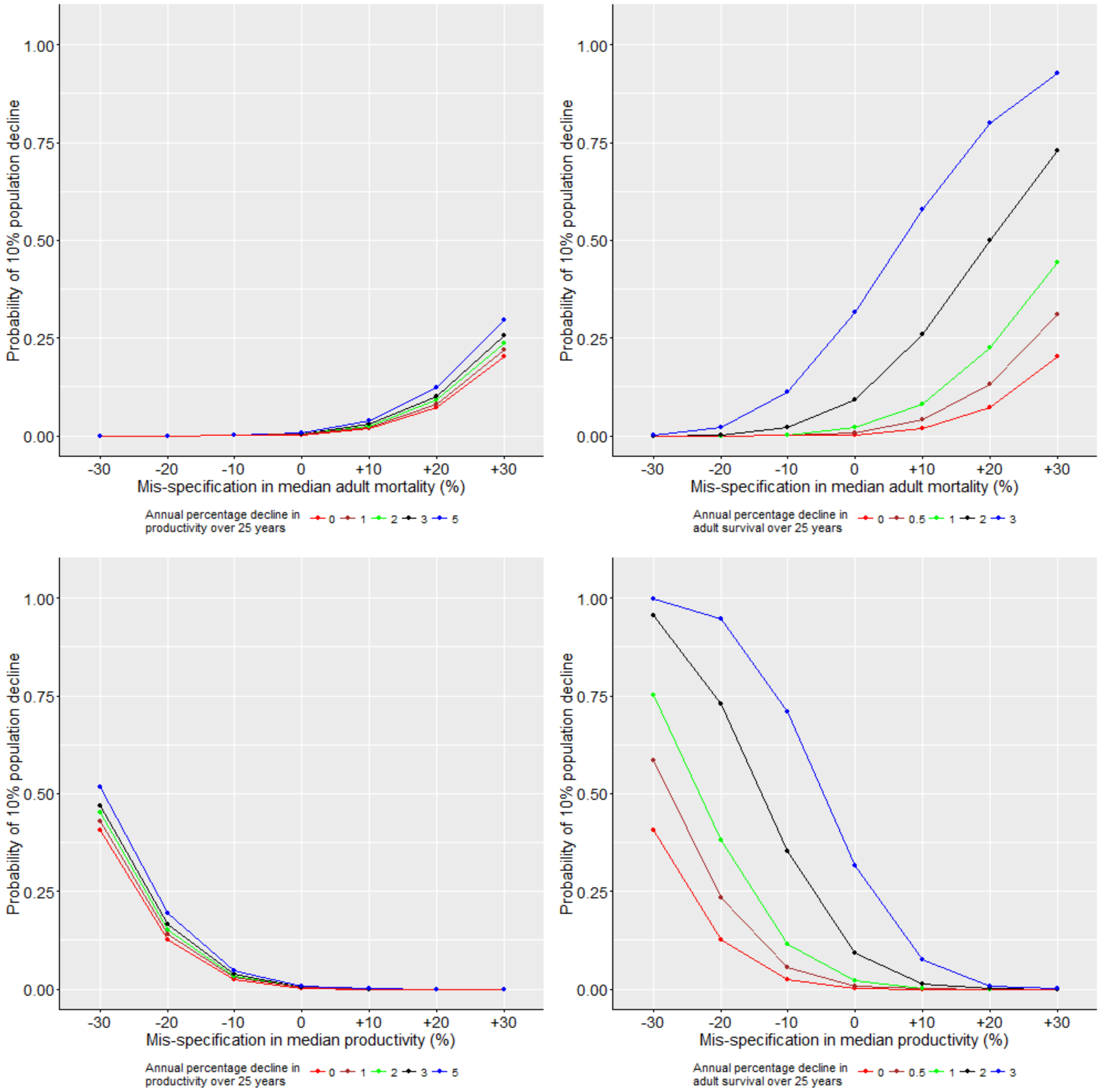


Figure A2.6f. PVA Metric E2 for St Abb's Guillemots – probability of population decline greater than 25% from 2016-2041.

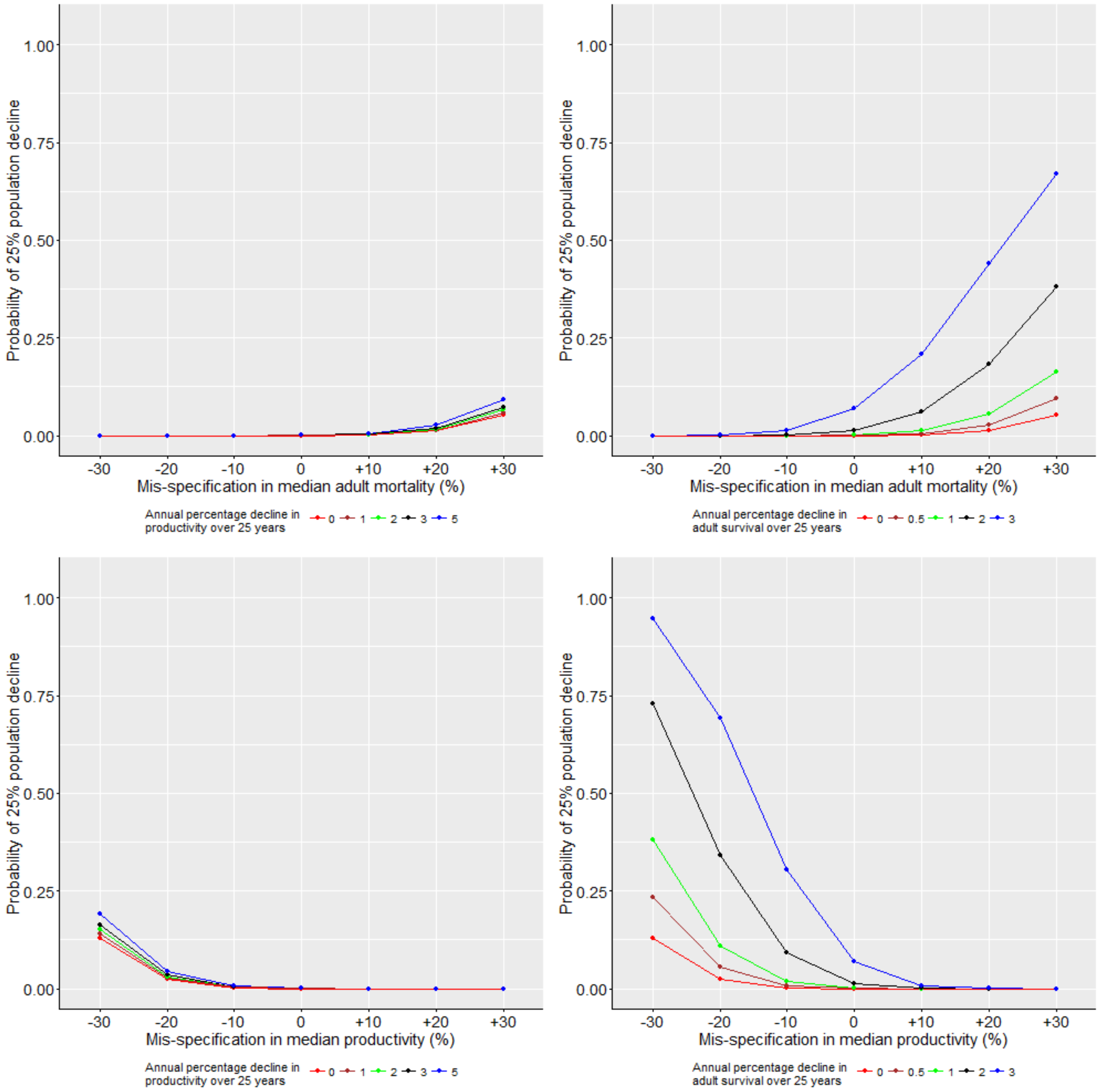


Figure A2.6g. PVA Metric E3 for St Abb’s Guillemots – probability of population decline greater than 50% from 2016-2041.

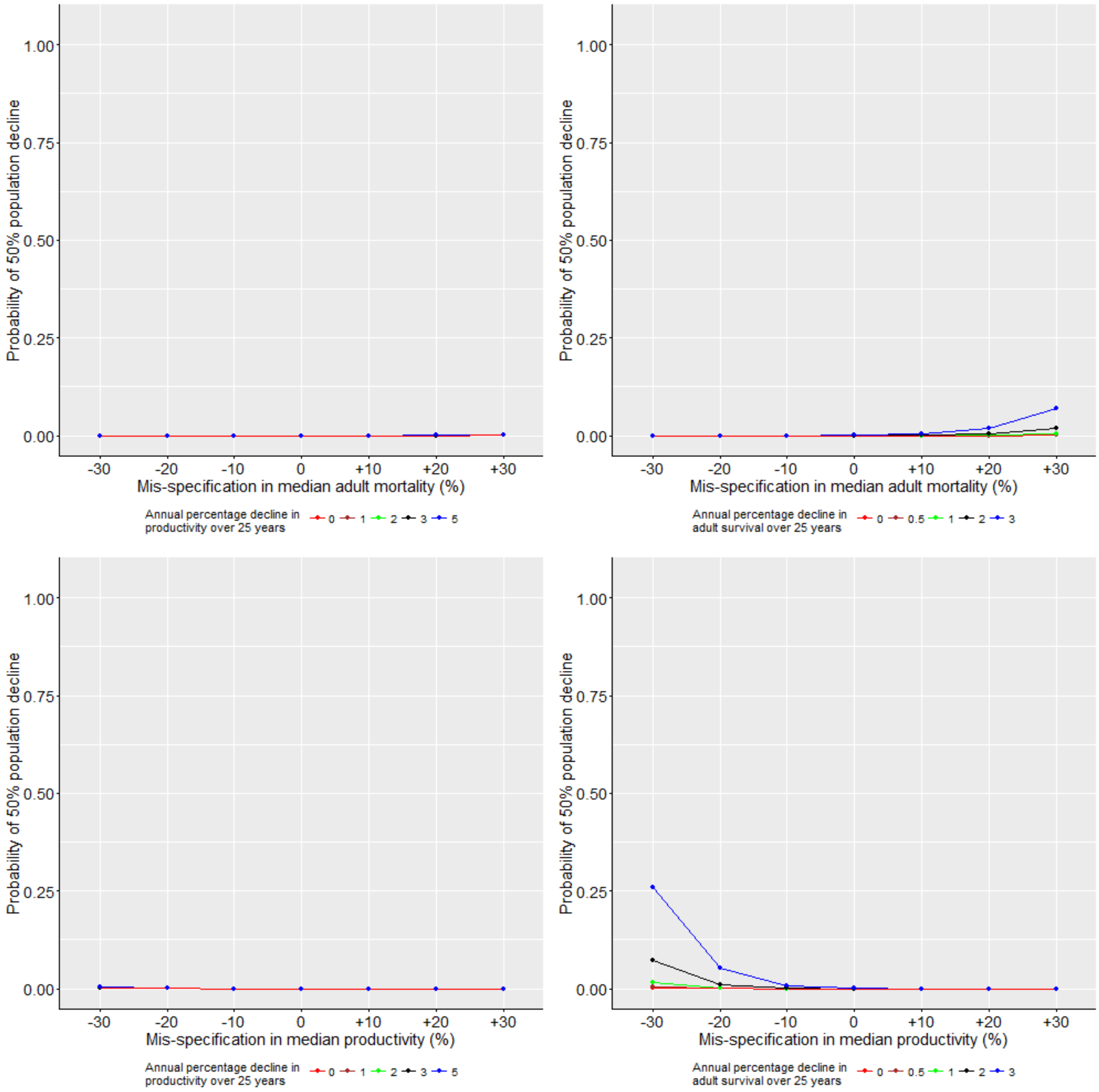
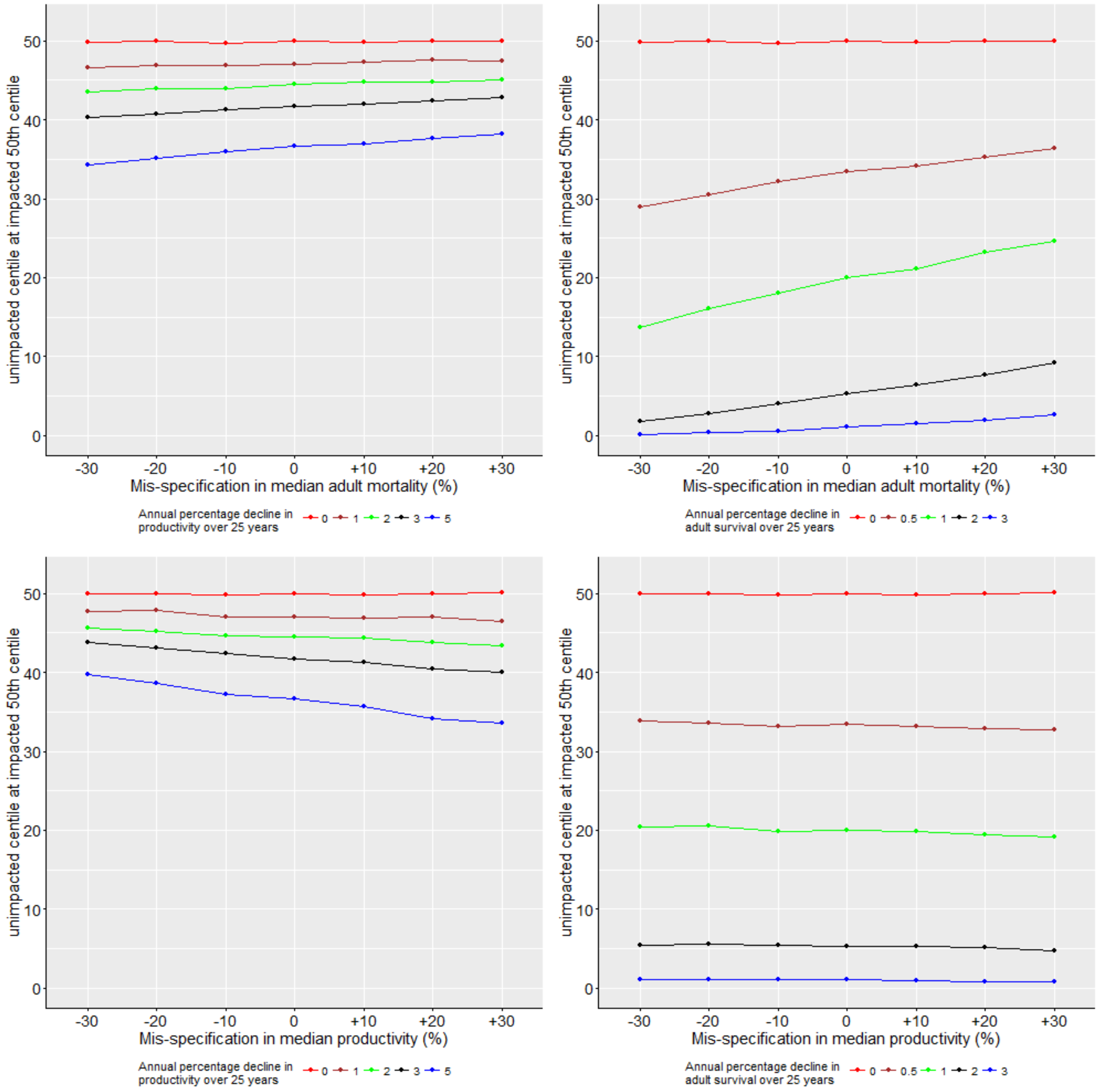


Figure A2.6h. PVA Metric F for St Abb’s Guillemots – centile from un-impacted population size equal to the 50th centile of the impacted population size, at 2041.



7. **Guillemots at Fowlsheugh SPA**:

Figure A2.7a. PVA Metric A for Fowlsheugh Guillemots – ratio of population growth rate from 2016-2041, comparing impacted population vs. un-impacted population.

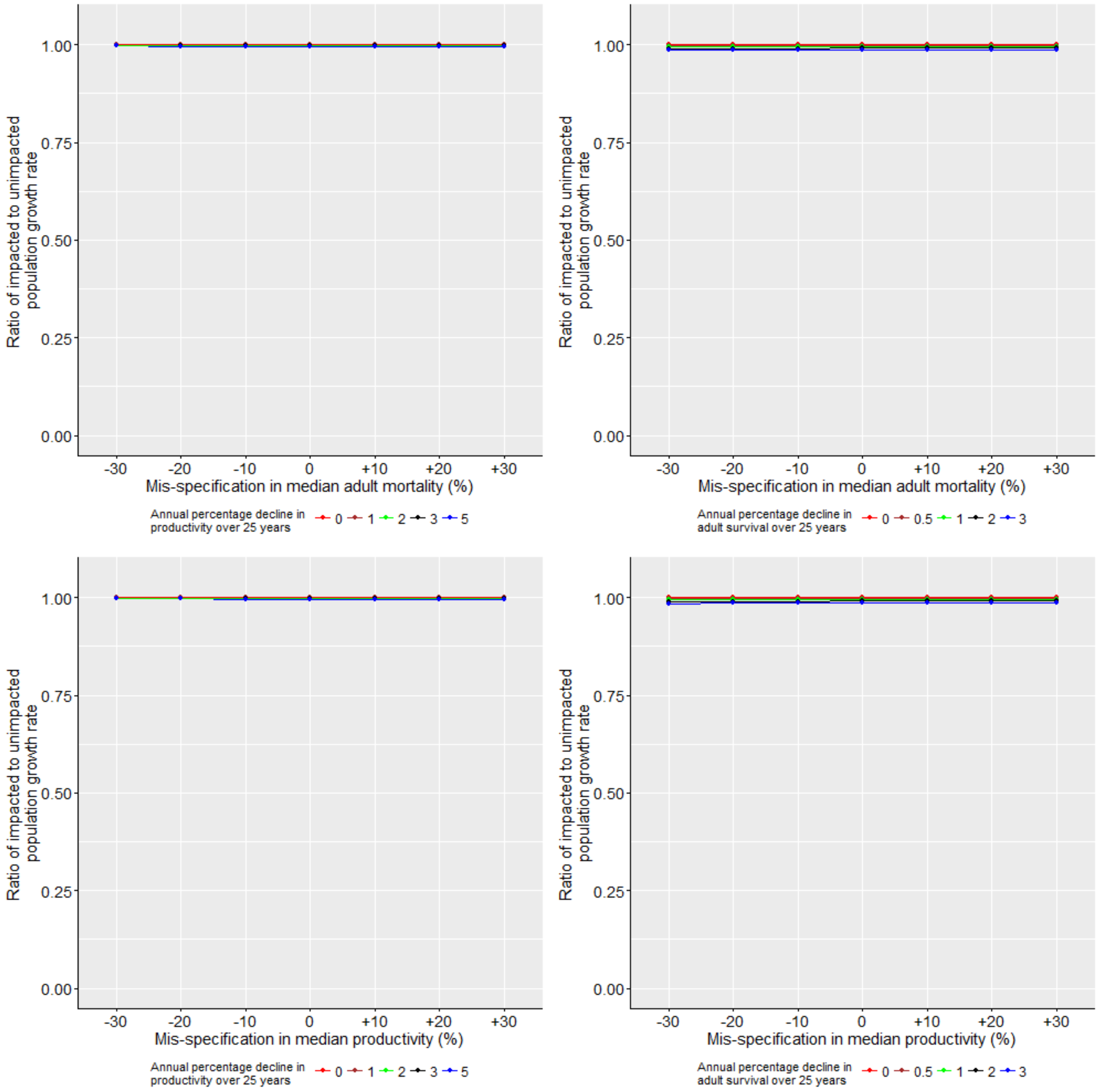


Figure A2.7b. PVA Metric B for Fowlsheugh Guillemots – ratio of population size at 2041, comparing impacted population vs. un-impacted population.

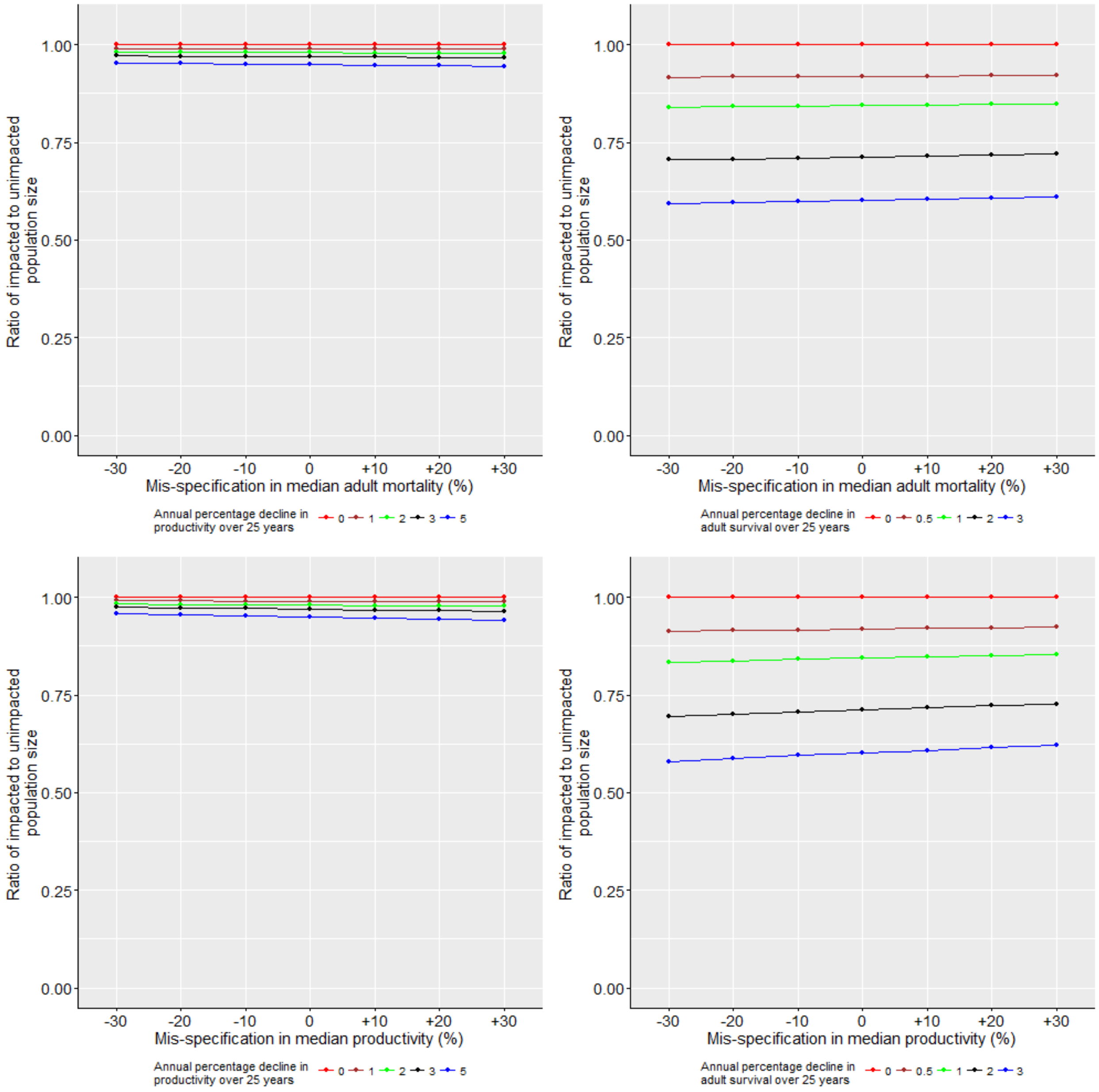


Figure A2.7c. PVA Metric C for Fowlsheugh Guillemots – difference in population growth rate from 2016-2041, comparing impacted population vs. un-impacted population.

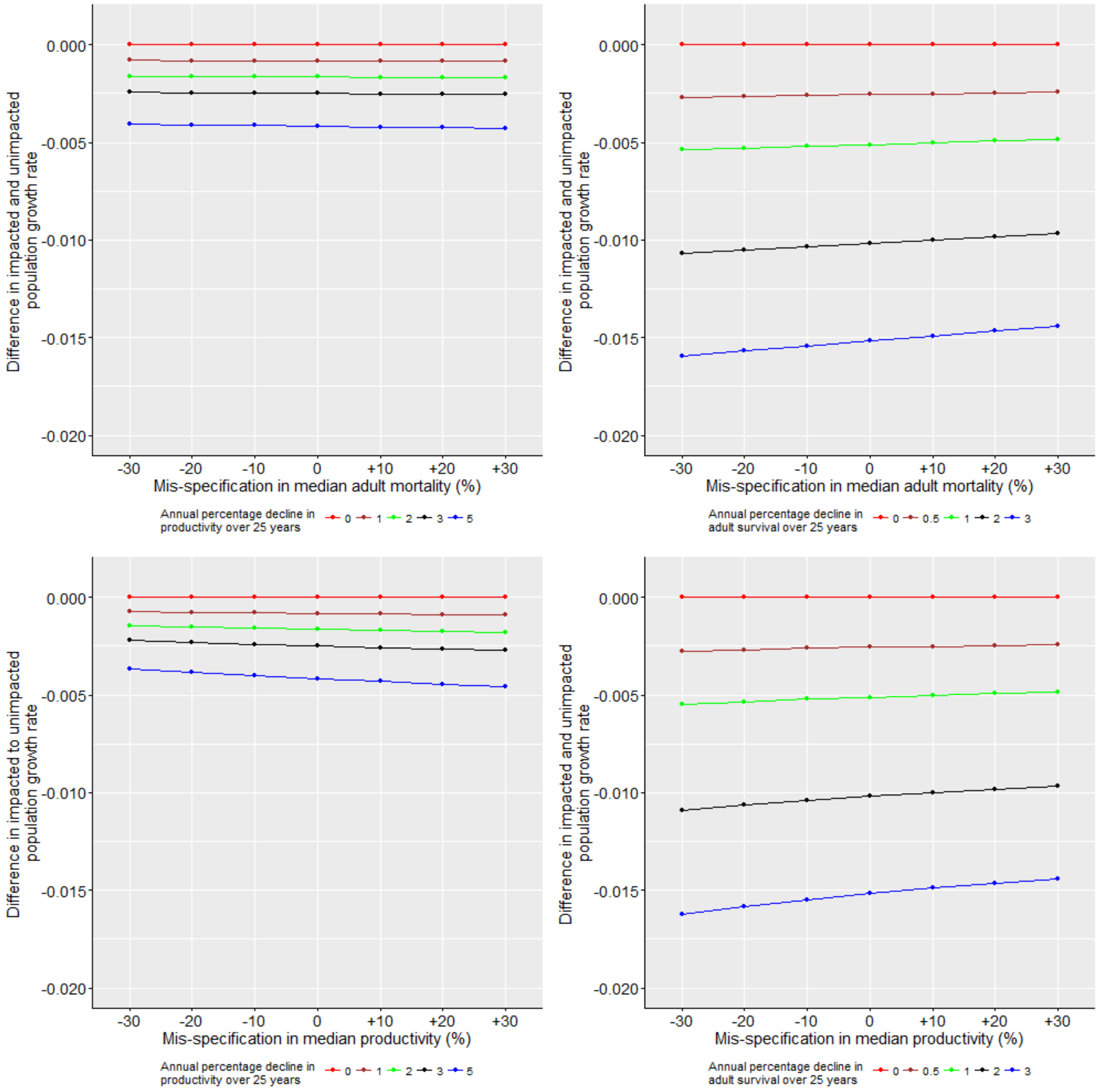


Figure A2.7d. PVA Metric D for Fowlsheugh Guillemots – difference in population size at 2041, comparing impacted population vs. un-impacted population.

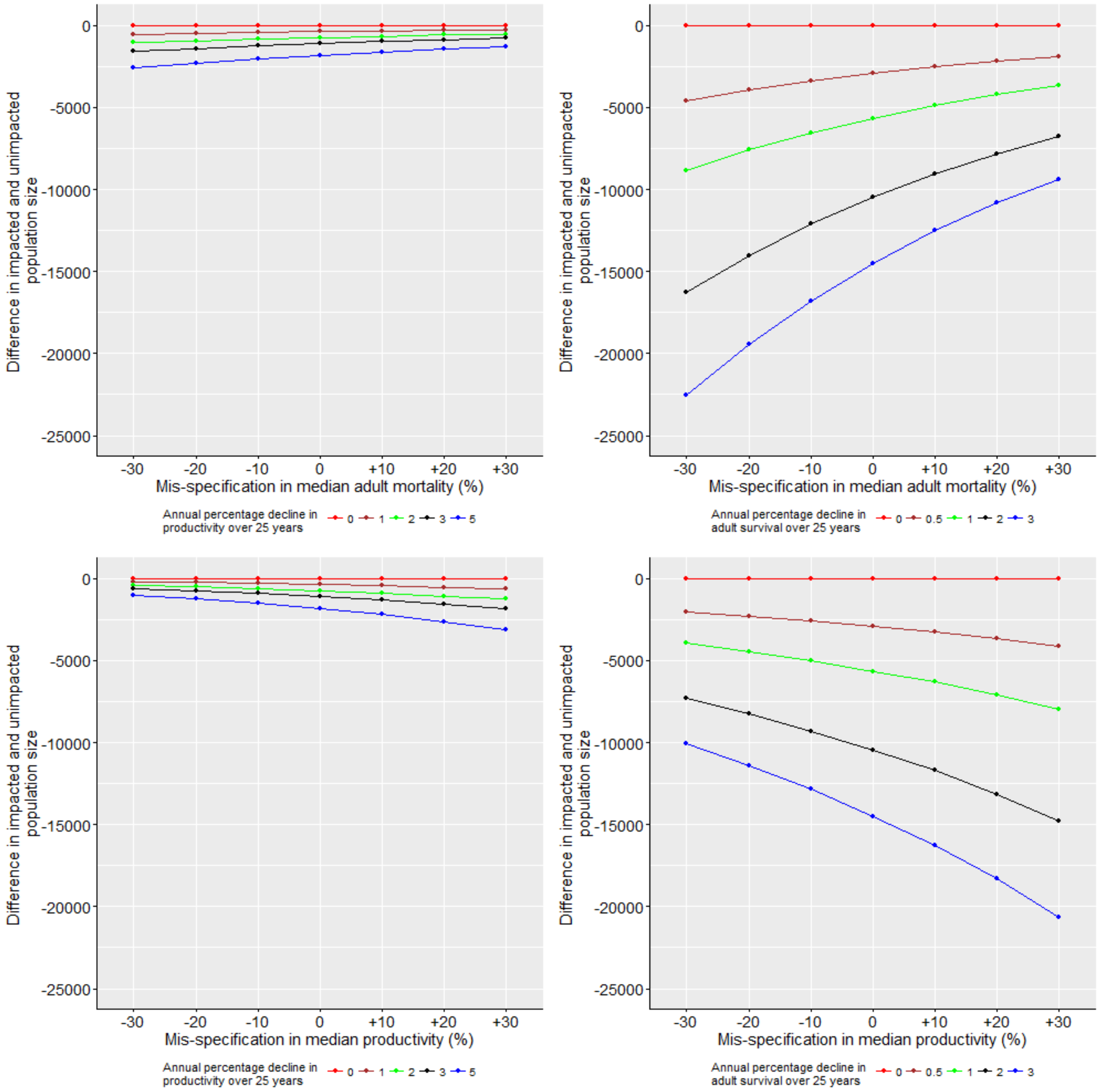


Figure A2.7e. PVA Metric E1 for Fowlsheugh Guillemots – probability of population decline greater than 10% from 2016-2041.

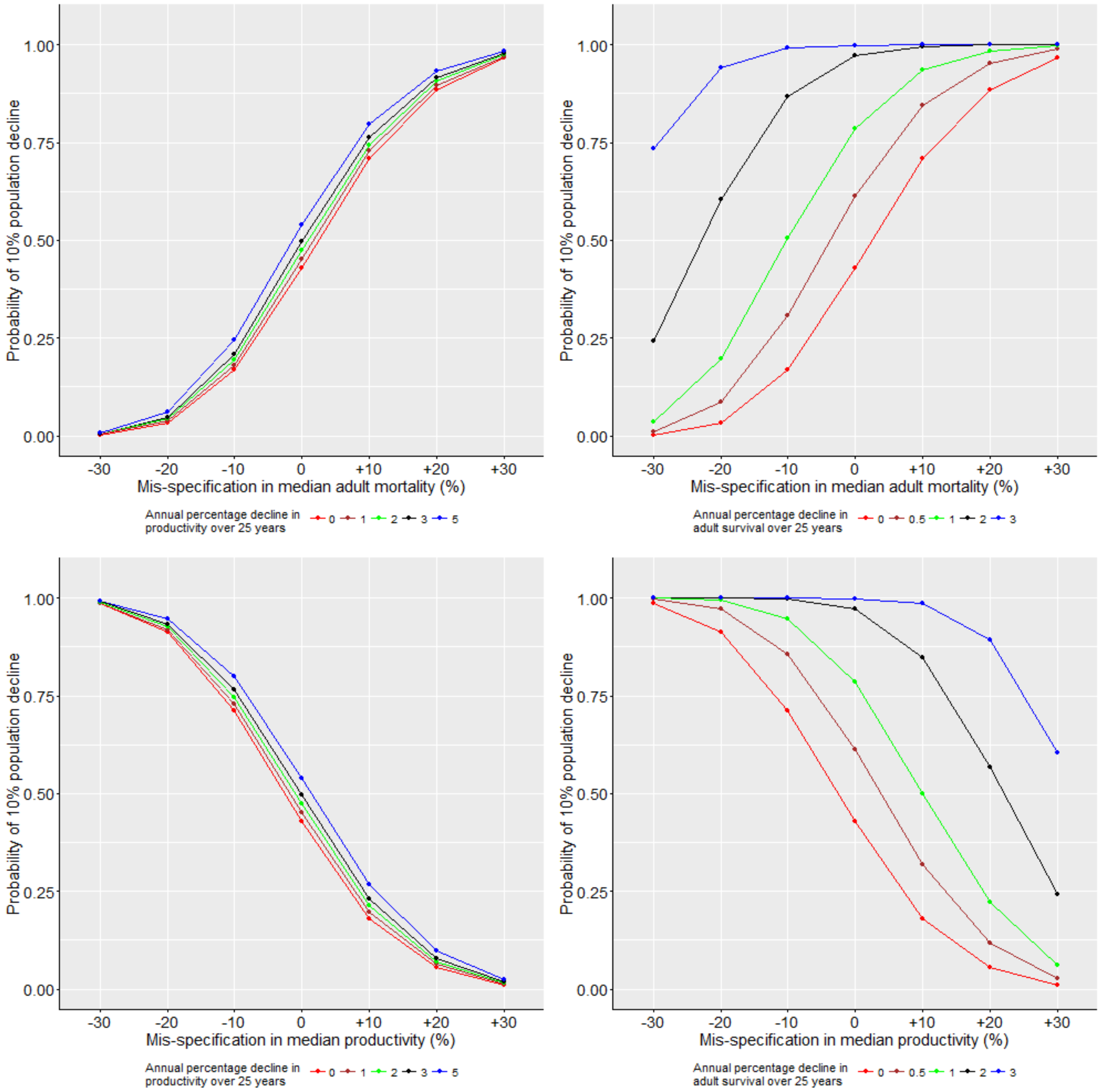


Figure A2.7f. PVA Metric E2 for Fowlsheugh Guillemots – probability of population decline greater than 25% from 2016-2041.

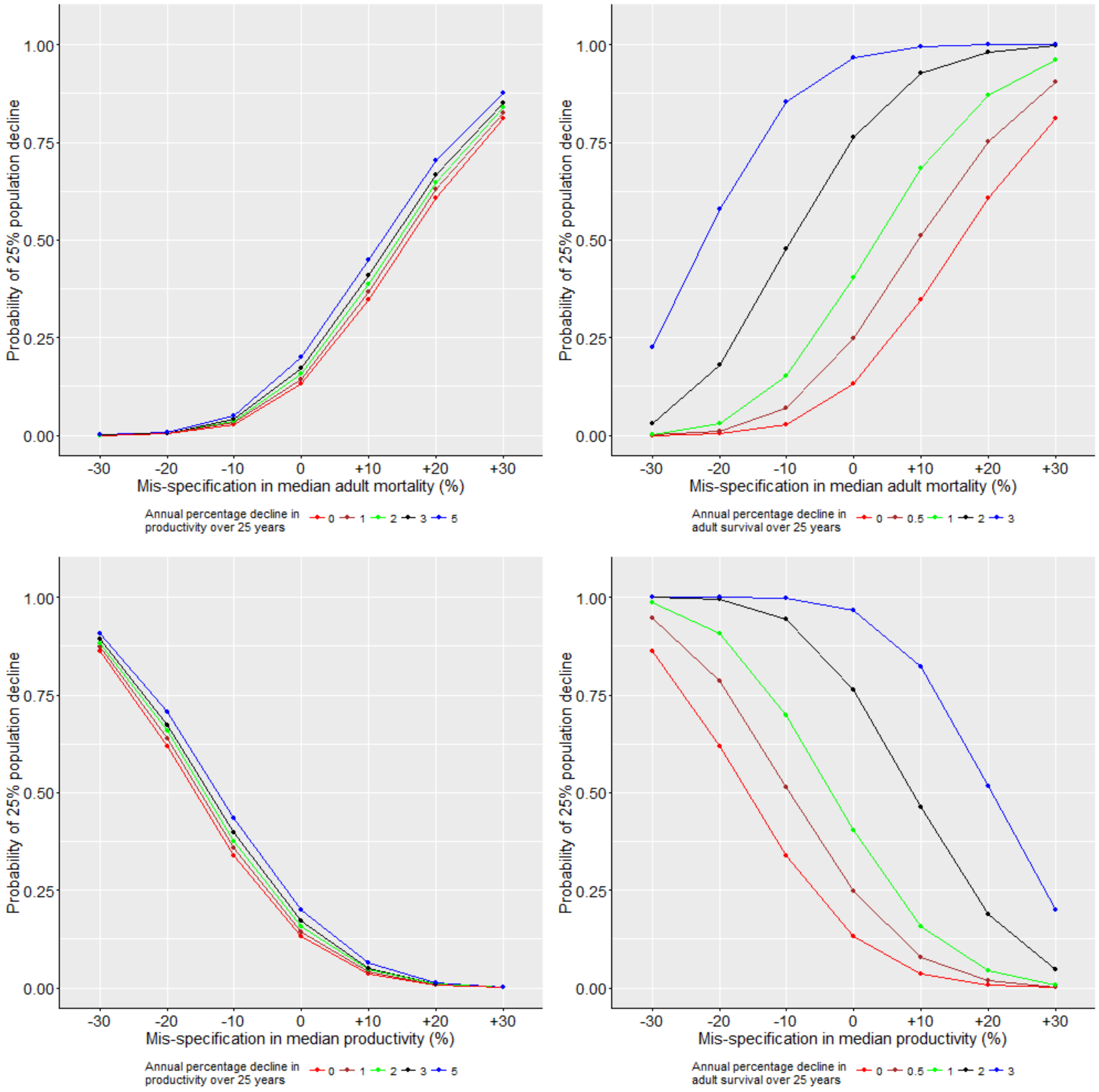


Figure A2.7g. PVA Metric E3 for Fowlsheugh Guillemots – probability of population decline greater than 50% from 2016-2041.

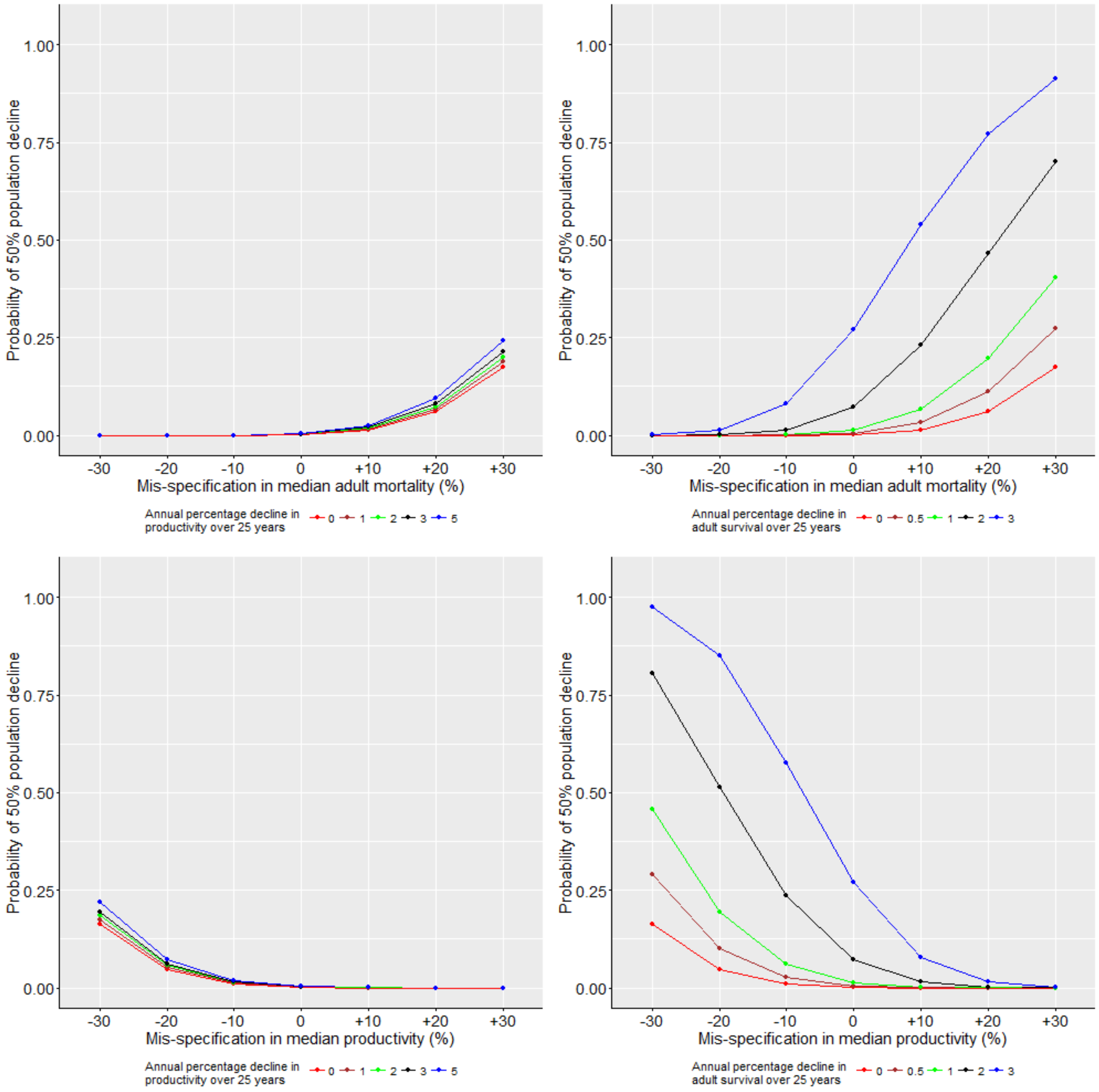
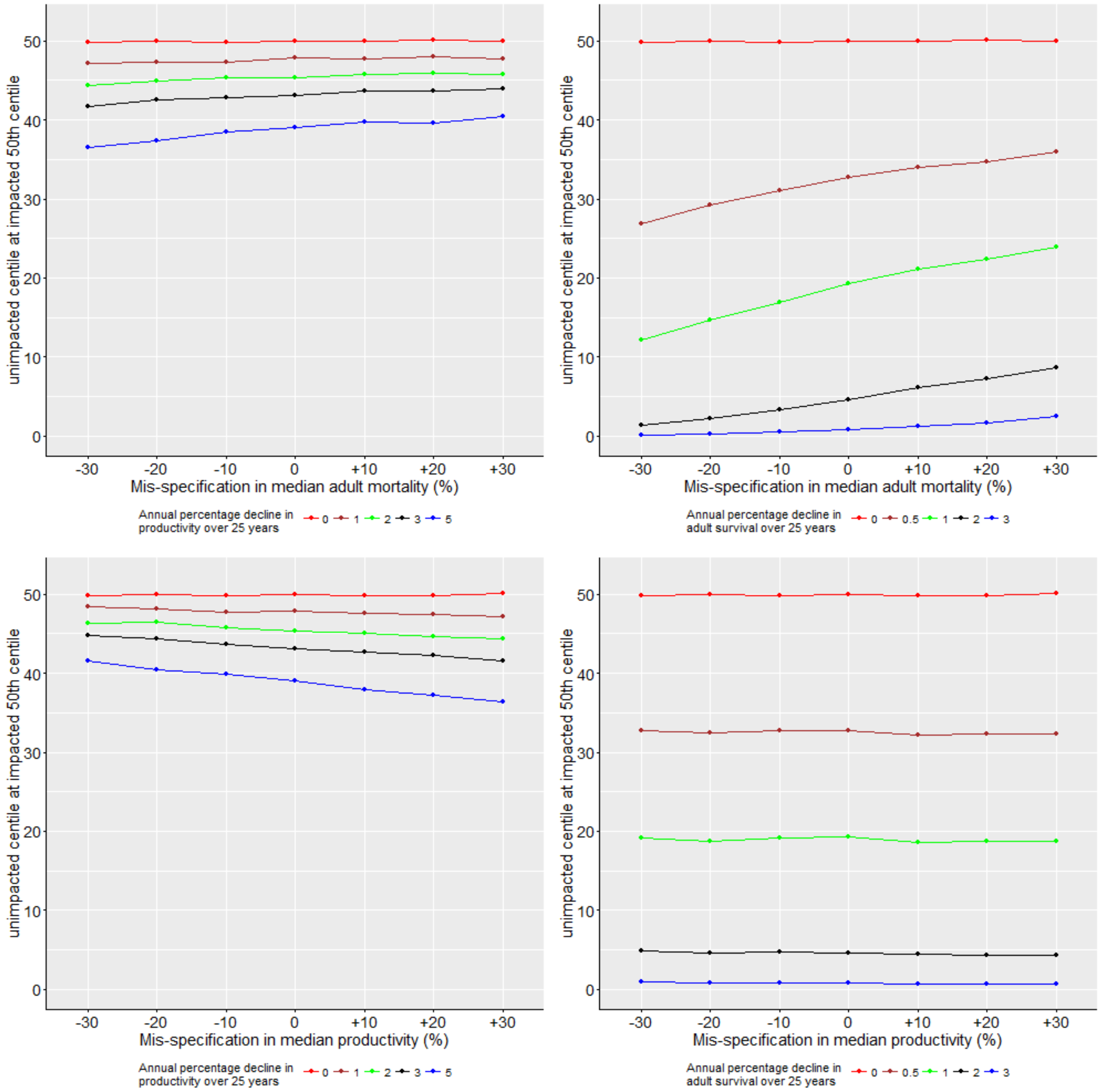


Figure A2.7h. PVA Metric F for Fowlsheugh Guillemots – centile from un-impacted population size equal to the 50th centile of the impacted population size, at 2041.



8. **Guillemots at Buchan Ness to Collieston Coast SPA**:

Figure A2.8a. PVA Metric A for Buchan Ness Guillemots – ratio of population growth rate from 2016-2041, comparing impacted population vs. un-impacted population.

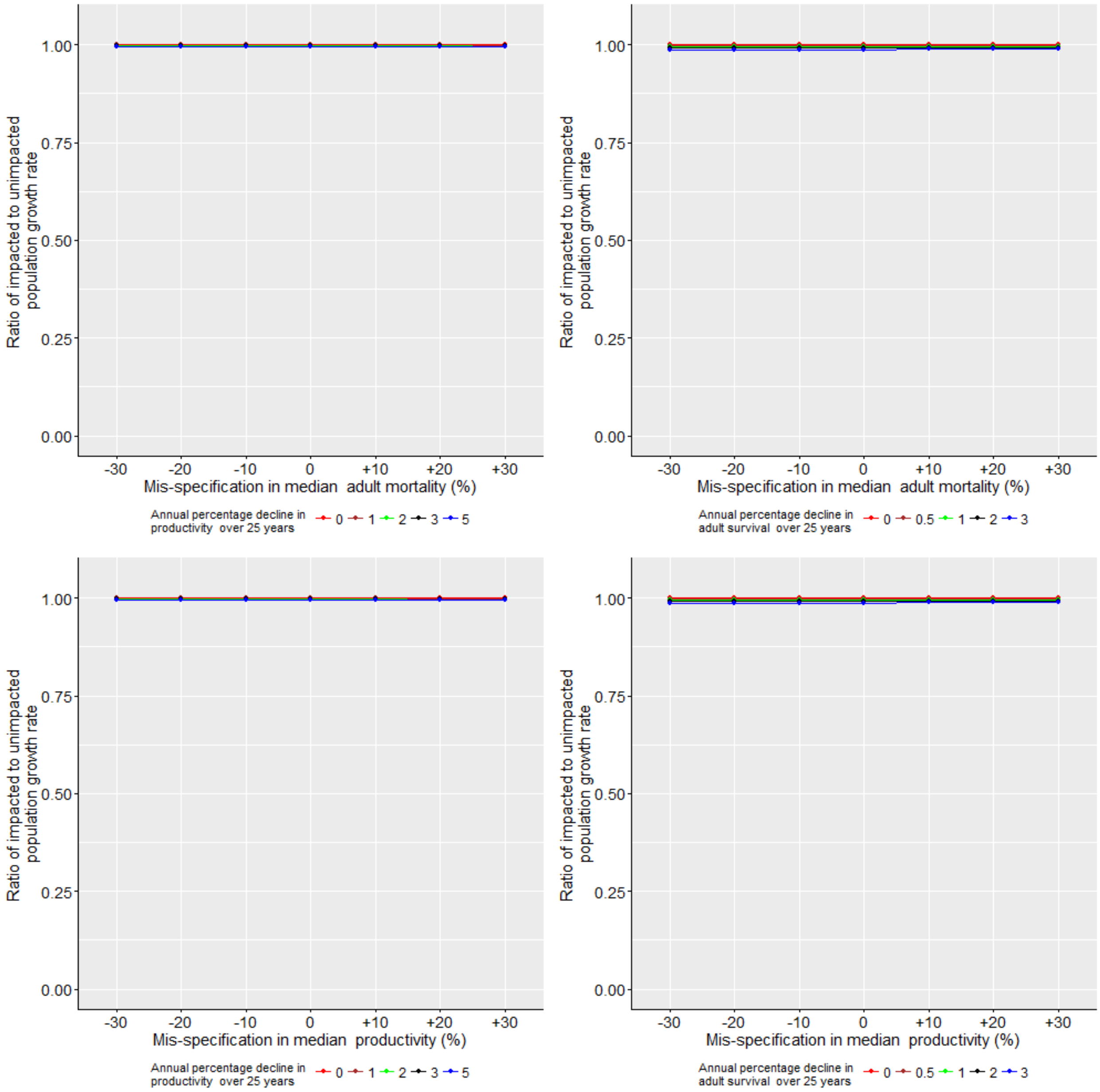


Figure A2.8b. PVA Metric B for Buchan Ness Guillemots – ratio of population size at 2041, comparing impacted population vs. un-impacted population.

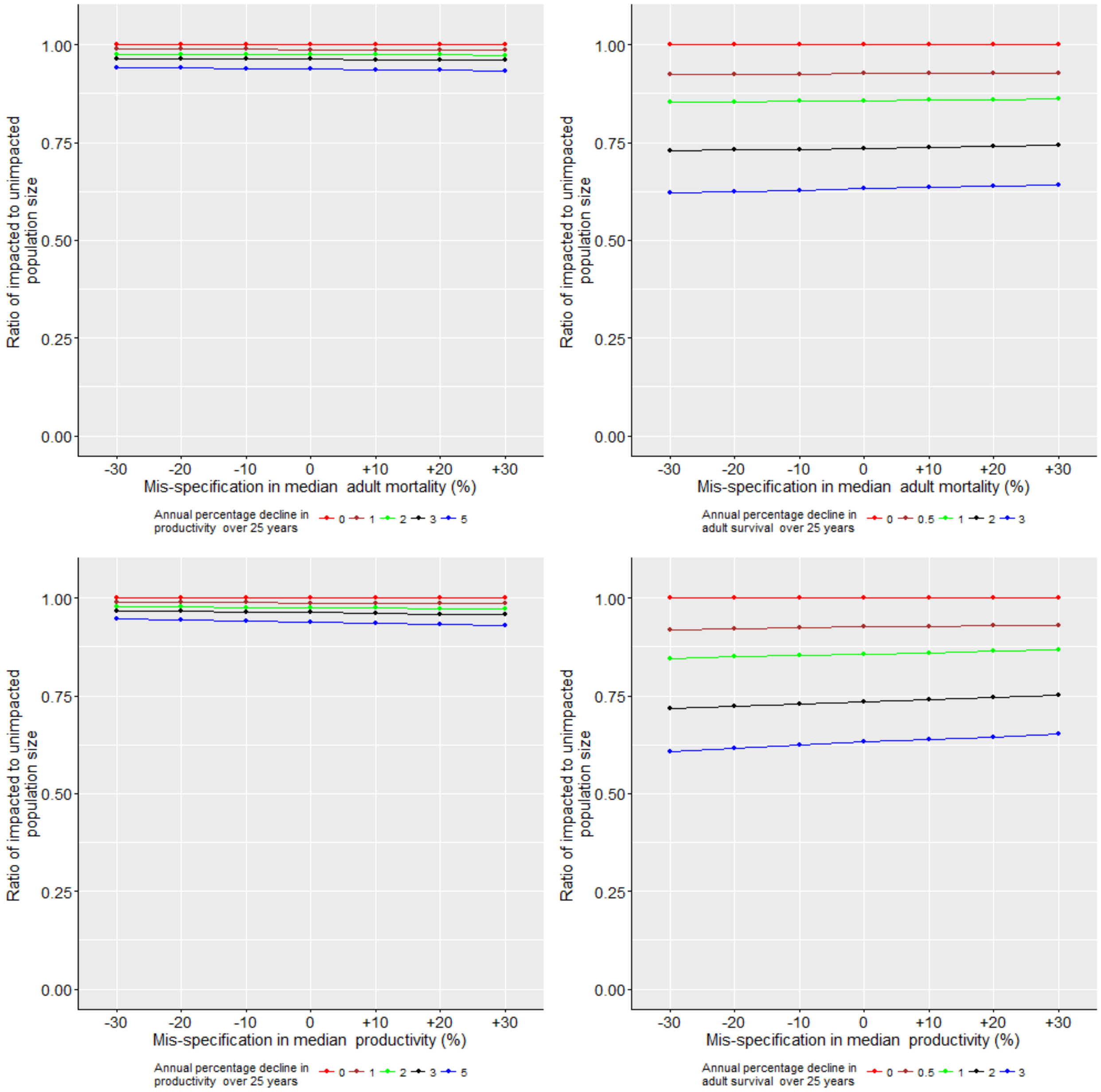


Figure A2.8c. PVA Metric C for Buchan Ness Guillemots – difference in population growth rate from 2016-2041, comparing impacted population vs. un-impacted population.

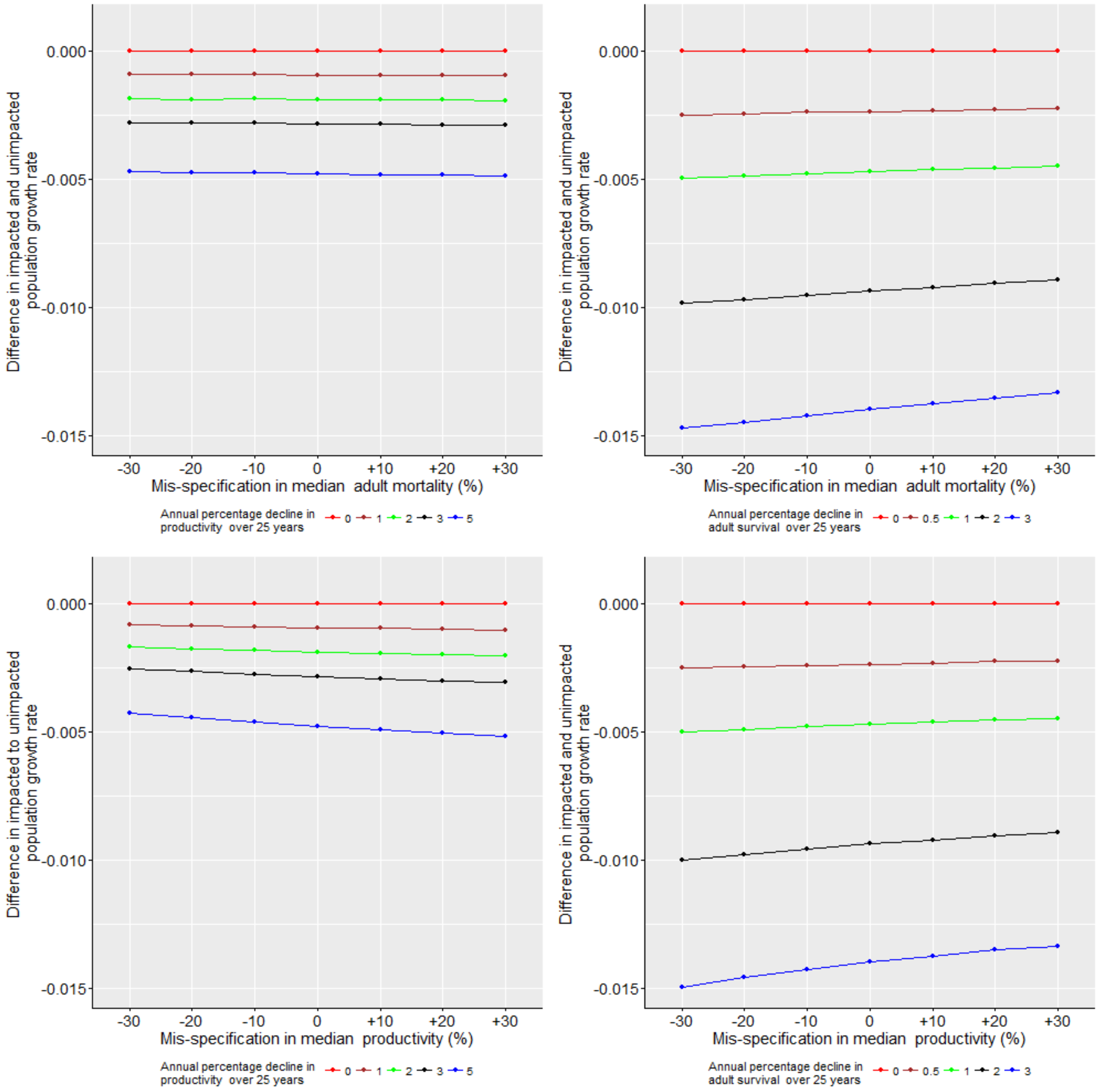


Figure A2.8d. PVA Metric D for Buchan Ness Guillemots – difference in population size at 2041, comparing impacted population vs. un-impacted population.

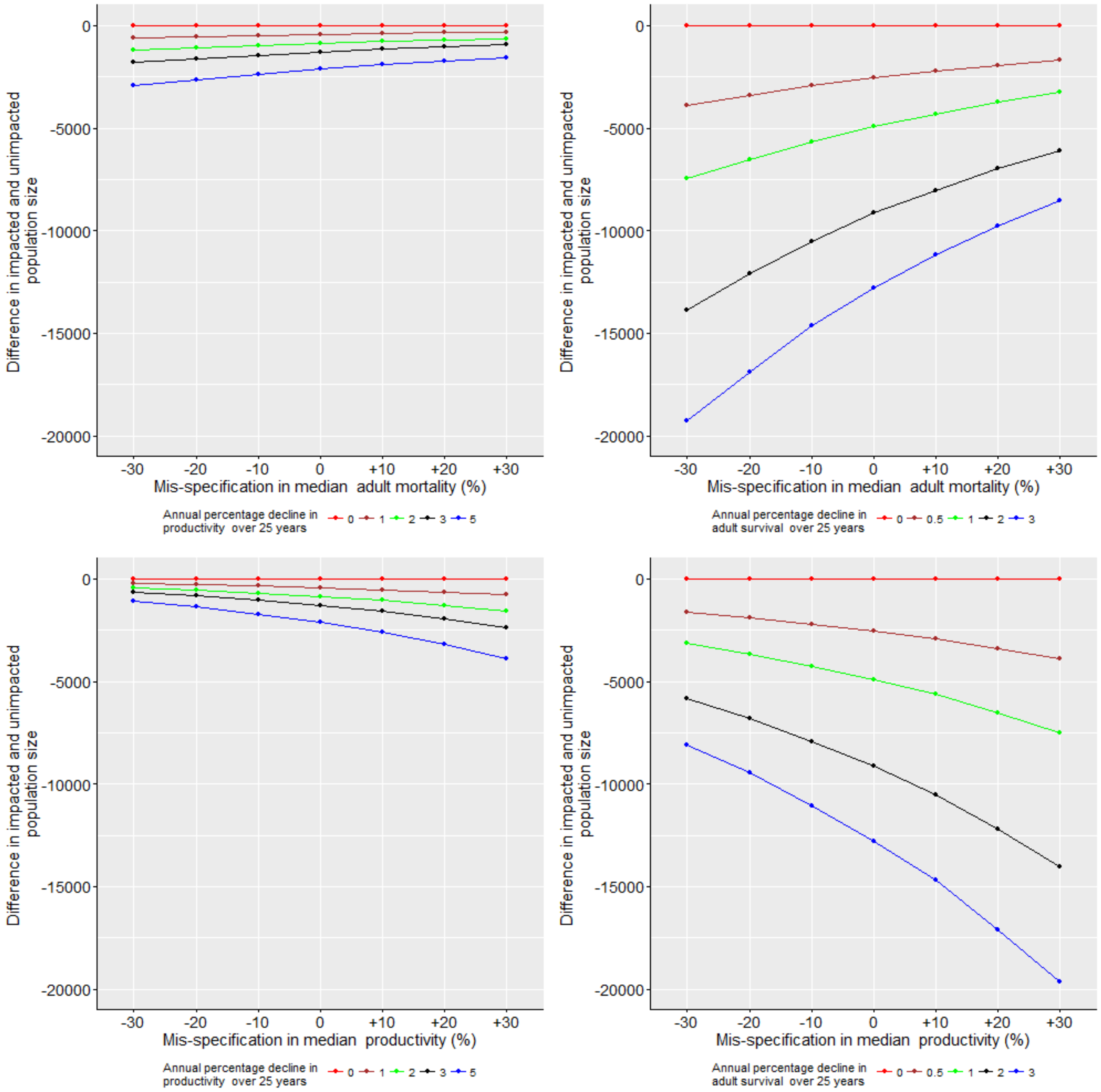


Figure A2.8e. PVA Metric E1 for Buchan Ness Guillemots – probability of population decline greater than 10% from 2016-2041.

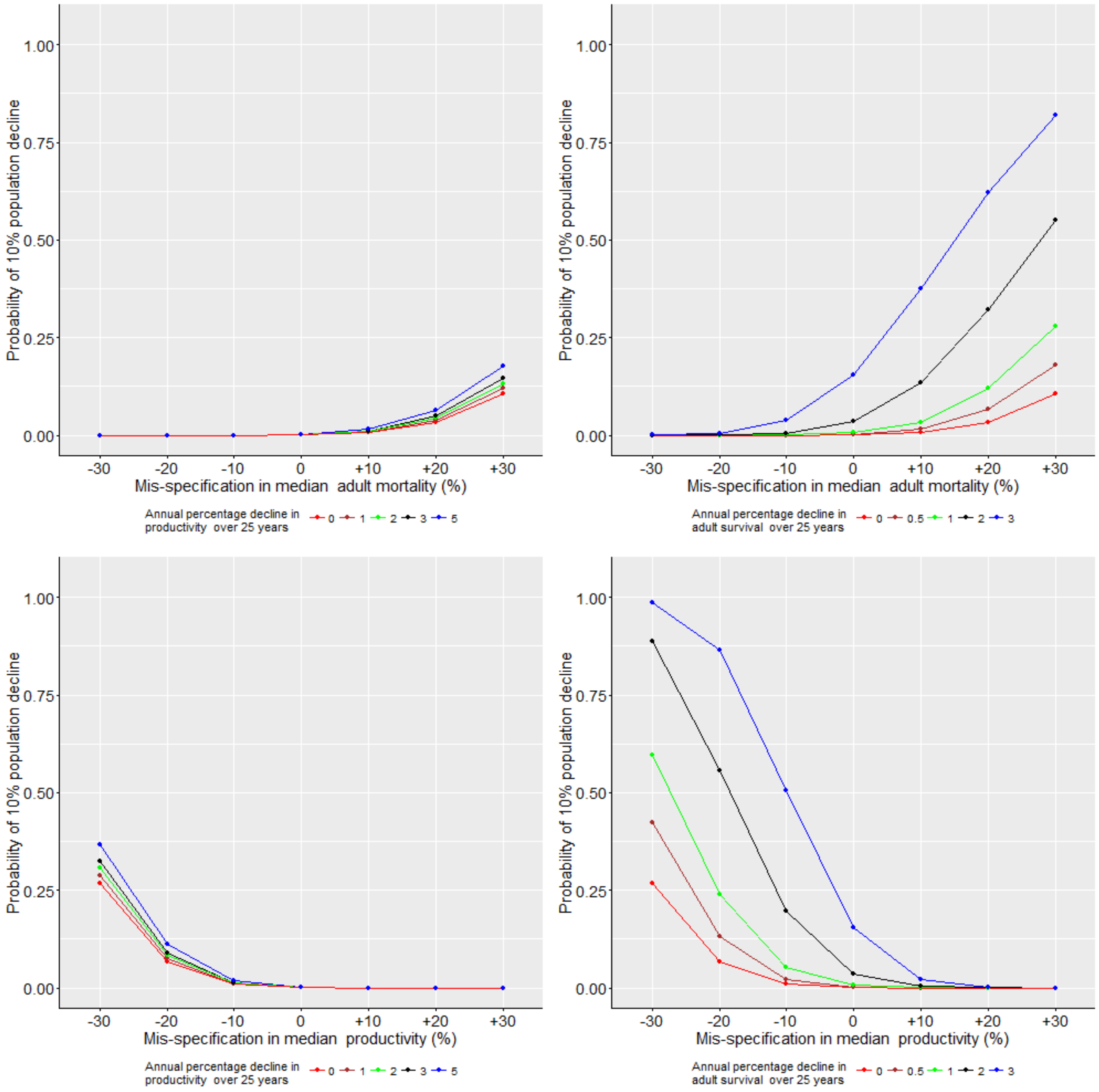


Figure A2.8f. PVA Metric E2 for Buchan Ness Guillemots – probability of population decline greater than 25% from 2016-2041.

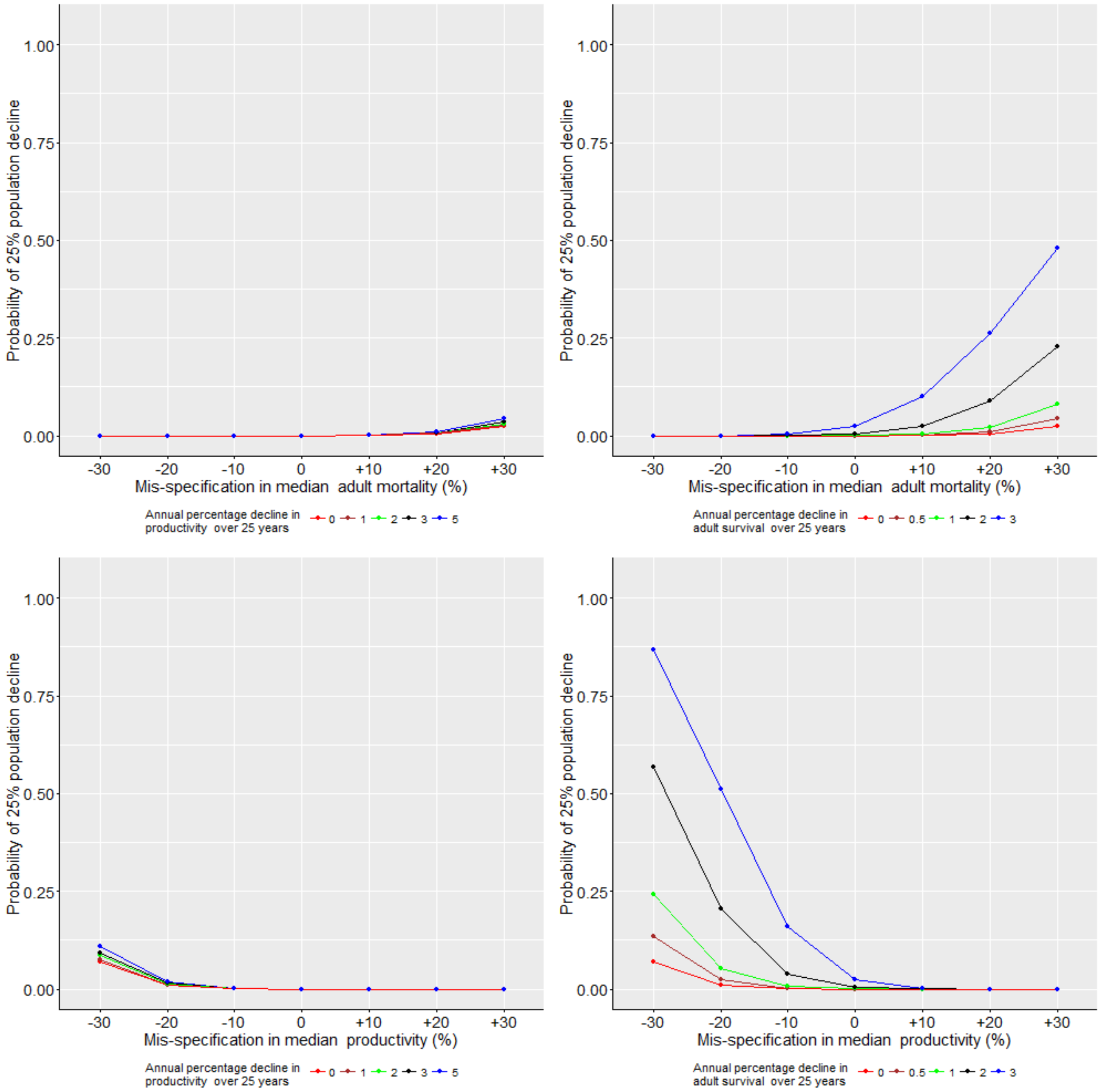


Figure A2.8g. PVA Metric E3 for Buchan Ness Guillemots – probability of population decline greater than 50% from 2016-2041.

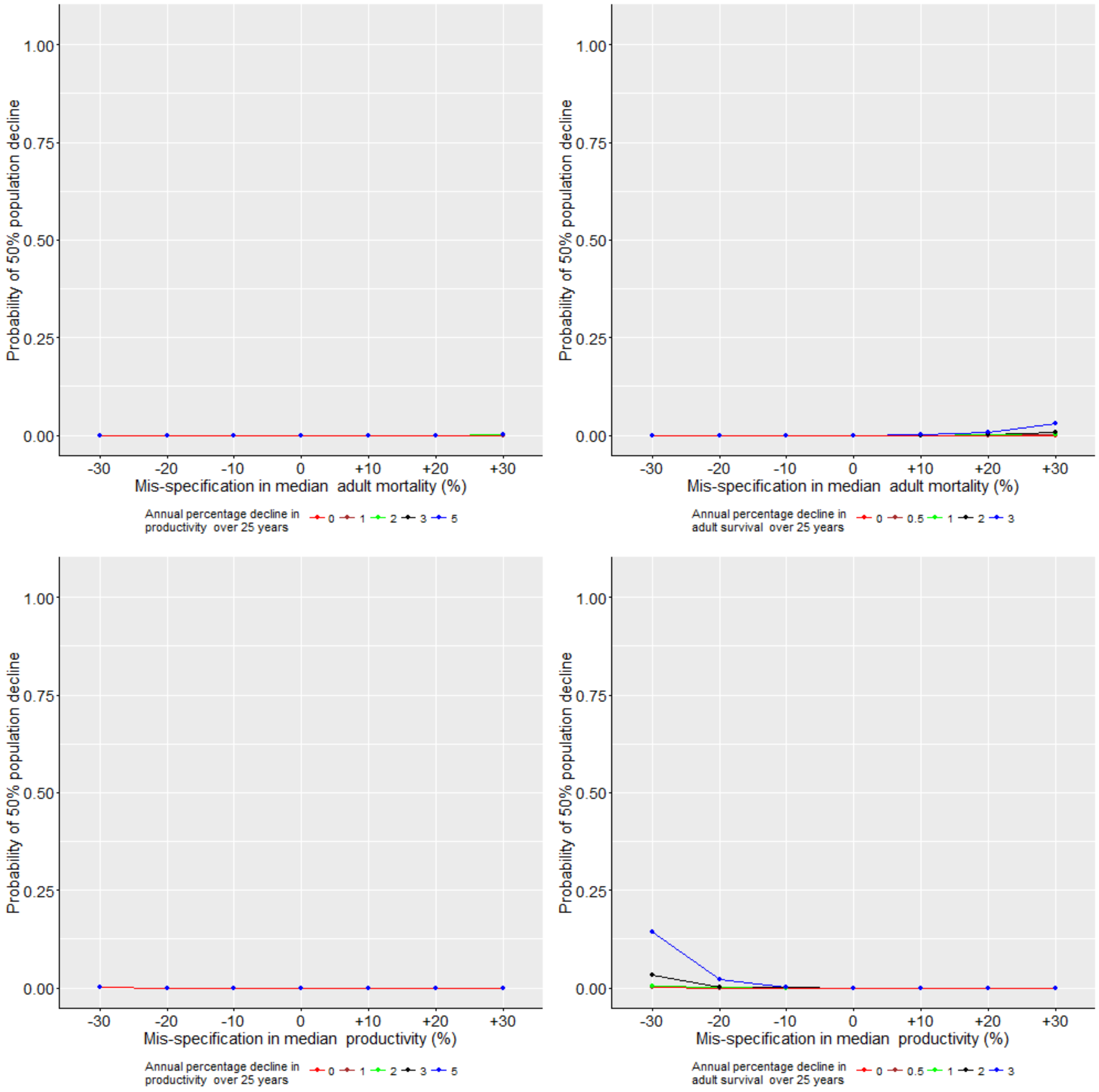
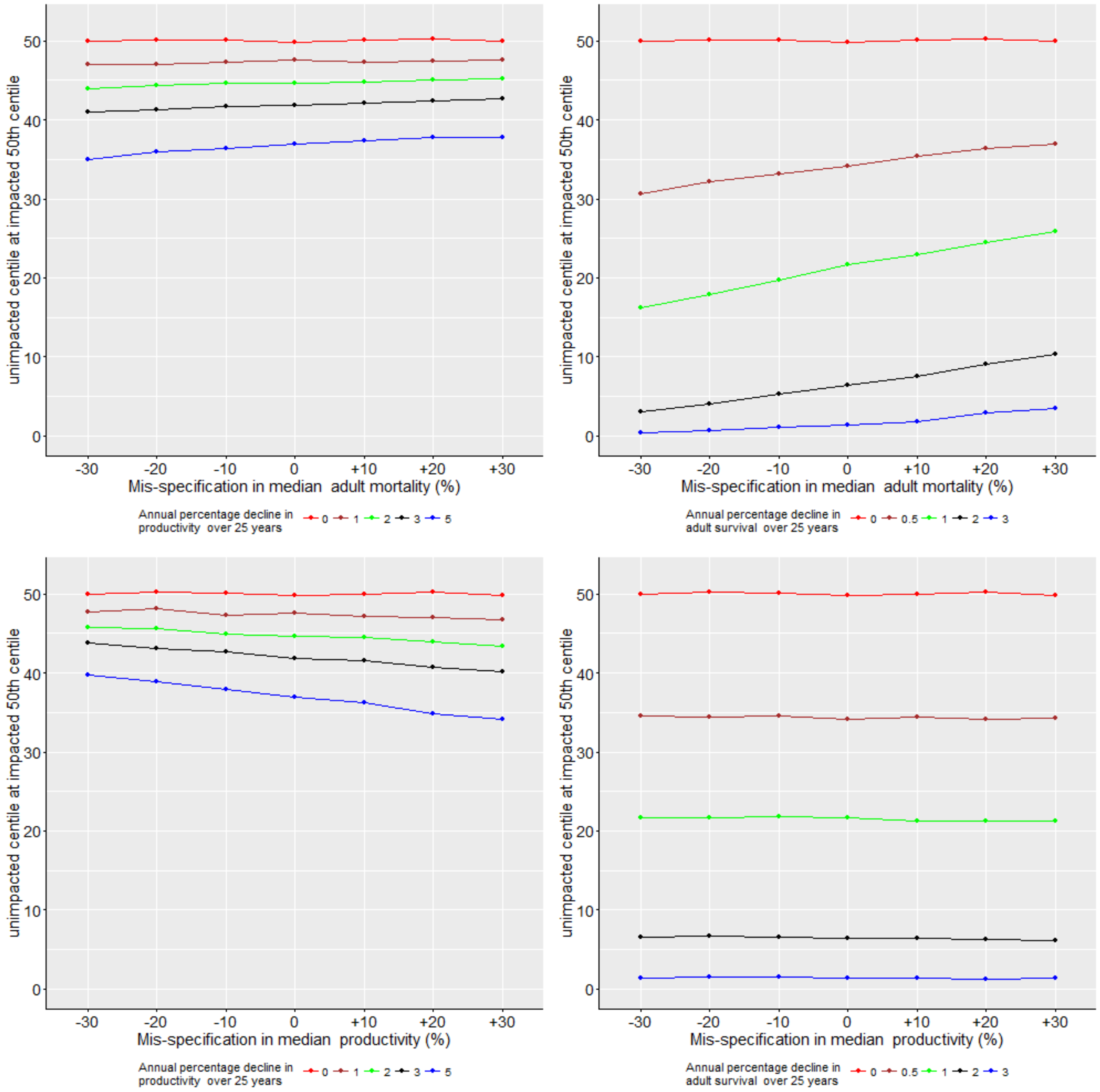


Figure A2.8h. PVA Metric F for Buchan Ness Guillemots – centile from un-impacted population size equal to the 50th centile of the impacted population size, at 2041.



9. **Razorbills at Forth Islands SPA**:

Figure A2.9a. PVA Metric A for Forth Razorbills – ratio of population growth rate from 2016-2041, comparing impacted population vs. un-impacted population.

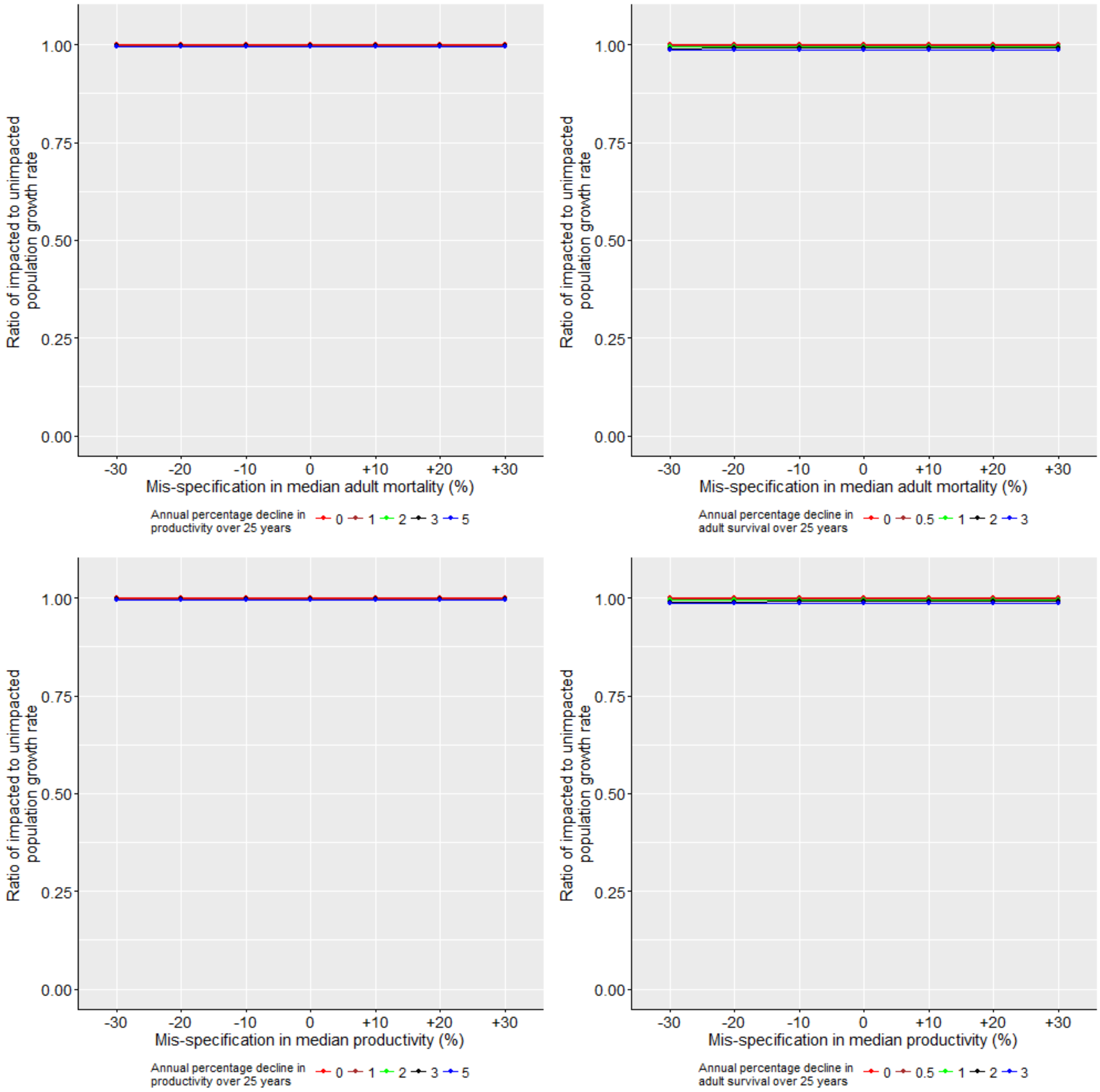


Figure A2.9b. PVA Metric B for Forth Razorbills – ratio of population size at 2041, comparing impacted population vs. un-impacted population.

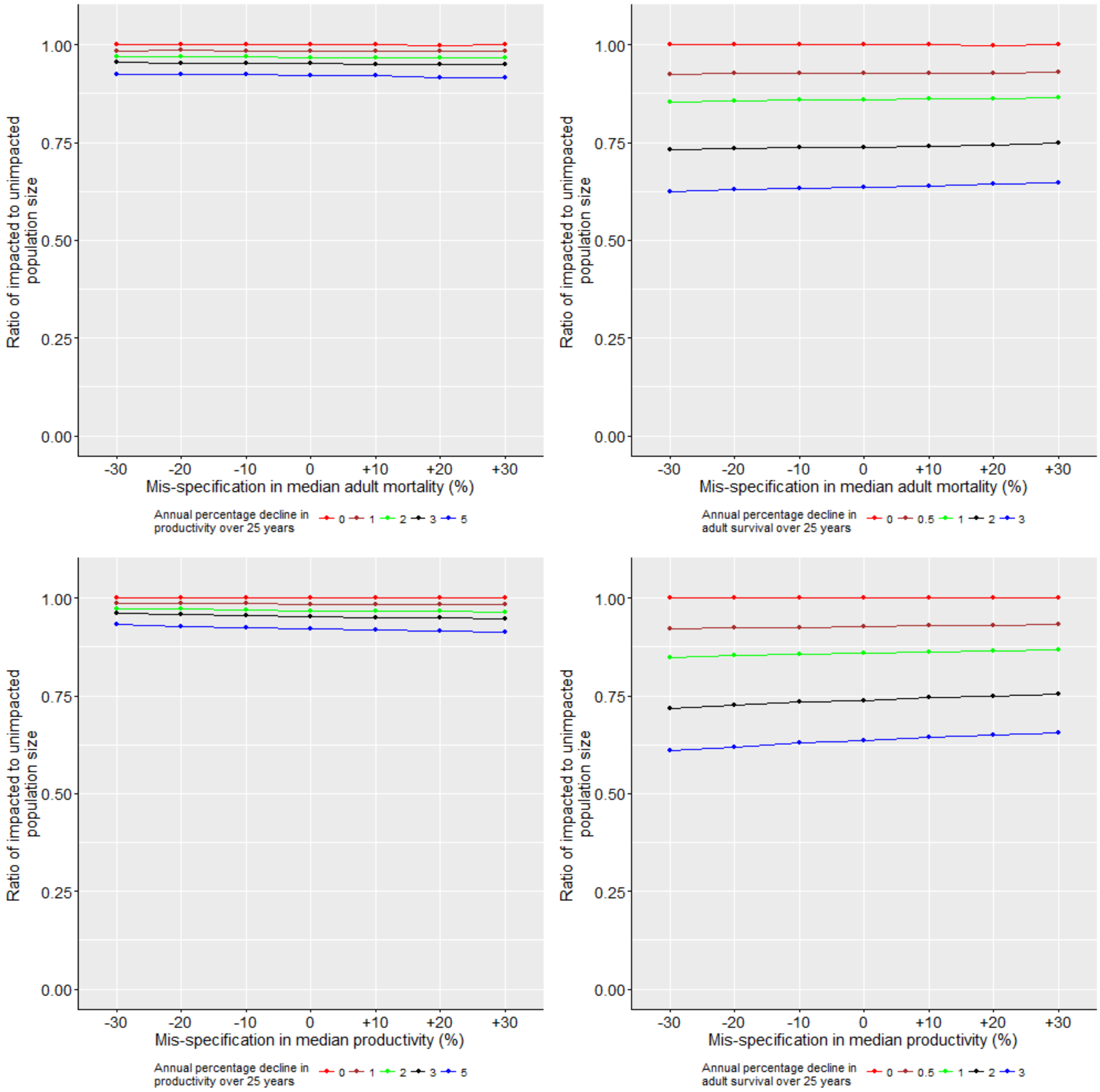


Figure A2.9c. PVA Metric C for Forth Razorbills – difference in population growth rate from 2016-2041, comparing impacted population vs. un-impacted population.

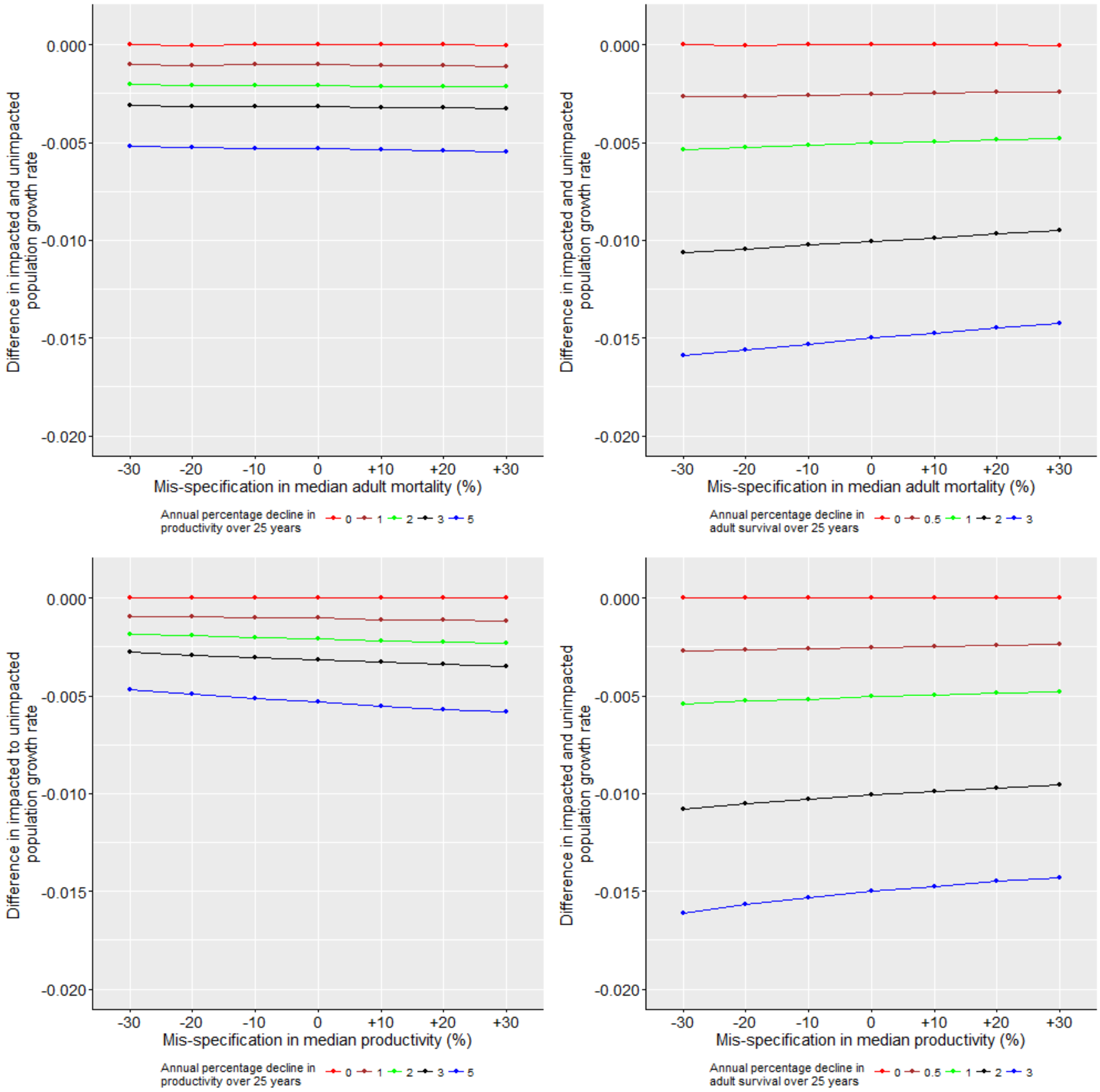


Figure A2.9d. PVA Metric D for Forth Razorbills – difference in population size at 2041, comparing impacted population vs. un-impacted population.

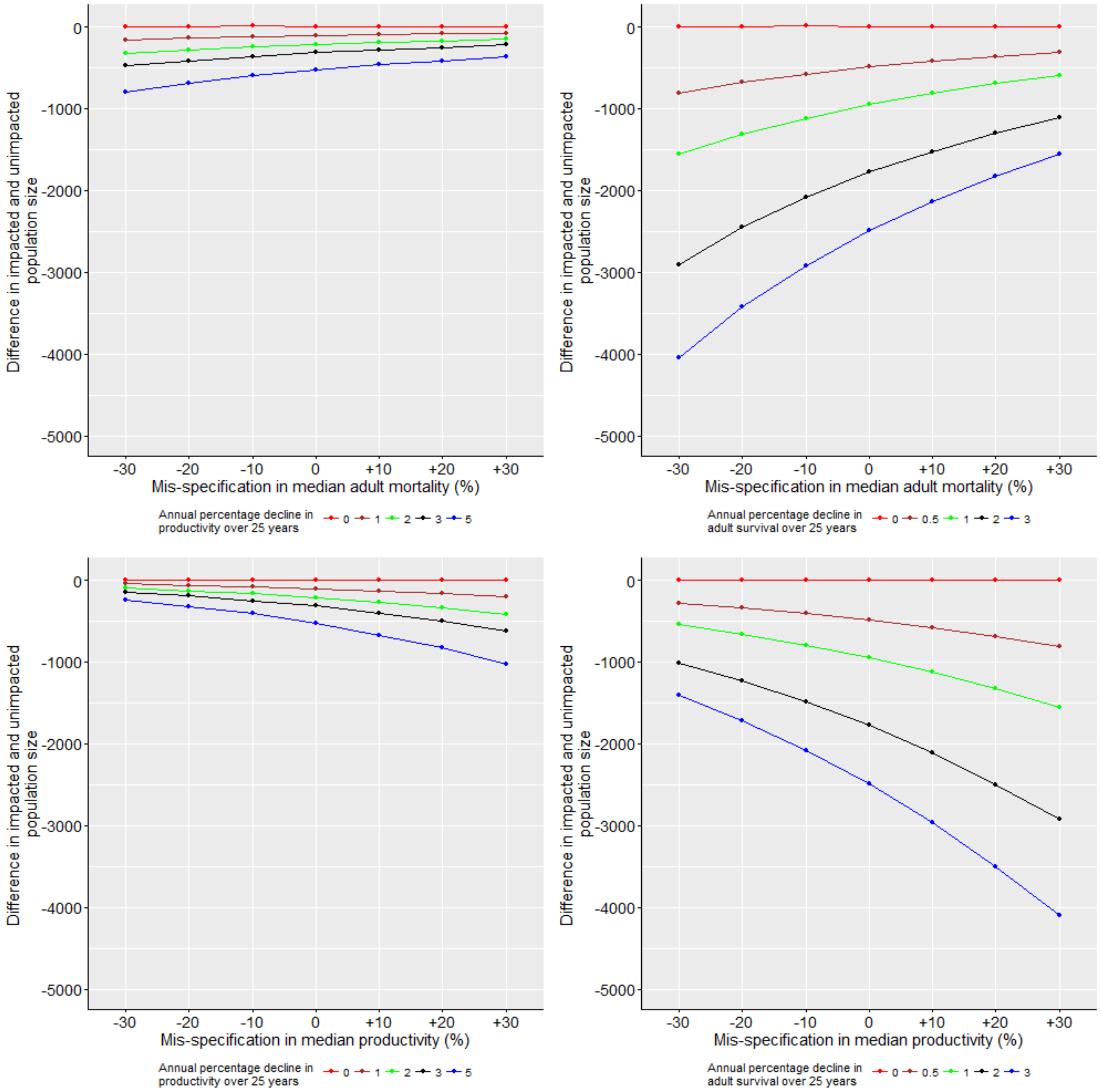


Figure A2.9e. PVA Metric E1 for Forth Razorbills – probability of population decline greater than 10% from 2016-2041.

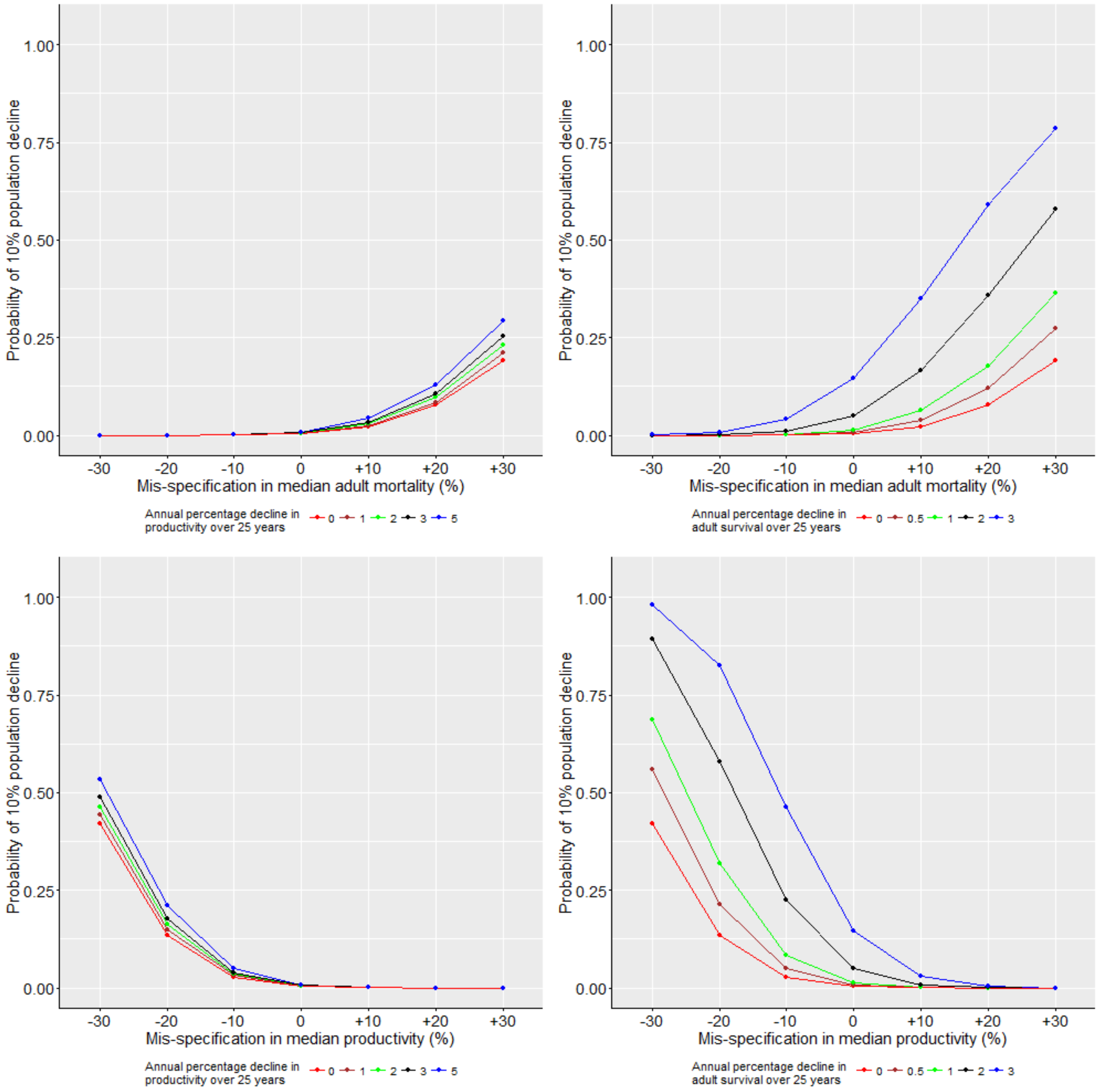


Figure A2.9f. PVA Metric E2 for Forth Razorbills – probability of population decline greater than 25% from 2016-2041.

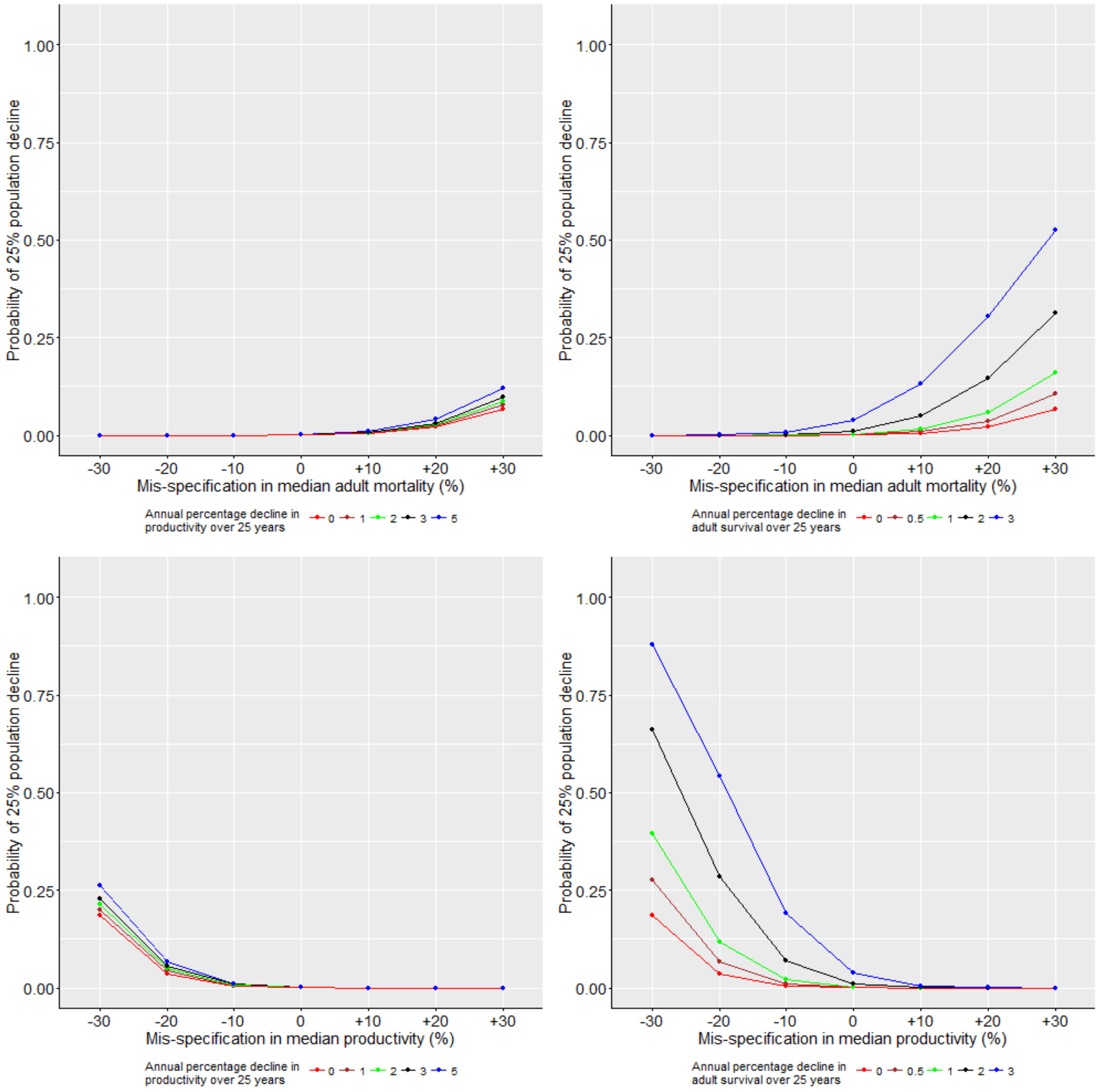


Figure A2.9g. PVA Metric E3 for Forth Razorbills – probability of population decline greater than 50% from 2016-2041.

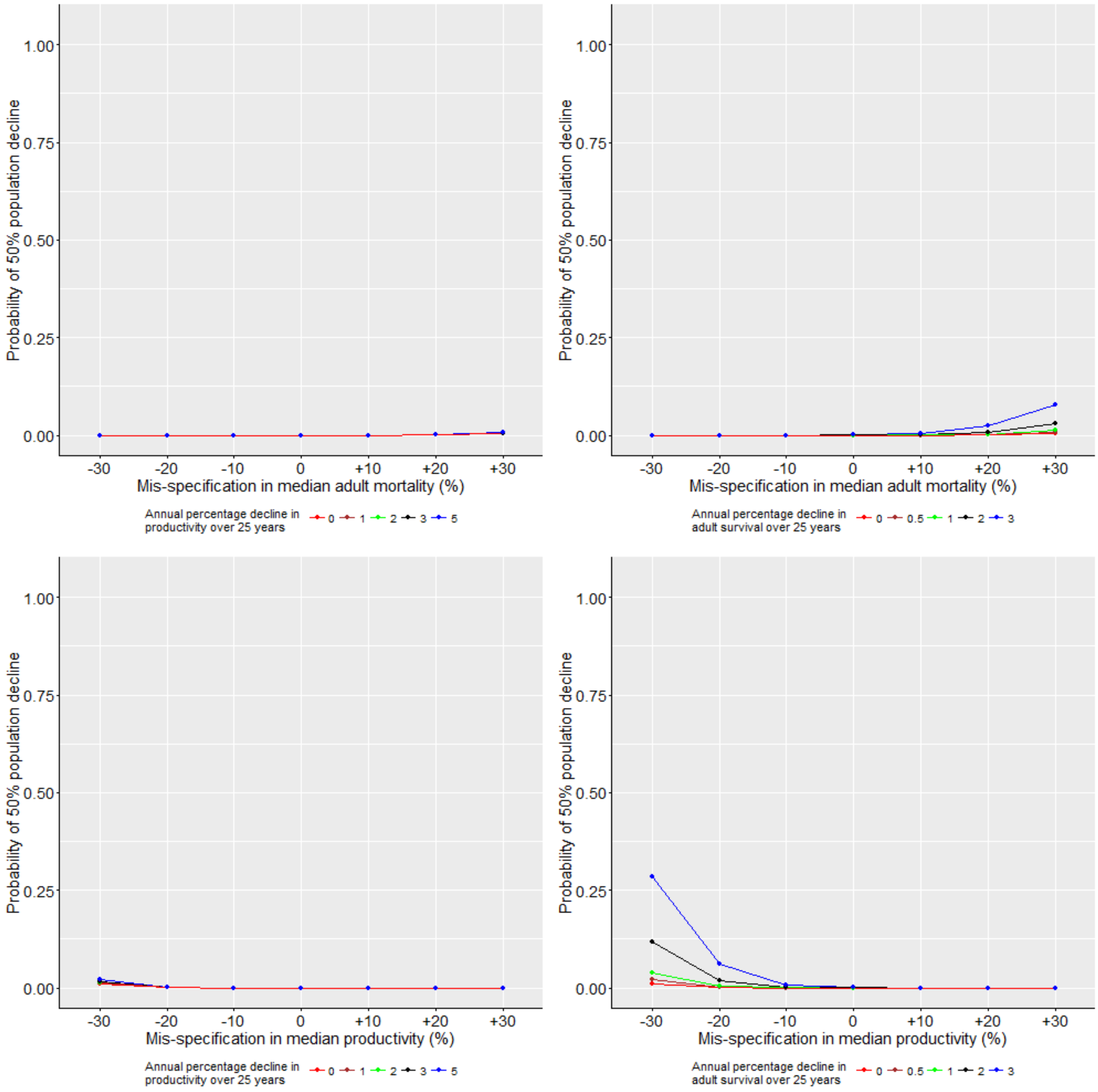
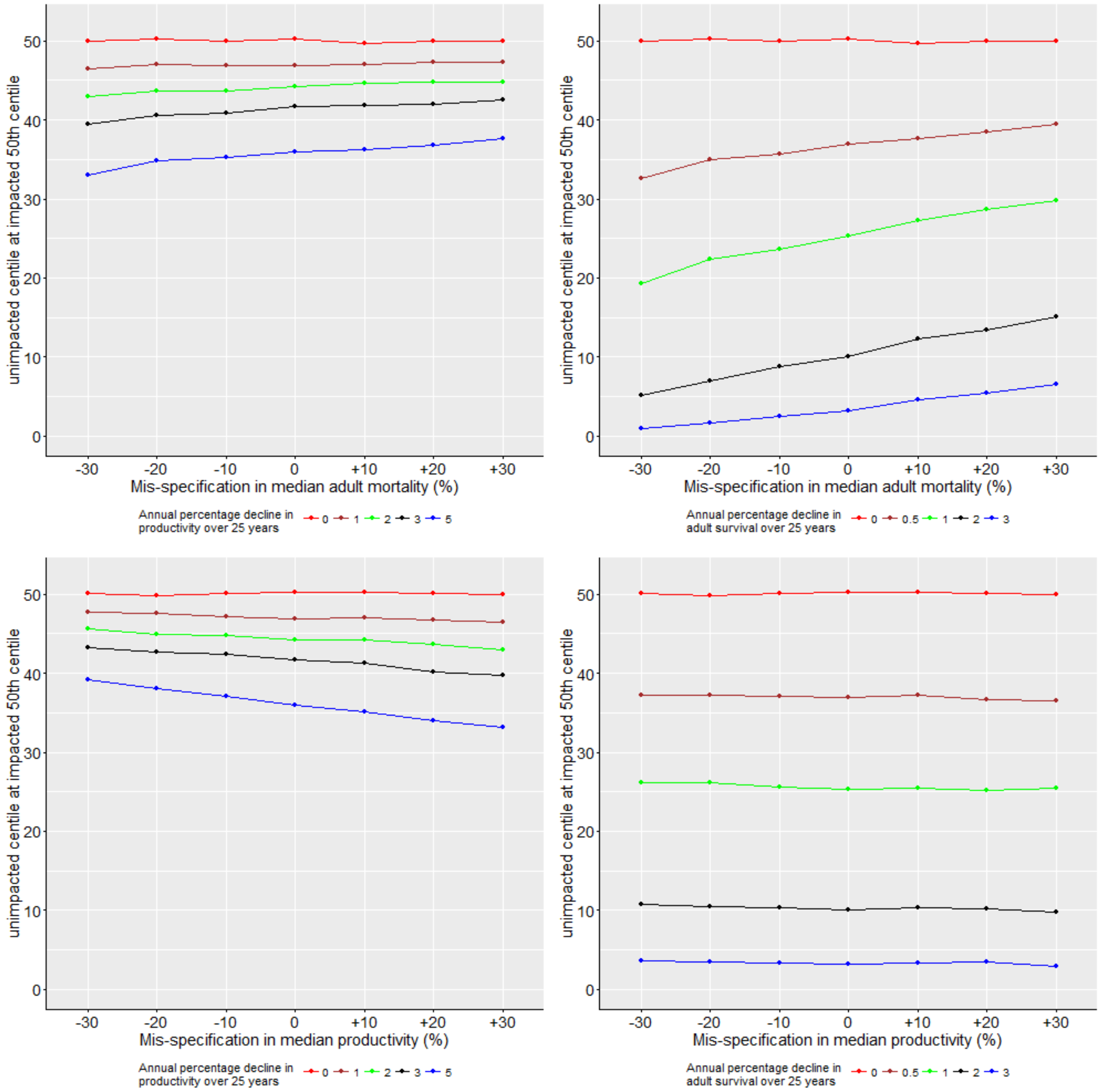


Figure A2.9h. PVA Metric F for Forth Razorbills – centile from un-impacted population size equal to the 50th centile of the impacted population size, at 2041.



10. **Razorbills at St Abb’s Head SPA**:

Figure A2.10a. PVA Metric A for St Abb’s Razorbills – ratio of population growth rate from 2016-2041, comparing impacted population vs. un-impacted population.

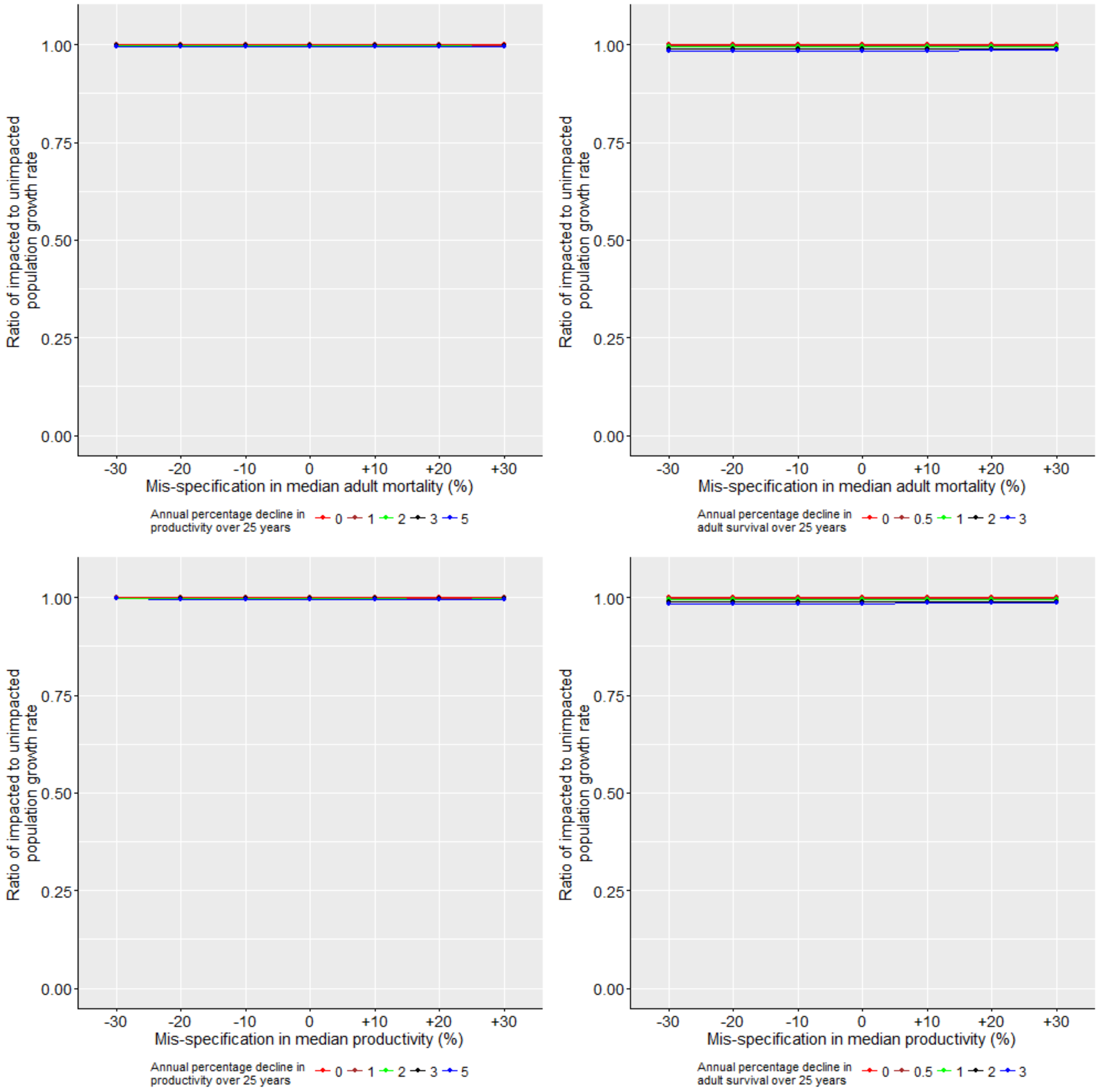


Figure A2.10b. PVA Metric B for St Abb's Razorbills – ratio of population size at 2041, comparing impacted population vs. un-impacted population.

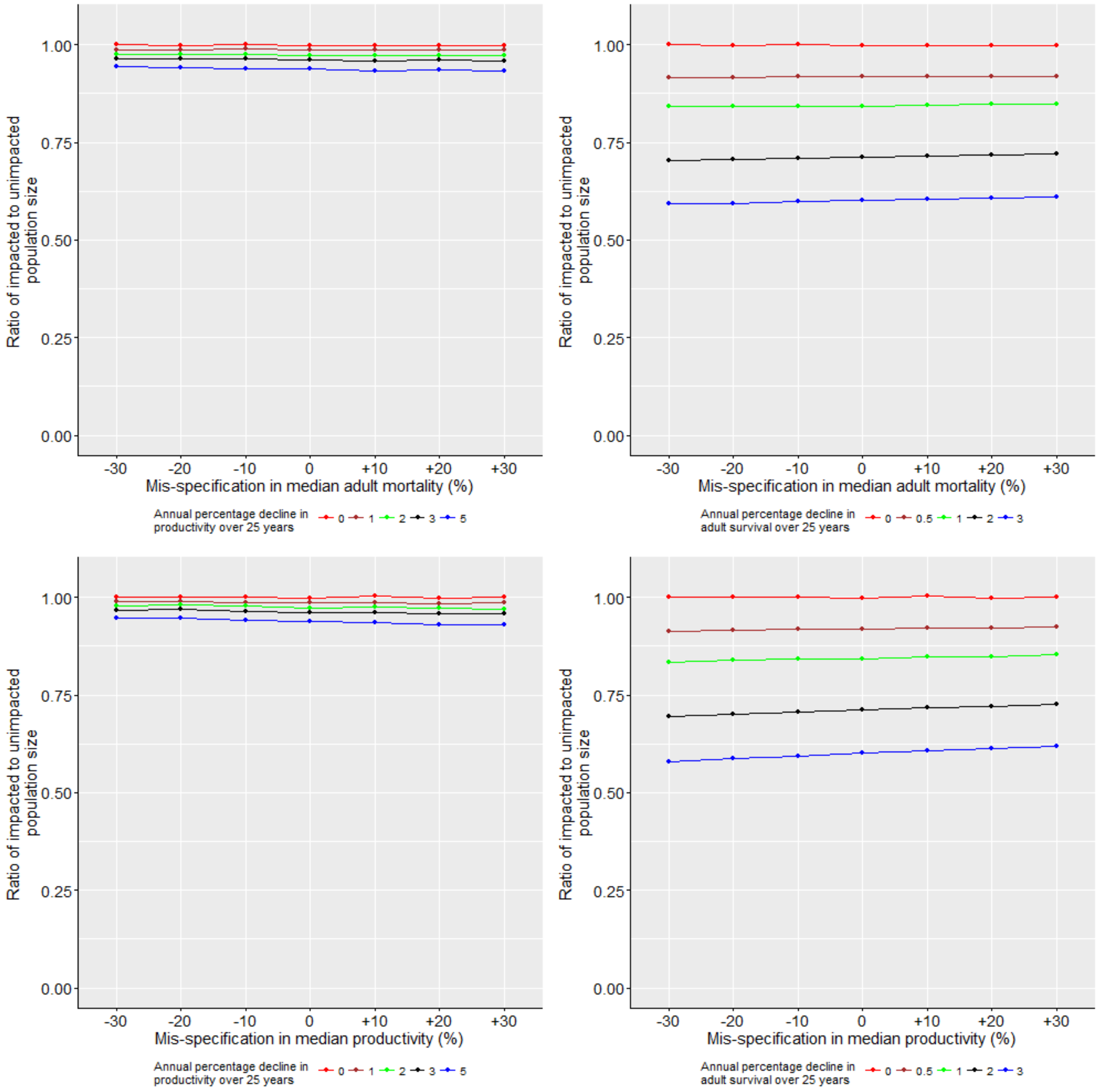


Figure A2.10c. PVA Metric C for St Abb's Razorbills – difference in population growth rate from 2016-2041, comparing impacted population vs. un-impacted population.

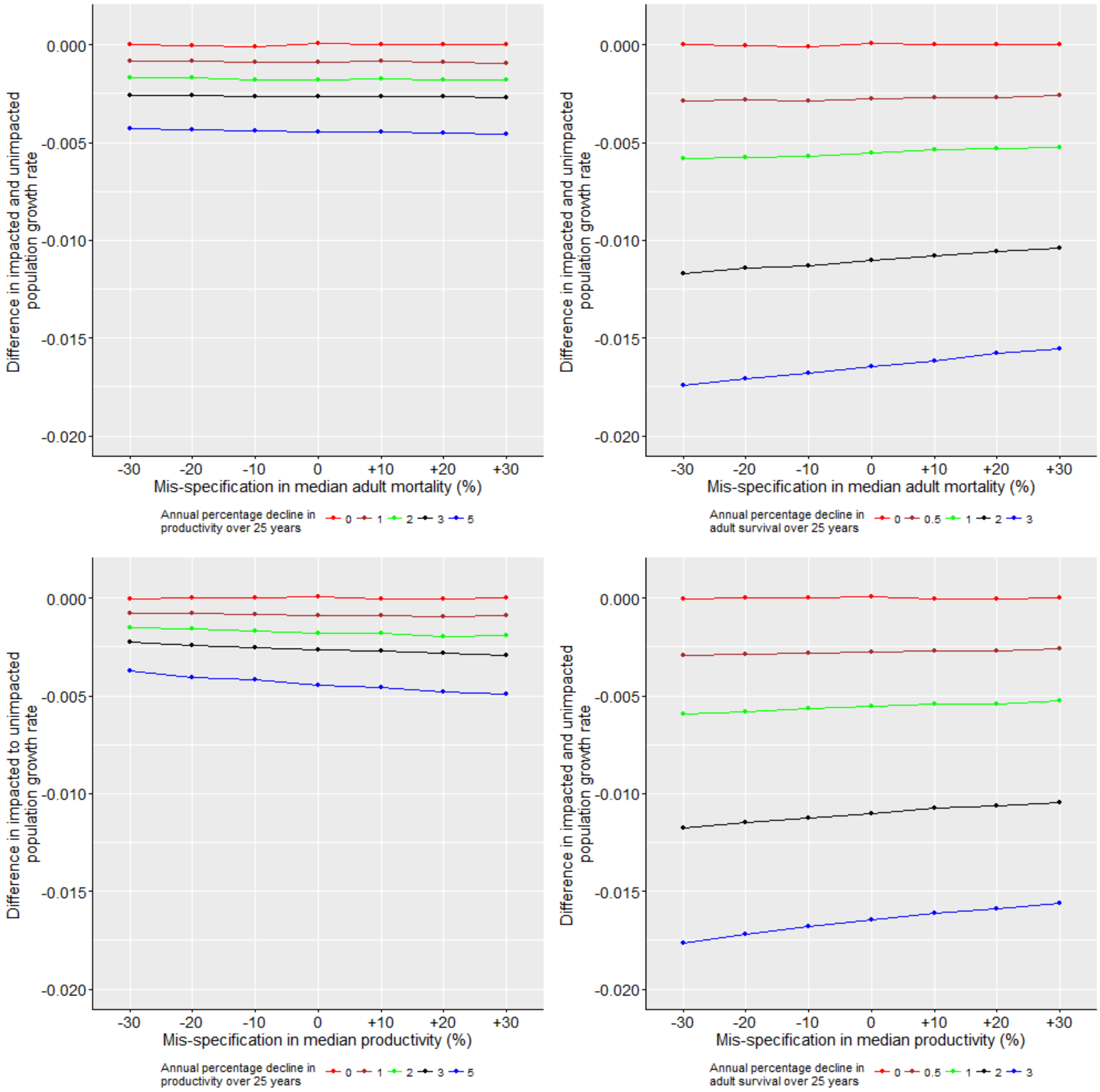


Figure A2.10d. PVA Metric D for St Abb’s Razorbills – difference in population size at 2041, comparing impacted population vs. un-impacted population.

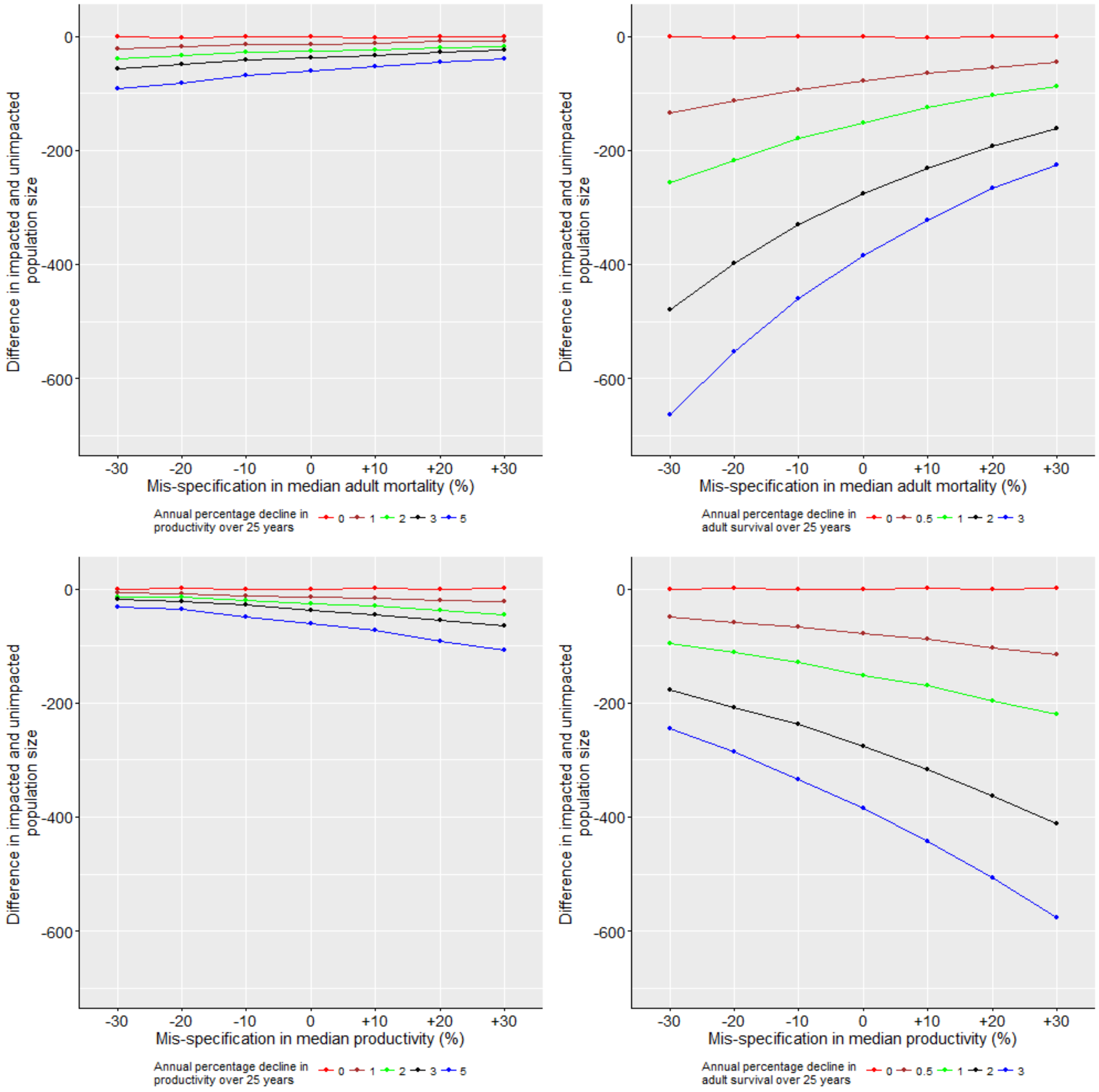


Figure A2.10e. PVA Metric E1 for St Abb’s Razorbills – probability of population decline greater than 10% from 2016-2041.

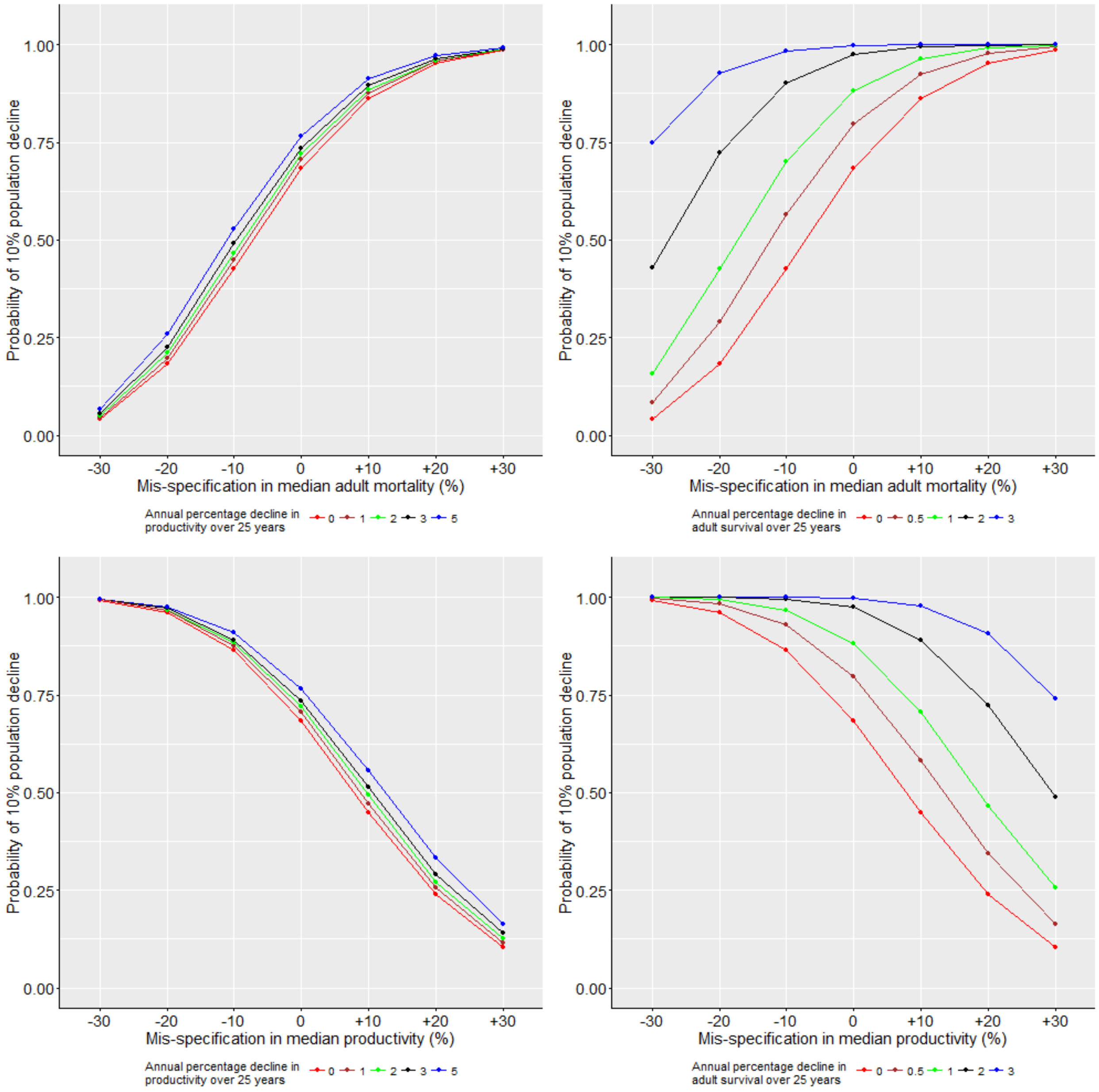


Figure A2.10f. PVA Metric E2 for St Abb's Razorbills – probability of population decline greater than 25% from 2016-2041.

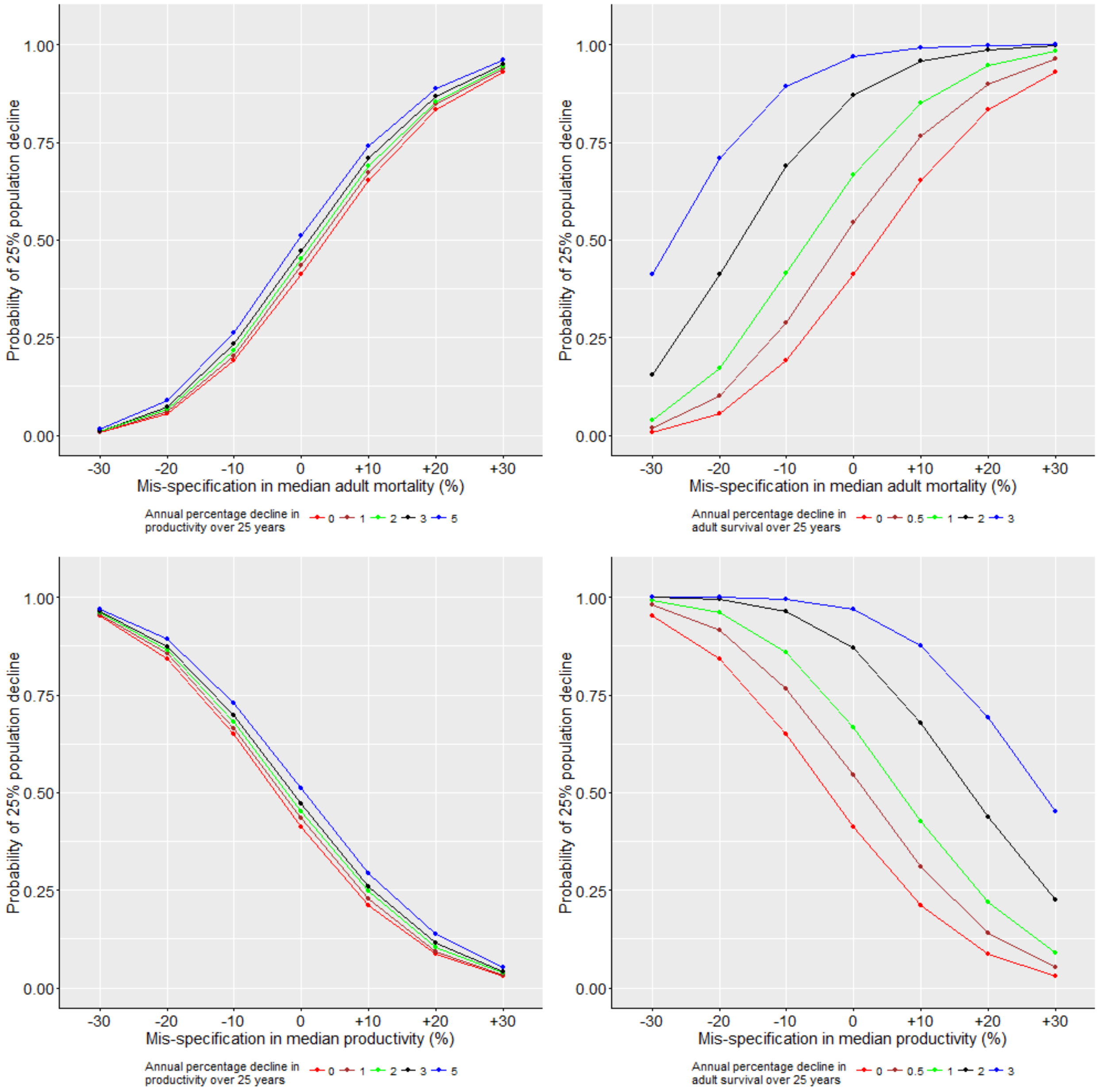


Figure A2.10g. PVA Metric E3 for St Abb’s Razorbills – probability of population decline greater than 50% from 2016-2041.

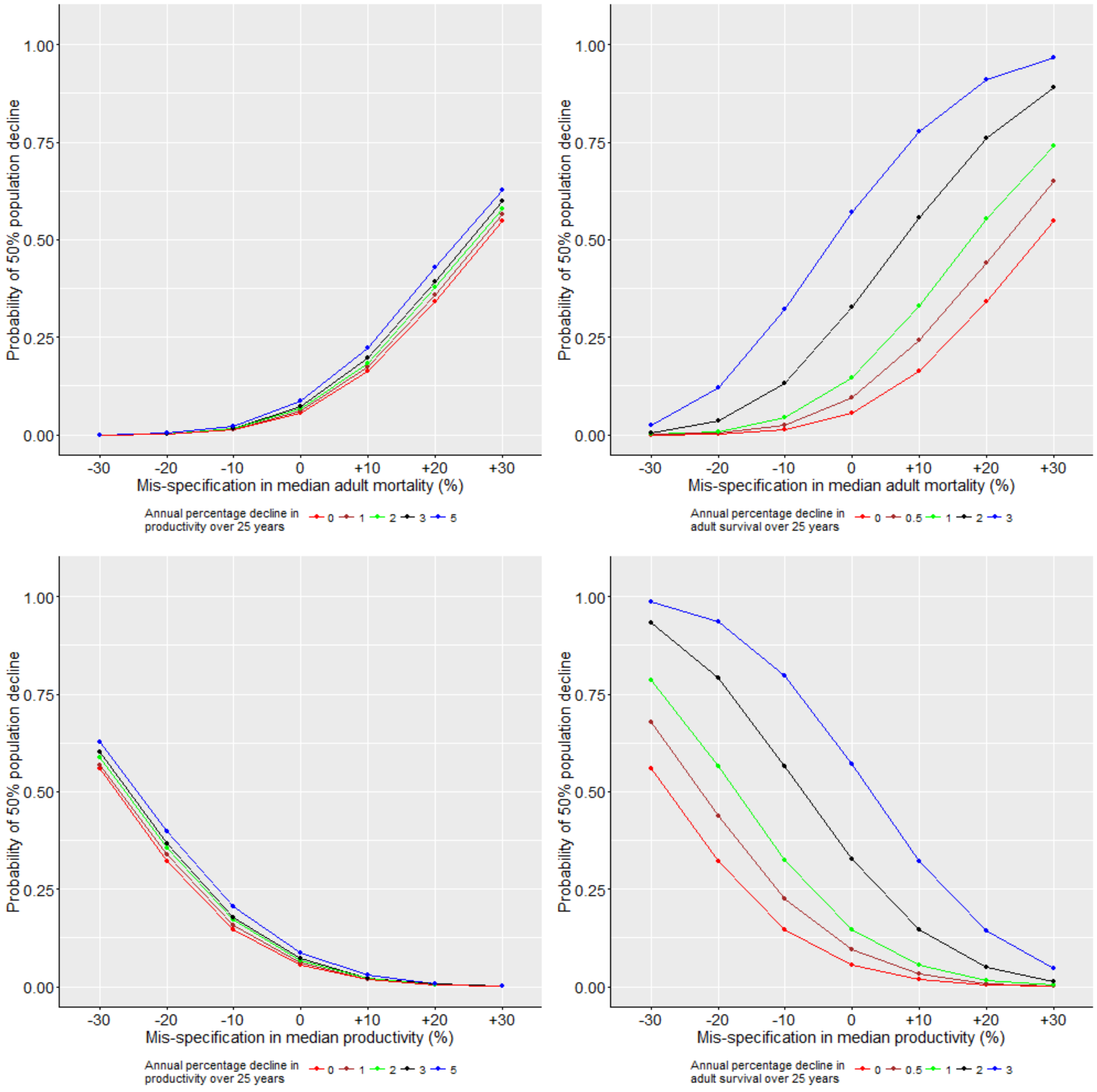
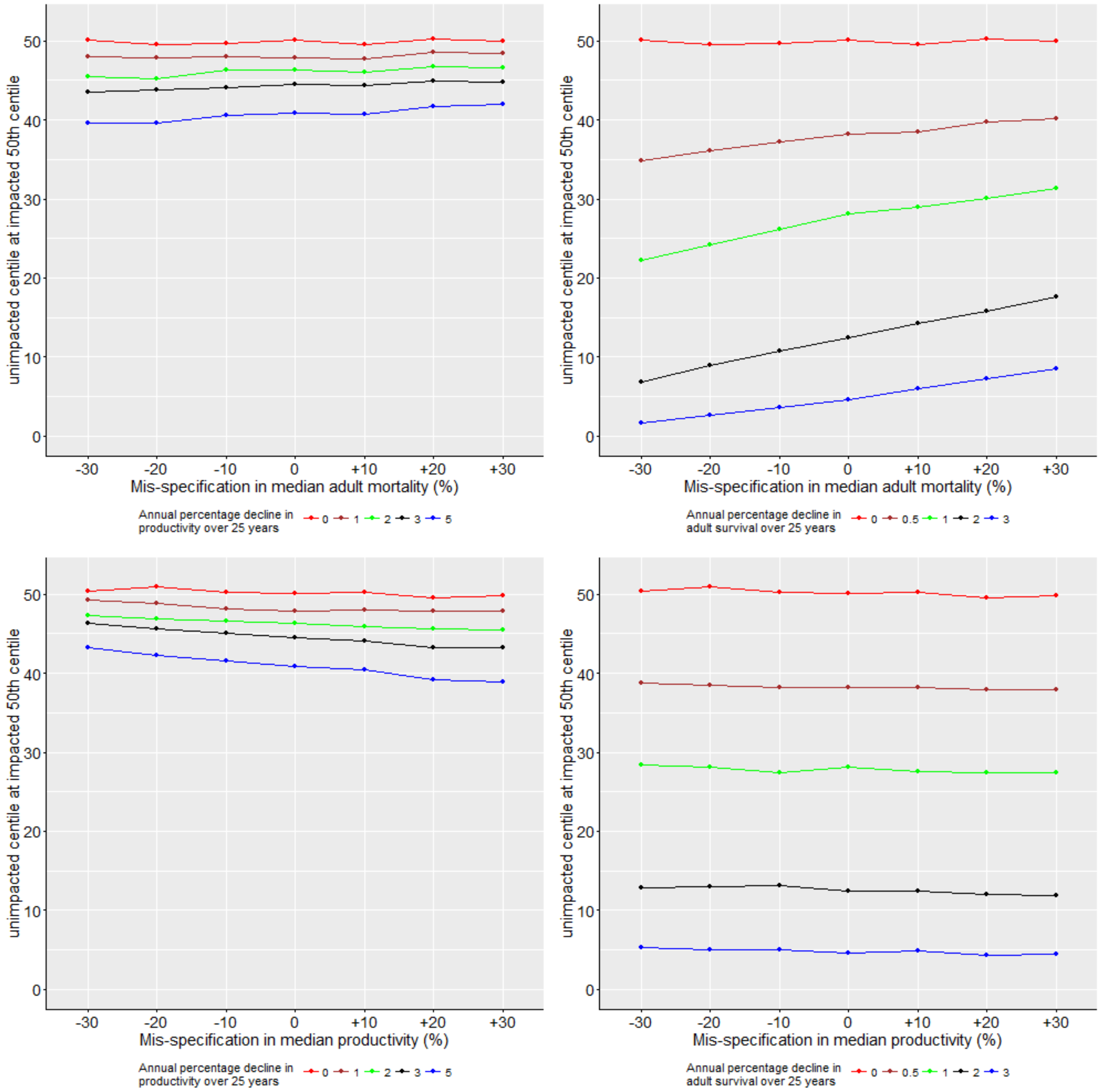


Figure A2.10h. PVA Metric F for St Abb's Razorbills – centile from un-impacted population size equal to the 50th centile of the impacted population size, at 2041.



11. **Razorbills at Fowlsheugh SPA**:

Figure A2.11a. PVA Metric A for Fowlsheugh Razorbills – ratio of population growth rate from 2016-2041, comparing impacted population vs. un-impacted population.

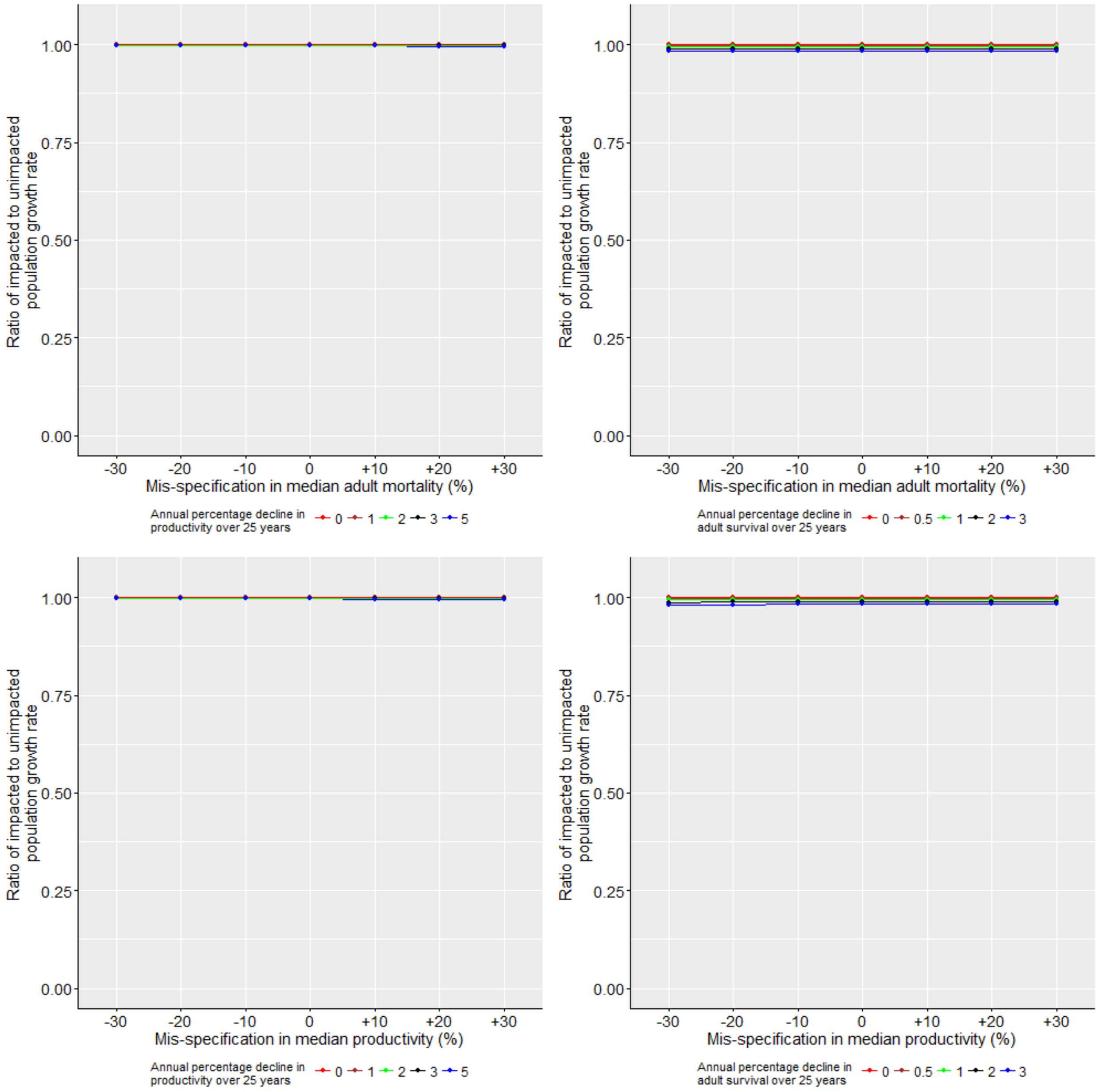


Figure A2.11b. PVA Metric B for Fowlsheugh Razorbills – ratio of population size at 2041, comparing impacted population vs. un-impacted population.

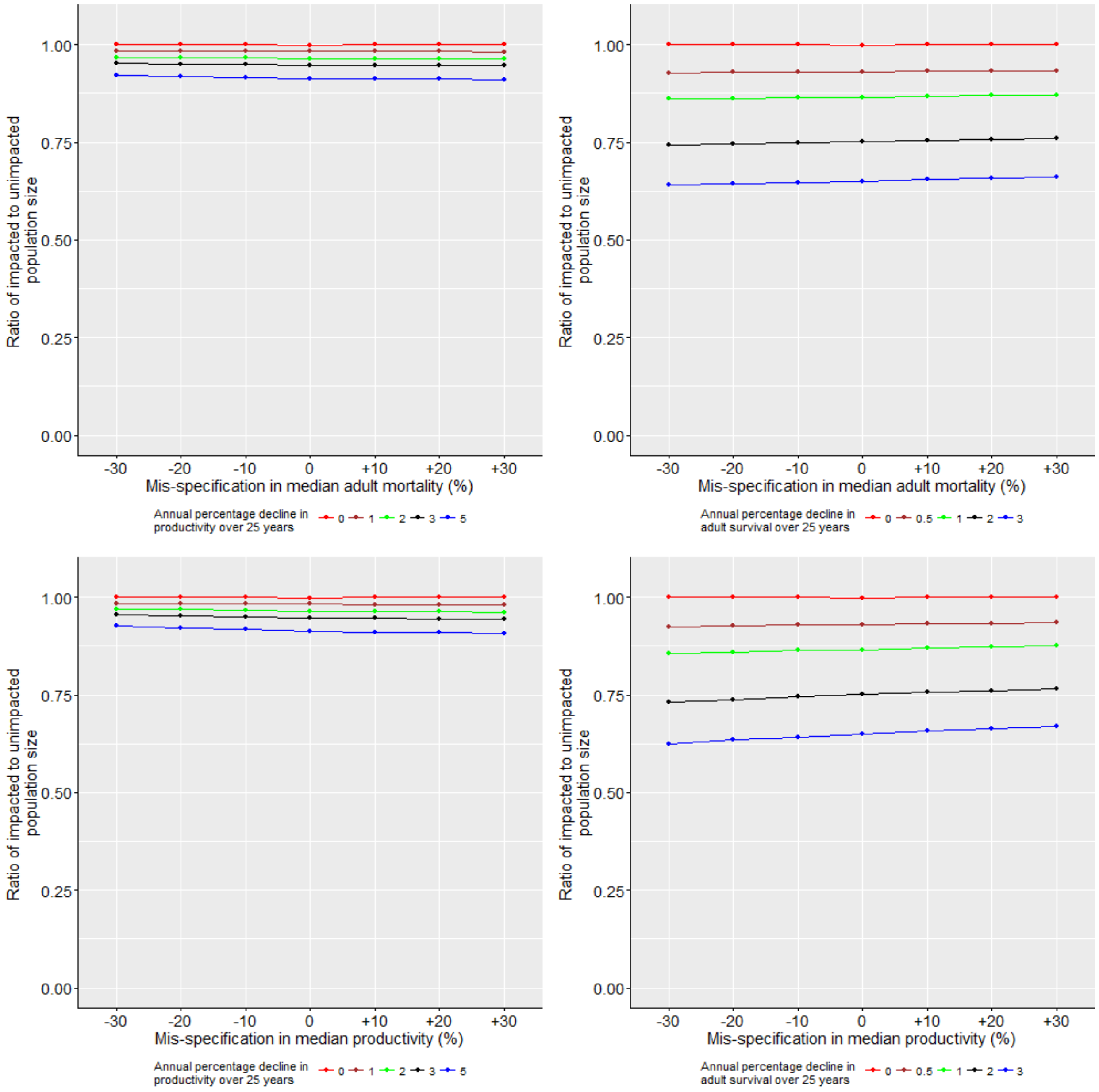


Figure A2.11c. PVA Metric C for Fowlsheugh Razorbills – difference in population growth rate from 2016-2041, comparing impacted population vs. un-impacted population.

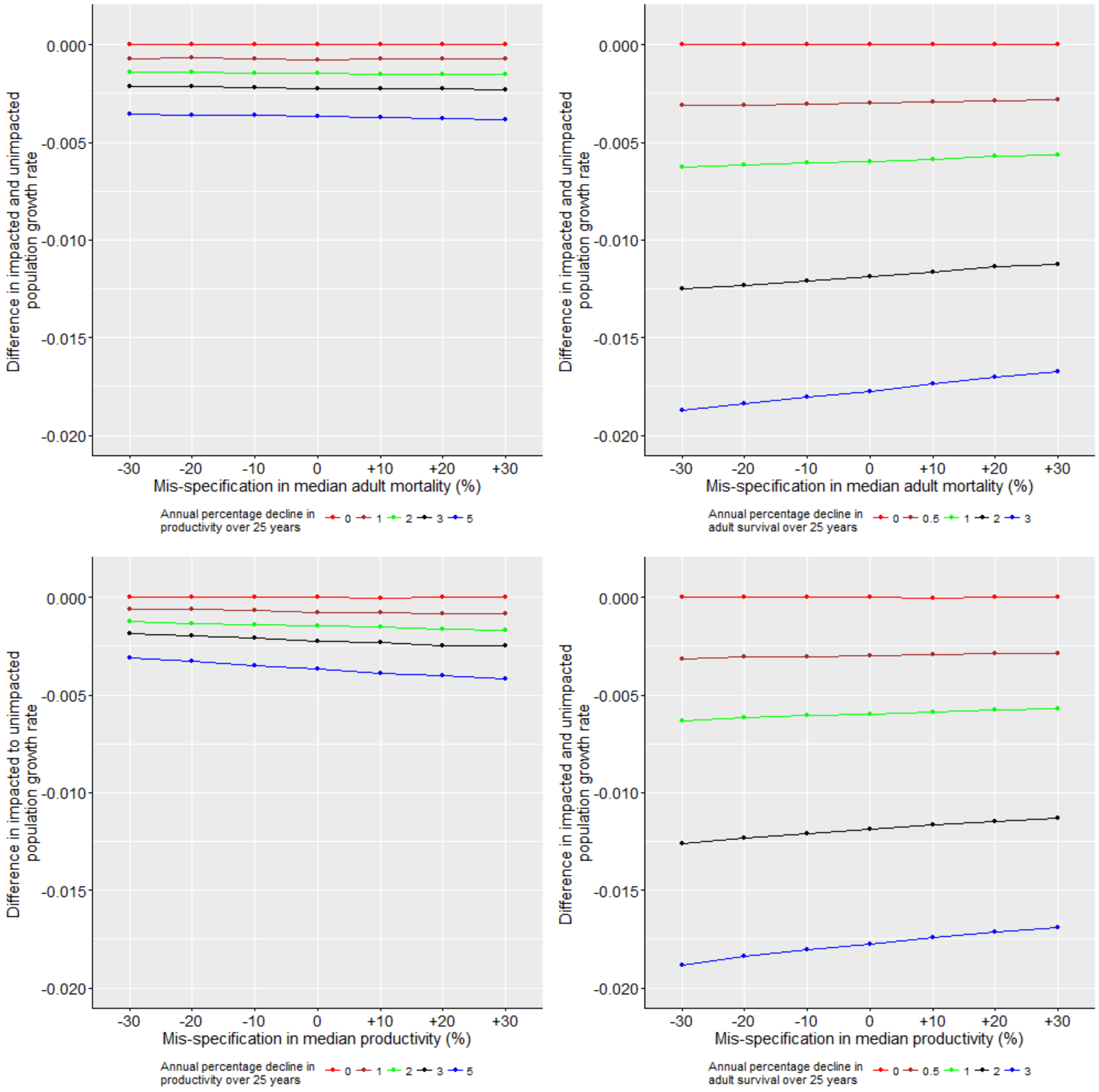


Figure A2.11d. PVA Metric D for Fowlsheugh Razorbills – difference in population size at 2041, comparing impacted population vs. un-impacted population.

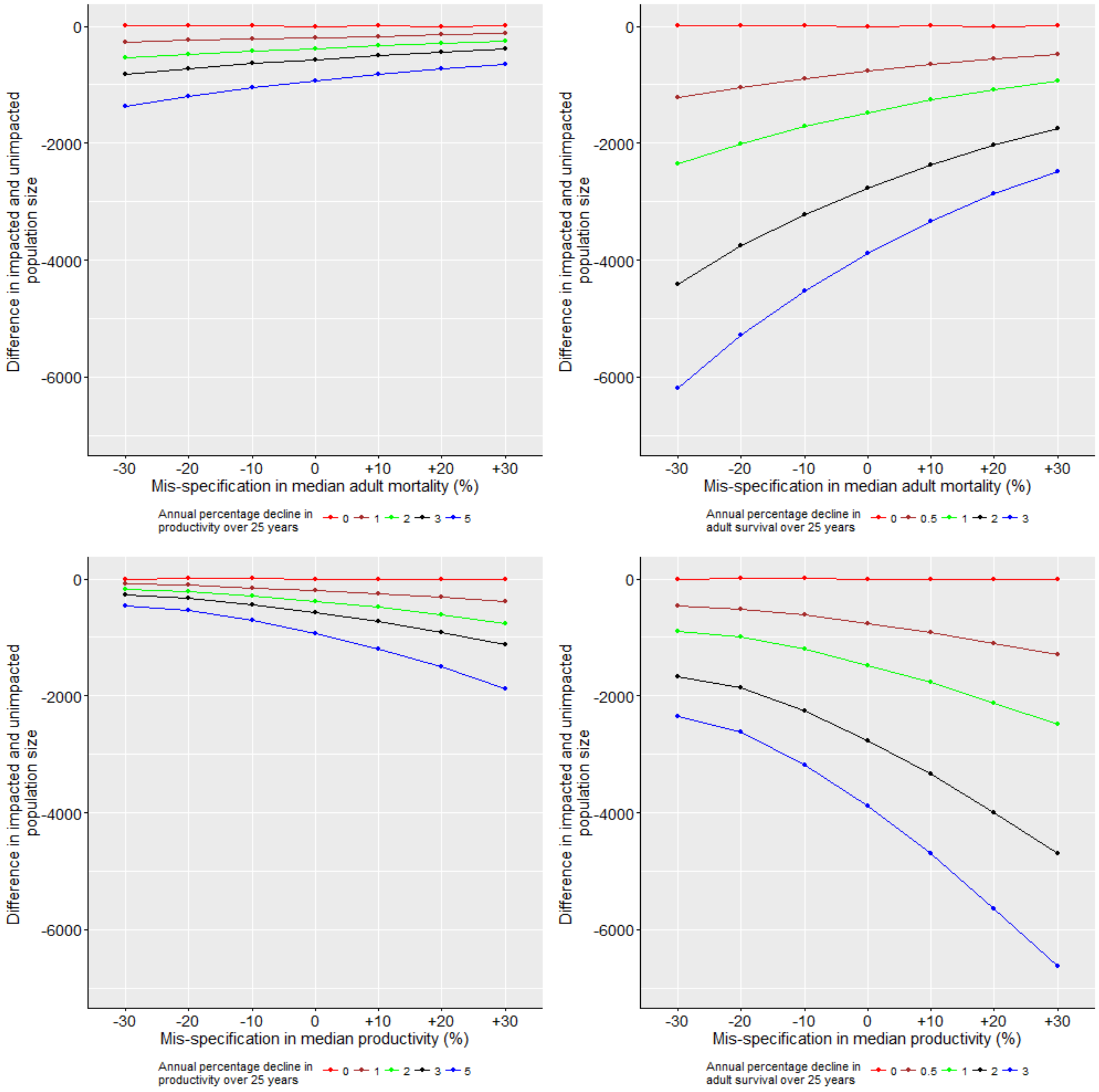


Figure A2.11e. PVA Metric E1 for Fowlsheugh Razorbills – probability of population decline greater than 10% from 2016-2041.

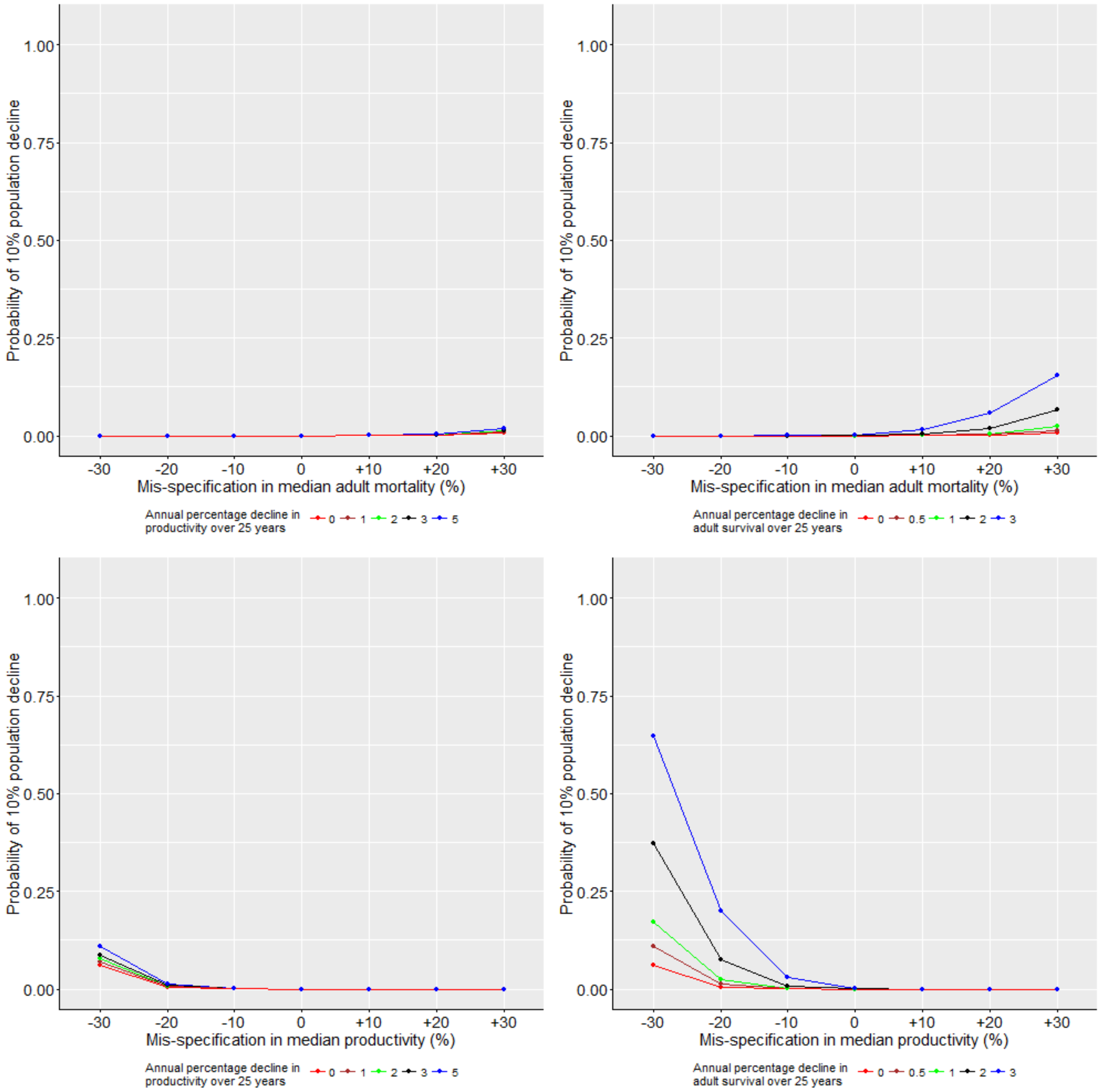


Figure A2.11f. PVA Metric E2 for Fowlsheugh Razorbills – probability of population decline greater than 25% from 2016-2041.

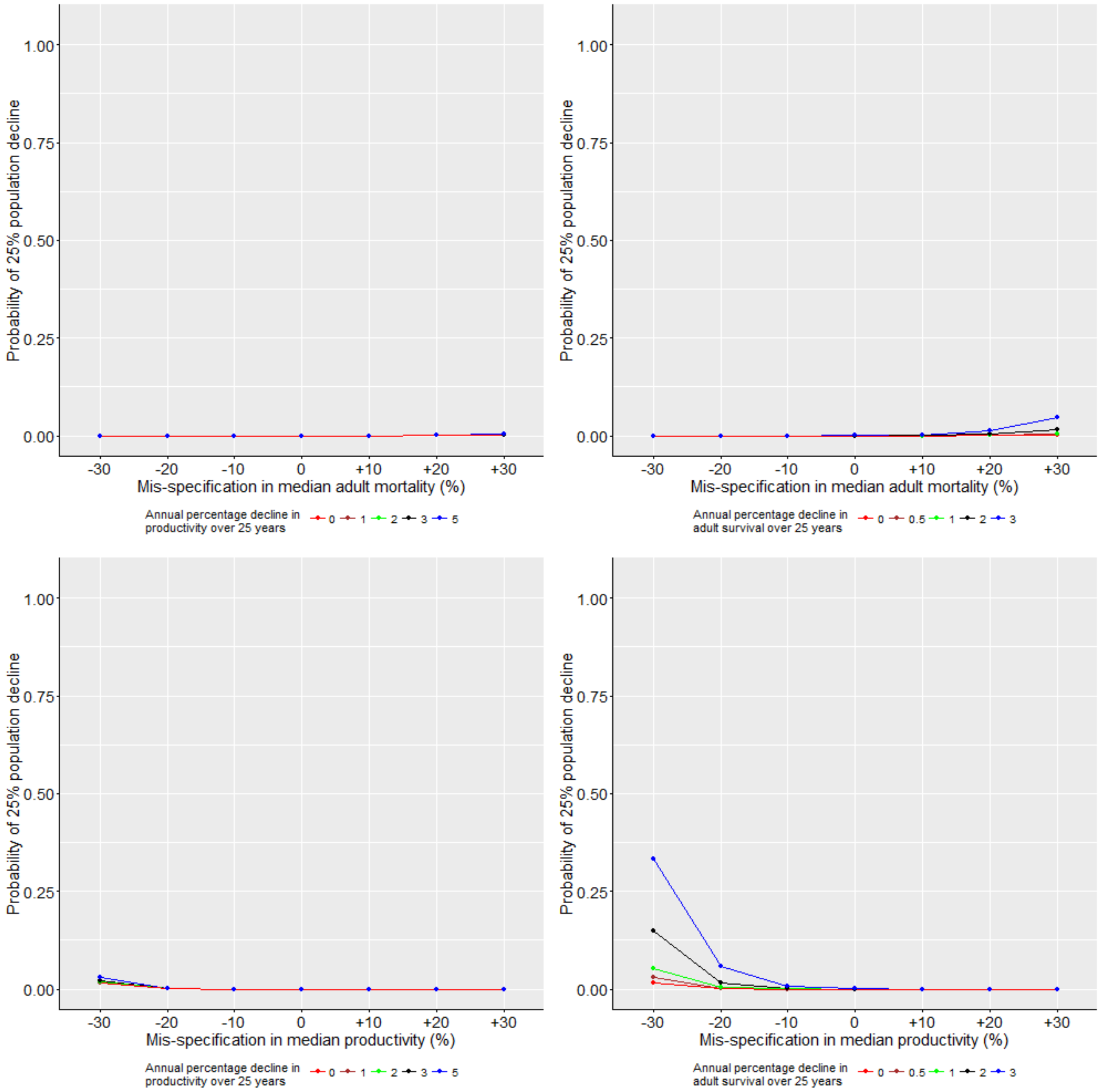


Figure A2.11g. PVA Metric E3 for Fowlsheugh Razorbills – probability of population decline greater than 50% from 2016-2041.

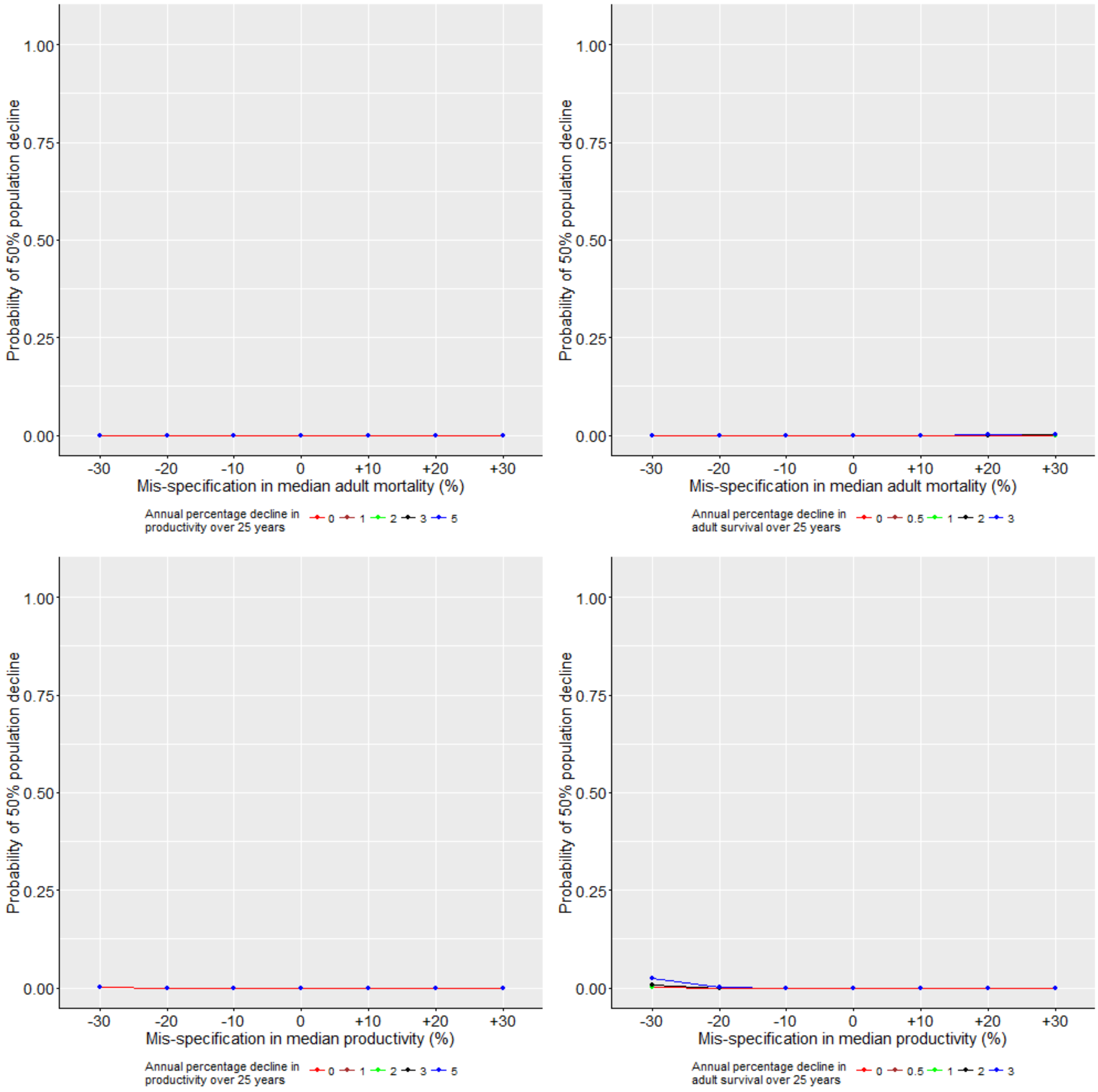
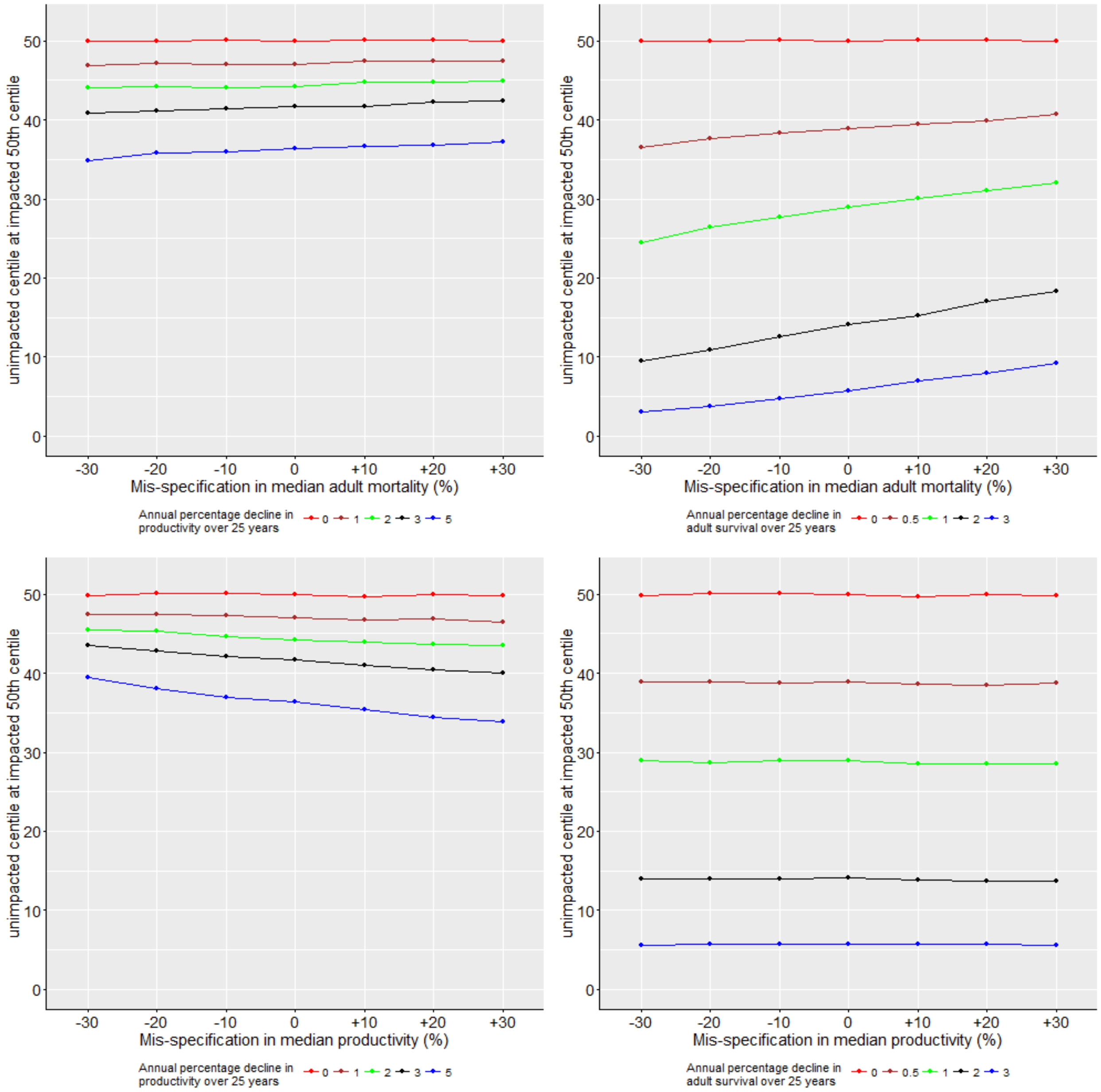


Figure A2.11h. PVA Metric F for Fowlsheugh Razorbills – centile from un-impacted population size equal to the 50th centile of the impacted population size, at 2041.



12. **Shags at Forth Islands SPA**:

Figure A2.12a. PVA Metric A for Forth Shags – ratio of population growth rate from 2016-2041, comparing impacted population vs. un-impacted population.

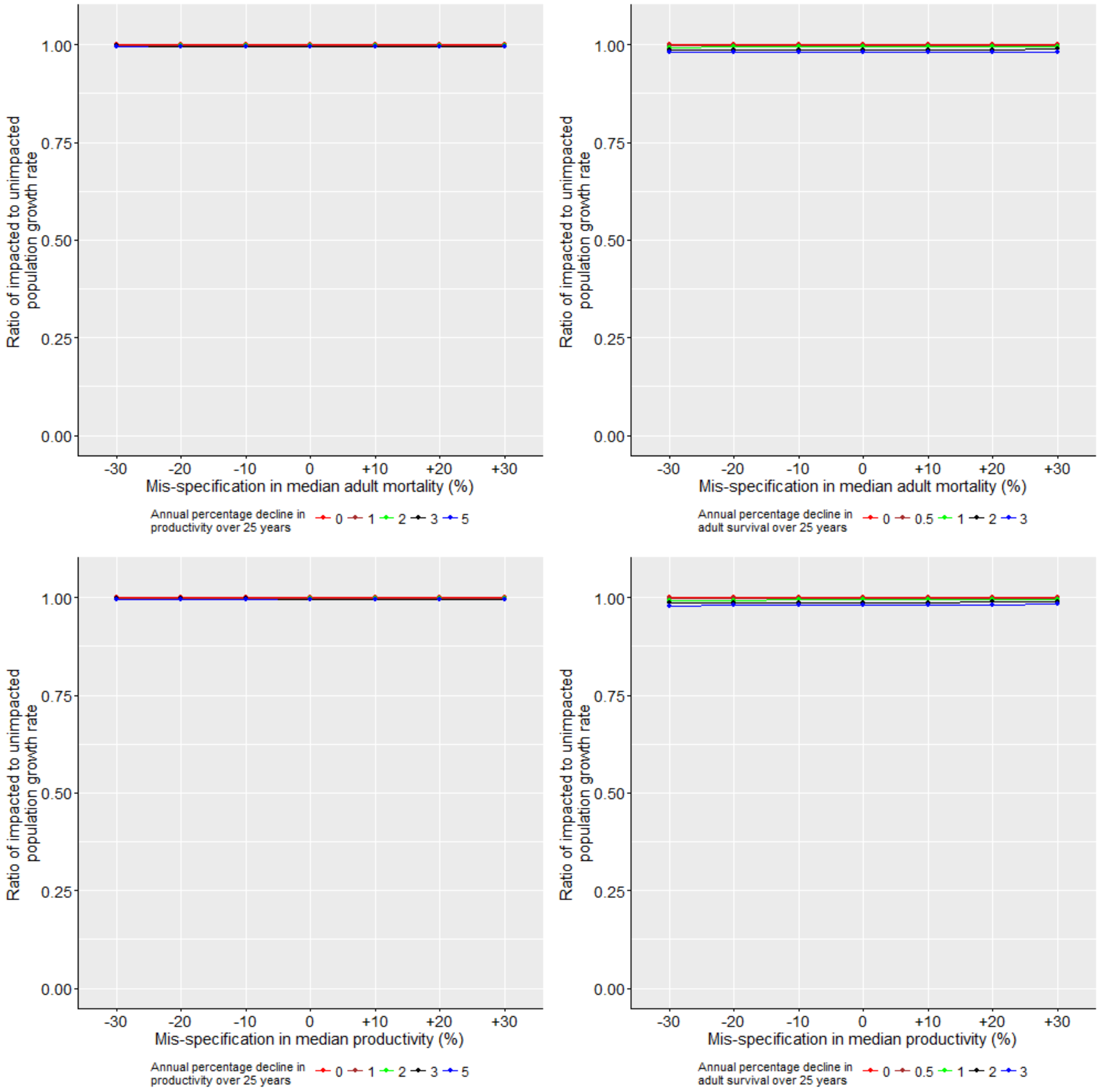


Figure A2.12b. PVA Metric B for Forth Shags – ratio of population size at 2041, comparing impacted population vs. un-impacted population.

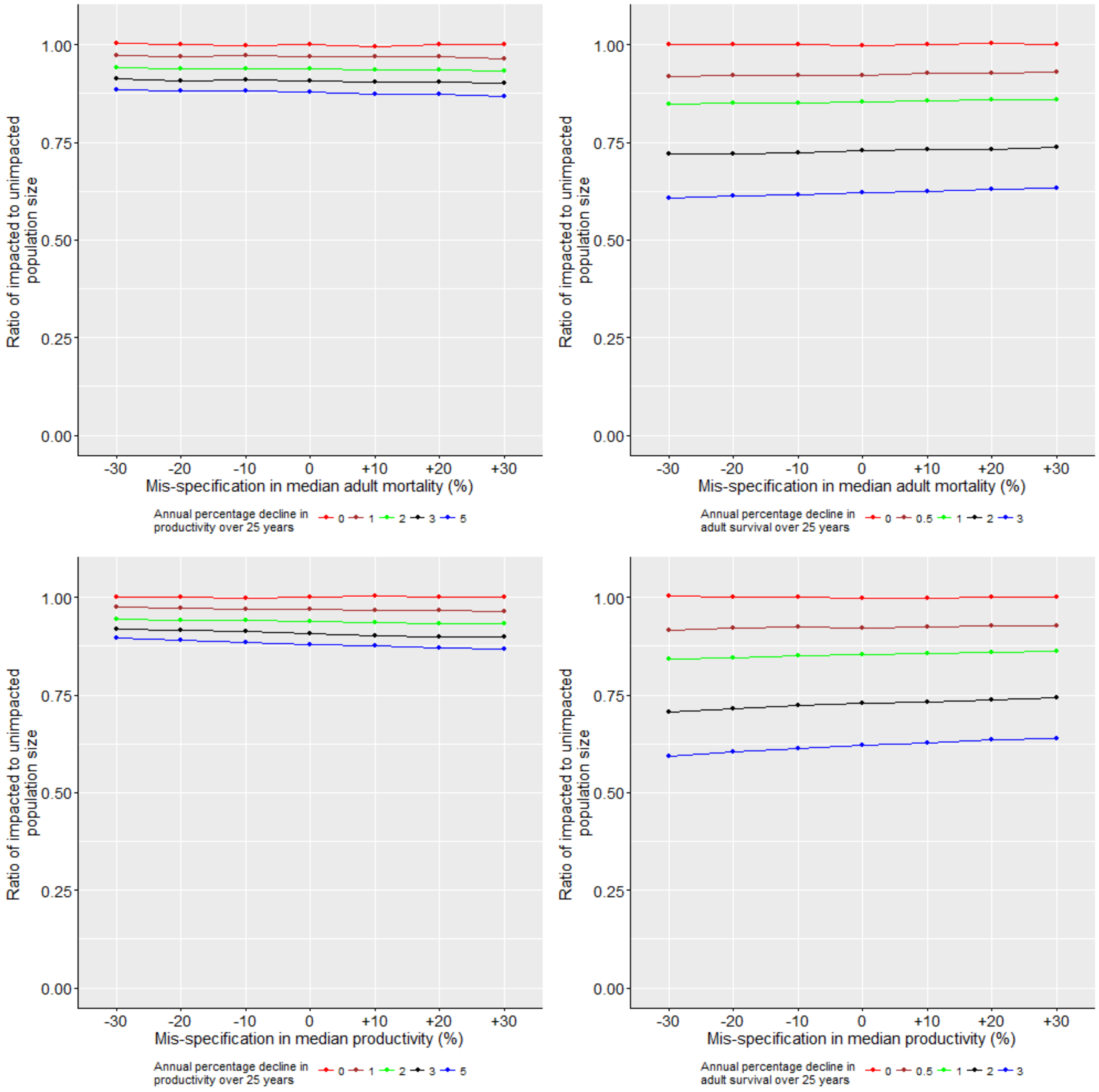


Figure A2.12c. PVA Metric C for Forth Shags – difference in population growth rate from 2016-2041, comparing impacted population vs. un-impacted population.

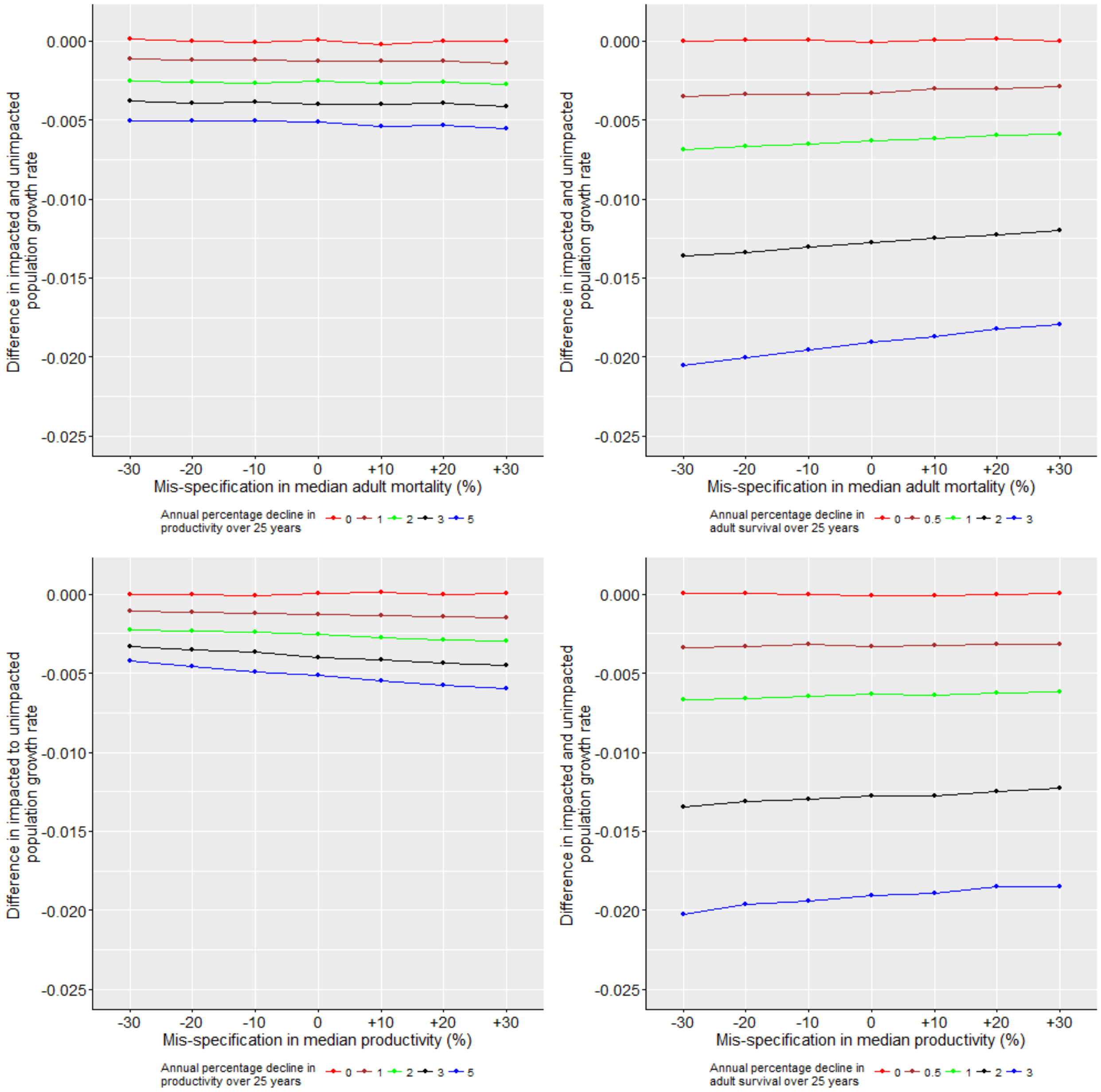


Figure A2.12d. PVA Metric D for Forth Shags – difference in population size at 2041, comparing impacted population vs. un-impacted population.

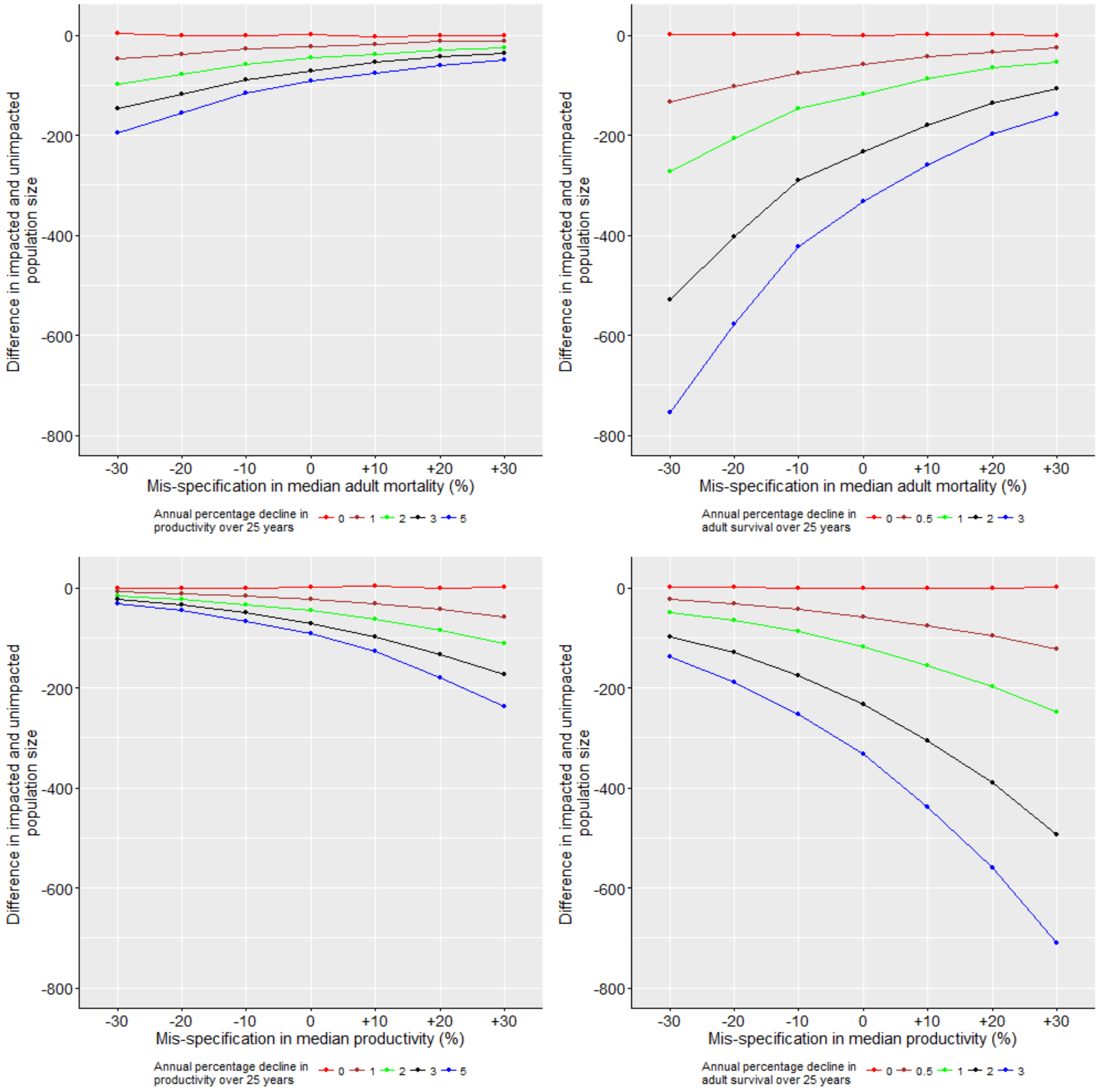


Figure A2.12e. PVA Metric E1 for Forth Shags – probability of population decline greater than 10% from 2016-2041.

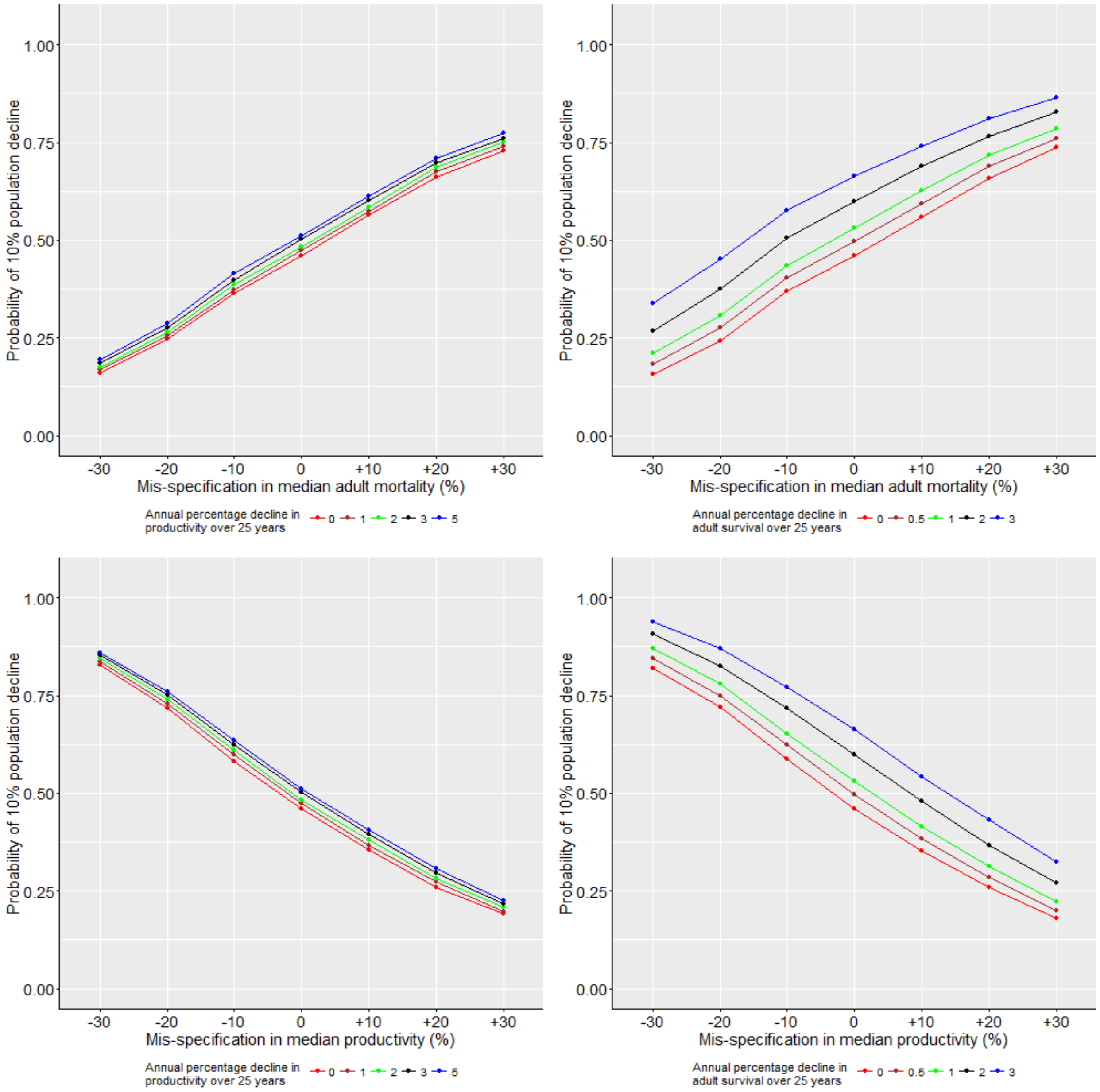


Figure A2.12f. PVA Metric E2 for Forth Shags – probability of population decline greater than 25% from 2016-2041.

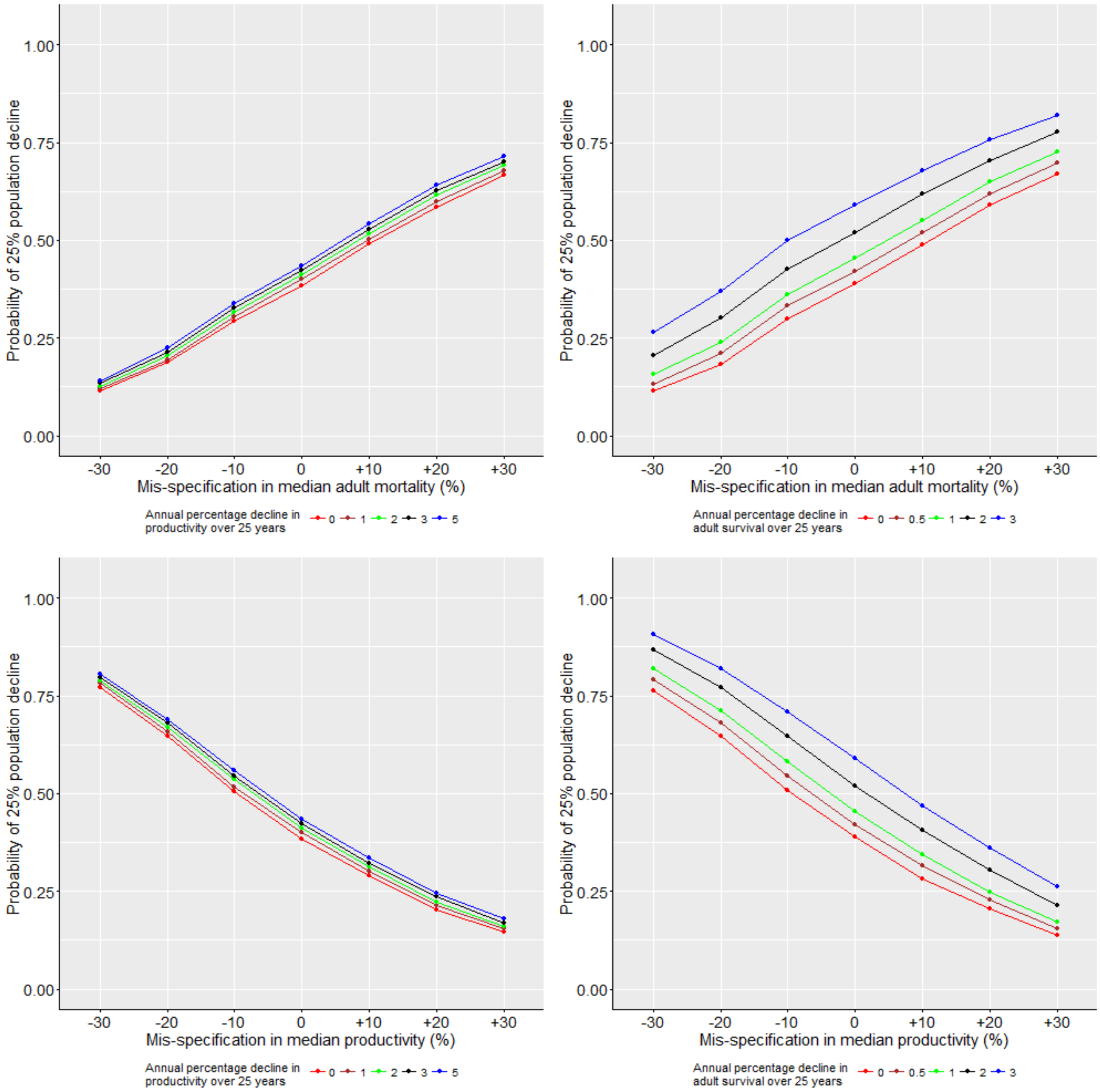


Figure A2.12g. PVA Metric E3 for Forth Shags – probability of population decline greater than 50% from 2016-2041.

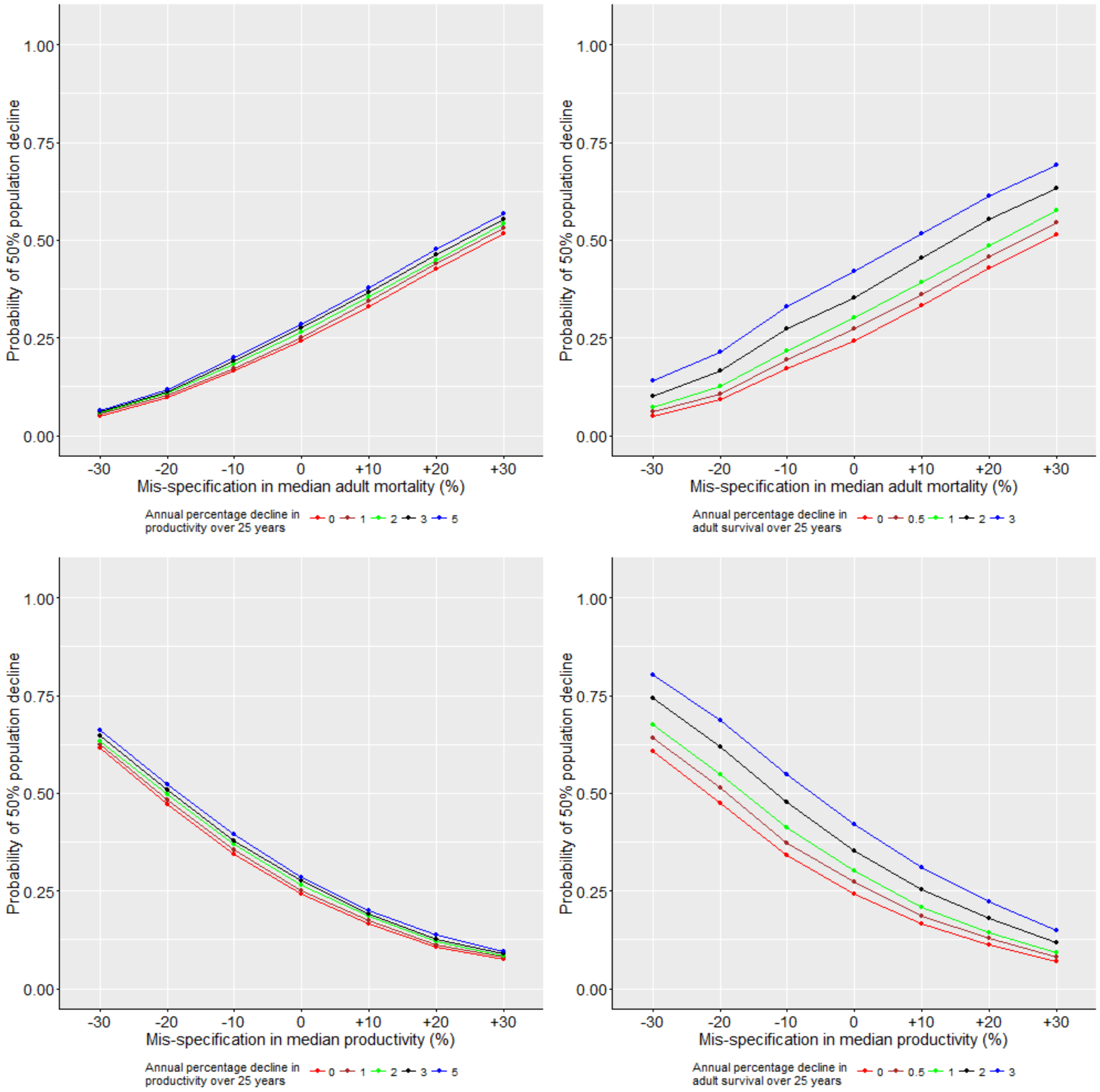
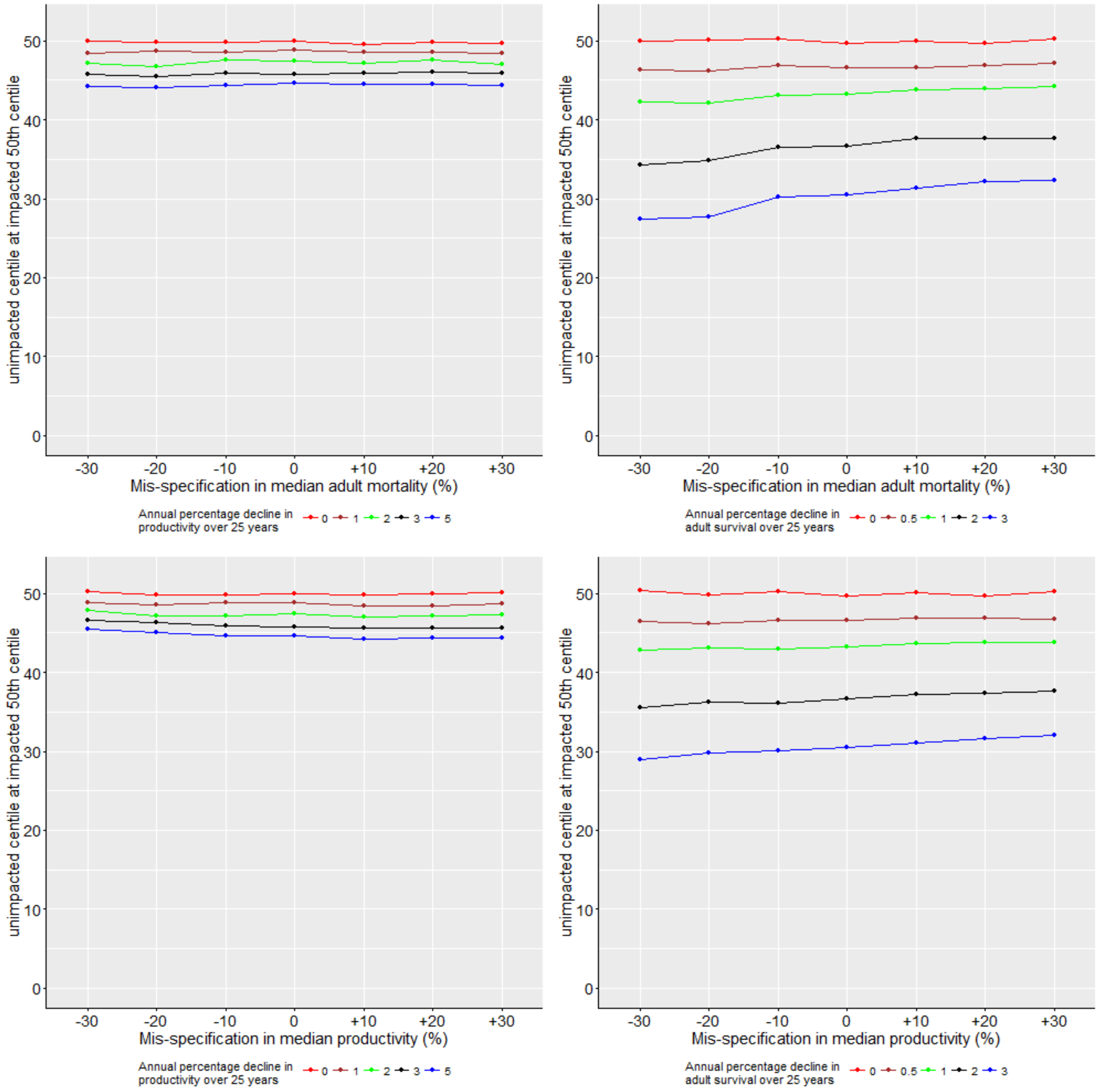


Figure A2.12h. PVA Metric F for Forth Shags – centile from un-impacted population size equal to the 50th centile of the impacted population size, at 2041.



13. **Shags at St Abb’s Head SPA**:

Figure A2.13a. PVA Metric A for St Abb’s Shags – ratio of population growth rate from 2016-2041, comparing impacted population vs. un-impacted population.

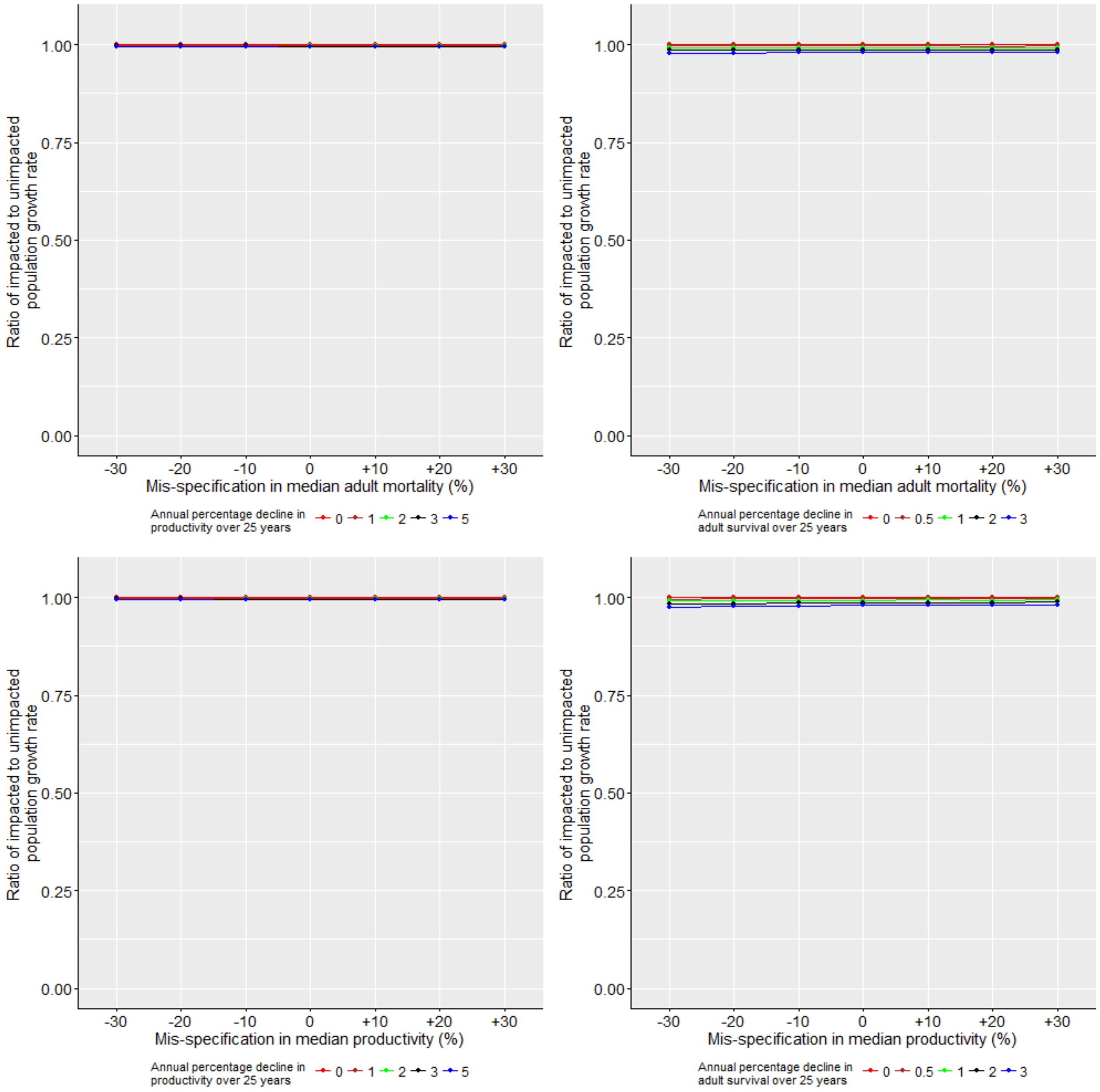


Figure A2.13b. PVA Metric B for St Abb’s Shags – ratio of population size at 2041, comparing impacted population vs. un-impacted population.

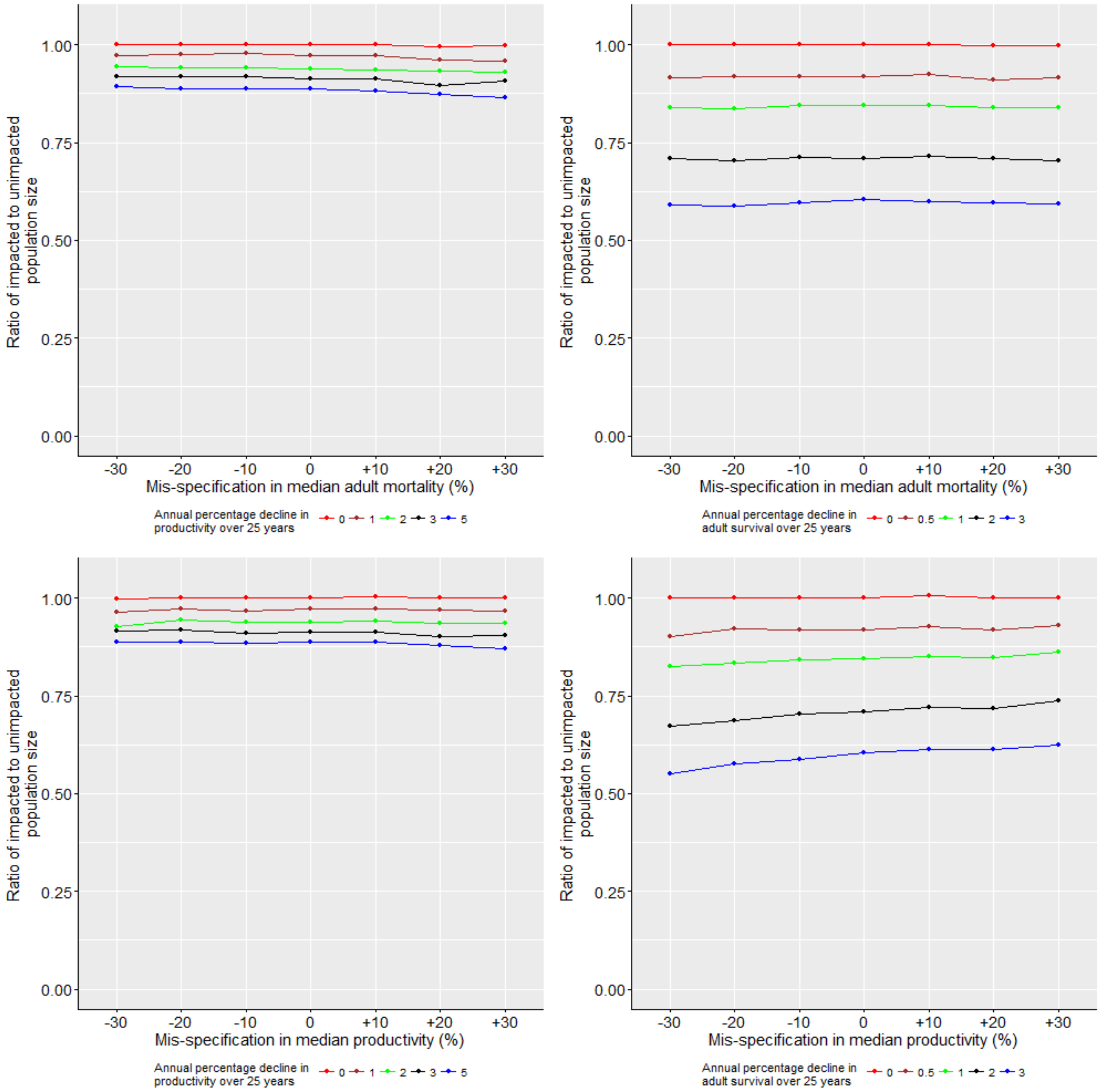


Figure A2.13c. PVA Metric C for St Abb’s Shags – difference in population growth rate from 2016-2041, comparing impacted population vs. un-impacted population.

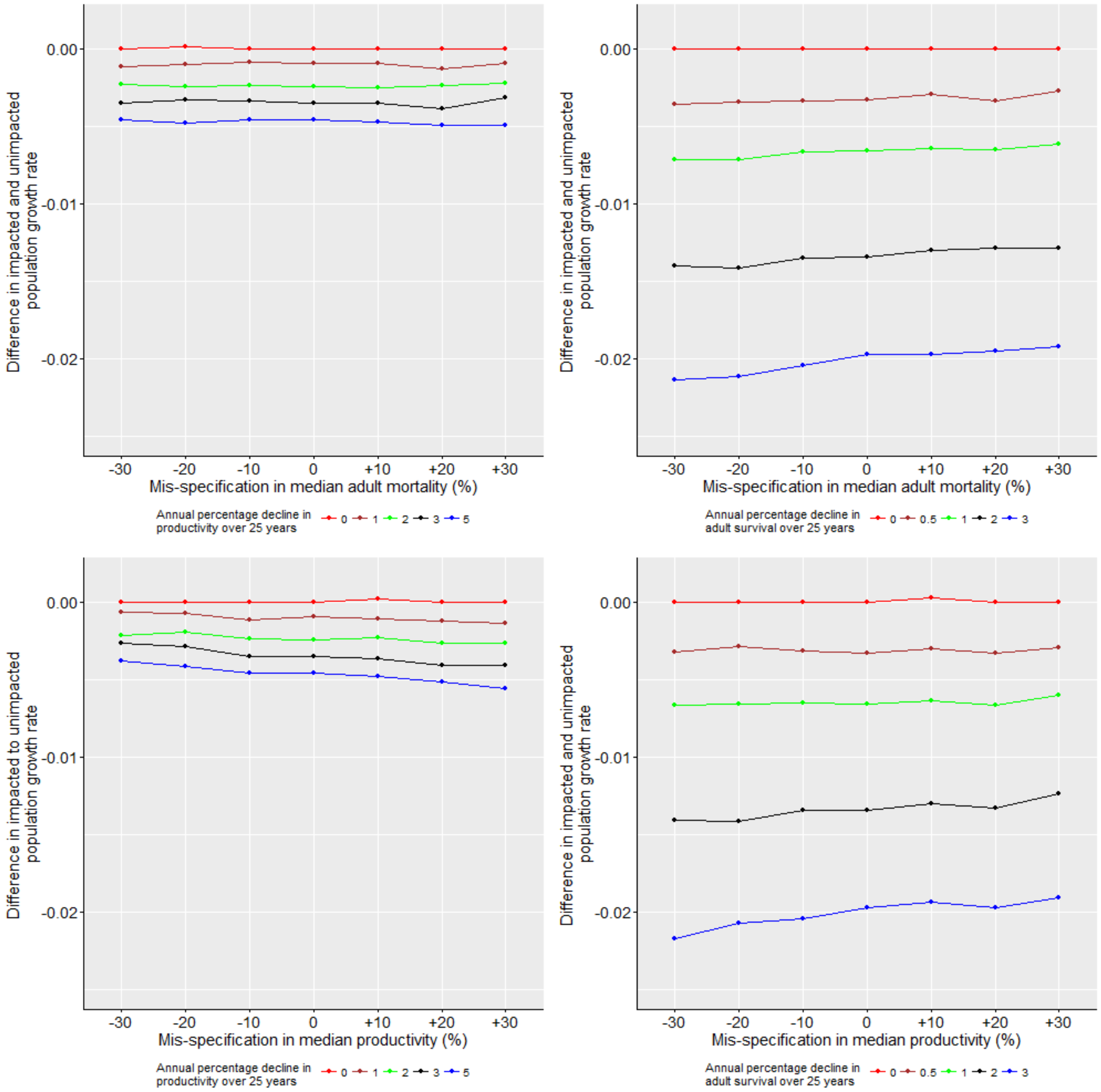


Figure A2.13d. PVA Metric D for St Abb’s Shags – difference in population size at 2041, comparing impacted population vs. un-impacted population.

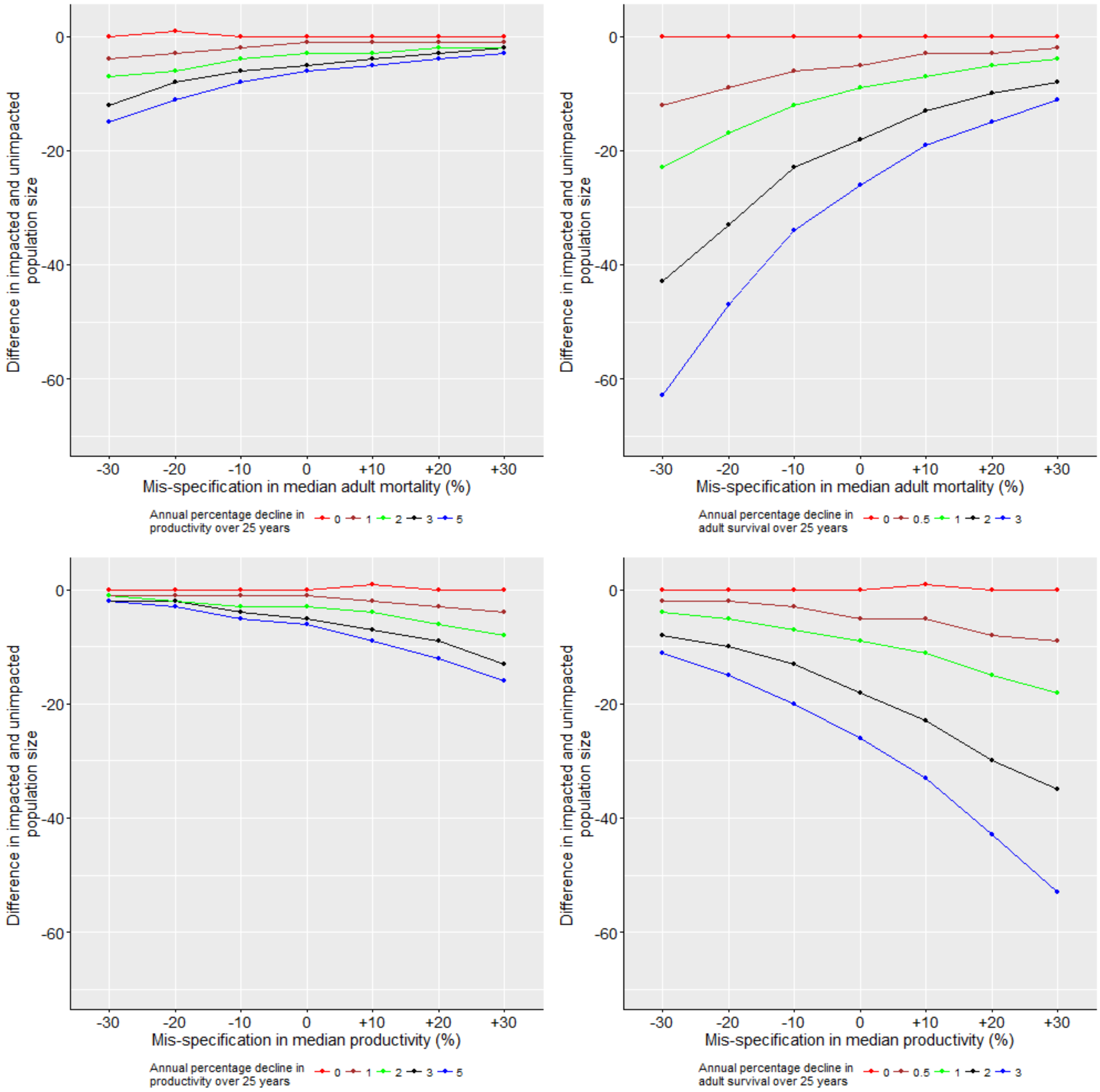


Figure A2.13e. PVA Metric E1 for St Abb’s Shags – probability of population decline greater than 10% from 2016-2041.

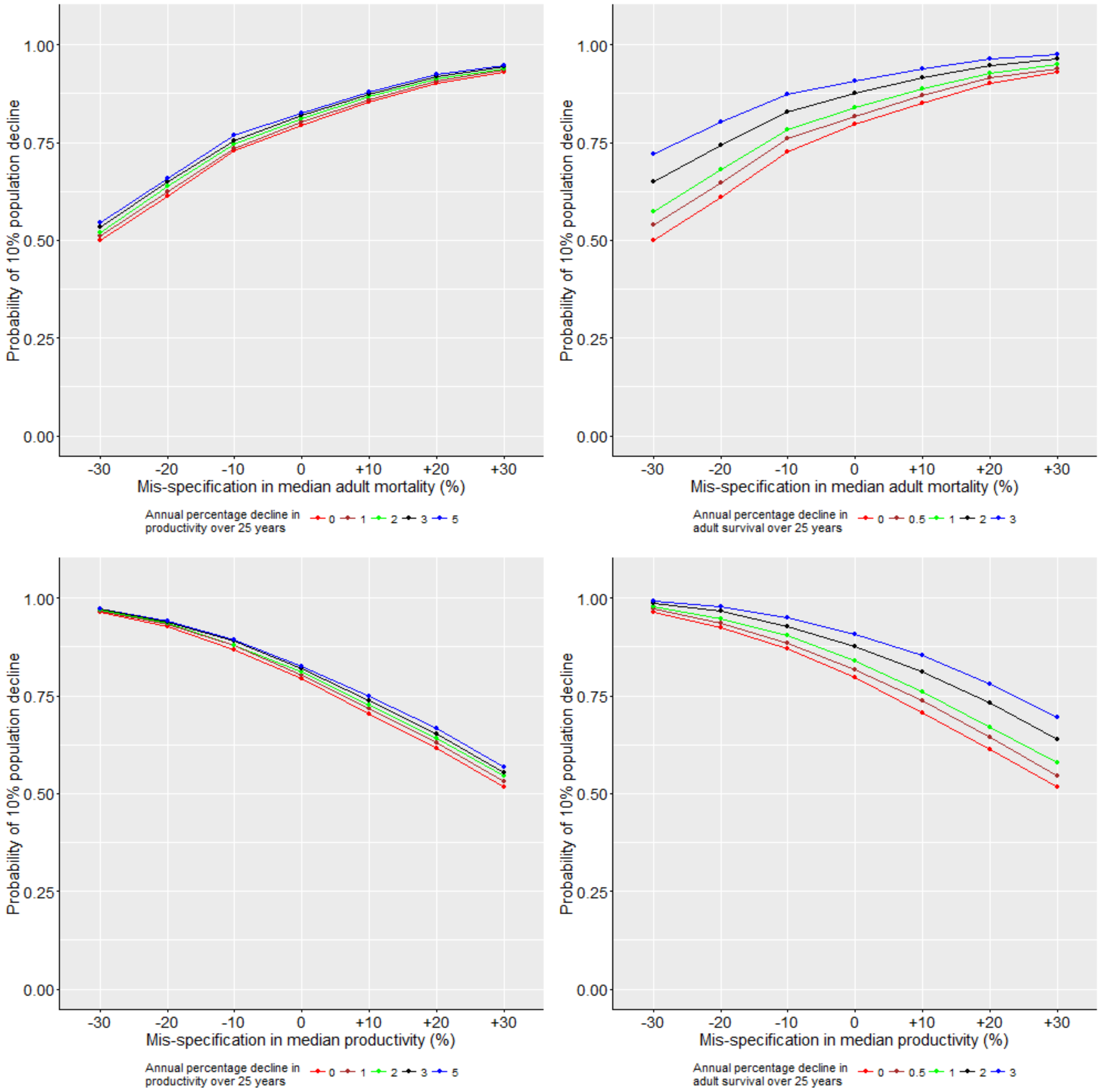


Figure A2.13f. PVA Metric E2 for St Abb’s Shags – probability of population decline greater than 25% from 2016-2041.

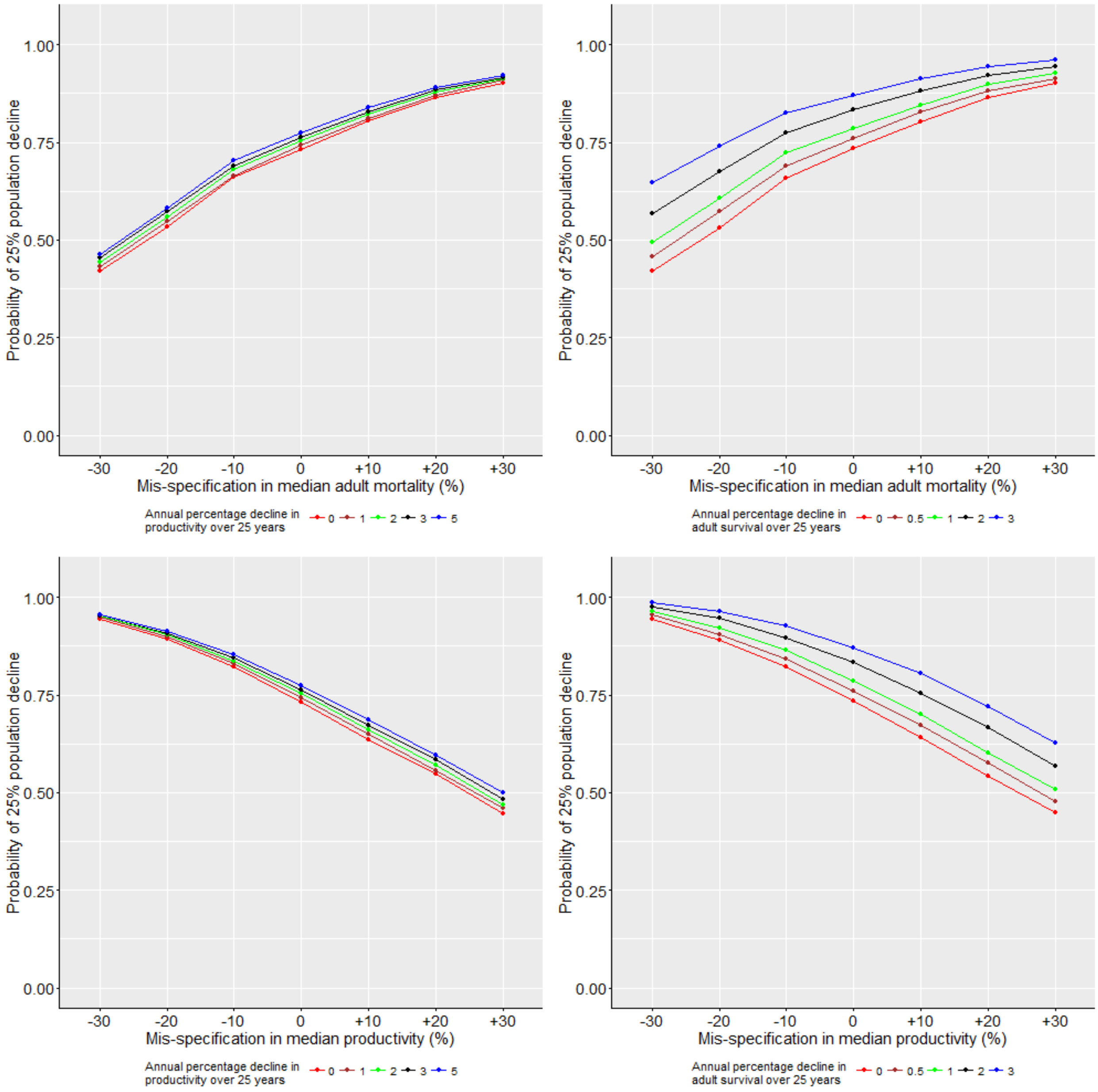


Figure A2.13g. PVA Metric E3 for St Abb's Shags – probability of population decline greater than 50% from 2016-2041.

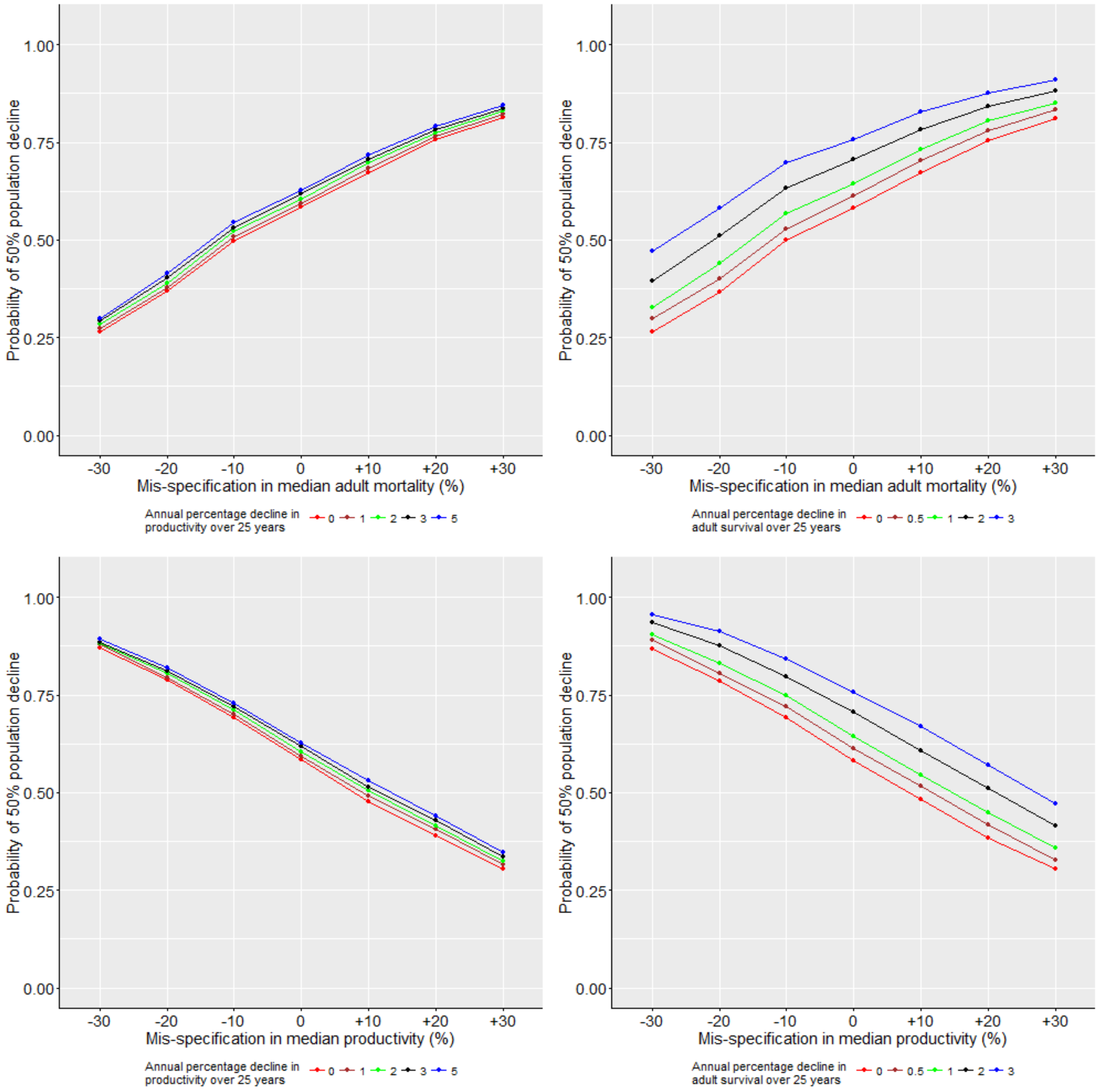
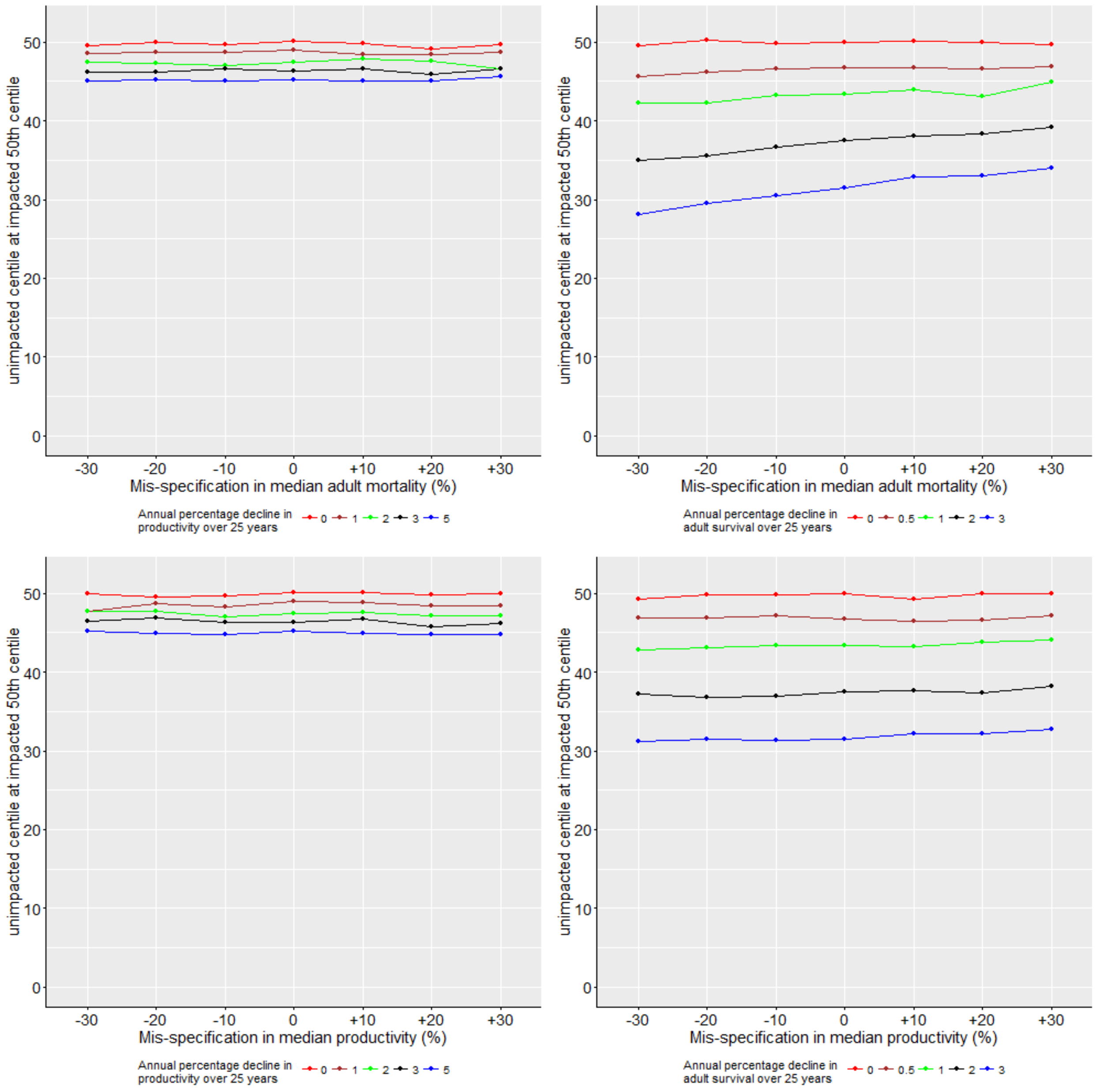


Figure A2.13h. PVA Metric F for St Abb’s Shags – centile from un-impacted population size equal to the 50th centile of the impacted population size, at 2041.



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This report presents the results of marine and freshwater scientific work carried out for Marine Scotland under external commission.

Developing an avian collision risk model to incorporate variability and uncertainty

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Part 1: Model Description

Background

As wind energy developments increase globally the potential associated environmental impacts are receiving considerable attention, particularly avian impacts. These potential impacts on bird populations can be grouped into three main types: direct mortality due to collision with turbines/infrastructure; physical habitat modification and/or loss; and behavioural responses of birds to turbines (Fox *et al.* 2006; Langston 2013). Focussing on avian collision, a variety of methods have been developed to aid the assessment of the risk of collision, including collision risk models.

After extensively reviewing both the peer-reviewed scientific literature and grey literature, 10 distinct collision risk models referring to birds and wind turbines were identified, the earliest dating back to 1996 (Tucker 1996). At their core, most avian collision risk models include a calculation of the probability of a collision occurring (assuming no evasive action or avoidance behaviour) and often also a measure of the number of birds at risk, if an estimate of likely collision events is to be calculated. The probability of collision is generally based on the probability of a turbine blade occupying the same space as the bird during the time that the bird takes to pass through the rotor swept area. This therefore relies upon information on both bird and wind turbine characteristics such as bird morphometrics and flight speed, turbine rotor speed and size, etc.

In the UK, the most frequently used avian collision risk model is commonly known as 'the Band model' (Band, Madders & Whitfield 2007) and was originally conceived in 1995. Since then it has undergone several iterations with the most recent associated with the Strategic Ornithological Support Services (SOSS) (Band 2012a; b). The Band model (Band 2012b) provides four different options for calculating collision risk.

- Option 1 - Basic model, i.e. assuming that a uniform distribution of flight heights between the lowest and the highest levels of the rotors and using the proportion of birds at risk height as derived from site survey.
- Option 2 - Basic model, but using the proportion of birds at risk height as derived from a generic flight height distribution provided.
- Option 3 - Extended model and using a generic flight height distribution.
- Option 4 - Extended model and using a flight height distribution generated from site survey.

The most recent update of the Band model guidance also provides an approach under which uncertainty can be expressed. However, this approach is relatively simplistic and can only be applied when the sources of variability are independent of one another. Furthermore, although provided, it is not routinely followed and so could be improved upon. From undertaking interviews with stakeholders (for summary see Appendix 1), it was established that a new collision risk model that was fundamentally different was not required by the industry and the Band model was considered generally fit for purpose. However, although the majority of the stakeholders questioned did not consider major changes necessary, the general opinion was that if it were possible to incorporate uncertainty into the modelling process, it would be beneficial. The main reasoning for this was that expressing collisions as a single number does not sufficiently represent the complexity of the situation. In addition, it is known that the Band model is sensitive to the choice of input parameters (Chamberlain *et al.* 2006). Variability in input parameters such as bird density, flight speed and turbine rotor speed are likely to contribute uncertainty to the final collision estimates. Sensitivity analyses of both the basic and extended options of the Band model are provided in Appendix 2.

General purpose of model update

The general purpose of this collision risk model update is to further develop the application of the Band model using a simulation approach to incorporate variability and uncertainty. In this report we refer to variability as the inherent heterogeneity of the environment and uncertainty as a lack of data or incomplete knowledge. The simulation model randomly samples from distributions for each of the model parameters and the simulations can then be used to derive average collision estimates, with associated confidence intervals. The model update will therefore allow for a better understanding of the uncertainty associated with the predicted collision impact of a wind farm development and provide confidence limits, something which has previously been absent. In addition, the incorporation of uncertainty would reduce the possibility that a collision estimate was driven by the choice of a single input parameter value. Ultimately, the update should aid streamlining of the planning/consenting stages of a development by providing information not only on the magnitude of collisions i.e. the number of collision events, but also the likelihood of that number of collisions occurring.

In this model update, variability and uncertainty are considered together in combination, rather than separately. Some model input parameters will have associated variability, for example bird body length, others may be expected to be point estimates with associated uncertainty, such as turbine rotor radius, and some parameters may have both variability and uncertainty. Ideally it would be possible to differentiate between variability and uncertainty but at present this is not possible due to a lack of data. However, including variability and uncertainty in combination in the model still provides a significant step forward.

The report describes the data required, and the methods used, to estimate collision risk. It is accompanied by a worked example and R code (available at http://dx.doi.org/10.7489/1657-1), which enables the collision risk calculations to be performed in a standardised and reproducible way.

Model format

Whereas previous iterations of the Band model have used Microsoft Excel, the collision risk model updated presented uses R http://www.r-project.org. Opinions given during stakeholder interviews (for summary see Appendix 1) were that the Excel spreadsheet was difficult to use at times and there was the potential for errors to be easily introduced into calculations, particularly if the spreadsheet did not update correctly when new input parameters were entered. In addition, the Excel spreadsheet does not allow results to be reproduced easily making auditing onerous, as values have to be entered manually for each occasion or scenario. Using R enables reproducible methods and results as code and data are provided along with the computational environment used. This improves understanding and allows verification of results, therefore increasing transparency.

Relationship to previous guidance on collision risk modelling

The model described and presented in this document is an update to the Band collision risk model (Band 2012b) which was most recently updated as part of SOSS. The mechanistic details of the Band model have not been altered and form the core of the model update described below.

The guidance (Band 2012b) states clearly that the collision estimate should be a best-estimate rather than a worst-case scenario.

“This guidance does not recommend use of ‘worst case’ assumptions at every stage. These can lead to an overly pessimistic result, and one in which the source of the difficulty is often concealed. Rather, it is recommended that ‘best estimates’ are deployed, and with them an analysis of the uncertainty or variability surrounding each estimate and the range within which the collision risk can be assessed with confidence. In stating such a range, the aspiration should be to pitch that at a 95% confidence level, that is, so that there is 95% likelihood that the collision risk falls within the specified range. However, given the uncertainties and variability in source data, and the limited firm information on bird avoidance behaviour, it seems likely that for many aspects the range of uncertainty may have to be the product of expert judgement, rather than derived from statistical analysis.”

The model update presented in this document follows this principle by using ranges of values rather than a single, ‘worst-case’ scenario.

The previous guidance (Band 2012b) presented a method to express overall uncertainty in collision estimates (stage F), as there are a large number of sources of variability or uncertainty. Cook *et al.* (2012) and Johnston *et al.* (2014) are key resources to include in this process as they provide data with confidence limits. However, the uptake of stage F in the collision estimation process appears to have been minimal. There have also been cases of its misapplication with estimates presented with implausible confidence limits such as 40±100 collisions, suggesting that negative numbers of collisions are possible. In addition, the method for expressing uncertainty suggested in stage F does so *post hoc*, rather than being integrated in to the model itself. Also, combining sources of uncertainty as suggested is only applicable when parameters are independent. The model update described in this document further develops the concepts presented in stage F of the previous guidance.

Before the most recent iteration of the Band model (Band 2012b) was conceived, McAdam (2005) produced a model which incorporated species specific flight height distributions. Variation in flight height has now been incorporated in to the most recent update (Band 2012b), however, the model produced by McAdam (2005) used Monte Carlo simulation to obtain 500 samples. Using Monte Carlo methods allowed for the production of summary statistics rather than single collision estimates, as well as probability distributions of events (numbers of collisions) occurring. It was also executed using R rather than Excel. In the model update presented below, the method of Monte Carlo simulation used by McAdam (2005) has been applied to the most recent version of the Band model to allow the incorporation of uncertain parameter values.

General data requirements

The model update is based on the Band model (Band 2012b) therefore the types of data required are the same:

- Bird survey – data on the number of birds flying through or around the site, and their flight height
- Bird specification – details on bird morphology and flight speed
- Turbine specification – details on the number, size and rotation speed of turbine blades
- Bird behaviour – prediction of likely change due to wind farm, e.g. avoidance

The crucial difference from previous iterations of the Band model is that rather than using a single value for a given input parameter, for example bird flight speed, this update of the model randomly samples from a distribution of values. Using the randomly sampled parameter values, a collision risk estimate is calculated. This process is then repeated numerous times to produce a distribution of collision estimate for which summary statistics i.e. average and spread, can be calculated. Whereas the previous guidance and methods provided a measure of uncertainty post hoc, uncertainty is now incorporated in the modelling procedure itself with this update. Therefore, information regarding uncertainty in the data is required to be entered into the model.

Where possible, and when suitable, a mean and standard deviation should be provided for input parameters. These should capture the uncertainty within the data. For example, if the maximum turbine blade width has not been decided upon but is likely to be 5 metres then a mean = 5 should be provided with a standard deviation which describes the uncertainty and possible values. For this example, a mean of 5 and standard deviation of 0.3 would give a minimum of approximately 4 metres, and a maximum of approximately 6 metres. If there is no uncertainty and it is definite that the maximum blade width is to be 5 metres then a value of 5 should be entered as the mean and either 0 entered as the standard deviation or it left blank.

To incorporate uncertainty into the collision risk estimate, a mean and standard deviation will be required for the following parameters. Attention should be paid to the units of measure.

Table 1: Bird-related parameters

Parameter	**Units**	**Description/Notes**
Length	m (metres)	
Wingspan	m (metres)	
Flight speed	m/sec	Available from telemetry data or wind tunnel experiments
Flight type		Flapping or gliding
Nocturnal activity	Proportion e.g. 0.5 for 50%	Available from telemetry data or visual observations
Proportion at collision risk height	Proportion e.g. 0.5 for 50%	
Flight height distribution	Proportion e.g. 0.5 for 50%	Distribution curves from which the proportion of birds flying within 1 metre height bands are calculated. Data provided by BTO (Johnston *et al.* 2014). (See below)
Avoidance rate	Proportion e.g. 0.5 for 50%	Suggested values available from MSS avoidance report
Bird density	Birds/km^2	Birds in flight in daytime, taken from survey data

Table 2: Turbine-related parameters

Parameter	**Units**	**Description/Notes**
Rotor radius	m (metres)	Measured from the axis of rotation to blade tip.
Hub height	m (metres)	Sum of rotor radius and minimum blade clearance above HAT. (See below)
Max. blade chord width	m (metres)	
Rotation speed	rpm	See below
Blade pitch	Degrees relative to rotor plane	See below
Turbine operation time	Proportion e.g. 0.5 for 50%	Requires both information on wind availability and maintenance down time.

This model requires information on flight height distributions, if options 2, 3 or 4 are to be used. A generic flight height distribution is presented with the SOSS guidance (Band 2012a; b; Cook *et al.* 2012), however this does not provide information on the uncertainty associated with the distribution. Johnston *et al.* (2014) used a bootstrapping technique to provide confidence limits associated with the generic flight distribution and these bootstraps can be used within this model update to provide uncertainty associated with the flight height distribution curve. For each iteration of the model, a curve produced from a bootstrap sample is re-sampled and used. It is possible to use this update to calculate a collision risk estimate using option 4, should site-specific data on flight height distributions be available.

The model also requires information on wind speed ($m.s^{-1}$) at the proposed site as well as the relationship between rotor speed and wind speed and turbine pitch and wind speed. This allows rotor speed and pitch to be linked both to the wind speed and also to each other. This is achieved through the provision of data similar to that in table 3 (below), describing the relationship between wind speed and rotor speed and pitch, as well as information on wind speed at the site.

Table 3: Example data describing relationship between wind speed, rotor speed and blade pitch.

Wind speed (m/s)	Rotor Speed (rpm)	Pitch (degrees)
0	0	90
1	0	90
2	0	90
3	6	0
4	6	0
5	6	2
6	8	4
…	…	…

The turbine operation time is wind availability minus maintenance down time. Wind availability should be provided as a constant i.e. proportion of time the wind conditions allow for turbine operation and should be available from meteorological data. Maintenance time should be provided as a monthly mean and standard variation as it is expected that there will be uncertainty and variability surrounding maintenance.

Hub height is the distance from highest astronomical tide (HAT) to the axis of rotation of the turbine. This distance comprises the rotor radius and the distance between the minimum rotor tip height and HAT. Therefore, as rotor radius is already entered into the model, it is importantly only the distance component from HAT to the minimum rotor height that is required here and not the total hub height.

Calculating collision risk

As stated previously, this model is an update of the Band model. For more information on the Band model refer to (Band 2012a; b) and associated information on the SOSS website http://www.bto.org/science/wetland-and-marine/soss/projects.

Monte Carlo simulation

The model update presented herein uses Monte Carlo simulation. Monte Carlo simulation is a computational technique that uses random sampling to produce numerical results, and in this model update, is used to obtain values for uncertain input parameters, for example flight speed or bird length. These values are then used in the Band model. For each set of random samples, a collision estimate is calculated. Therefore if the simulation is run for 100 iterations, 100 sets of random input parameters will be sampled and 100 collision risk estimates calculated, instead of a single value. Monte Carlo simulation therefore allows for the presentation of a range of possible outcomes, when there is uncertainty surrounding the input data, and produces distributions of possible collision estimates. The distribution data can then be further re-sampled and used in stochastic population models, should this be required.

Sampling distributions

With the exception of rotor speed, pitch and flight height distributions, input values for the Band model are sampled from probability distributions. These distributions are parameterised using data provided by the user and have been constrained to the Normal distribution, or in cases where negative values are not plausible, the truncated Normal distribution. The user defines the mean or expected value and a standard deviation to describe the variation about the mean. Values in the middle near the mean are most likely to occur. The decision to use the Normal distribution was made on the basis of ease of parameterisation for the user as well as suitability. The Normal distribution was considered more suitable that a uniform distribution because in most cases it is expected that there will be a more likely value, and the uniform distribution, where all values are equally likely, would therefore enter more uncertainty than realistic into the model. It is however accepted that in all cases, the Normal distribution may not be the most suitable distribution, but there is a balance to be achieved between suitability and ease of use.

Collision risk options

The Band model provides four different options for calculating collision risk (Band 2012b). Options 1, 2 and 3 are the most frequently used. The model update calculates estimates for both the basic (options 1 and 2) and extended (option 3) versions of the Band model. It is possible to use this update to calculate a collision risk estimate using option 4, should site-specific data on flight height distributions be available. However, this would require a large amount of data collection, to provide information on variation in flight height distribution therefore the default option does not include option 4.

Running the model

As well as being designed to run numerous simulations of the Band model, this update is designed to loop through multiple species and multiple turbine designs automatically. Therefore once the initial user information is entered and the model begins, the user is not required to enter any further information and the results will be saved automatically to the location specified by the user. The number of results obtained will depend on the number of different turbine designs and species entered.

Model Output

The model outputs information on the expected numbers of collisions. The information is provided both as tables and figures. Descriptions of the outputs are listed below and illustrated examples are provided in the worked example.

TABLES

1. Overall summary table of collisions by species, turbine and model option. Results are presented as mean, standard deviation (SD) and coefficient of variation (CV), and median and inter quartile range (IQR).
2. Monthly summaries of collisions. Separate tables are produced according to species, turbine and model option for example 6_Black_legged_Kittiwake_monthlySummaryOpt3.csv. Results are presented as mean, standard deviation (SD) and coefficient of variation (CV), and median and inter quartile range (IQR).
3. Summary of sampled bird parameters by species, turbine and model option presented as mean and standard deviation (SD), and median and inter quartile range (IQR).
4. Summary of sampled turbine parameters by species, turbine and model option presented as mean and standard deviation (SD), and median and inter quartile range (IQR).

FIGURES

1. 3-panel boxplots of monthly collisions for model options 1, 2 and 3 by species, and turbine type.
2. Density plots of numbers of collisions by species, and turbine type. A density curve is plotted for each of the 3 model options.
3. If 2 or more turbine models are included, then a 3-panel figure will be produced for each species, with the panels representing model options 1, 2 and 3 and each panel containing density plots for the different turbines included.

In addition to the collision estimates, the model also saves a copy of the input files which were entered into the model, as well as a summary of the randomly sampled input parameter values. This would therefore allow for the model to be re-run and results verified (if required). It also outputs a text file stating the time elapsed between the start and the end of the model, the number of iterations, the species for which the model was run and also the different turbines i.e. 6MW, 8MW, etc. if more than one turbine type was specified.

Future work

During this project, an update to the Band collision risk model (Band 2012b) has been developed, however it is accepted that there are still aspects which could be improved further in the future with additional updates, particularly with improved data collection methods and understanding of the interactions between birds and wind farms. These are listed below.

1. Wind speed data: This model update has taken a step forward from previous iterations of the Band model by including the relationship between wind speed data and both rotor speed and rotor pitch, however there are still improvements which could be made. Due to a lack of clarity in the availability and format of site-specific wind speed data which is available to developers it was decided that in this model update, wind speed would be sampled from a truncated Normal distribution, parameterised by a mean and standard deviation set by the user. In the future, if consensus could be reached on wind data availability and format, a summary of the raw wind speed data could be used, rather than using it to parameterise a sampling distribution. If this were the case, then it would also be possible to programme the model to automatically calculate wind availability from the wind speed data, rather than this being entered manually.

2. Monthly vs. annual input parameters: The current model uses annual estimates for the majority of input parameters such as bird flight speed and percentage of nocturnal activity. It is possible that these may differ between the breeding and non-breeding season, and vary monthly, and in response to wind speed. However, at present it was considered that data of sufficient quality were not available for enough parameters on a monthly basis to warrant including this in the model for all. Should this be the case, including monthly values for all parameters could introduce unrealistic precision into the model; therefore only monthly values were included for bird density and turbine operation time. In the future it might be more appropriate to consider all input parameters on a monthly basis.

3. Linking wind speed and flight speed: Within this model update bird flight speed was not linked to wind speed. This alteration could improve the model, however little data is available regarding bird flight speeds, especially in relation to wind speed, though more flight speed data are becoming available as the number of projects using telemetry e.g. GPS tags, increases . The link between flight speed and wind speed was however included in the model produced by McAdam (2005), therefore it would be possible to include this relationship in future updates, should sufficient data become available.

4. Validate the model: Due to the difficulties associated with collecting collision data offshore, as yet, it has not been possible to validate this model update. This is the case for previous versions of the Band model and also collision risk models in general. In the guidance supplied alongside the 2012 update to the Band model, Band (2012b) highlights that there is likely to be uncertainty as a result of simplifications in the model itself. As an estimate, it is suggested that this may be in the region of 20%. By using the results of projects, such as the bird collision avoidance component of the Offshore Renewables Joint Industry Programme

(ORJIP) in the UK, to validate the model, it may be possible to quantify this uncertainty more accurately and reduce it through further refinements to the model.

5. Sensitivity analysis: Whilst it is possible to perform a manual sensitivity analysis on the model update (results available in Appendix 2), it would be useful in the future to have the utility to perform a sensitivity analysis as a matter of course during the assessment of collision risk. This would offer users the ability to highlight which parameters had the strongest influence on the final collision estimates and consider how best to target data collection in order to reduce uncertainty. It may also enable developers to plan mitigation strategies, for example by demonstrating how using fewer, larger turbines may reduce collision risk.

Part 2: Worked Example

Downloading R

Whereas previous iterations of the Band model have used Microsoft Excel, the collision risk model update presented uses R. To get the most recent version of R, go to the R website http://www.r-project.org and click the 'CRAN' link on the left hand side. Select a mirror site near you from the list provided, click the 'Windows' link on the next page, then the 'base' link on the following page, and then download the R installer from the link 'Download R 3.1.2 for Windows' (note that the version number you see may be different than this example).

The collision risk model (available at http://dx.doi.org/10.7489/1657-1) requires the installation of an R package “msm” to allow sampling from a truncated normal distribution. This should be installed before the model is run.

To install the required package type the following

```
install.packages(“msm”)
```

into the R console and press return.

Once installed, this step should not be required again and the package will be loaded from within the model script.

Imagined scenario

This example is fictitious. The results are not characteristic of collision risks at any particular site.

A wind farm is planned for the North Sea. The imagined area has a width of 10 km and it is intended that the planned wind farm will generate 600MW. The location of the development area is at 55.8 degrees latitude and has a tidal offset of 2.5 metres. One of the seabirds present at the site and considered sensitive to collision risk is the black-legged kittiwake (*Rissa tridactyla*). This worked example estimates a collision risk estimate for kittiwake.

The wind farm is still in the design phase and there is some uncertainty surrounding the turbine design.

The collision risk model

The collision risk model update is provided as R code. The majority of the model is written as functions which are provided in individual files in a folder named ‘scripts’. These files do not need to be altered. In terms of code, the ONLY file that needs to be altered in order to run the collision risk model is "BandModel.R". All other files are called from R within this script.

In addition to a folder called ‘scripts’ and the code BandModel.R, there should also be a folder named ‘data’ that contains the data files required to run the model. The folder should contain the following:

*BirdData.csv - contains the biometric & flight speed data for species of interest. At present data for 12 species are included (Fulmar, Gannet, Lesser Black-backed Gull, Herring Gull, Great Black-backed Gull, Black-legged Kittiwake, Guillemot, Razorbill, Little Auk, and Puffin)

Species can be added as required, however, it is not necessary to remove data for species which are not being considered. It is important that the species names used are taken from the file BirdData.csv and are used throughout, otherwise the model will not recognise the species. The species names used in the code are as follows:

Species (scientific name)	**Code name used in model**
Fulmar (*Fulmarus glacialis*)	Northern_Fulmar
Gannet (*Morus bassanus*)	Northern_Gannet
Lesser Black-backed Gull (*Larus fuscus*)	Lesser_Black_Backed_Gull
Herring Gull (*Larus argentatus*)	Herring_Gull
Great Black-backed Gull (*Larus marinus*)	Great_Black_backed_Gull
Black-legged Kittiwake (*Rissa tridactyla*)	Black_legged_Kittiwake
Guillemot (*Uria aalge*)	Common_Guillemot
Razorbill (*Alca torda*)	Razorbill
Little Auk (*Alle alle*)	Little_Auk
Puffin (*Fratercula arctica*)	Atlantic_Puffin
Arctic skua (*Stercorarius parasiticus*)	Arctic_Skua
Great skua (*Stercorarius skua*)	Great_Skua

*CountData.csv - count data for the species of interest, should be entered as densities (Birds/km^2). Make sure that biometric data for each species included here is available in BirdData.csv

*FlightHeight.csv - modelled flight height distributions for 21 species of seabird/seaduck etc. taken from Johnston *et al.* (2014)

*TurbineData.csv - should contain information on the size etc. of the turbines being considered. Each row represents a different turbine model to be considered. The name should be the MW rating of the turbine.

* [insert species name]_ht.csv – should contain bootstrapped flight height distribution data from Johnston *et al*. (2014) for the species required. A separate file is required for each species.

* windpower_[insert turbine model name].csv – should contain the relationship between wind speed, rotor speed and pitch. A separate file is required for each turbine model/type.

Data requirements

BIRD DATA: these data are entered in to BirdData.csv

Bird length (Body_Length, Body_LengthSD): the bird body length data were taken from the Concise Birds of the Western Palearctic (OUP, 1994), BTO bird facts and other sources. A mean and standard deviation were estimated from these sources. Mean = 0.39 metres, standard deviation = 0.005.

Wing span (Wingspan, WingspanSD): the wing span data were taken from the Concise Birds of the Western Palearctic (OUP, 1994), BTO bird facts and other sources. A mean and standard deviation were estimated from these sources. Mean = 1.08 metres, standard deviation = 0.04.

Flight speed data (Flight_Speed, Flight_SpeedSD): the flight speed data used in this worked example were taken from the RSPB FAME project and were collected using GPS tags. The data were used to parameterise a normal distribution. A mean flight speed for each tagged bird was estimated (to remove variation within individual) and then an overall mean of these values was calculated. Mean = 7.26 $m.s^{-1}$, standard deviation = 1.5.

Nocturnal activity (Nocturnal_Activity, Nocturnal_ActivitySD): the data used in this worked example to estimate the proportion of nocturnal activity were taken from the RSPB FAME project and were collected using GPS tags. All records away from the nest and travelling at speeds considered to be flying were used. The data were used to parameterise a normal distribution. Mean = 0.033, standard deviation = 0.0045.

Proportion at collision risk height (Prop_CRH_Obs, Prop_CRH_ObsSD): these data are required for the basic model (option 1). A mean and standard deviation were calculated using the data within Black_legged_Kittiwake_ht.csv. For each bootstrapped distribution curve, the proportion of birds between the average minimum and average maximum rotor tip height was summed. From these values, a mean and standard deviation was calculated. Mean = 0.06, standard deviation = 0.009.

Avoidance (AvoidanceBasic, AvoidanceBasicSD, AvoidanceExtended, AvoidanceExtendedSD): This is the probability that a bird on a collision course with a turbine will take evading action to avoid collision. The data were taken from Cook *et al.* (2014) and for kittiwake, we used the 'all gulls' avoidance rate. The avoidance rate differs between the basic and extended models and a mean and standard deviation are required for both. Mean (basic) = 0.9893 and standard deviation (basic) = 0.0007; mean (extended) = 0.9672 and standard deviation (extended) = 0.0018.

COUNT DATA: these data are entered in to CountData.csv and are the number of birds in flight in the daytime presented as birds/km^2. A mean and standard deviation are required. For this worked example, data were taken from the Creyke Beck A Environmental Statement.

Month	**Mean**	**SD**
Jan	0.97	0.67
Feb	1.04	0.75
Mar	1.15	0.78
Apr	0.48	0.36
May	0.56	0.58
Jun	0.63	0.45
Jul	0.68	0.47
Aug	0.64	0.47
Sep	0.53	0.39
Oct	1.20	0.78
Nov	1.02	0.61
Dec	0.99	0.7

FLIGHT HEIGHT DISTRIBUTION DATA: these data are contained within FlightHeight.csv and the individual species files such as Black_legged_Kittiwake_ht.csv and are required for options 2 and 3. The data within FlightHeight.csv are provided by the British Trust for Ornithology (Johnston *et al*, 2014) and are generic flight height distributions. The file provides data on the proportion of birds within 1m height bands for 21 species. For example,

Height (m)	Black_legged_Kittiwake
1	0.089904
2	0.081107
3	0.072538
4	0.067068
5	0.061008
6	0.055579
7	0.050707
…	…

Confidence intervals were calculated around these best fit/generic flight height distributions using a bootstrapping approach, randomly sampling from the original dataset each time. These bootstraps are provided for each species in separate files, for example Black_legged_Kittiwake_ht.csv and are used to include uncertainty surrounding the flight height distribution.

TURBINE DATA: these data are entered into TurbineData.csv and are also contained within the turbine-specific files such as windpower_6. Names in parentheses are the names in the data files and scripts.

Rotor radius (RotorRadius, RotorRadiusSD): this is measured from the axis of rotation to blade tip and was taken from expert opinion. Mean = 80 and Standard deviation = 5

Hub height (HubHeightAdd, HubHeightAddSD): This is the measure that in addition to rotor radius sums to give the distance from HAT to the axis of rotation and was taken from expert opinion. Mean = 26.5 and Standard deviation = 2.

Maximum blade chord width (BladeWidth, BladeWidthSD): This is the maximum width of the rotor blade and was taken from expert opinion. Mean = 5.5 and Standard deviation = 0.3.

Turbine operation time (example for January is JanOp, JanOpMean, JanOpSD): This included both information on wind availability (JanOp) which is considered a constant and maintenance down time (JanOpMean, JanOpSD) which included uncertainty. Data were taken from the Inch Cape Environmental Statement.

Month	**Wind availability**	**Mean maintenance**	**SD maintenance**
Jan	96.28	6.3	2
Feb	96.53	6.3	2
Mar	95.83	6.3	2
Apr	92.78	6.3	2
May	90.86	6.3	2
Jun	92.22	6.3	2
Jul	89.11	6.3	2
Aug	89.92	6.3	2
Sep	93.71	6.3	2
Oct	96.14	6.3	2
Nov	97.14	6.3	2
Dec	96.41	6.3	2

Rotation speed (RotationSpeed, RotationSpeedSD): It is possible to specify a mean rotor speed and standard deviation, however the default model uses the relationship between wind speed and rotor speed provided to calculate the specific rotor speeds, therefore RotationSpeed and RotationSpeedSD in TurbineData.csv should be left blank. The relationship should be provided in a csv file named windpower_[insert model name here] for example windpower_6.csv

The table below shows an example relationship between wind speed, rotor speed and pitch. It was constructed using expert opinion as an example table and used within this worked example. It does not relate to a specific turbine specification currently available.

Wind speed (m/s)	Rotor Speed	Pitch
0	0	90
1	0	90
2	0	90
3	6	0
4	6	0
5	6	2
6	8	4
…	…	…

Blade pitch (Pitch, PitchSD): As for rotor speed, it is possible to specify a mean pitch and standard deviation, however the default model uses the relationship between wind speed and pitch provided to calculate the specific rotor pitch, therefore Pitch and PitchSD in TurbineData.csv should be left blank. The relationship should be provided in a csv file named windpower_[insert model name here] for example windpower_6.csv (as above).

Model set up

Before the model can be run it requires information to be entered into the file "BandModel.txt".

1. Set working directory

First, set the working directory. This is the location where the folders ‘scripts’ and ‘data’ have been saved as well as ‘BandModel.R’. For example,

```
setwd("F:\\BAND CRM For R")
```

This step directs R to all the files and data that are required to run the model therefore all the files required much be within this directory.

2. Set results folder

The model will save output to a folder. Set the name of the results folder, for example, the name of the development. For example,

```
results_folder <- "windfarm1"
```

If no name is specified the model will default to the date. WARNING: If the model is run several times on the same day and no folder name is specified, it will over-write files in the folder.

3. Set model components

Next set the model components. These include:

The number of iterations the model simulation will execute, for example 1000

```
iter<- 1000
```

The species to include, for example kittiwake

```
CRSpecies = "Black_legged_Kittiwake"
```

If more species were to be included this would look like

```
CRSpecies = c("Black_legged_Kittiwake", "Northern_Gannet", "Arctic_Skua")
```

The target power (in MW) to be generated within wind farm, for example 600MW. This is used in conjunction with the turbine name i.e. 6 if a 6MW turbine, to calculate the number of turbines in the array.

```
TPower = 600
```

Large array correction (Yes/No), for example

```
LargeArrayCorrection = "yes"
```

The wind farm width (km), for example 10km

```
WFWidth = 10
```

The proportion of bird flights up/downwind, for example 50%

```
Prop_Upwind = 0.5
```

The latitude of the wind farm in decimal degrees, for example 55.8 degrees. This is used to calculate day length at the site location throughout the year.

```
Latitude = 55.8
```

The tidal offset in metres (to correct for flight heights being calculated in relation to mean sea-level and turbine dimensions being calculated in relation to Highest Astronomical Tide), for example 2.5 metres

```
TideOff = 2.5
```

4. Parameterise wind speed sampling distribution

The model uses wind speed data to calculate rotor speed and pitch. Wind speed data are therefore required. At the time of production it was unclear what format wind speed data would be available to wind farm developers. To avoid inconsistencies, the model samples wind speed from a truncated normal distribution parameterised by the user. The mean wind speed ($m.s^{-1}$) and standard deviation are required to be set, for example

```
windSpeedMean<- 7.74
windSpeedSD<- 3.2
```

It is expected that these will be obtained from met mast data or other sources of wind speed data such as NASA's Modern-Era Retrospective Analysis for Research and Applications (MERRA).

5. Running the model…

Once you have set the working directory and entered all of the necessary information, all that is needed to run the model is to copy and paste all of BandModel.R into the R console, or alternatively type

```
source("*****EnterMyDirectoryHere****\\BandModel.R")
```

in the R console, and press return.

The code is designed to loop through multiple species and multiple turbine designs in a single step. The number of results obtained will depend on the number of different turbine designs entered in TurbineData.csv and the number of different species for which data are entered and listed.

A progress bar will provide an indication of progress and at the end of the model, the time elapsed since the model was started will be displayed.

Model Output

The model outputs information on the expected numbers of collisions. The information is provided both as tables and figures, and according to the species and turbine designs entered into the model. Results are provided for the basic (options 1 and 2) and extended (option 3) versions of the model.

- Option 1 - using the basic model, i.e. assuming that a uniform distribution of flight heights between the lowest and the highest levels of the rotors and using the proportion of birds at risk height as derived from site survey.
- Option 2 - again using the basic model, but using the proportion of birds at risk height as derived from the generic flight height distribution provided.
- Option 3 - using the extended model and using the generic flight height distribution.

Tables

The file names of the tables indicate the type of information contained.

1. CollisionEstimates.csv: Overall summary table of collisions by species, turbine and model option. Results are presented as mean, standard deviation (SD) and coefficient of variation (CV), and median and inter quartile range (IQR).

Species	Turbine	Option	Mean	SD	CV	Median	IQR
Black_legged_Kittiwake	6	1	45.6455	12.5448	27.4831	45.1206	16.5493
Black_legged_Kittiwake	6	2	42.4292	14.5621	34.3210	40.3032	18.671
Black_legged_Kittiwake	6	3	35.4131	13.1271	37.0685	33.0278	16.466

2. 6_Black_legged_Kittiwake_monthlySummaryOpt1.csv : monthly summaries of collisions. Separate tables are produced according to species, turbine and model option. Results are presented as mean, standard deviation (SD) and coefficient of variation (CV), and median and inter quartile range (IQR).

Month	Mean	SD	CV	Median	IQR
Jan	3.080382	1.753642	56.92937	2.970413	2.441757
Feb	3.590156	2.012736	56.06263	3.439408	3.043374
Mar	4.884338	2.76244	56.55711	4.716011	4.134139
Apr	2.890070	1.763726	61.02710	2.648057	2.297968
May	4.255352	2.845274	66.86343	3.771635	3.664073
Jun	4.458946	2.63927	59.19044	4.172458	3.708436
Jul	4.529034	2.769847	61.15756	4.132559	3.751684
Aug	4.034511	2.423247	60.06296	3.738152	3.239942
Sep	2.720596	1.625044	59.73115	2.546583	2.131306
Oct	4.650331	2.451314	52.71267	4.471203	3.288374
Nov	3.484131	1.787375	51.30045	3.386141	2.557487
Dec	3.067742	1.625422	52.98432	2.964345	2.258359

3. 6_Black_legged_Kittiwake_sampledBirdParameters.csv: Summary of sampled bird parameters by species, turbine & model option presented as mean & standard deviation (SD), and median & interquartile range (IQR).

Parameter	Mean	SD	Median	IQR
AvoidanceBasic	0.989328	0.000696	0.989341	0.000986
AvoidanceExtended	0.967215	0.00176	0.967172	0.00244
WingSpan	1.079481	0.040021	1.078701	0.054751
BodyLength	0.3898	0.005124	0.389929	0.007007
PCH	0.06006	0.009073	0.06025	0.012644
FlightSpeed	7.242055	1.472594	7.200565	1.91039
NocturnalActivity	0.033239	0.004627	0.033195	0.005833

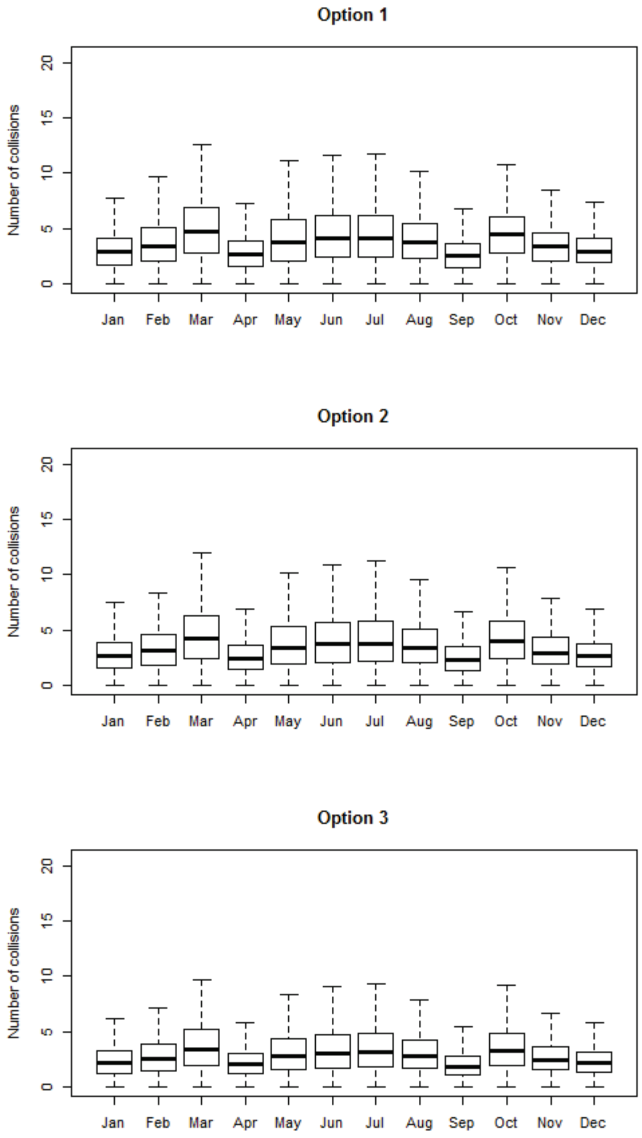
4. 6_Black_legged_Kittiwake_sampledTurbineParameters.csv: Summary of sampled turbine parameters by species, turbine and model option presented as mean and standard deviation (SD), and median and inter quartile range (IQR).

Parameter	Mean	SD	Median	IQR
RotorSpeed	7.8795	1.187502	6.8	2.3
RotorRadius	80.02557	4.641109	79.93026	6.275585
HubHeight	106.5836	4.981272	106.4803	6.57322
BladeWidth	5.492034	0.293631	5.499434	0.414762
Pitch	1.091	2.912594	0	0
JanOp	90.04184	1.992513	90.08147	2.699385
FebOp	90.32209	2.00625	90.27184	2.864814
MarOp	89.53487	1.983321	89.52079	2.567127
AprOp	86.5725	2.050986	86.58052	2.786068
MayOp	84.5539	1.99842	84.5696	2.744052
JunOp	86.04224	2.070166	86.06471	2.915727
JulOp	82.7399	2.025924	82.68036	2.643739

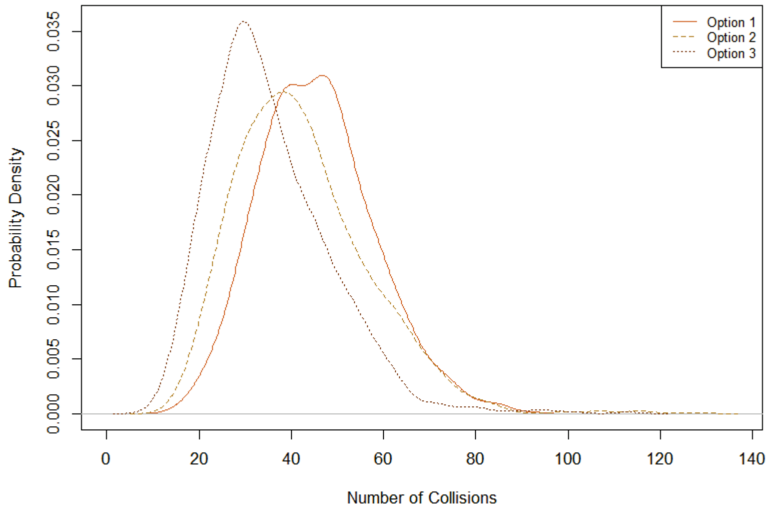
AugOp	83.69396	2.0276	83.71622	2.59691
SepOp	87.46471	2.045078	87.38634	2.880023
OctOp	89.81123	1.95635	89.78423	2.731423
NovOp	90.89083	1.979187	90.93791	2.677976
DecOp	90.04899	2.107647	90.1124	2.786412

Figures

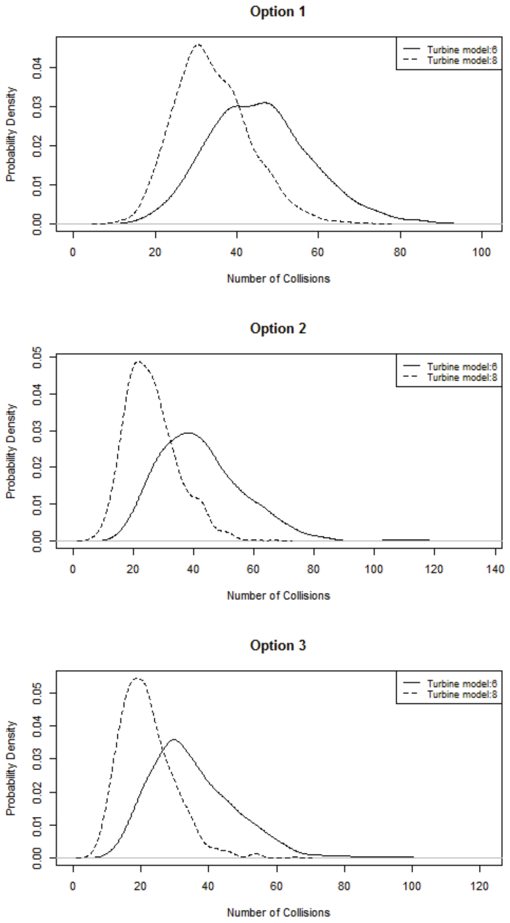
1. 6_Black_legged_Kittiwake .jpg: 3-panel boxplots of monthly collisions for model options 1, 2 and 3 by species, and turbine type (included in the file name).



2. 6_Black_legged_Kittiwake_density.jpg: Density plots of numbers of collisions by species, and turbine type (specified in file name). A density curve is plotted for each of the 3 model options.



3. Black_legged_Kittiwake.jpg: If 2 or more turbine models are included, then a 3-panel figure will be produced for each species, with the panels representing model options 1, 2 and 3 and each panel containing probability density plots for the different turbines.



In addition to the collision estimates, the model also saves a copy of the input files which were entered into the model, as well as a summary of the randomly sampled input parameter values. This would therefore allow for the model to be re-run and results verified (if required). It also outputs a text file (run.time.txt) stating the time elapsed between the start and the end of the model, the number of iterations, the species for which the model was run and also the different turbines i.e. 6MW, 8MW, etc. if more than one turbine type was specified.

```
Time difference of 1.936949 hours
"The model ran 1000 iterations"
"The following species were modelled:"
"Black_legged_Kittiwake"
"The following turbines were modelled:"
6 8
```

<u>A note on comparisons of results</u>: Whilst differences in the density curves (specifically the spread) can be compared within each model option, they should not be compared between different model options. This is because of differences in the way in which variation and uncertainty are introduced into each model option. For example, in the basic model (options 1 & 2) uncertainty in the proportion of birds at risk height is only introduced when estimating the flux rate. However, in the extended model (option 3), the uncertainty in the flight height distribution is introduced when calculating the collision integral (the extended model equivalent of the probability of collision).

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Appendix 1: Stakeholder Interviews

Purpose of interviews

To obtain views and opinions of a wide range of stakeholders involved in offshore wind, on collision risk models and modelling, particularly in relation to uncertainty and variability.

Interview questions

Conducted telephone interviews based around the following questions:

1. How much experience do you have, relating to collision risk models/modelling?
2. What collision risk models do you most regularly use or have experience of?
3. What uncertainties exist in the collision risk models that you have used?
4. What are the key uncertainties in input parameters?
5. What parameters do you think have the greatest influence on the outputs of collision risk modelling?
6. If you could, how would you improve collision risk models/modelling?
7. Would the explicit reporting of variability and uncertainty in outputs from collision risk models benefit the consenting process and discussions with regulators?

Interviews were approximately 20-30 minutes each.

Interviewees

I contacted 30 people from a range of stakeholder groups and from those I conducted 20 interviews with people from the following organisations:

BTO
DONG Energy
EDPR
MacArthur Green
Natural England
NIRAS
PMSS
Scottish Natural Heritage
Sue King Consulting
CEH
ECON
Joint Nature Conservation Committee
Marine Scotland Science
Natural Power
Pelagica
Royal Society for the Protection of Birds
Statkraft/Forewind
The Crown Estate
…and Bill Band

Results

Experience of interviewees

Question 1: How much experience do you have, relating to collision risk models/modelling?

The experience of interviewees varied from ‘intelligent client’ to model creator. All interviewees had a good understanding of the general modelling process and the use of model output though not all had conducted the modelling and run the models themselves. One person declined the offer of being interviewed because they thought they didn’t have enough experience to contribute constructively.

Question 2: What collision risk models do you most regularly use or have experience of?

All people interviewed (20) used the Band model and the associated updates. Of these, most people mentioned options 1 and 3 rather than 2 and 4. Additionally, 5 people used the Folkerts model, though less regularly, and one had an understanding of the Tucker model. These were the only models mentioned.

Uncertainties in collision risk modelling

Question 3: What uncertainties exist in the collision risk models that you have used?

This question was targeted at the broader uncertainties surrounding collision risk modelling. The following opinions were given more than once:

- Data collection methods including number and timing of surveys and the fact that surveys only occur in good weather least to a density estimate which may not capture the variability in the environment.
- The use of the Rochdale Envelope and therefore wide ranges for turbine parameters.
- How much precaution should be included?
- Bird behaviour and avoidance
- Which option of the model (or in most cases, which option of Band) is acceptable?
- Little empirical data and also no validation or comparison with post-construction data.
- The appropriate use of the model and output. The collision estimate is considered as definitive and black and white when it is supposed to be a collision risk tool.
- In the case of the Band model, what is the latest version of the model and flight height data sets to use?

Question 4: What are the key uncertainties in input parameters?

All of the input parameters were discussed and raised by the interviewees as a whole but those that occurred more than once and in descending order (most frequently highlighted first):

- Flight height data
- Avoidance
- Density
- Nocturnal activity
- Flight speed
- Rotor speed

Question 5: What parameters do you think have the greatest influence on the outputs of collision risk modelling?

Most of the input parameters were discussed and raised by the interviewees as a whole but those that occurred more than once and in descending order (most frequently highlighted first):

- Avoidance rate
- Flight height data
- Rotor Speed
- Density
- Number of turbines
- Which Band option used
- Operation time

Changes or updates to model

Question 6: If you could, how would you improve collision risk models/modelling?

There were many different opinions on how to improve collision risk modelling but generally they did not involve making large changes to the mechanics of the model itself but rather to the input data or presentation of data and outputs. Comments that were raised more than once and in descending order (most frequently highlighted first) included:

- Present a covering/summary sheet with input data values to ensure parameters are clearly set out and defined.
- Stop presenting single numbers as black and white and also provide context.
- Take data from existing sites to validate the model and also use post-construction monitoring.
- Have a standard approach to derive turbine parameters and bird parameters including consistently defining breeding season periods.
- More studies/data on bird behaviour around turbines and avoidance behaviour.

- More and clearer guidance on the model and model use and intended use, especially on the tidal offset.
- Collect flight height data objectively, not just human observation/estimation but using rangefinders.
- Factor uncertainty into estimates.
- Use R code rather than excel to make modelling process more reproducible.
- Better interpretation of model outputs.
- Single location to have the most up to date version of model and email updates.

These can then be split into comments that were more input data-related:

- Present a covering/summary sheet with input data values to ensure parameters are clearly set out and defined.
- Have a standard approach to derive turbine parameters and bird parameters including consistently defining breeding season periods.
- More studies/data on bird behaviour around turbines and avoidance behaviour.
- Collect flight height data objectively, not just human observation/estimation but using rangefinders.

Or those which were model or output data-related:

- Stop presenting single numbers as black and white and also provide context.
- Take data from existing sites to validate the model and also use post-construction monitoring.
- More and clearer guidance on the model and model use and intended use, especially on the tidal offset.
- Factor uncertainty into estimates.
- Use R code rather than excel to make modelling process more reproducible.
- Better interpretation of model outputs.
- Single location to have the most up to date version of model and email updates.

Question 7: Would the explicit reporting of variability and uncertainty in outputs from collision risk models benefit the consenting process and discussions with regulators?

When asked more specifically about including variability and uncertainty in CRMs interviewees gave a wide range of responses but these were not consistent within different stakeholder groups. Of the 20 people interviewed, 13 agreed that including variability and uncertainty in outputs from collision risk models would benefit the consenting process and discussions with regulators, however 7 people disagreed. Of those 7, all said that they disagreed because of the consenting and assessment process and that in principle it would be better to include variability and uncertainty, but they though that the system did not allow for it. A recurrent comment was that interviewees were unsure of how variability and uncertainty could be included in outputs and still fit in with the Habitats Regulations.

Some comments and themes that were raised in the interviews are listed below:

- Scientifically there is a benefit to making clear what the uncertainties are.
- Accounting for uncertainty in data collection methods and survey data would be useful.
- I am uncomfortable with presenting a value that is apparently so precise.
- There is an absolute fixation on single numbers which is dangerous.
- We need greater acceptance that we live and work in an uncertain world and things are grey, not black and white.
- We need a way of showing that some scenarios are more likely than others.
- Decision makers have to be confident that they are making the right decisions so they need to an understanding of uncertainty around the single numbers.
- We need to weigh up risk (or use a risk assessment process) and we can’t do that currently with CRM, though it happens more regularly with PVA.
- The current approach is too precautionary and always uses the most precautionary values.
- If the system were to change, including variability and uncertainty is a more useful approach.
- Any outputs need to be suitable to be taken forward through the assessment process.
- The risk is that it complicates the process even more than already because the more the risks are explicit the more difficult it is to explain to the planning inspectorate.
- There is probably too much uncertainty in the system to make it useful to include it.

There was a wide range of views on some topics, for example opinions on using probability distributions:

- Presenting probability distributions would help a lot because regulators often have a background of understanding risk probabilities.
- Using probability distributions might help with presentation but it might not help with interpretation of outputs, especially if people don’t understand how to interpret probability distributions.
- Distributions are probably more helpful but people need to understand them.
- Scientists are used to dealing with probabilities but legislation is binary.

This probably stems from uncertainty and/or inconsistency in (the understanding of) how decisions are made and the lack of a strategic decision on a standard method for presenting data which is most informative for the decision makers.

Appendix 2: Sensitivity analyses

Chamberlain *et al.* (2006) previously documented that the Band model was sensitive to input parameters. Following on from this, the sensitivity of the Band model update produced during the SOSS project (Band 2012) was assessed, both for the basic and extended versions. Similar to Chamberlain *et al.* (2006), the effect of a 10% change in the input parameters was assessed but in addition, a more realistic parameter range was also assessed.

When assessing the effect of a 10% change in the flight height distribution for the extended model, we increased the proportions of birds at heights between the minimum and maximum rotor tip heights by 10%.

The following data sources were used for the input parameters. For turbine-related parameters, expert opinion within the project group was used to assess reasonable parameters ranges and those likely to be built out in the near future.

Bird-related Parameter	**Data description**
Length	Taken from Concise Birds of the Western Palearctic (Cramp and Perrins, 1993) and other sources
Wingspan	Taken from Concise Birds of the Western Palearctic (Cramp and Perrins, 1993) and other sources
Flight speed	RSPB telemetry data (breeding season only)
Nocturnal activity	RSPB telemetry data (breeding season only)
Proportion at collision risk height	Generic flight height curve provided with the Band model and data provided by BTO (Johnston *et al.* 2014).
Flight height distributions	Data provided by BTO (Johnston *et al.* 2014).
Avoidance	‘All gulls’ rate available from Marine Scotland Science avoidance report (Cook et al. 2014)
Bird density	Taken from Creyke Beck A Environmental Statement

Turbine-related Parameter	**Description/Notes**
Rotor radius	Expert opinion
Hub height	Expert opinion
Max. blade chord width	Expert opinion
Rotation speed	Expert opinion (example relationship between wind speed and rotation speed)
Blade pitch	Expert opinion (example relationship between wind speed and pitch)
Turbine operation time	Taken from Inch Cape Environmental Statement

10% change

BASIC MODEL (Option 1)

Input variable	Baseline	Baseline ± 10% (increases mortality)	Collision risk (in absence of avoidance)	Revised number of collisions	% increase in number of collisions
Avoidance rate	0.9893	0.8904	0.065	439	921
Non-avoidance rate	0.0107	0.0118	0.065	47	9
% at collision risk height	6	6.6	0.065	47	9
Bird density (birds/km^2)	9.89	10.879	0.065	47	9
Flight speed (m.s^{-1})	7.26	7.986	0.063	46	7
% nocturnal flight	3.3	3.63	0.065	43	0
Bird length (cm)	39	42.9	0.067	44	2
Wing span (cm)	108	118.8	0.065	43	0
Number of turbines	100	110	0.056	47	9
Rotor radius (m)	80	88	0.061	44	2
Hub height (m)	125	112.5	0.065	43	0
Rotation speed (rpm)	7.74	8.514	0.067	44	2
Blade width (m)	5.5	6.05	0.069	46	7
Blade pitch (degrees)	0	-	-	-	-
% time operational	87.61	96.371	0.065	47	9

Effects of 10% variation in input parameters on predicted mortality rates of black-legged kittiwakes using hypothetical wind farm parameters (100 turbines). Original collision risk was 0.065 and the original number of predicted collisions per year was 43. Collisions are presented as integers.

EXTENDED MODEL (Option3)

Input variable	Baseline	Baseline ± 10% (whichever increases mortality)	Collision risk (in absence of avoidance)	Revised number of collisions	% increase in number of collisions
Avoidance rate	0.9672	0.8705	0.065	9	350
Non-avoidance rate	0.0328	0.0361	0.065	3	50
% at collision risk height	6	6.6	0.065	3	50
Bird density (birds/km^2)	9.89	10.879	0.065	3	50
Flight speed (m.s^{-1})	7.26	7.986	0.063	2	0
% nocturnal flight	3.3	3.63	0.065	2	0
Bird length (cm)	39	42.9	0.067	2	0
Wing span (cm)	108	118.8	0.065	2	0
Number of turbines	100	110	0.065	3	50
Rotor radius (m)	80	88	0.061	5	150
Hub height (m)	125	112.5	0.065	8	300
Rotation speed (rpm)	7.74	8.514	0.067	2	0
Blade width (m)	5.5	6.05	0.069	2	0
Blade pitch (degrees)	0	-	-	-	-
% time operational	87.61	96.371	0.065	3	50

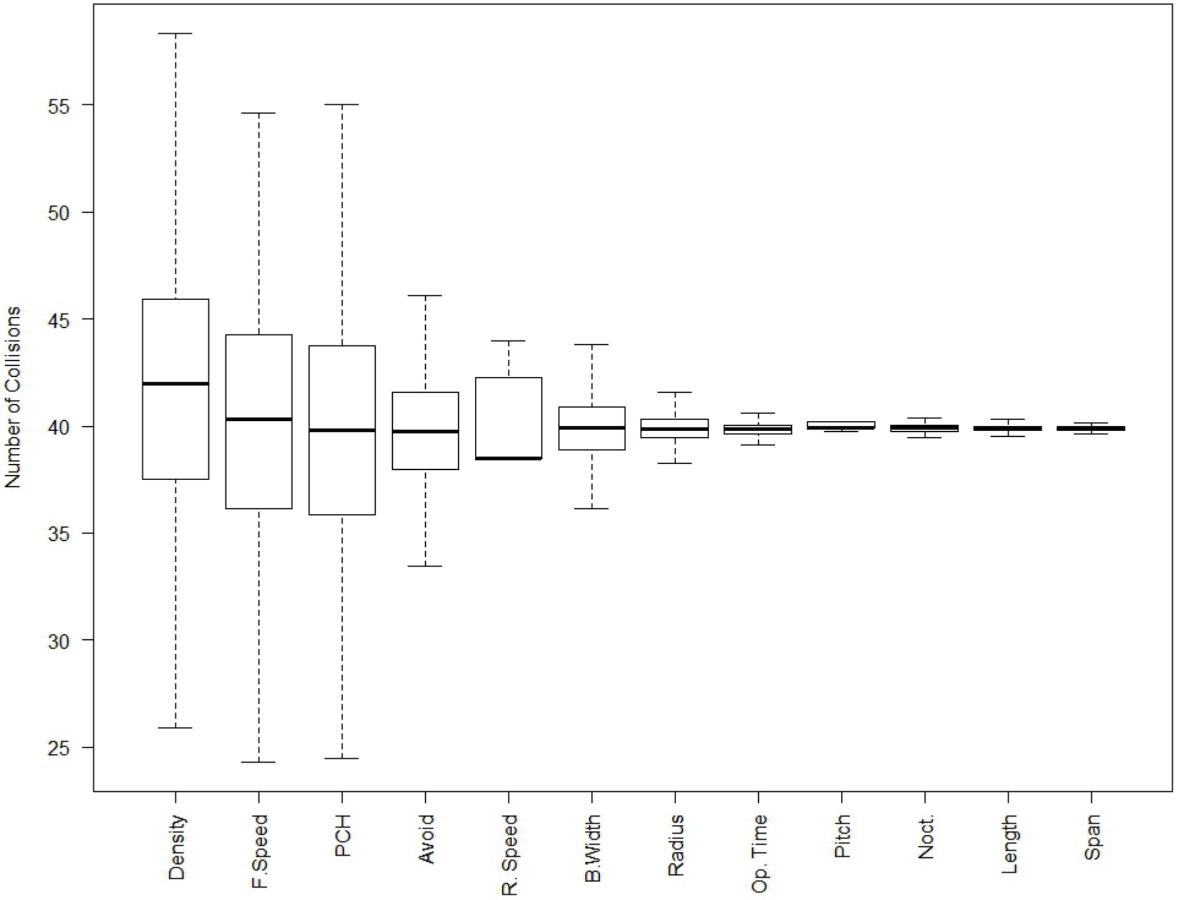
Effects of 10% variation in input parameters on predicted mortality rates of black-legged kittiwakes using the hypothetical wind farm parameters (100 turbines). The original collision risk was 0.065 and the original number of predicted collisions per year was 2. Collisions are presented as integers, therefore the % increase in the number of collisions is greatly influenced by rounding.

Real data range

BASIC MODEL (Option 1)

Input variable	Input variability	Mean collisions (SD)	Median collisions (IQR)
Avoidance rate	N(0.9893,0.0007)	39.76 (2.55)	39.72 (3.61)
% at collision risk height	N(6, 0.9)	39.77 (6.06)	39.77 (7.86)
Bird density (birds/km^2)	tN(monthly mean, monthly SD)	41.86 (6.40)	41.99 (8.39)
Flight speed (m.s^{-1})	N(7.26, 1.50)	40.25 (5.73)	40.30 (8.09)
% nocturnal flight	N(3.3, 0.45)	39.90 (0.19)	39.91 (0.26)
Bird length (cm)	N(39, 0.5)	39.89 (0.15)	39.89 (0.21)
Wing span (cm)	N(108, 4)	39.89 (0.09)	39.89 (0.12)
Rotor radius (m)	N(80, 5)	39.89 (0.67)	39.84 (0.88)
Hub height (m)	Rotor radius + N(26.5, 2)	39.89 (0)	39.89 (0)
Rotation speed (rpm	Relationship to wind speed	40.15 (1.81)	38.51 (3.76)
Blade width (m)	N(5.5,0.3)	39.91 (1.39)	39.90 (1.97)
Blade pitch (degrees)	Relationship to wind speed	40.50 (1.66)	39.89 (0.32)
% time operational	Wind availability-tN(6.3, 2)	39.91 (1.39)	39.90 (1.97)

Effects of variation in input parameters on predicted mortality rates of black-legged kittiwakes using the hypothetical wind farm parameters (100 turbines). 500 iterations. The original collision risk was 0.065 and the original number of predicted collisions per year was 40. N is normal distribution. N(mean, SD). tN is truncated normal distribution. Hub height does not affect calculations in option 1, therefore the values were constant across all iterations.

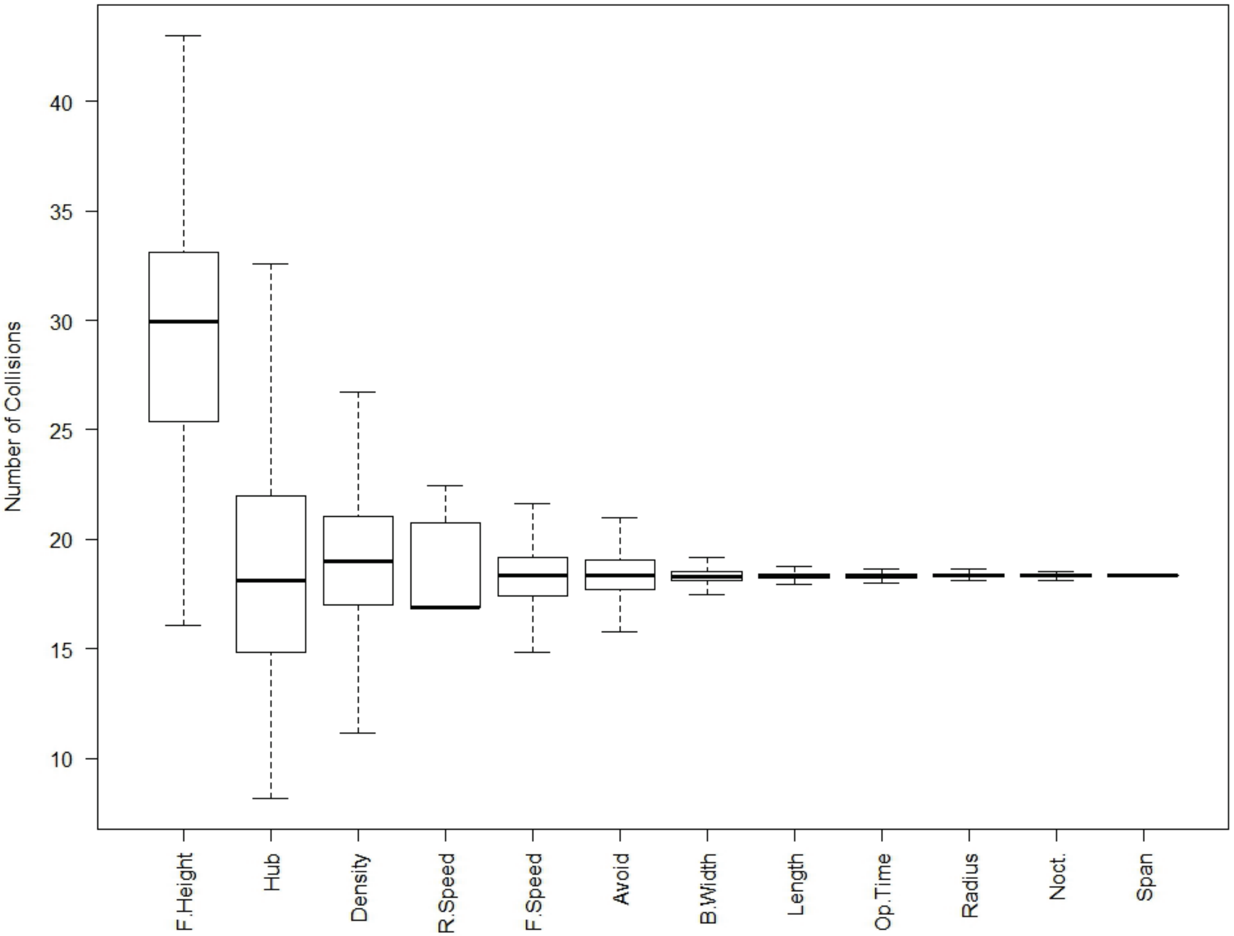


Effects of variation in input parameters on predicted collision mortality of black-legged kittiwakes using the basic Band model. Density values are slightly skewed due to need for use of truncated normal distribution as negative density values are not possible.

EXTENDED MODEL (Option3)

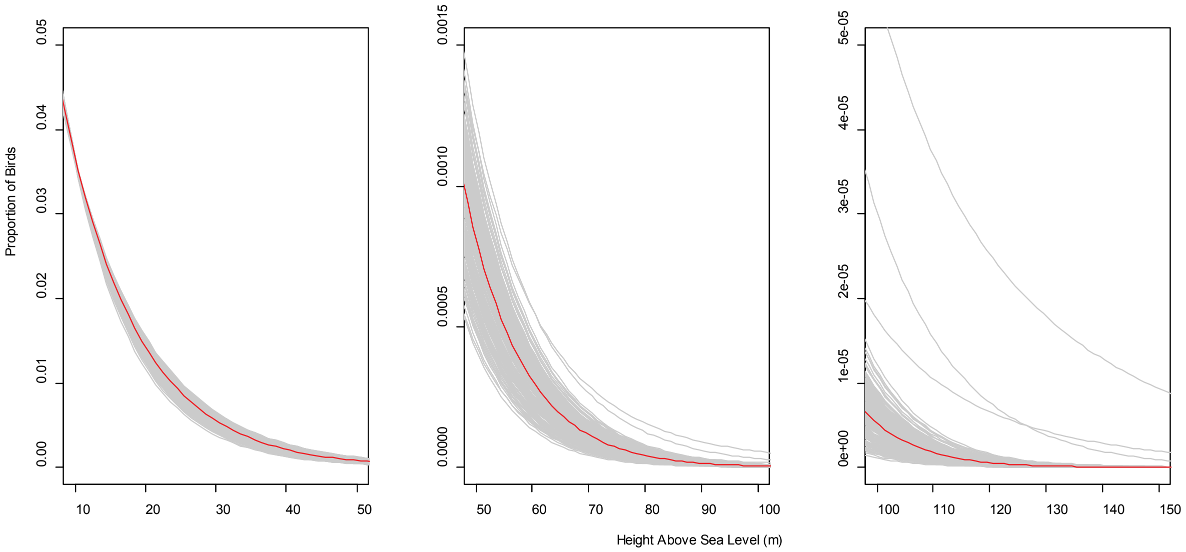
Input variable	Input variability	Mean collisions (SD)	Median collisions (IQR)
Avoidance rate	N(0.9672, 0.0018)	18.34(1.03)	18.33 (1.35)
% at collision risk height	Data from BTO	29.60 (5.92)	29.94 (7.70)
Bird density (birds/km^2)	tN(monthly mean, monthly SD)	19.06 (3.03)	18.97 (4.04)
Flight speed (m.s^{-1})	N(7.26, 1.50)	18.31 (1.38)	18.32 (1.75)
% nocturnal flight	N(3.3, 0.45)	18.32 (0.09)	18.32 (0.12)
Bird length (cm)	N(39, 0.5)	18.32 (0.16)	18.32 (0.22)
Wing span (cm)	N(108, 4)	18.32 (0.16)	18.32 (0.22)
Rotor radius (m)	N(80, 5)	18.35 (0.10)	18.13 (0.14)
Hub height (m)	Rotor radius + N(26.5, 2)	18.72 (5.33)	18.11 (7.08)
Rotation speed (rpm)	Relationship to wind speed	18.57 (1.87)	16.86 (3.89)
Blade width (m)	N(5.5,0.3)	18.31 (0.32)	18.32 (0.44)
Blade pitch (degrees)	Relationship to wind speed	18.32 (0.00074)	18.32 (0.00014)
% time operational	Wind availability-tN(6.3, 2)	18.32 (0.14)	18.32 (0.18)

Effects of variation in input parameters on predicted mortality rates of black-legged kittiwakes using the hypothetical wind farm parameters (100 turbines). 500 iterations. The original collision risk was 0.065 and the original number of predicted collisions per year was 18. N is normal distribution. N(mean, SD). tN is truncated normal distribution.

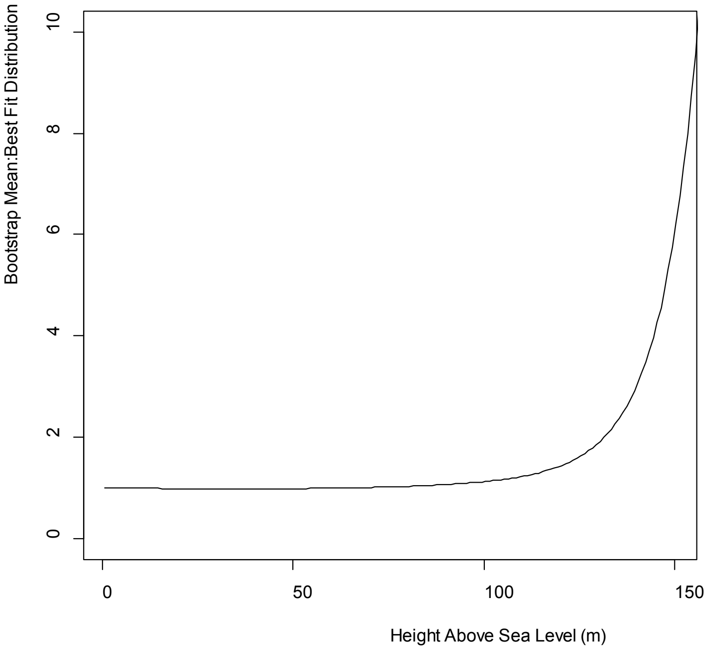


Effects of variation in input parameters on predicted collision mortality of black-legged kittiwakes using the extended Band model. Density values are slightly skewed due to need for use of truncated normal distribution as negative density values are not possible.

A note on variation in flight height: It is noticeable that when variation in the flight height distribution used for the extended model (option 3) is considered, it results in a very different average value to that obtained using the best fit distribution. This should be expected. Flight height distributions are estimated following the methodology set out in Johnston et al. (2014). The best fit distribution is estimated from the complete flight height dataset, and is that which best fits the available data. Confidence intervals were calculated around this distribution using a bootstrapping approach, randomly sampling from the original dataset each time. As a result, each individual bootstrap reflects the shape of the distribution would be if some of the data were excluded. It is not meaningful to compare the mean values obtained from the bootstraps to the best-fit distribution because they are a series of sub-samples. On closer examination, it is clear that the best fit distribution predicts a lower proportion of birds at collision risk height than is obtained from the mean across all bootstraps, and that crucially, this difference is greatest towards the centre of the rotor-swept area, where collision risk is greatest. As a result, the mean collision rate predicted from the bootstraps is greater than collision rate predicted from the best fit distribution.



Comparison of the best fit (red) and bootstrapped (grey) flight height distributions for kittiwake. The best fit distribution does not pass through the centre of the bootstrapped distributions as would be expected if it were directly comparable to the mean. Instead, as height above sea level increases, the proportion of birds predicted by the best-fit distribution moves towards the lower end of the proportion predicted by the bootstraps. The difference is most apparent at heights of around 100 m, which roughly corresponds to the centre of the rotor sweep, the point at which collision risk is greatest. This can be seen more clearly by examining the ratio of the best fit distribution to the mean of the bootstrap distribution at 1 m intervals.


Bootstrap Mean:Best Fit Distribution
0
2
4
6
8
10
0
50
100
150
Height Above Sea Level (m)

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Research article

Operational offshore wind farms and associated ship traffic cause profound changes in distribution patterns of Loons (*Gavia* spp.)

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ABSTRACT

Seabirds select suitable habitats at sea, but these habitats may be strongly impacted by marine spatial planning, including the construction of offshore wind farms (OWFs) and the associated ship traffic. Loons (*Gavia* spp.) are particularly vulnerable to anthropogenic activities and are also of high conservation status, making them particularly relevant to marine planning processes. We investigated the effects of OWF construction and ship traffic on Loon distributions in the German North Sea on a large spatial scale, using a 'before–after' control impact analysis approach and a long-term data set. Many OWFs were built in or close to core areas of Loon distributions. Loons showed significant shifts in their distribution in the 'after' period and subsequently aggregated between two OWF clusters, indicating the remaining suitable habitat. The decrease in Loon abundance became significant as far as about 16 km from the closest OWF. Ship traffic also had a significant negative impact on Loons, indicating that OWFs deterred Loons through the combined effect of ship traffic and the wind turbines themselves. This study provides the first analysis of the extensive effects of OWFs and ships on Loons on a large spatial scale. The results provide an essential baseline for future marine spatial planning processes in the German North Sea and elsewhere.

1. Introduction

Shallow-shelf sea areas have long been used by humans. The North Sea is amongst the most-intensively utilised sea areas worldwide for activities including fishing, transport, oil and gas drilling, and gravel extraction (Emeis et al., 2015; Halpern et al., 2008). The installation of offshore wind farms (OWFs) in many sea areas throughout Europe and elsewhere represents a relatively new human use requiring considerable attention in terms of the marine planning process. In order to meet their climate goals, many European governments have started to install and plan further OWFs within relatively large sea areas (e.g. Breton and Moe, 2009; Langston, 2010). Germany intends to extend its offshore power generation to 6,500 MW by 2020 and to 15,000 MW by 2030, leading to a large increase in the number of OWF sites, mainly in the German North Sea, making Germany one of the countries with the most extensive plans for OWF installations (Beiersdorf and Radecke, 2014). Seventeen OWFs are currently (2018) in operation, with five further ones under construction and several more being approved in German sea areas (BSH, 2017).

In terms of the process of marine spatial planning, these permanent installations at sea represent a major addition to other types of marine human activities, whilst competing with sea areas assigned for nature conservation (Emeis et al., 2015; Moksness et al., 2009; Nolte, 2010) and potentially overlapping with areas used by resting and foraging seabirds. Previous studies have pointed out contrasting effects (negative or positive) of OWFs on seabirds that vary strongly among areas and species (Dierschke et al., 2016; Drewitt and Langston, 2006; Fox and Petersen, 2006; Furness et al., 2013; Garthe and Hüppop, 2004; Masden et al., 2009). In addition, the construction and maintenance of OWFs is further associated with a strong increase in shipping activities in and around OWFs (Exo et al., 2003).

OWFs may have direct effects on birds such as collision of individuals with the turbines, with subsequent impacts on the whole population (Fox et al., 2006; Goodale and Milman, 2014; Masden et al., 2009). Furthermore, the energy budget and condition of individual birds may also be affected indirectly through the effects of OWFs on habitat loss and reduced food availability (Drewitt and Langston, 2006; Fox et al., 2006; Stienen et al., 2007), though the long-term effects of these indirect effects at the population level are hard to estimate (Fox et al., 2006; Goodale and Milman, 2014; Searle et al., 2017). However,

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birds have been shown to lose suitable resting and foraging habitats or to select less suitable sea areas (Stienen et al., 2007). Furthermore, they may need to increase their flight time by flying around OWFs on their way to suitable foraging sites (Drewitt and Langston, 2006; Masden et al., 2009). This study aimed to quantify the indirect effects (i.e. habitat loss by OWFs and associated ship traffic) on Loons (*Gavia* spp.) to provide baseline data for future studies that might address population consequences.

Loons belong to the most sensitive species group with respect to the avoidance of OWFs, as shown for single OWF sites in the North Sea (e.g. Dierschke et al., 2012, 2016; Leopold et al., 2010; Mendel et al., 2014; Petersen et al., 2006a, b; Welcker and Nehls, 2016). Furthermore, Red-throated Loons (*Gavia stellata*) are also very sensitive to ship traffic, demonstrating long flush distances in front of approaching vessels (Bellebaum et al., 2006) and significantly lower densities in areas with permanently higher ship traffic (Hüppop et al., 1994; Schwemmer et al., 2011). Their sensitive nature and the fact that a significant proportion of the biogeographic population occurs in European waters means that Loons are listed in Annex I of the EU Birds Directive and are considered to be particularly threatened with respect to human activities (e.g. Furness et al., 2013; Garthe and Hüppop, 2004). Negative effects on Loons at both the individual and population levels as a result of avoidance of OWFs cannot be ruled out (Dierschke et al., 2016, 2017), and Loons are therefore currently rated as a species group requiring particular consideration with respect to marine spatial planning in Germany and the UK (Busch et al., 2013).

Most Loons in the North Sea are Red-throated Loons (90%), with a minor proportion of Black-throated Loons (*G. arctica*; 10%) (Dierschke et al., 2012; Garthe et al., 2007). The German North Sea represents one of the most important resting sites for Loons with internationally important numbers, especially during spring migration (Garthe et al., 2007, 2015; Mendel et al., 2008; Skov et al., 1995), when around 20,200 Loons use German waters (Garthe et al., 2015). The 'Eastern German Bight' Special Protection Area (SPA) has been established to acknowledge the importance of this resting site and the high sensitivity of Loons with respect to human disturbances (Fig. 1). However, there is a potential conflict with the 'Butendiek' OWF, which was approved before but installed after the establishment of the SPA (Garthe et al., 2012), while further OWFs ('Helgoland Cluster') are located just south of the border of the SPA (Fig. 1).

Information on the long-term and large-scale effects of OWFs on Loons is currently limited and there has been no long-term comparison of their distributions before and after the installation of OWFs. Furthermore, the effects of increasing construction- and maintenance-related ship traffic have rarely been considered (Boon et al., 2010; Christensen et al., 2003).

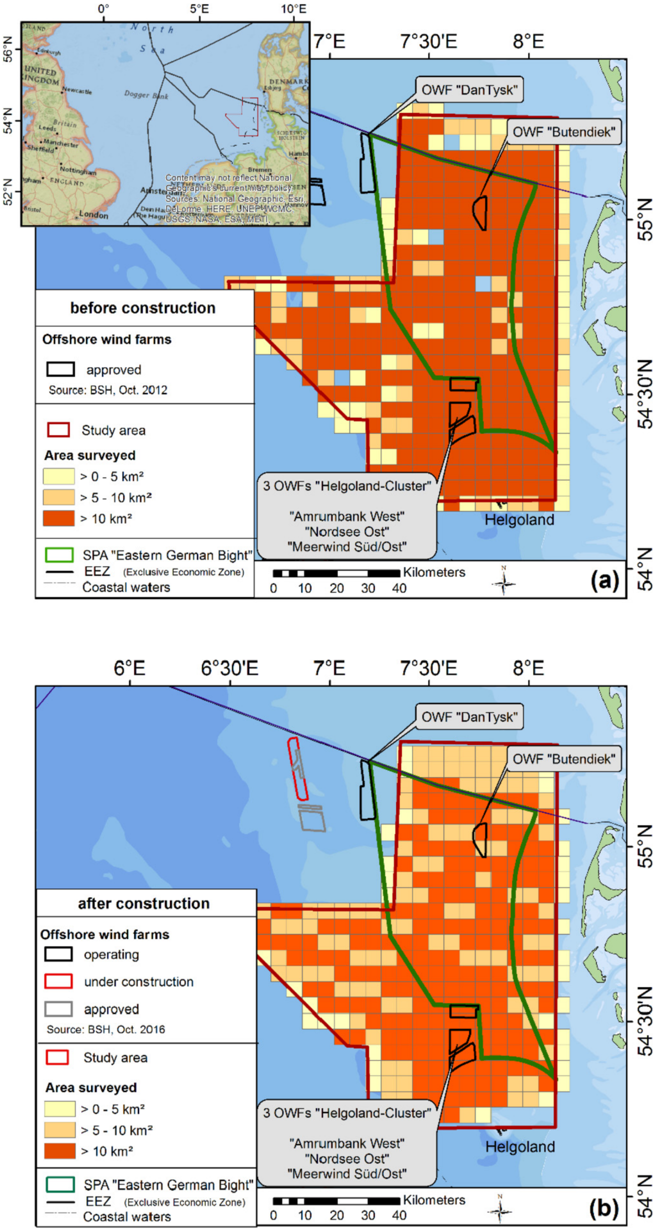

Fig. 1. Location of the study site within the south-eastern North Sea (inserted map in Fig. 1a) and in the eastern German Bight (North Sea) with locations of the different OWFs and the area surveyed for Loon abundance (yellow to red squares) across the 'Eastern German Bight' Special Protection Area (SPA; bold green line) for the 'before' (a) and 'after' periods of the analysis (b). Start of construction: 'Nordsee Ost' OWF during summer 2012; end of all construction works: 'Butendiek' OWF during summer 2015. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

We therefore hypothesized that Loons would avoid OWF areas and that their distribution patterns would differ before and after the installation of OWFs. We also hypothesized that the ship traffic associated with OWF sites would cause avoidance reactions among Loons. Against this background, this study aimed to shed light on five specific topics. (1) We had access to a long-term dataset covering the 14-year period before the installation of the OWFs ('before'). We therefore aimed to compare this information directly with the distribution of Loons after the installation of OWFs ('after'), using a long-term perspective not achievable in most previous studies. Mandatory operational monitoring of the four offshore windfarms in focus is still ongoing. (2) Most previous studies of the potential effects of OWFs on Loons have focussed on the effects of single OWF sites and their direct vicinities (see Dierschke et al., 2016). These therefore only allowed the reactions of Loons to be studied on a relatively small spatial scale, and could only show that Loon numbers were impacted within the respective site but could not show where they had moved to (Rexstad and Buckland, 2012). In contrast, the current study aimed to analyse the large-scale effects of multiple OWFs on Loon distribution, considering potential shifts between the 'before' and 'after' periods. (3) There is currently a need to disentangle the potential effects of OWFs from the effects of natural habitat characteristics that determine the distribution of Loons (Garthe, 1997; Winiarski et al., 2014). We therefore developed a model including stable natural parameters such as water depth and distance to land, as well as anthropogenic predictors such as distance to closest OWF and shipping traffic. (4) Given that the installation and maintenance of OWFs is associated with large increases in ship traffic, the effects of shipping need to be quantified and separated from the effects of the OWFs themselves. To date, this only has been analysed based on general ship densities (e.g. APEM, 2013; 2016; Leopold et al., 2014), while OWF ships present a dynamic source of disturbance for Loons.

This study therefore aimed to relate Loon and ship distributions at very high spatial and temporal scales by relating ship distributions derived from the Automatic Identification System (AIS) with Loon abundance assessed during aerial surveys. (5) Given a negative effect of OWFs on Loons, we aimed to quantify the avoidance distance to OWFs to draw conclusions about the degree of resulting (permanent) habitat loss.

In this study, we adopted two different approaches to analyse different aspects of the effects of OWFs on Loons: we used ‘before’ data to demonstrate the importance of the OWF areas before construction, and also focused on the simultaneous effects of OWFs and ships associated with OWFs after construction. The combined interpretation of these approaches allowed a comprehensive evaluation of the effects of OWFs on Loons.

2. Methods

2.1. Study area

The study was conducted within the eastern part of the Exclusive Economic Zone of the German North Sea, south of 55°17′ N, north of 54°11′, east of 6°30′ E, and west of 8°9′ E (Fig. 1a). The study site was located within an area 8–100 km off the Wadden Sea islands of northern Germany. The water depth ranged from 10 to 40 m. Loon distribution was recorded within the SPA ‘Eastern German Bight’ and beyond, and the study site therefore covered the core area of highest Loon densities within German waters (Garthe et al., 2015). The ‘Butendiek’ OWF is located in the core area of the SPA, while the ‘Helgoland Cluster’ OWFs are located at the border of the SPA and south of the core Loon distribution (Fig. 1a).

2.2. Recording Loon distribution and data processing

Loon distribution was recorded, both, in the period prior to OWF construction and in the period after construction:

(1) Before construction: These data cover the months of spring migration (i.e. March to April) of the years 2000–2013 and are the similar database as used by Garthe et al. (2015). The records originated from environmental impact assessment studies required for licensing procedures of offshore wind farms in the German EEZ and from seabird monitoring and research programmes (for details see Garthe et al., 2015; Fig. 1a). The data were recorded using visual aerial and ship-based surveys. Briefly, Loons were counted along transects of a known area, which allowed the densities to be computed (see Diederichs et al., 2002; Garthe et al., 2002 for a full description of both recording methods).

(2) After construction: These data also cover the months of spring migration (i.e. mainly March to April, but including the last week of February and the first week of May to enhance the sample size of surveys) of the years 2015–2017. Data originated from ongoing mandatory monitoring of the wind farms during operation, and from the ‘Helbird’ research project funded by the German Federal Ministry for Economic Affairs and Energy. Overall, data for the after period were based on 10 digital aerial surveys in 2015–2017 (Fig. 1b). Those data were obtained by video-based digital recordings instead of visual observations. Briefly, an aircraft sampled a transect of a known area using a video camera and all seabirds found were recorded and used to compute overall densities (for a detailed description of the method see Buckland et al., 2012; Thaxter and Burton, 2009). A change from visual to digital survey methods was mandatory for safety reasons because the flight altitude needed to be higher during the construction and operational phases of the turbines (168 m, instead of 91 m for visual observations), which excluded visual recordings.

(3) During construction: No data were considered in this study, as disturbance during the construction of the OWF is temporary and mainly associated with construction ships, and its contribution to the overall effect of the OWF on the Loon population was assumed to be of low importance in relation to the expected lifetime of the OWF (Christensen et al., 2003).

Visual observations of seabird distributions are known to underestimate birds in parts of the transect further from the observer (Buckland et al., 2001, 2015). We therefore applied a species-specific correction factor for aerial and ship-based observations, respectively (see Garthe et al., 2015 for details). However, no distance correction was necessary for the video-based digital surveys because the probability of detecting a bird was equal across the whole transect.

All three recording methods relied on the principle that transect sampling of birds could be used to compute densities. However, we did not compare absolute density values between the ‘before’ and ‘after’ periods, because the visual and digital methods have not been confirmed to produce the same absolute values (Buckland et al., 2012; Skov et al., 2016); this could only be tested by performing both methods at the same time, and no such dataset is currently available. Thus, both periods were compared by computing the relative deviance from the maximum density in each period in %, and using this to compare the distributions and locations of high-density areas of Loons between the two periods.

Data were spatially pooled in a grid with cells of 2.5 × 2.5 km for the ‘before’ and ‘after’ periods, for each of the three methods (visual aerial and ship-based surveys, video-based digital recordings), respectively. Bird numbers and monitored areas were each summed per grid cell, and eventually used to compute mean densities for each period, while geographical coordinates were averaged for each cell.

2.3. Integrating covariates for the ‘before–after’ control impact (BACI) approach

We related the average distribution data for Loons with environmental variables using ArcGIS (version 10.3; Environmental System Research Institute, 2016). The environmental variables included: (1) *dist_coast* = minimum distance to the mainland and larger islands (except Helgoland); (2) *dist_helgoland* = minimum distance to Helgoland; (3) *dist_owf* = minimum distance to the border of the OWF; and (4) *mean_depth* = mean water depth.

This first model, hereafter named the BACI approach, did not consider the effect of ships because ship data at a sufficiently high spatiotemporal resolution were only available for the ‘after’ period. To distinguish between the effect of the OWFs and the effect of ship traffic on Loons, we therefore developed a second model (ship model) using only the data from the ‘after’ period.

To merge the environmental variables with the bird-count data in an optimal way, we first pooled the covariates to a spatial grid of 2.5 × 2.5 km, and then fitted each covariate with a generalised additive model (GAM) using the function gam() in the R-package mgcv (R Core Team, 2017; R version 3.4.2; Wood, 2006). We used only latitude and longitude as a smooth 2D-predictor based on cubic splines with the maximal degree of freedom, so that the result represents a cubing interpolation on the given (possibly irregular) grid. Thirdly, we used the predict() function to predict the values straight to the coordinates as given in the pooled bird-count data. Finally, the additional categorical variable *owf_zone* for ‘inside OWF-affected area’ vs. ‘outside OWF-affected area’ was defined for two different zones: 1) inside: ≤ 3 km vs. outside: > 3 km (measured from the nearest turbine), given that OWF-associated ships operate mainly within a 3 km radius around the OWF and this distance class has been used in previous studies of the impact of single OWFs (Vanermen et al., 2015a; Welcker and Nehls, 2016); and 2) inside: ≤ 10 km vs. outside: > 10 km, because an initial analysis showed the greatest decrease in Loon densities up to a distance of 10 km from the turbines.

2.4. Set up and validation of regression models for the BACI approach

The BACI approach is based on surveying a potentially impacted situation and a control situation before the impact (variable '*period*'), and relative comparisons of spatial and temporal differences can then be used to extract the unbiased impact (Schwarz, 2014; Smith, 2002). We formulated the BACI approach within the framework of generalised additive mixed models (GAMMs), which are known to describe biological count data appropriately (Zuur et al., 2007, 2009; 2012). We used a continuous linear or smooth predictor measuring the distance to the border of the next OWF. This allowed us to estimate how the abundance of Loons changed in relation to the distance from the OWF and to estimate avoidance distances. Notably, we introduced a variable for the observation method ('visual ship-based surveys' vs. 'visual aerial surveys' vs. 'digital aerial surveys') as a random intercept to account for differences in detection among these methods. We were aware that this variable was partially collinear with the variable 'period' because only digital aerial surveys were used 'after' and only visual surveys were performed 'before'. Importantly, the estimation of the interaction term 'period x wind_farm' (see below) representing the BACI approach was not influenced by this, because only relative differences in Loon densities were evaluated.

This approach produced the following full model for the BACI approach (not yet thinned regarding its predictors; see below):

$$\log(y_{ij}) = \beta_0 + u_i + \mathrm{f(mean\_depth_j)} + \mathrm{f(dist\_coast_j)} + \mathrm{f(dist\_helgoland_j)} + \mathrm{s(latitude,longitude)} + [\mathrm{wind\ farm_j}] + \mathrm{period_j} + [\mathrm{wind\ farm_j}] \times \mathrm{period_j} + \mathrm{offset}(\log(\mathrm{area_j})) + \varepsilon_{ij} \quad (1)$$

where $\varepsilon_{ij} \sim N(0, \sigma^2)$ and $u_i \sim N(0, \sigma_u^2)$ were independent and identically distributed. Here, y_{ij} is the vector of bird numbers, where the index j refers to the observation number and i is related to the method-ID. *f*() depicts either a linear term or a cubic regression spline *s*() (tested during predictor selection), where, in the case of a spline, the optimal number of knots was estimated via cross-validation. The variable [*wind_farm*$_j$] was either considered as a linear term, *dist_owf*$_j$ measuring the distance to the next wind turbine, as an additive smoother, *s*(*dist_owf*$_j$), or as a bivariate variable, *owf_zone*$_j$, the latter distinguishing between 'inside OWF-affected area' and 'outside OWF-affected area'. For each model, an appropriate probability distribution was selected for y_{ij} via Akaike Information Criterion (AIC; Akaike, 1973) analysis (see below).

We modified the common selection and validation strategies to validate the optimal GAMM model (Field et al., 2012; Korner-Nievergelt et al., 2015; Zuur, 2012; Zuur et al., 2009, 2010; 2012) using the following steps: (1) Based on the entire model (1), we selected an appropriate probability distribution/stochastic part of the model using the AIC. Namely, we compared Poisson-, negative binomial-, Tweedie-, zero-inflated Poisson distribution, and observation-level random intercept Poisson models. All five probability distributions are known to describe the stochastic part in regression models of (overdispersed) count data reasonably well (Kokonendji et al., 2004; Korner-Nievergelt et al., 2015; Linden and Maentyniemi, 2011; Zuur et al., 2012). (2) The optimal model regarding the set of fixed-effect predictors was selected from the full model by comparing 16 different models. (3) Model validation was carried out by visual inspection of the residual plots to assess all the required model assumptions (Zuur et al., 2010). Corresponding auto-correlation structures were added to the model if required.

AIC favoured a negative-binomial distribution, and subsequent predictor selection produced the following final model:

$$\log(y_{ij}) = \beta_0 + u_i + \beta_1\mathrm{dist\_coast_j} + \mathrm{s(latitude,longitude)} + [\mathrm{wind\_farm_j}] + \mathrm{period_j} + [\mathrm{wind\_farm_j}] \times \mathrm{period_j} + \mathrm{offset}(\log(\mathrm{area_j})) + \varepsilon_{ij} \quad (2)$$

Residual analysis revealed no violation of linearity, homogeneity, independence, or normality of the random intercept.

2.5. Integrating covariates for the ship model

Ship traffic has been shown to have a significant effect on Loon distribution (Bellebaum et al., 2006; Schwemmer et al., 2011), and ship traffic in the study area has increased greatly due to the construction and maintenance of OWFs. It is therefore important to disentangle the effects of these two sources of anthropogenic activities (OWFs and ship traffic) on Loons. Ship traffic shows temporal inhomogeneity, with more traffic in the morning and evening hours, and it was therefore necessary to consider the data spatio-temporally instead of purely spatially, as with the BACI approach. Data were only used for five digital-survey flights from the 'after' period because no real-time ship data were available for the 'before' period or for any other survey days during the 'after' period. Bird data were spatially assigned to an optimal grid of 2.5 × 2.5 km for each survey day separately and treated as described above. To consider the time, we also calculated the mean time at which the Loon observations were recorded for each grid cell.

Data on ship distributions were recorded in parallel with the digital-survey flights to record Loon distribution using an AIS spotter (www.aisspotter.com). Because the ship data consisted of irregular position data in terms of time and space, they were linearly interpolated to obtain positions at least every minute. To merge the ship data with the Loon-distribution data, it was assumed that all ships within the time interval $[t - \delta_t, t]$ and within a circle around (x, y) with radius r may influence bird density, for each time point t and each pair of spatial coordinates (x, y). Given that the optimal values δ_t and r are not known *a priori*, we tested all existing combinations between δ_t ε {2, 60, 120, 180, 250, 300, 350, 400, 600, ∞} sec and r ε {1, 2, 3, 4, 5, 6, 7, 8, 9, 10} km, and created a separate variable counting all ships within the given time and space interval for each of the 100 combinations. Here, $\delta_t = \infty$ depicts a case where all available ship data have only been spatially correlated to bird-count data without considering temporal distance to the observations. We subsequently compared 100 resulting regression models (see below) to find the optimal values of δ_t and r. However, the AIC value was not appropriate for selecting the optimal model because the ship densities and OWF-related variables were collinear, and the model with only one of both variable types would be favoured due to the parsimony of the AIC-based selection. In contrast, we aimed to consider both (collinear) variables to distinguish explicitly between the unique effects of ships and wind turbines on Loon abundance. An appropriate measure should thus relate the effect size of the ship-dependent variable with its reliability. Hence, we selected the model with the highest $|\beta|/SE_\beta$ value, where β is the ship-related regression coefficient and SE_β is its standard error.

2.6. Set up and validation of regression models for the ship model

The GAMMs were set-up as described above for the BACI approach. Notably, the ID of the digital-survey flight was introduced as a random intercept to account for different numbers of birds or different monitoring conditions between surveys.

This produced the following GAMM structure of the ship model (not yet thinned regarding its predictors):

$$\log(y_{ij})\beta_0 + u_i + \mathrm{f(mean\_depth_j)} + \mathrm{f(dist\_coast_j)} + \mathrm{f(dist\_helgoland_j)} + [\mathrm{wind\_farm_j}] + [\mathrm{ship\_number_j}] + \mathrm{offset}(\log(\mathrm{area_j})) + \varepsilon_{ij} \quad (3)$$

where $\varepsilon_{ij} \sim N(0, \sigma^2)$ and $u_i \sim N(0, \sigma_u^2)$ were independent and identically distributed. Here, y_{ij} is the vector of bird numbers, where the index j refers to the observation number and i is related to the survey flight ID. *f*() depicts a linear or smooth predictor (tested during AIC-based predictor selection). The variable [*wind_farm*$_j$] was either considered as a binomial predictor ('inside' vs. 'outside'), a linear term (distance to the

OWF border), or a cubic regression spline depending on the latter. The variable [*ship_number$_j$*] was considered as the total number of temporally and spatially related ships, additionally depending on the *a priori* defined parameters δ_t and r (see above). In contrast to the BACI approach, we did not consider a spatial smooth because this predictor would interfere with the correct estimation of [*wind_farm$_j$*]. GAMM-model selection and validation strategies were performed as described for the BACI approach (see above), including integration of the appropriate autoregression structures (if required).

AIC-based selection of the probability distribution again favoured a negative-binomial distribution. The optimal values of δ_t and r required to blend the observation and ship data showed that the highest (β/SE)-values (indicating high precision of the ship-related regression coefficient) were $\delta_t = 5$ min and $r = 5$ km. Subsequent predictor selection revealed the following final model:

$$\log(y_{ij}) = \beta_0 + u_i + \mathrm{f}(\mathrm{mean\_depth_j}) + \beta_1 \mathrm{dist\_coast_j} + \mathrm{s}(\mathrm{dist\_helgoland_j}) + [\mathrm{wind\_farm_j}] + [\mathrm{ship\_number_j}] + \mathrm{offset}(\log(\mathrm{area_j})) + \varepsilon_{ij} \tag{4}$$

where $s()$ depicts the cubic regression splines with optimal degrees of freedom estimated via cross-validation.

Analysis using different sizes of the underlying spatial grid for spatio-temporal pooling revealed an optimal grid size of 2.5 × 2.5 km, leading to a temporal autocorrelation of model residuals of order 2 (in contrast to the model based on raw data, where the autoregressive order (AR order) was > 30). Model-validation plots indicated no violation of linearity or homogeneity, spatial residual plots and a semi-variogram indicated no violation of spatial independence, and a plot of the partial autocorrelation function (pACF-plot) revealed a temporal autocorrelation of approximately order 2, which was integrated as an AR(2)-structure into the model.

3. Results

3.1. Loon abundance before and after OWF installation

The spatial distribution patterns of Loons changed profoundly between the 'before' and 'after periods (Fig. 2). During the 'before' period, the core area with the highest Loon densities clearly overlapped the area of the planned 'Butendiek' wind farm, while moderately high densities stretched out to the area of the planned 'Helgoland Cluster'. In contrast, there was a clear shift to the area located between these two OWF sites during the 'after' period (Fig. 2). The areas of the OWFs themselves, as well as the immediate vicinities, showed extremely low abundances of Loons during the 'after' period. The core area of Loons during the 'after' period was thus still located in the centre of the SPA, but the birds were more aggregated within the still-undisturbed sea area.

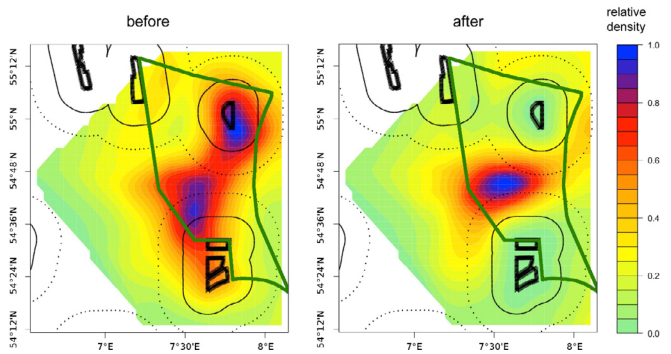

Fig. 2. Spatial density plots of predicted Loon distributions 'before' vs. 'after' the construction of OWFs, based on the BACI-GAMM. Bold black lines: OWFs; thin black lines: 10 km distance buffer; dotted black lines: 20 km distance buffer; bold green line: Special Protection Area.

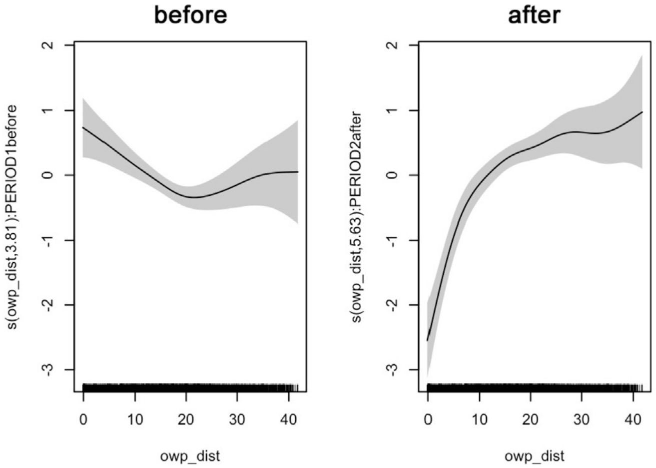

Fig. 3. Change in Loon abundance in relation to distance from the closest OWF site 'before' (left) and 'after' construction of the OWFs. Smoothed curve: predicted number of Loons at a given distance from the closest OWF; shaded area: 95% confidence interval; small lines on the x-axis: observations of Loons at a given distance from the OWF.

We also introduced the distance from the wind farm as a smooth term, estimated separately for each period. This revealed a striking difference between the two periods (Fig. 3): the 'before' plot suggested that the future wind farm areas were sites with naturally increased Loon abundance, while the 'after' plot showed a strong decline in Loon abundance due to the OWFs (Fig. 3). The start of this decline was already visible at > 20 km from the OWFs (see also dotted black lines in the 'after'-plot in Fig. 2b). To determine the distance from the wind farm at which the decline in abundance was significant, we approximated the first derivative of the corresponding smooth (Fig. 3 'after') by calculating its first finite difference.

To determine the distance at which the change in Loon density became significant, we calculated confidence intervals for the first derivatives via bootstrap analysis and subsequently evaluated where the lower confidence interval intersected with zero. This occurred at around 16.5 km from the OWFs (Fig. 4). However, the greatest decline in density was at distances within 10 km from the OWF (Figs. 3 and 4). Avoidance of wind farms within 10 km was also clearly visible in the distribution maps (solid black lines in Fig. 2b).

Additionally, the binomial wind farm-related variable *owf_zone* was highly significant for both radii (3 or 10 km, respectively). The abundance of Loons decreased highly significantly by 94.5% inside the 3 km zone around the OWFs within the study site (interaction term in Table 1; $\beta = -2.9$, $p < 0.001$), while the abundance was still decreased by 83.7% inside the 10 km zone (Table 2, $\beta = -1.8$, $p < 0.001$). The distance to land (*dist_coast*) had no significant effect on Loon densities (Table 1; Table 2).

3.2. Distinguishing between effects of ships and OWFs

Loon densities were still reduced if ships were included in the overall model as a predictor for the 'after' period, as was the case without considering the effect of ships, as shown above. Applying a 3 km radius around the wind farms, OWFs alone reduced the Loon density by 70.8% compared with the sea areas outside the OWFs ($p < 0.001$; Table 3). If the radius was extended to 10 km around the OWFs, the Loon density was still reduced by 44.5% ($p < 0.001$) by the OWFs alone.

When ships as single predictor were removed from the model, the estimated effect of OWFs (now combined with the effect of the ships) on Loons was 84% using a 3 km radius ($p < 0.001$). This suggested that

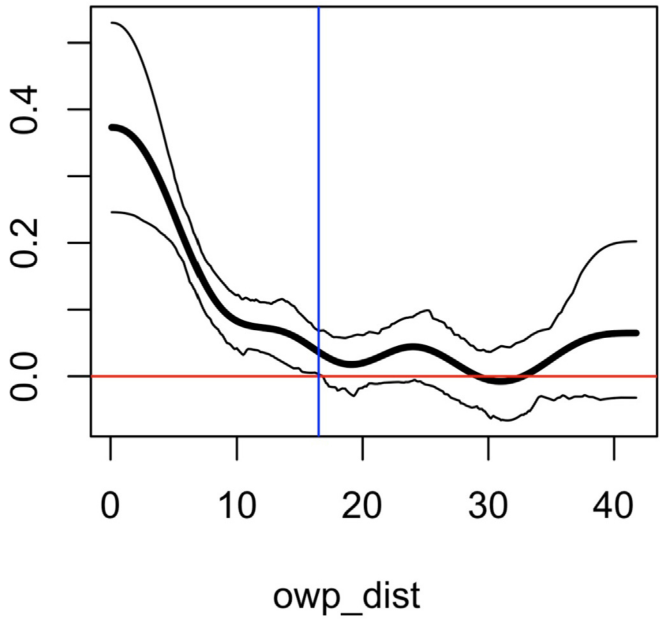

Fig. 4. First finite difference of the smooth depending on the distance from the closest OWF, partially evaluated for the 'after' period. Red line indicates a derivative of zero, blue line indicates distance at which the derivative was significant. Thick black line corresponds to the first derivative; thin black lines depict 95% confidence intervals.

Table 1
Regression results of the BACI approach–GAMM using the binomial variable 'inside wind farm' vs. 'outside wind farm' (*owf_zone*) for a radius of 3 km.

	Estimate	Std. Error	z value	Pr(> |z|)
(Intercept)	−1.05	1.17	−0.90	0.37
period[after]	0.40	0.85	0.47	0.64
owf_zone[a][inside]	0.70	0.13	5.07	< 0.001
dist_coast[b]	0.02	0.02	0.70	0.43
period[after]xowf_zone[inside]	−2.90	0.22	−13.16	< 0.001

[a] Offshore wind farm zone.
[b] Distance to coast.

Table 2
Regression results of the BACI approach–GAMM using the binomial variable 'inside wind farm' vs. 'outside wind farm' (*owf_zone*) for a radius of 3 km.

	Estimate	Std. Error	z value	Pr(> |z|)
(Intercept)	−1.41	1.17	−1.21	0.23
period[after]	0.73	0.95	0.76	0.45
owf_zone[a][inside]	0.66	0.12	5.59	< 0.001
dist_coast[b]	0.02	0.02	1.01	0.31
period[after] xowf_zone[inside]	−1.81	0.12	−15.26	< 0.001

[a] Offshore wind farm zone.
[b] Distance to coast.

Table 3
Regression results for the *ship-owf*-approach–GAMM distinguishing between the effect of ships and the effect of OWFs in the 'after' period for a radius of 3 km.

	Estimate	Std. Error	z value	Pr(> |z|)
(Intercept)	0.29	0.58	0.5	0.62
owf_zone[a][inside]	−1.23	0.31	−4.03	< 0.001
dist_coast[b]	−0.01	0.01	−0.55	0.58
n_ships[c]	−0.37	0.08	−4.82	< 0.001

[a] Offshore wind farm zone.
[b] Distance to coast.
[c] Number of ships.

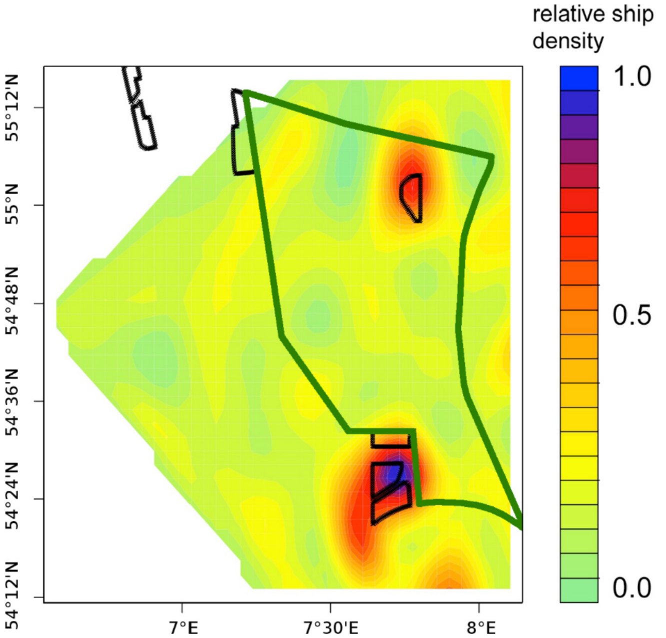

Fig. 5. Spatial density plot of ship distribution in the 'after' period based on AIS data.

ships also had a strong negative effect on Loon abundance, accounting for at least 14% of the joint OWF–ship effect.

Thus, in the ship model, the effect of OWFs alone was not as strong as estimated by the BACI approach (i.e. without considering ship traffic; > 94% and > 84%, respectively). There are two possible explanations for these different estimations. (1) the ship model was only fitted using data from the 'after' period because no ship data were available for the 'before' period. Hence, the estimated reduction in effect does not take account of the fact that bird densities within the OWFs showed the highest Loon abundances before the construction of the farms (see above), leading to a strong underestimation of the reduction effect. (2) Although the ship model considered the effect of ships, these were at least partially correlated with OWF location (Fig. 5). Thus the BACI approach actually estimated the joint reduction effect of OWFs and ships, whereas the ship model evaluated both impacts separately, which may have led to a reduction in the OWF effect compared with the BACI approach.

Indeed, the ship model showed a significant negative impact of ships on Loon abundance (Tables 3–4), with a highly significant decline of 31% in abundance for each additional ship in the spatio-temporal range of the Loons (i.e. 5 min and 5 km from the Loon sighting; see Methods) ($p < 0.001$). This suggests that one in three Loons left the area as one ship approached. The spatial component of ship disturbance was much stronger than the temporal component; i.e. our regression models selecting for the optimal δ_t and r revealed that ships within 5 km had a strong impact on Loon abundance, whereas the time lag between

Table 4
Regression results for the *ship-owf*-approach–GAMM distinguishing between the effect of ships and the effect of OWFs in the 'after' period for a radius of 10 km.

	Estimate	Std. Error	z value	Pr(> |z|)
(Intercept)	0.73	0.58	1.26	0.20
owf_zone[a][inside]	−0.59	0.17	−3.51	< 0.001
dist_coast[b]	−0.01	0.01	−1.00	0.32
n_ships[c]	−0.48	0.07	−6.44	< 0.001

[a] Offshore wind farm zone.
[b] Distance to coast.
[c] Number of ships.

the Loon sighting and the AIS signal of the ship was less relevant (with an optimum at approx. 5 min). This suggests that ships may affect Loons most strongly at a distance of ≤5 km.

As seen with the BACI approach, the distance to land had no significant influence on Loon abundance (Tables 3–4).

4. Discussion

4.1. Distribution patterns before and after OWF installation

Our results demonstrated that the distribution patterns of Loons, which had remained stable over a period of many years (Garthe et al., 2015), were substantially altered at both small and large spatial scales by the installation of OWFs in the German North Sea. We developed our BACI approach on a solid database including 14 years of large-scale surveys in the period 'before' OWF installation. To the best of our knowledge, all previous reports have been based on a maximum of 1–3 years of data prior to the construction of OWFs, and have mostly focused on the effect of a single OWF (e.g. Leopold et al., 2013; Petersen et al., 2014). Although we were unable to compute absolute differences in Loon populations between the two periods due to a change in survey methods, our results demonstrated profound large-scale shifts in distribution patterns, as well as significant avoidance of the OWF areas.

We observed a shift in the Loon-abundance hotspot to the western–central area of the SPA that remained undisturbed by OWFs in the 'after' period. This hotspot is located about 20 km distant from all surrounding OWFs. Several previous studies have highlighted the environmental parameters that are most important for determining Loon distribution patterns. Frontal systems are expected to increase prey availability for Loons (Skov and Prins, 2001), while nearshore and shallow sandy sea areas also play a major role (O'Brien et al., 2008; Skov and Prins, 2001; Skov et al., 2016). Our results suggest that the area of the 'Butendiek' OWF, which was installed in the northern part of the SPA, was of particular importance for Loons before the construction of this OWF, given that this was the area of maximum Loon abundance during the 'before' period. The 'Helgoland-Cluster' OWFs are located south-west of the border of the SPA, and our results showed that, in contrast to the 'Butendiek' area, Loon abundances in the 'before' period were significantly lower compared with abundances within the SPA. However, Loons are known to occur here regularly (Garthe et al., 2015).

One aim of this study was to disentangle the importances of natural habitat structures and anthropogenic pressures on Loons. Our modelling approach showed that natural habitat predictors, such as distance to the coast/Helgoland and water depth did not play major roles compared with the effects of OWFs and shipping (see below). This suggests that anthropogenic pressures are the most important factors driving the distribution patterns of Loons within their natural hot spots.

Still, we cannot completely rule out that undetected changes in ecological conditions might have additionally led to the shift in distribution patterns. For instance, it could be assumed that Loons might have followed shifts in their prey community. However, given that Loons are known to feed on a variety of fish species (Guse et al., 2009), a shift in fish distribution that could account for the change in Loon distribution seems highly unlikely. The reef effect is even known to likely increase benthic and fish communities inside OWFs (e.g. Vandendriessche et al., 2015; Vanermen et al., 2015a) which in turn may enhance the quality of these sites for piscivorous seabirds. However, given that these sites were avoided by Loons, despite of a likely higher fish availability and as Loon distribution patterns had been stable over a period of many years in the 'before' period (Garthe et al., 2015), it seems to be convincing that OWFs and associated ship traffic are the main factors explaining the shifts in distribution patterns.

Incorporating distance from the nearest OWF as a smoothed term in the model allowed us to highlight the fact that Loons reacted as far as 20 km from OWFs, with significant changes in densities at a distance of 16.5 km and the greatest changes in abundance within 10 km. These values were higher than those reported in previous studies (summarized in Dierschke et al., 2016; Welcker and Nehls, 2016). However, most previous studies only investigated local avoidance effects (often only up to 4 km distance; Leopold et al., 2013; Petersen and Fox, 2007, Petersen et al., 2006a,b; Welcker and Nehls, 2016) and were therefore unable to detect any larger-scale avoidance reactions. This highlights the importance of a sufficiently large-scale approach and the inclusion of multiple OWF sites (Rexstad and Buckland, 2012), as in the current study. To emphasize the importance of scale, we quantified the effects of OWFs on Loons by defining the affected sea areas by both 3 km and 10 km radii.

The 3 km distance class was chosen based on previous studies that showed avoidance distances for single OWFs up to this value (Vanermen et al., 2015a; Welcker and Nehls, 2016). However, our results suggest that this distance was too short, based on the effects of multiple OWFs on a larger spatial scale.

The reason for the relatively large-scale effect of OWFs on Loons detected in the current study is not completely clear. It is possible that visual cues are not the only reason for the large disturbance distance. Previous studies showed that OWFs not only affected seabirds and other marine wildlife directly (Bergström et al., 2014; Goodale and Milman, 2014; Lindeboom et al., 2011), but may additionally cause changes in the abiotic environment, such as sediment properties and water stratification due to turbulence caused by the piles (Carpenter et al., 2016; Nagel et al., 2018). Carpenter et al. (2016) pointed out that an individual OWF may enhance mixing of the water column, with a cascade of effects on the whole ecosystem in an area of 10–20 km from the OWF, though the physical–biological interactions remain unclear. This was in accordance with the disturbance distance of Loons found in the current study. Petersen et al. (2014) also showed significantly lower Loon abundances up to 13 km from OWFs, which also matched the results of the current larger-scale approach.

Finally, it is important to critically explore the question of the power of the data used in this study. For the type of data used, previous investigations have shown that high survey intensities are required to safely trace declines in seabird populations, mainly as a result of high variability in distribution patterns (e.g. MacLean et al., 2013; Vanermen et al., 2015b). However, compared to our study that was conducted over a large sea area, both studies mentioned above focussed on rather small study sites, likely enhancing small-scale variability in counting data. According to Vanermen et al. (2015b) the statistical power after 10 years of survey was sufficiently high to detect reliable changes. For the 'before' period, 13 years of data were available for our BACI approach, indicating a valid data base. In contrast, the 'after' period only consists of 10 aerial surveys over a period of three years, suggesting that the data base for the 'after' period may still be too weak. However, the significant negative and consistent effects of OWFs and associated ship traffic on Loon distribution during all surveys of the 'after' period indicates that the data base is sufficient to yield valid results. Nevertheless, it will be necessary to enhance the data base for the 'after' period by future surveys to confirm the results and to enhance the statistical power.

4.2. Distinguishing between the effect of ships and OWFs

The installation of OWFs causes a substantial increase in ship traffic in the surrounding area due to maintenance and service activities (Exo et al., 2003). Although ship traffic is known to affect the distribution patterns of seabirds and particularly of Loons (Bellebaum et al., 2006; Schwemmer et al., 2011), the combined effect of OWFs and their associated ship traffic has rarely been reported; however, the few available studies noted a significant impact of ship traffic on Loon distribution (APEM, 2013, 2016; Leopold et al., 2014; Skov et al., 2016). Loons have been shown to exhibit a behavioural response to approaching ships, and flight distances of up to 2 km have been documented (Bellebaum et al., 2006; Schwemmer et al., 2011). This corresponds to the current results, which suggested a significant reduction in Loon densities within a radius of up to 5 km from the vicinity of ships, with the temporal aspect of ship distribution having little effect.

Inclusion of ship abundance in the model showed a reduced density of Loons of up to 70% based on the 3 km distance zone. This reduction could be considered to reflect the effect of the OWFs alone. In contrast, the joint effect of OWFs and ships led to a reduction of 84%, indicating the additional negative impact of ships on Loon densities. The exact reduction in densities due to ships alone could not be computed reliably because of the collinearity of ship traffic and OWFs. Importantly, their mobile nature means that ships are both spatially and temporally variable predictors, and a reliable estimation of their overall effects on birds will always be biased. This issue will remain difficult to address even in future studies, given that ships aggregate strongly in the vicinity of OWFs and present no fixed predictor.

The greater reduction in Loon densities following inclusion of ship traffic in the model demonstrates the importance of reviewing the cumulative impact of multiple anthropogenic pressures in the marine environment. Previous studies have focussed on cumulative effects simply by investigating the combined effects of multiple OWFs (Busch et al., 2013; Desholm, 2009; Dierschke et al., 2003, 2006, Fox et al., 2006; King et al., 2009; Mendel and Garthe, 2010). However, given the strong effect of ships on Loon abundance, it seems necessary to include other anthropogenic pressures in estimates of cumulative effects on Loon abundance in general.

4.3. Conclusions

The large-scale avoidance effects of OWFs (and ships) on Loons suggest that Loons are unlikely to suffer from enhanced direct mortality, e.g. because of collisions (Leopold et al., 2010; Petersen et al., 2006a,b; this study). Furthermore, a low flight altitude of only up to 10 m above the sea surface (Van Bemmelen et al., 2011) reduces the collision risk for Loons. Indirect effects, such as habitat loss, are thus likely to be key factors affecting Loons in relation to OWFs. However, the consequences of such indirect effects e.g. on population levels of seabirds, and density-dependent effects are hard to assess, and appropriate methodologies are largely lacking (Green et al., 2016; Horswill et al., 2017). When assessing the consequences of habitat loss due to the installation of OWFs and the associated enhancements in ship traffic, it is essential to consider which alternative sea areas could be used as resting and foraging grounds. In the current case, alternative sites seemed to be very limited because the SPA was virtually surrounded by OWFs. This might explain why Loons tended to concentrate in the centre of the SPA rather than moving outside it.

Although it was not possible to compute absolute differences in abundance between the 'before' and 'after' periods in this study, it is hoped that this issue will be resolved when enough data become available from parallel digital and visual surveys of sea areas where visual observations are still allowed. However, the relative reduced densities of Loons with respect to OWFs and ship traffic as well as the avoidance distances provided in the current study will serve as a baseline for further studies. A suitable approach for quantifying the overall habitat loss for Loons would involve computing the relative proportion of habitat loss within a certain area (e.g. within the SPA). Dierschke et al. (2006) suggested summing the total OWF areas and adding an additional buffer zone to assess the overall habitat loss. Applying this approach to the current study allowed the minimum habitat loss due to the OWFs in the SPA to be computed, indicating that complete loss of the sea area within a 3 km radius around the OWFs for Loons (as strongly supported by the current study) would equate to a loss of 8.8% of the SPA (overall size 3,135 km^2) for Loons. This should be regarded as an absolute minimum, given that our results clearly showed that the density of Loons was greatly reduced beyond 3 km from the nearest OWF.

Although we are not able to compare absolute density values between the 'before' and 'after' periods, our results indicated that Loons aggregated in the centre of the SPA after OWF installation, representing an increase in Loon density in a much smaller sea area. Given that Loons tend to occur in comparatively small flocks, only occasionally exceeding 5–10 individuals/km^2 (Garthe et al., 2015; O'Brien et al., 2012), this change in distribution might promote density-dependent effects (Blanc et al., 2006; Horswill et al., 2017; Lewis et al., 2001). A possible shift towards suboptimal habitats may lead to suboptimal body conditions prior to breeding, which could in turn reduce the reproductive success and enhance mortality in adult birds (Coulson et al., 1983; Hüppop, 1995). Even a slight increase in the mortality of adult Loons of only 0.3% can have significant negative effects on population levels (Rebke, 2005).

To assess the role of habitat loss on Loons, it is crucial to know if habituation to OWFs will occur or if the habitat loss will be permanent. Although studies from the UK and The Netherlands have indicated slight (though insignificant) increases in Loon abundances after 4–5 years since construction, studies from Denmark have shown no signs of habituation (Petersen and Fox, 2007; Petersen et al., 2008). Similarly, the current study found no habituation 3 years after construction. However, the monitoring of the operating wind farms is still ongoing and thus results on habituation are preliminary. Given that the degree of habituation remains very unclear, we strongly recommend the need for long-term monitoring to assess any potential large-scale effects of cumulative anthropogenic drivers on Loon distribution, particularly within the most relevant sea areas for Loons (e.g. Vanermen et al., 2015a,b).

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To find out more about JNCC visit http://www.jncc.defra.gov.uk/page-1729.

Table 1 Numbers of coastal-breeding Lesser Black-backed Gulls (AON) in Britain and Ireland 1969–2002. Numbers breeding inland and on the coast during Seabird 2000 (1998–2002) are also given.

	Coastal colonies only[1]						
Administrative area or country	*Operation Seafarer (1969–70)*	*SCR Census (1985–88)*[3]	*Seabird 2000 (1998–2002)*	*Percentage change since Seafarer*	*Percentage change since SCR*	*Annual percentage change since SCR*	*Seabird 2000 inland and coastal*
Shetland	541	488	341	−37%	−30%	−2.4%	341
Orkney	910	1,714	1,045	15%	−39%	−3.6%	1,045
North coast Caithness		5	2		−60%	−6.3%	2
East coast Caithness		11		−100%	−100%		
Caithness total	*16*	*16*	*2*	*−88%*	*−88%*	*−13.8%*	*2*
Northwest coast Sutherland		73	44		−40%	−3.8%	44
East coast Sutherland		250	1		−100%	−33.6%	1
Sutherland total	*373*	*323*	*45*	*−88%*	*−86%*	*−13.7%*	*45*
West coast Ross & Cromarty		176	41		−77%	−9.8%	41
East coast Ross & Cromarty			7				7
Ross & Cromarty total	*258*	*176*	*48*	*−81%*	*−73%*	*−8.8%*	*48*
Inverness			6				18
Nairn	6			−100%			
Badenoch & Strathspey							1
Moray	42	10		−100%	−100%		129
Banff & Buchan	15	23	10	−33%	−57%	−5.5%	10
Gordon	1	45	6	500%	−87%	−12.6%	6
City of Aberdeen			154				154
Kincardine & Deeside	3	27	8	167%	−70%	−8.7%	8
Angus		16	7		−56%	−5.5%	7
Perth & Kinross							1,120
City of Dundee			65				65
North East Fife	300	520	1,203	301%	131%	6.2%	1,203
Kirkcaldy	2,000	1,753	3,282	64%	87%	4.8%	3,282
Dunfermline	10	730	1,262	12520%	73%	4.3%	1,262
Clackmannan							88
Falkirk			113				179
City of Edinburgh	2	70	323	16050%	361%	13.7%	323
East Lothian	335	1,130	1,470	339%	30%	2.0%	1,470
Midlothian							1
Berwickshire	7	5	1	−86%	−80%	−11.4%	1
Nithsdale			2				2
Stewartry	645	1,523	1,025	59%	−33%	−3.5%	1,025
Wigtown	6	4	10	67%	150%	6.8%	11
Kyle & Carrick	434	1,925	1,721	297%	−11%	−0.8%	1,721
Cumnock & Doon Valley							4
Clydesdale							40
Cunninghame	1,045	4,573	4,588	83%	0%	0.0%	4,588
Kilmarnock & Loudoun							134
Eastwood							40
East Kilbride							250
Motherwell							34
City of Glasgow			209				640
Renfrew			340				518
Inverclyde		31	190		513%	15.8%	190
Clydebank			30				31
Bearsden & Milngavie							118
Strathkelvin							329
Cumbernauld & Kilsyth							449
Dumbarton		119	143		20%	1.5%	209
Argyll & Bute	4,191	3,203	3,235	−23%	1%	0.1%	3,235
Lochaber	152	165	88	−42%	−47%	−4.7%	88
Skye & Lochalsh	47	351	41	−13%	−88%	−14.7%	41

	Coastal colonies only[1]						
Administrative area or country	*Operation Seafarer (1969–70)*	*SCR Census (1985–88)*[3]	*Seabird 2000 (1998–2002)*	*Percentage change since Seafarer*	*Percentage change since SCR*	*Annual percentage change since SCR*	*Seabird 2000 inland and coastal*
Western Isles– Comhairle nan eilean	692	584	552	−20%	−5%	−0.4%	552
Scotland Total	**12,031**	**19,524**	**21,565**	**57%**	**10%**		**25,057**
Northumberland	1,583	1,035	850	−46%	−18%	−1.4%	858
Tyne and Wear		3	4		33%	2.0%	11
Cleveland		9	90		900%	20.4%	90
North Yorkshire			1				1
Humberside			1				1
Cambridgeshire							5
Norfolk		22	1,605		7195%	34.5%	1,605
Suffolk	150	5,000	6,956	4537%	39%	2.4%	6,956
Essex		3	155		5067%	28.5%	157
Greater London		1	195		19400%	47.5%	257
Kent			75				75
East Sussex	2	1	38	1800%	3700%	32.1%	38
West Sussex		1	36		3500%	30.5%	36
Hampshire			2				2
Isle of Wight	1	2	1	0%	−50%	−4.5%	1
Dorset	1	2	10	900%	400%	12.8%	10
Channel Islands	304	778	1,734	470%	123%	6.7%	1,734
Cornwall	13	11	39	200%	255%	10.0%	39
Isles of Scilly	2,500	3,762	3,603	44%	−4%	−0.4%	3,603
Devon	101	180	426	322%	137%	6.4%	426
Somerset	171	1	64	−63%	6300%	43.3%	74
Avon	1,730	350	768	−56%	119%	6.7%	1,823
Gloucestershire		115	2,224		1834%	23.7%	2,350
Hereford & Worcester							79
West Midlands							40
Derbyshire							30
Merseyside			51				51
Lancashire	7	2[2]	4,167	59429%	208250%	69.9%	22,685
Cumbria	18,175	11,806	22,772	25%	93%	5.2%	22,905
Isle of Man	54	99	114	111%	15%	1.1%	114
England, Isle of Man and Channel Islands Total	**24,792**	**23,183**	**45,981**	**85%**	**98%**	**5.2%**	**66,056**
Gwent	10	50	36	260%	−28%	−2.2%	65
South Glamorgan		2,096	3,381		61%	3.1%	3,381
Mid Glamorgan			0				0
West Glamorgan	1		94	9300%			94
Dyfed	7,782	16,007	15,588	100%	−3%	−0.2%	15,588
Powys							1
Gwynedd	3,727	1,883	1,556	−58%	−17%	−1.3%	1566
Clwyd	9	7	27	200%	286%	8.8%	27
Wales Total	**11,529**	**20,043**	**20,682**	**79%**	**3%**	**0.2%**	**20,722**
Great Britain, Isle of Man and Channel Islands total	**48,352**	**62,750**	**88,228**	**77%**	**41%**		**111,835**
Co. Londonderry							42
Co. Antrim	64	160	485	658%	203%	7.7%	647
Co. Down	159	288	548	245%	90%	4.9%	548
Co. Armagh							164
Co. Fermanagh							447
Co. Tyrone							125
Co. Monaghan							6
Co. Dublin	45	158	334	642%	111%	5.9%	334

Table 1 continued

Administrative area or country	*Coastal colonies only*[1]						*Seabird 2000 inland and coastal*
	Operation Seafarer (1969–70)	*SCR Census (1985–88)*[3]	*Seabird 2000 (1998–2002)*	*Percentage change since Seafarer*	*Percentage change since SCR*	*Annual percentage change since SCR*	
Co. Wexford	468	126	165	−65%	31%	2.1%	165
Co. Waterford	25			−100%			
Co. Cork	132	339	227	72%	−33%	−2.8%	227
Co. Kerry	587	519	792	35%	53%	3.4%	792
Co. Clare	24		66	175%			66
Co. Galway	68	5	37	−46%	640%	18.9%	43
Co. Mayo	60	39	237	295%	508%	15.0%	539
Co. Sligo	16	23	36	125%	57%	3.3%	36
Co. Donegal	35	10	168	380%	1580%	24.0%	668
All-Ireland Total	1,683	1,667	3,095	84%	86%	4.8%	4,849
Britain and Ireland Total	50,035	64,417	91,323	83%	42%		116,684

Notes

[1] Operation Seafarer and the SCR Census did not survey inland.

[2] The Ribble Estuary (Lancashire) colony was not counted during the SCR.

[3] The figures for the SCR are actual counts and do not include adjustments to totals made in order to account for unsurveyed colonies (see Lloyd *et al.*, 1991)

Table 2 Changes in the number of breeding Lesser Black-backed Gulls (AON) at major colonies in Britain and Ireland between the SCR Census (1985–88) and Seabird 2000 (1998–2002). Major colonies are those that contained the top 75% of the British population during the SCR and/or Seabird 2000 or the top 50% of the Irish population during the SCR and/or Seabird 2000. ID corresponds to colony symbols in Fig. 2.

ID	*Colony*	*SCR Census (1985–88)*	*Seabird 2000 (1998–2002)*	*Percentage change since SCR*	*Annual percentage change since SCR*	*Percentage of population in Britain or Ireland (1998–2002)*
1	Skomer	11,760	10,007	−15%	−1.2%	9.2%
2	South Walney	10,000	19,487	95%	5.3%	17.9%
3	Tarnbrook Fell[1]	7,984	18,518	132%	5.4%	17.0%
4	Orfordness	5,000	5,500	10%	0.7%	5.0%
5	Isles of Scilly	3,762	3,603	−4%	−0.4%	3.3%
6	Skokholm	3,531	2,419	−31%	−2.9%	2.2%
7	Little Cumbrae Island	3,000	1,200	−60%	−6.8%	1.1%
8	Flat Holm	1,800	3,309	84%	4.5%	3.0%
9	Ailsa Craig	1,800	400	−78%	−10.2%	0.4%
10	Rockcliffe Marsh[3]	1,791	2,400	34%	2.7%	2.2%
11	Inchkeith	1,753	3,276	87%	4.3%	3.0%
12	Almorness	1,500	1,023	−32%	−3.8%	0.9%
13	Meall A'choire Odhair	1,200	0	−100%		0.0%
14	Craigleith	930	782	−16%	−1.3%	0.7%
15	Inchmarnock Island	825	200	−76%	−8.5%	0.2%
16	Inchcolm	730	1,221	67%	3.5%	1.1%
17	Horse Island	641	2,677	318%	10.7%	2.5%
18	Isle of May	520	1,203	131%	6.2%	1.1%
19	Bristol[2]	518	850	64%	3.5%	0.8%
20	The Skerries	453	574	27%	1.7%	0.5%
21	Gloucester (incl. RAF Quedgeley)[1,2]	450	2,250	400%	10.4%	2.1%
22	Cardigan Island	375	1,648	339%	11.1%	1.5%

ID	*Colony*	*SCR Census (1985–88)*	*Seabird 2000 (1998–2002)*	*Percentage change since SCR*	*Annual percentage change since SCR*	*Percentage of population in Britain or Ireland (1998–2002)*
23	Bardsey Island	225	634	182%	7.7%	0.6%
24	Sark	154	555	260%	10.8%	0.5%
25	Fidra	140	599	328%	12.8%	0.5%
26	Lady Isle	125	1,000	700%	12.3%	0.9%
27	Hesketh and Banks Marshes, Ribble Estuary	nc	4,100			3.8%
28	Outer Trial Bank		1,378			1.3%
29	St Serfs Island, Loch Leven[1]		1,103			1.0%
30	Lowestoft[2]		750			0.7%
31	Haverigg Prison[1,2]		430			0.4%
32	Lough Neagh (SPA)[1]	449	494	10%	0.7%	10.2%
33	Lough Mask[1]	447	286	−36%	−4.0%	5.9%
34	Old Lighthouse Island and Mew Island	200	240	20%	1.1%	4.9%
35	Rathlin Island	155	127	−18%	−1.4%	2.6%
36	Lambay Island	150	309	106%	6.2%	6.4%
37	Blasket Sound Islands (Beginish Island, Young's Island)	148	162	9%	0.8%	3.3%
38	Lough Corrib[1]	123	6	−95%	−22.3%	0.1%
39	Cape Clear Island	103	204	98%	5.4%	4.2%
40	Great Saltee	80	144	80%	4.7%	3.0%
41	Puffin Island	55	139	153%	6.4%	2.9%
42	Inishvickillane	30	156	420%	14.6%	3.2%
43	Inishgoosk	nc	500			10.3%
44	Lower Lough Erne[1]	nc	444			9.2%
45	Scariff Island		97			2.0%

Notes

[1] Inland

[2] Roof-nesting

[3] SCR 1988 count estimated from a mixed count of 2,500 AON of Herring Gulls and Lesser Black-backed Gulls using a ratio of 2.53LBBG:1HG measured in 1990.

nc not counted

Table 3 Numbers of roof-nesting Lesser Black-backed Gulls (AON) in Britain and Ireland 1976–2002.

	1976[1]		*1993–95*[2]		*1999–2002*			
Administrative area or country	*AON*	*Colonies*[3]	*AON*	*Colonies*[3]	*AON*	*Colonies*	*Percentage change since 1976*	*Percentage change since 1993–95*
East coast Sutherland					*1*	*1*		
Sutherland total					1	1		
Inverness					6	1		
Moray					28	1		
Banff & Buchan			5	2	4	1		−20%
City of Aberdeen			50	1	154	1		208%
Angus					7	2		
City of Dundee			7	1	65	1		829%
Kirkcaldy					6	1		
Dunfermline					38	1		
Clackmannan					88	2		
Falkirk			4	1	179	5		4375%
West Lothian			15	2	nc			
City of Edinburgh			107	3	nc			
East Lothian			1	1	nc			
Kyle & Carrick			120	2	311	4		159%
Cunninghame			20	1	97	1		385%
Kilmarnock & Loudoun					134	1		
East Kilbride					250	1		
Motherwell					28	2		
Monklands					6	1		
City of Glasgow			349	1	1,553	1		345%
Inverclyde			141	1	189	2		34%
Strathkelvin			2	1	119	1		5850%
Cumbernauld & Kilsyth			350	1	449	2		28%
Dumbarton			175	1	134	2		−23%
Scotland Total			1,346	19	3,846	35		186%
Northumberland			4	1	6	1		50%
Tyne & Wear	8	2	46	2	11	1	38%	−76%
Cleveland			9	2	90	3		900%
North Yorkshire			4	2	1	1		−75%
Humberside					1	1		
Suffolk			1	1	1,149	3		114800%
Essex					5	2		
Greater London			11	1	204	1		1755%
Kent			25	2	50	3		100%
East Sussex	1	1	2	1	18	6	1700%	800%
West Sussex					29	4		
Hampshire					1	1		
Dorset			1	1	5	2		400%
Channel Islands			2	1				
Cornwall					14	6		
Devon			6	3	17	3		183%
Somerset			6	1	47	7		683%
Avon	38	2	420	2	1,421	3	3639%	238%
Gloucestershire	80	1	255	1	2,350	4	2838%	822%
Wiltshire			5	1				
Hereford & Worcester			28	2	77	4		175%
West Midlands					40	1		
Derbyshire					30	1		
Merseyside			18	1	51	2		183%
Lancashire			7	2	56	4		700%
Cumbria			104	4	877	5		743%
England,Isle of Man and Channel Islands Total	127	6	954	31	6,550	69	5057%	587%

	1976[1]		*1993–95*[2]		*1999–2002*			
Administrative area or country	*AON*	*Colonies*[3]	*AON*	*Colonies*[3]	*AON*	*Colonies*	*Percentage change since 1976*	*Percentage change since 1993–95*
Gwent	9	1	14	4	65	2	622%	364%
South Glamorgan	170	2	101	3	72	3	−58%	−29%
Mid Glamorgan	19	2						
West Glamorgan			71	2	94	2		32%
Dyfed					112	5		
Gwynedd			9	3	25	6		178%
Clwyd			6	1	26	3		333%
Wales Total	198	5	201	13	394	21	99%	96%
Great Britain, Isle of Man and Channel Islands total	325	11	2,501	63	10,790	125	3220%	331%
Co. Antrim			8	1	63	1		688%
Co. Dublin					21	1		
All-Ireland Total			8	1	84	2		950%
Britain and Ireland Total	325	11	2,509	64	10,874	127	3246%	333%

Notes

[1] Source data: Monaghan & Coulson (1977)

[2] Source data: Raven & Coulson (1997)

[3] The numbers of colonies do not include those in 1976 and 1993–95 that were not counted.

nc not counted

Table 4 International context.

		Number of pairs			
Country or region	*Subspecies*	*Min*	*Max*	*Year*	*Source*
Great Britain, Isle of Man and Channel Isles	*graellsii*	117,000	117,000	1999–2002	Seabird 2000
All Ireland	*graellsii*	4,800	4,800	1999–2002	Seabird 2000
Belgium	*intermedius*	180	180	1996	BirdLife International /EBCC (2000)
Denmark	*intermedius*	4,400	4,400	1993–96	BirdLife International /EBCC (2000)
Estonia	*fuscus*	250	300		BirdLife International /EBCC (2000)
Faeroes	*graellsii*	9,000	9,000	1995	BirdLife International /EBCC (2000)
Finland	*fuscus*	6,000	8,000	1990–95	BirdLife International /EBCC (2000)
France	*graellsii*	22,655	22,655	1997–2001	Cadiou *et al.* (in press)
Germany	*intermedius*	3,700	9,300	1993	BirdLife International /EBCC (2000)
Greenland	*graellsii*	10	20	1996	BirdLife International /EBCC (2000)
Iceland	*graellsii*	25,000	25,000	1990	BirdLife International /EBCC (2000)
Netherlands	*intermedius*	32,000	57,000	1992–97	Bijlsma *et al.* (2001)
Norway	*fuscus & intermedius*	25,000	36,000	1990	BirdLife International /EBCC (2000)
Poland	*fuscus*	0	10		BirdLife International /EBCC (2000)
Portugal	*graellsii*	5	10	1989	BirdLife International /EBCC (2000)
Russia	*fuscus*	2,120	2,300	1998	BirdLife International /EBCC (2000)
Spain	*graellsii*	240	300	1990	BirdLife International /EBCC (2000)
Sweden	*fuscus & intermedius*	15,000	20,000	1990	BirdLife International /EBCC (2000)

Biogeographic region		*Min*	*Max*	*Min % GB*	*Max % GB*	*Min % Ireland*	*Max % Ireland*
Greenland, Iceland, Faeroes, Britain, Ireland, France, Spain and Portugal*	*graellsii*	179,000	179,000	65.4%	65.4%	2.7%	2.7%
World	all	267,000	316,000	37.0%	43.8%	1.5%	1.8%

*Stroud *et al.* (2001)

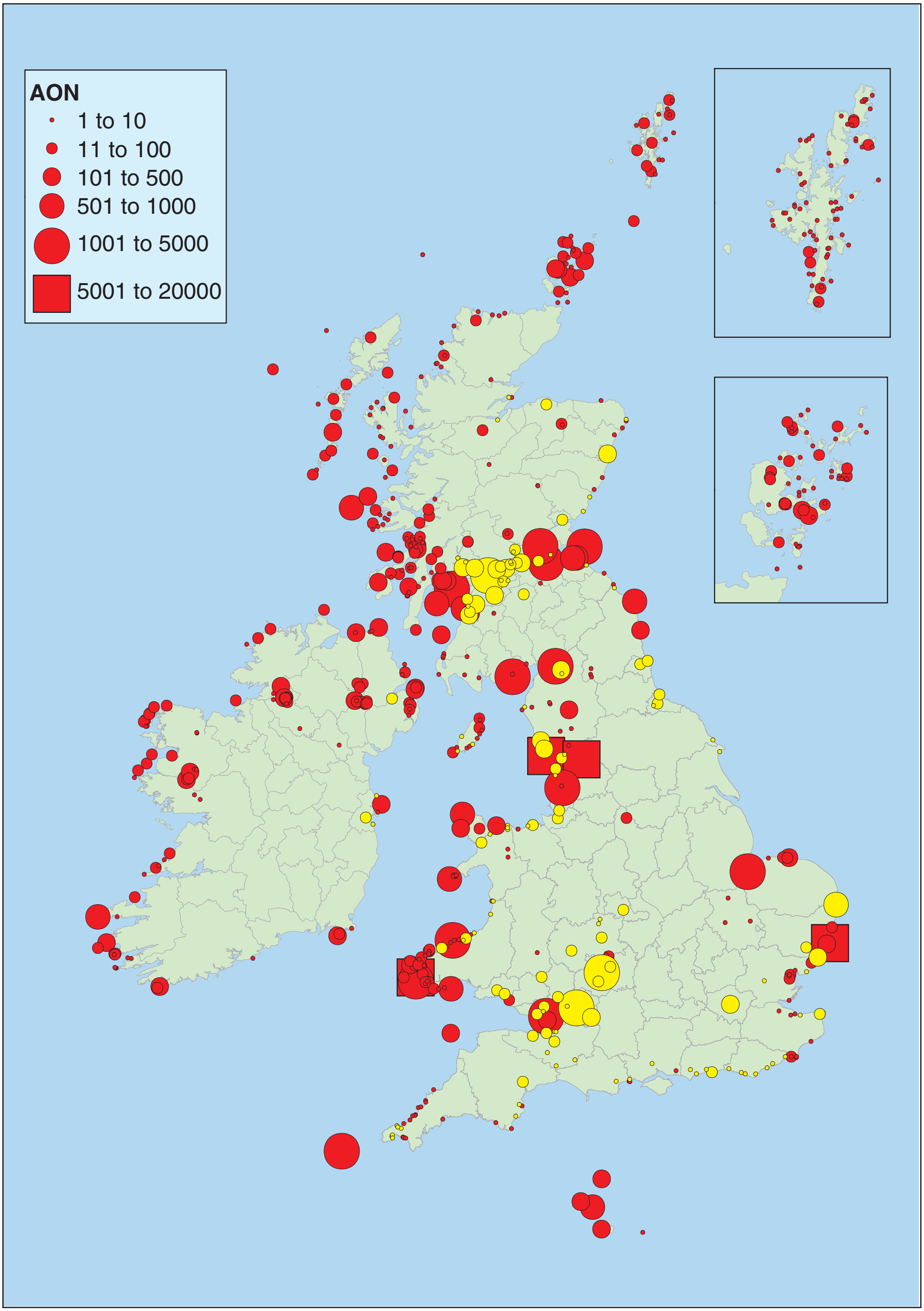

Figure 1 *Abundance and distribution of breeding Lesser Black-backed Gulls in Britain and Ireland 1998–2002. Natural sites are shown in red and man-made site (e.g. rooftops) are in yellow (the scale is the same for both types of site).*

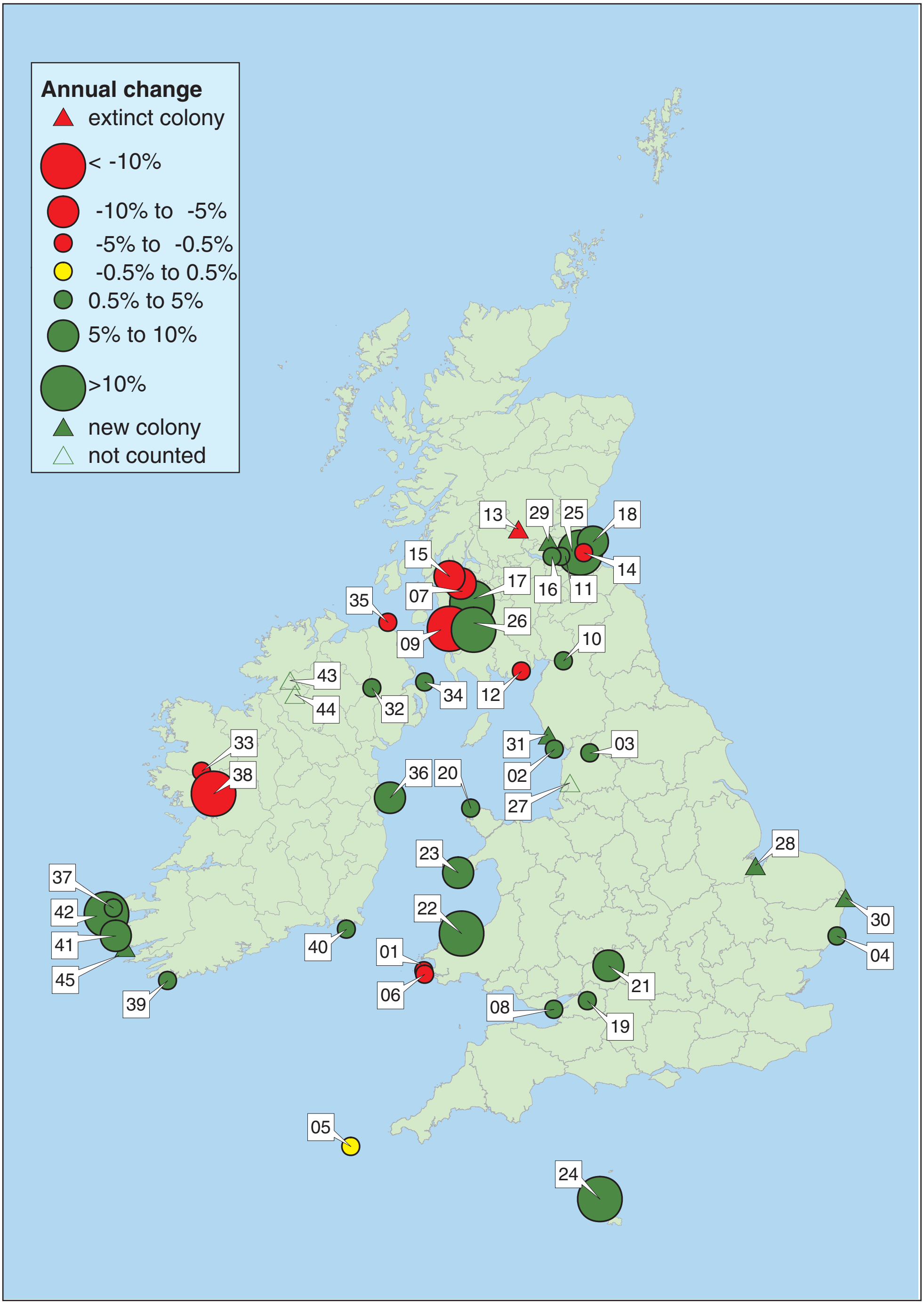

Figure 2 *Changes in the number of breeding Lesser Black-backed Gulls (AON) at major colonies in Britain and Ireland between the SCR Census (1985–88) and Seabird 2000 (1998–2002). Major colonies are those that contained the top 75% of the British population during the SCR and/or Seabird 2000 or the top 50% of the Irish population during the SCR and/or Seabird 2000. Numbers correspond to colonies listed in Table 2.*

Scottish Natural Heritage
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INTERIM GUIDANCE ON APPORTIONING IMPACTS FROM MARINE RENEWABLE DEVELOPMENTS TO BREEDING SEABIRD POPULATIONS IN SPECIAL PROTECTION AREAS

Updated November 2018

Introduction

A key element of impact assessments for proposed marine renewable (offshore wind, wave and tidal) energy sites is the possible impact that such developments will have on breeding seabird populations. Many of the seabirds using marine renewables sites will breed in protected areas, including Special Protection Areas (SPAs). The questions which then follow are: from which SPAs do these birds originate and in what proportion?

This guidance is based on original work by Mark Trinder (MacArthur Green) and Andy Douse (SNH). It focuses solely on effects on seabirds during the breeding season. The Statutory Nature Conservation Bodies (SNCBs) are also considering issues surrounding the assessment of impacts on seabirds outwith the breeding season, which will lead to further guidance for developers and regulators.

Scope of this paper

We present a range of methods to determine connectivity with breeding seabird SPAs. These include both data-led empirical approaches and a theoretical approach for estimating how birds using marine renewables development sites can be 'apportioned' to multiple source colonies.

Future Approaches to Apportioning
Marine Scotland have been developing a tool to use for attributing birds at sea to origin colonies (see https://www2.gov.scot/Topics/Research/About/EBAR/research-opportunities/AttributingSeabirdsAtSeaToAppropriateBreedingColon#). This approach uses seabird density predictions based on the work of Ewan Wakefield an others (Wakefield et al 2017). These predictive maps are available for four seabird species Shag, Common Guillemot, Kittiwake and Razorbill. When the outputs of this work are available the approach outlined in this guidance may no longer be relevant, at least for those species.

A theoretical approach

Connectivity to SPAs is largely based on determining seabird foraging ranges. This theoretical approach to apportioning uses published seabird foraging range information and generalised models. Empirically derived approaches (see below) use site-specific field-derived data on foraging ranges and locations. In the absence of sufficient survey or tagging data, theoretical approaches are the only option.

The simplest approach, that has been adopted at some inland development sites, is to assess impacts against each SPA within foraging range separately and in turn, assuming each time that all the birds within the development site originate from the SPA in question. For all its simplicity, this approach is fundamentally flawed in that impacts will inevitably be overestimated at every SPA and particularly at SPA sites with relatively small species populations. We do not recommend this approach in the marine environment where large foraging ranges mean that multiple SPAs will often be included in such an analysis.

In this theoretical approach, as available data are currently limited, we make some necessary and very general assumptions. We acknowledge that the resulting picture of seabird distribution and habitat use at sea is not accurate, but the approximations are aimed at creating a workable tool for the assessment of impacts on SPAs. Our aim is to present a practical solution that has an agreed basis and can be widely adopted. In future we hope that further data will be available to help refine this approach.

The method is based on foraging range and three colony-specific weighting factors:
(i) Colony size (in individuals);
(ii) Distance of colony from the development site[1]; and
(iii) Sea area (the areal extent of the open sea within the foraging range of the relevant species).

Foraging Range

Breeding seabirds are central place foragers i.e. they have to leave the colony (the central place) and disperse to find food before returning to the breeding colony. There is an energetic and time advantage in feeding as close to the colony as possible (all else being equal) and a limit to how far birds will travel, as beyond this distance a bird becomes unable to find enough food fast enough to bring back to the nest in order to breed successfully. This suggests that there is a limit to the distance travelled which will be approximated by the maximum recorded foraging range for that species (in that location). As seabirds generally avoid crossing any substantial land-mass the effective distance from colony should be measured as the ‘by-sea’ route.

To determine the SPAs for which there may be connectivity (i.e. which are within foraging range), we recommend using the single mean maximum value from Table 1 of Thaxter et al 2012 (which itself is a review of information across many studies available at the time). In some cases, where no estimate of foraging range for a species has yet been published, it may be justifiable to use values from a related species with strong ecological similarities when those are available.

The HRA of the Sectoral Plans included a systematic review of foraging ranges for seabirds in Scottish SPAs. The summary report is online. Table E1 of Appendix E in the report gives value of maximum ranges for seabird species which may be useful in some cases.

The amount of information on seabird foraging range is expanding rapidly due to the increasing numbers of tracking studies (e.g. the FAME[2] project) using technology such as GPS tags. These studies provide temporally and spatially explicit data on the ranging and behaviour of key species. Although at present we advise use of the ranges published in Thaxter et al 2012, we recommend that developers conduct a thorough review of all the available data and contact SNH seabird specialists to agree appropriate foraging ranges if newer data are available.

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[1] Normally using the geometric centre of both and without crossing substantial areas of land

[2] Future of the Atlantic Marine Environment (FAME). See http://www.fameproject.eu/en/

Weighting by colony size

Large colonies will contribute more individuals to the number of birds found within a development site, all other factors being equal. Population size is best measured in numbers of individuals (which can be converted from breeding pairs or apparently occupied site). As long as count units are consistent between colonies, the actual unit is not particularly important, as weighting values are relative.

All colonies being considered should have been counted concurrently or at the very least at a similar time. A suitable reference point for most species will be Seabird 2000 data, and though we acknowledge that these data are now relatively old, at present this is the recommended dataset. A new seabird census is planned and will provide newer data in a few years' time. Where count data are old , species-specific change metrics can be used to correct population counts to approximate the current likely population sizes but there must be clear justification for doing so. Such change metrics will usually be based on regional population trends, so the resultant weighting factor is insensitive to absolute values of population size.

For colonies that occupy long stretches of coastline (and may consist of separate 'sub-sections') a single weighting factor may not be appropriate. In such cases different sub-sections may be treated as separate colonies. Most seabird colonies will have separate count sections (see Seabird Colony Register)[3]. If a single SPA is made up of several sub sections or colonies the combined SPA impact can be reconstructed after the weighting for each sub section is completed.

The birds using proposed development areas will often include a mix of birds from SPA and non-SPA colonies. Non-SPA birds must be included in the analyses; otherwise impacts on SPA breeding birds will be exaggerated. This will require knowledge of the sizes of non-SPA as well as SPA colonies in the area of search.

If the apportioning is to relate to breeding adults only, the proportion of adult birds that do not breed in any one year (so called breeding sabbaticals) may be taken into account if known. More significantly, there will be a proportion of non-breeding age birds present in the proposed development area, and this will require determination of the age of seabirds during field surveys of the site. Where birds cannot be aged in the field, published life-tables may be used to estimate of the expected proportion of adults different age classes. Assessors should be aware that for some species, sub-adult and immature age classes may use different geographical areas to those frequented by adult birds. A good knowledge of seabird ecology is fundamental to such analyses.

Weighting by distance from the colony

The distance of the colony should be measured as the distance between the geometric centre of the development to the geometric centre of the colony. We acknowledge that colonies with complex boundaries might be more sensibly measured from a different location, which should be specified. As an example if it is known that a species has a strongly skewed distribution within a large SPA it might be more suitable to measure from the centre of that distribution. A cautious approach would be to use the 'nearest boundary to nearest boundary' for an individual SPA, although this would then have impacts on the apportioning to other sites being considered.

Development sites that are distant from a seabird colony/s might be expected to have fewer birds on them from that colony than development sites close to a colony/SPA. Thus the

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[3] For Seabird Colony Register data, see http://jncc.defra.gov.uk/page-1776

contribution of any particular colony to the number of birds seen at a given development site will be inversely related to distance between the development site and the colony.

As birds radiate out from a colony density will decrease by a factor proportional to 1/distance^2 as area increases proportionally by $\pi.r^{2.}$ For simplicity a weighting factor based on 1/distance$_i^2$ is used here. The foraging range should be applied to the by-sea or sea route that birds would take to reach foraging locations. This prevents the implausible situation where species (particularly with very large foraging ranges e.g Gannet, Fulmar) from a colony on the west coast would be expected to be found at an east coast location by traversing the land between.

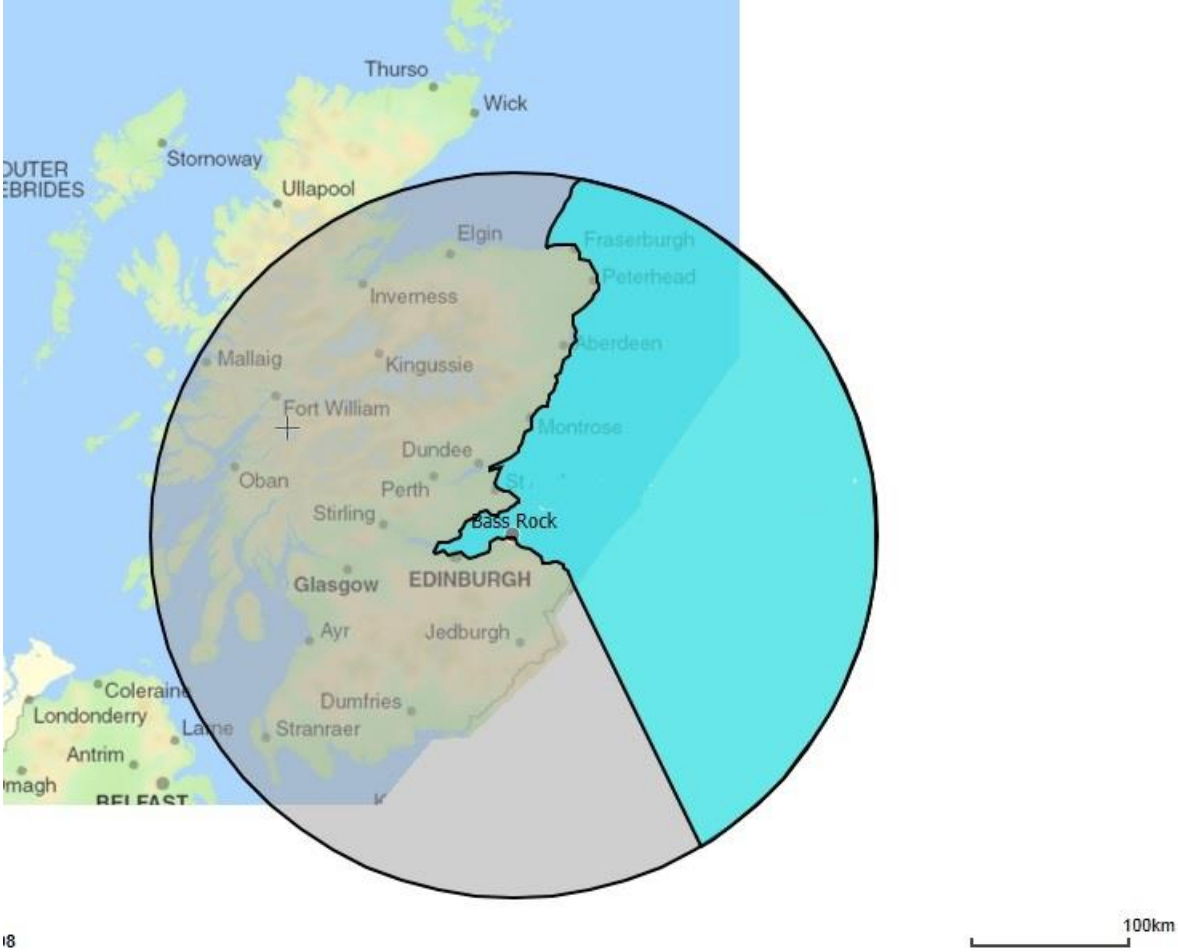


Figure 1. Illustration of effect of sea-route foraging range versus 'straight-line' foraging range. The blue area is the true area of sea accessible to a species with a foraging range of 230km from Bas Rock, whereas the grey circle shows the result of applying the straight-line distance, which includes sea areas on the west coast which would not be accessible.

Weighting by sea area

We suggest that a correction is also made for the available sea area. The logic behind this is that for a colony on an island, seabird foragers may have all 360^o of marine habitat to forage over, whereas for a mainland colony, birds may only have about 180^o of marine habitat. The available sea area can be measured by plotting a circle defined by species-specific foraging range round the colony in GIS and calculating the area of sea available to each seabird species. The fraction of the disc centred on the colony that is occupied by sea surface is then expressed as a decimal. As the density of birds will increase as the area of available forage area decreases this should be used in the formula as 1/estimated area.

We acknowledge that seabirds are very unlikely to be distributed evenly across all the available sea area; the distribution will be patchy and likely related to prey availability. However without specific evidence and in the interests of not overcomplicating any estimation this theoretical approach cannot provide finer scale differentiation.

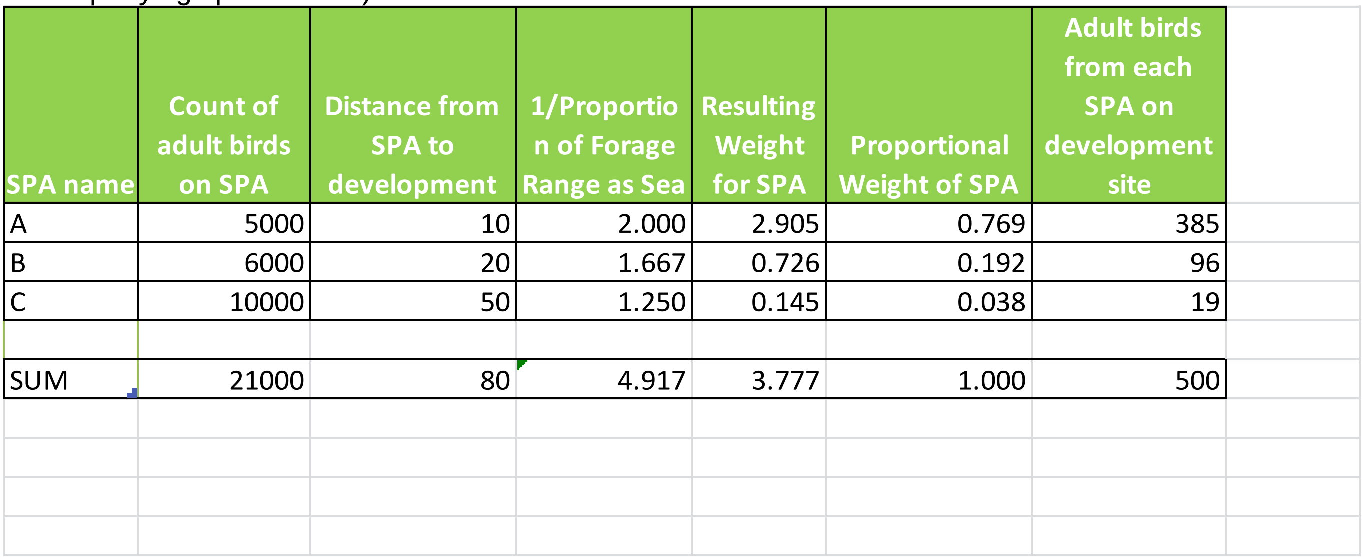
Combining the three weighting factors

The weighting factors should be combined to produce an overall weighting. Each factor is given equal weight in the combined weighting, as there appears to be no compelling reason or specific evidence for any other treatment.

Example 1

This illustrates the apportioning approach for three colonies A, B and C for an imaginary bird species. Data on populations, distances from a theoretical development site and sea areas are given in the table, along with the resulting weightings and the combined weighting. Here the population considered is number of breeding adults - immature birds will not be included. As noted above if necessary the development site counts should be adjusted to account for this. In this case the maximum number of birds on the development site is 725 birds. 225 are aged as sub-adult or immature, with 500 birds in adult plumage. The calculations estimate the contribution of each SPA to this total.

A table using and embedded excel spread sheet showing these calculations is presented below. Double click on the table to edit values of the SPA population count, distance to the development and the proportion of the foraging area that is sea to change the value of weighting and the resultant calculation of the number of birds allocated to each SPA on the development site. (This function not available in pdf format version of this document use the accompanying spreadsheet)



SPA name	Count of adult birds on SPA	Distance from SPA to development	1/Proportion of Forage Range as Sea	Resulting Weight for SPA	Proportional Weight of SPA	Adult birds from each SPA on development site
A	5000	10	2.000	2.905	0.769	385
B	6000	20	1.667	0.726	0.192	96
C	10000	50	1.250	0.145	0.038	19
SUM	21000	80	4.917	3.777	1.000	500

The calculation is made as follows:

Weight = (Colony Population / Sum of Populations) ***** (Sum of Distance^2 / Colony Distance^2) ***** (1/Colony Sea Proportion / Sum of 1/Sea Proportions).

(Note that the second part of the calculation (distance) is inverted as there is an inverse relationship with distance, following the inverse-square law.)

In the first line of the table (for colony A) the figures are:

(5000/21000)*(3000/100)*((2.0)/4.197) = 2.905

The weighting is then used to calculate the proportion of birds attributed to that SPA (SPA Prop) by calculating (SPA weight / sum of all weights) – in this case 2.905/3.777 = 0.769, this proportion is then used to calculate the number of birds on the site (multiply the total number on site by the proportion allocated to this SPA) 0.769 * 500 = 385 birds.

Disadvantages of the theoretical approach

This theoretical approach is a great simplification of a complex set of factors. It involves assumptions that are unlikely to be a good reflection of reality, but which have the merit of being simple and applicable through an easily understood working model. This approach offers a logical, potentially standard, basis for estimating the relevant contribution of birds from differing source colonies to the numbers seen in a given area where development(s) may be proposed. We suggest this approach is used until a more complex, evidence-based model is developed and agreed.

Some of the simplifications that underlie this calculation are

- It assumes that birds are evenly distributed at sea. This is extremely unlikely, when key prey species (e.g. lesser sandeel) are known to be patchily distributed. Indeed, tracking studies have shown that some seabirds will travel very long distances, to *particular* foraging locations. In most studies the distributions of birds and prey in the area under investigation will be unknown.
- It assumes that seabird colonies are ‘independent’ of one another. There is some evidence that seabird colonies may ‘interfere’ with one another, thus affecting foraging range (Furness & Birkhead 1984, Cairns 1989, Wakefield et al 2012) but taking this into account for the purpose of apportioning impacts arising from marine renewables developments is likely to be impractical. Studies which show this effect are few or have small samples of tracked birds, although the theory is quite feasible. There is some contrary evidence that birds from different adjacent colonies can share foraging areas (Dean et al 2013) or a hybrid of separate areas and overlapping areas (Ainsley et al 2004).
- Seabirds breeding at large colonies may have larger foraging ranges, due in part to competition and prey depletion near to the colony. This effect, known as the Storer-Ashmole’s Halo (Elliot *et al.* 2009), has rarely been demonstrated in practice, though it has a sound theoretical basis (Gaston et al 2007). It is not known whether there is evidence for this from current bird-tagging studies, although something similar does appear to be found in bumble bees (Goulson and Osborne 2009). At present we regard it as impractical to factor Storer-Ashmole’s Halo into this analysis.

Advantages of the theoretical approach

- Seabird foraging ranges can vary considerably within and between years. The theoretical approach represents an ‘averaging’ over a long time period and across sites, in a way that tagging studies and other empirical approaches do not.
- The relative simplicity of the data needed and their application provides cost savings over empirical approaches

- It provides a mechanism for considering all species, from all relevant colonies even when the number of colonies to be considered is large, in which case empirical approaches are unlikely to be viable.

Requirements for theoretical apportioning impacts to SPAs

In summary to undertake this analysis the following is required.

- Use of proposed development site by seabirds, obtained from field survey (should such data not already exist)
- Agreed foraging range value for all species of interest (from which the proportion of sea area within foraging range will be calculated)
- Concurrent population estimates (in numbers of individuals) for all seabird colonies within foraging ranges
- Accurate determination of distances from colonies to proposed development site

Data led empirical approaches

In principle it is advantageous to apply field data for a specific development to help to determine the origin of birds within the development boundary. There are several methods of collecting data and supporting evidence. The results could be used jointly with the theoretical approach in some cases.

Tagging studies (GPS equipped loggers or satellite tags)

Data from birds tagged[4] at different colonies can be used to estimate the proportion of birds from each colony using proposed marine renewable development sites. There is a need to correct for sample sizes of birds tagged from each colony, and to weight for the population size of each colony. Observations can also be weighted by time spent in the development site. We have already noted that seabird foraging ranges can vary considerably within and between years. The theoretical approach overcomes this by using ‘typical’ ranges derived from many studies over a long time period and across sites. To overcome this variation with site specific data would require a large number of tagged birds, from concurrent studies at all (or most) colonies under consideration, at each stage of the breeding season and over a number of years with considerable analysis costs. Almost all studies use only a relatively small number of tags, often deployed over a limited timescale. Most are behavioural or ecological studies, not strictly targeting the questions of apportioning origins of birds

Tagging studies have a number of significant advantages over the theoretical method:

- Assumptions about how far birds travel, or indeed where they travel to, do not have to be made.
- Weighting by time spent in the development area is possible for GPS tagged birds.
- Time spent in the development area (and potentially elsewhere) can lead to estimates of turnover.
- Data can be analysed at smaller timescales than across a whole breeding season, depending on the quantity available. This means that the ‘contribution’ each SPA makes to birds present within a proposed marine renewables development site can be calculated at different stages of the breeding cycle if sufficient data exist.

Disadvantages include:

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[4] Cheaper GPS tags are widely available now, though require birds to be re-caught and tags removed. Tags that down-load data automatically to a nearby base-station are also available but are more expensive.

- Capture and handling for tagging studies may cause considerable disturbance and stress to breeding birds. As a result, and due to the added burden of carrying a tag, tagged birds may not behave normally. Some species are less amenable to tagging than others (Baron et al 2010, Vandebeele et al 2011).
- Tagging studies may be difficult or even impossible to conduct safely at some colonies due to topography.
- The demanding resource requirements. For any study a sufficient number of birds must be tracked (of potentially multiple species) at each stage of the breeding season and across more than one year. Furthermore, when there are several colonies within foraging range, studies are needed from all SPA (and non-SPA) sites under consideration. In addition, tagging studies must be carried out by trained and experienced personnel.
- Tagging studies are likely to be viable only in situations where there are just a few colonies, each of which can be well-studied. In complex areas with multiple SPAs and other non-SPA colonies within range, tagging studies will be challenging and expensive to undertake.

Feeding locations can also be determined by triangulation of radio-tagged birds, though this technique quickly becomes unreliable as foraging distance increases.

Field Surveys to Support Theoretical and Empirical Methods

These survey results could be used to ‘sense check’ the theoretical apportioning, or to add support to tagging studies. As more tagging data are obtained these can be used to describe seabird abundance at sea. Longer attachment periods will bring the added benefit of being able to develop predictive models of densities beyond the breeding season.

a. Gradients of Survey density data

For surveys that cover very large areas (all surveys should cover a buffer area beyond the development site boundaries), then analyses of the gradient of bird densities away from the development site in the direction of each relevant seabird colony SPA (and non-SPA) may yield information on the origin of birds coming from each SPA. The premise is that if an SPA is ‘contributing’ birds to a marine renewables development site, then survey data extending beyond the development site boundary would be expected to show an increasing density of birds in the direction of the relevant colony or colonies. Aerial survey data could be suitable for such analyses, given the ability of planes to cover large areas of sea in short timescales. However, as aerial surveys are usually only samples based on flying transects, it is also probable that geospatial statistical techniques will be needed to underpin the analytical approach, in order to derive a two-dimensional density space.

Key advantages from such an approach are:

- The use of real survey data over a wide spatial area, some of which may have been gathered already as part of other survey work.
- It may be suitable for species that have very large foraging ranges

However, disadvantages include:

- The potential requirement for complex analytical statistical methods.
- The likelihood that, even if connectivity can be established with particular seabird colonies, assessing their relative contribution would be technically demanding and probably subject to a higher degree of uncertainty than from tracking data.
- The need for surveys to extend over a significantly greater area than the usual ‘development area plus buffer’. Although as an additional cost on top of an existing

survey programme this would probably prove considerably less expensive than a separate tagging study.

For these reasons we do not consider that this technique can be used to allocate apportioning alone, however it does have a place in providing some evidence of origin of birds at sea.

We are not aware of this method being used in any practical situation. Developers wishing to develop an approach using this method should discuss this with SNH.

b. Flight line data

This makes use of the fact that foraging seabirds generally return to a colony along a direct route after successful foraging. Before the widespread use of cheap, recoverable GPS technology on birds, this was the only suitable technique. Flight lines may be observed visually and small radio-transmitters can provide better data on departure and arrival directions (with the advantage that radio tags don't need to be retrieved unless there are welfare reasons. Establishing the proportion of birds returning to a colony from the direction of the development area will give an indication of:

- Whether they are likely to be using the development site (i.e. if birds do not return from the development site direction then this may not be an area used).
- The number or proportion of birds using the development site (or general area around the development site).

Flight directions of birds departing from a colony (heading off to forage) are less informative given that birds may have to spend time searching for a patchy and unreliable food resource. However, the recent use of boats (e.g. fast Rigid Inflatable Boats) to follow foraging terns has been very successful in establishing feeding areas for species that do not forage over long distances (Perrow *et al.* 2011).

Advantages of flight line data analyses are:

- That they may be relatively cheap to undertake, requiring little more than a cliff-top observer with a pair of binoculars and a compass for inshore developments.
- That a large volume of data can be gathered from multiple locations, across the breeding season and in a variety of weather conditions;

The technique also has significant disadvantages, in that:

- The direction of travel cannot identify the actual area over which birds were foraging, though combining directional data with time spent away from the colony by ringed or colour marked individuals, can (using flight speeds) identify *potential* foraging areas.
- Birds may forage over more than one area before returning to the colony, such that the direction of flight only indicates the *last* foraging location.
- Birds may not travel in a straight line back from a foraging area.
- Birds may gather in rafts on open water at some distance from the colony before returning to the colony (e.g. shearwaters), or head for loafing/preening/bathing sites before returning to the colony (e.g. kittiwakes at some colonies).

In theory this could produce similar results to tagging studies as flight line data should replicate data from tagged birds which indicate their origins, although surveys would need to identify popular feeding locations to achieve this . However in most cases this is the lowest quality data with large uncertainty attached to estimates of the connectivity to SPAs derived from these methods.

With Digital Aerial Surveys the direction of flight can be determined by the orientation of the bird's body. However as stated above it is birds travelling back to colonies (after successful provisioning foray) that are most likely to give more accurate indication of true direction of origin. It is thus only birds carrying fish or birds with full crops (if this can be ascertained - some species carry food in the stomach) that should be included in any such data-gathering exercise. This is usually only possible with land-based or boat-based visual surveys.

Given the large uncertainty associated with information from this type of survey, once again we consider it is useful supporting evidence, rather than a method to allocate proportions of birds on development sites.

Integrating Theoretical and Empirical approaches

It is probable that some colonies under consideration for a development will have empirical (tagging) data whereas others will not. Most likely the largest, most important colonies (e.g. SPAs) will get this attention. In this case it would be sensible to try and use this information.

We suggest at present that theoretical apportioning should be applied to all colonies under consideration, and then the apportioning that is allocated to sites with tagging data may be re-apportioned depending on the results of the tracking analysis. For instance, site-specific data may show that the birds' behaviour grossly violates assumptions e.g. they forage in a completely different direction.

Tracking data can give results for the number of birds entering the development area, the time spent within the development area, or the total number of tracks that reach the development site. To compare with on-site counts which use the mean peak numbers of birds on site, we suggest that a proportion calculated from the mean daily amount of time spent on the development site by tagged birds, weighted by the size of the colony, is used to apply to the proportion already allocated to the sites with tagged birds. This calculation would require knowledge of the time spent by each individual bird within the development area for each day of the tracking / tagging period.

Example 2

From Theoretical Apportioning (see Example 1 above) the resulting summed values are:

Non-tagged sites – combined proportion 0.65
Tagged sites - combined proportion 0.35

There are three sites with tracked birds. Using the data from each bird a calculated mean proportion of each day spent within the development area is known.

This requires information from each tag and the length of each foraging day (e.g. daylight period).

For each site the population size is known, the sample size of tagged birds is assumed to be adequate and selected from a random sample of breeding locations within the site. The tracking period should be representative of the breeding season and comparable for all sites.

Site	Colony Population	Mean proportion of day each tagged bird present on development area	Proportion of Day * Colony Population	Final proportion from tagging data	Theoretical proportion * Tagging Proportion
A	5000	0.187465	937.325	0.566	0.198
B	6000	0.11684	701.040	0.423	0.148
C	10000	0.001823	18.230	0.011	0.004
SUM			1656.595		

The table may be edited (to see effects of changing the values of proportion of day each bird is on development) by double clicking in any cell. In the pdf version of this document use the accompanying spreadsheet.

The calculation is made as follows:
Step 1 – Each Site weight is calculated : Colony Population *(Time birds active in development area /Sum of all time birds active)
Step 2 - Proportion for individual site within tagging group calculated: Site weight / Sum of all site weights
Step 3 – The final proportion calculated : site proportion within group * group proportion

> In the first line of the table above (for colony A) the figures are:
> Site weight : (5000*(305/1625)) = 937
> Within group proportion : 937 /1657 = 0.566
> Final proportion: 0.566*0.35 = 0.198

Concluding remarks

The process of apportioning impacts across multiple SPAs is necessarily an approximate one, ideally undertaken with empirically derived site-specific data. However, the data required to account for intra- and inter-annual variation in foraging behaviour will be considerable in most situations and onerous to collect. In the absence of good data, we recommend a theoretical approach though supplementary site-specific data may allow more realistic apportioning. However, tracking studies have shown that some seabirds will travel very long distances, to *particular* foraging locations. FAME project data have shown that seabirds from some Orkney breeding colonies are known to travel south into the outer Moray Firth, to feed in very specific areas (most likely targeting prey aggregations). Manx shearwaters breeding on Rum travel long distances south into the Irish Sea, whereas many fewer birds venture far north. These studies illustrate such models of feeding distribution are likely to be over-simplifications.

Our collective understanding of the journeys made by foraging seabirds is improving rapidly, and new techniques based on this improving information may soon replace the approach advocated here. At present we recommend that apportioning impacts arising from marine renewables developments are assessed using the methods described above, even if they are relatively simple approximations.

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Breeding density, fine-scale tracking, and large-scale modeling reveal the regional distribution of four seabird species

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Abstract. Population-level estimates of species' distributions can reveal fundamental ecological processes and facilitate conservation. However, these may be difficult to obtain for mobile species, especially colonial central-place foragers (CCPFs; e.g., bats, corvids, social insects), because it is often impractical to determine the provenance of individuals observed beyond breeding sites. Moreover, some CCPFs, especially in the marine realm (e.g., pinnipeds, turtles, and seabirds) are difficult to observe because they range tens to ten thousands of kilometers from their colonies. It is hypothesized that the distribution of CCPFs depends largely on habitat availability and intraspecific competition. Modeling these effects may therefore allow distributions to be estimated from samples of individual spatial usage. Such data can be obtained for an increasing number of species using tracking technology. However, techniques for estimating population-level distributions using the telemetry data are poorly developed. This is of concern because many marine CCPFs, such as seabirds, are threatened by anthropogenic activities. Here, we aim to estimate the distribution at sea of four seabird species, foraging from approximately 5,500 breeding sites in Britain and Ireland. To do so, we GPS-tracked a sample of 230 European Shags *Phalacrocorax aristotelis*, 464 Black-legged Kittiwakes *Rissa tridactyla*, 178 Common Murres *Uria aalge*, and 281 Razorbills *Alca torda* from 13, 20, 12, and 14 colonies, respectively. Using Poisson point process habitat use models, we show that distribution at sea is dependent on (1) density-dependent competition among sympatric conspecifics (all species) and parapatric conspecifics (Kittiwakes and Murres); (2) habitat accessibility and coastal geometry, such that birds travel further from colonies with limited access to the sea; and (3) regional habitat availability. Using these models, we predict space use by birds from unobserved colonies and thereby map the distribution at sea of each species at both the colony and regional level. Space use by all four species' British breeding populations is concentrated in the coastal waters of Scotland, highlighting the need for robust conservation measures in this area. The techniques we present are applicable to any CCPF.

Key words: animal tracking; central-place foraging; coloniality; density dependence; habitat use; Poisson point process; species distribution models.

Introduction

Accurate distribution estimates are key to effective wildlife management yet many colonial central-place foragers (i.e., those that return regularly to a common breeding location or refuge) are difficult to observe because they range so widely. Innovations in telemetry are increasingly making it possible to track these species at the individual level (Wikelski et al. 2007, Hart and Hyrenbach 2010, O'Mara et al. 2014), but both theoretical and analytical advances are needed before unbiased, population-level, distribution estimates can be derived from the resulting data (Aarts et al. 2008, Hebblewhite and Haydon 2010). This is of concern because many

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Correction added November 30, 2017: copyright line was changed after original online publication.

colonial central-place foragers are currently suffering unsustainable declines due to human activities (Mickleburgh et al. 2002, Williams and Osborne 2009, Hamann et al. 2010).

Seabirds are one of the world's most endangered avian groups (Croxall et al. 2012). This is due to anthropogenic impacts including invasive species, fisheries bycatch, pollution, and direct exploitation. The distribution and size of seabird breeding colonies has been recorded directly in many regions. In contrast, the distribution of birds at sea is generally estimated from visual survey or more recently, tracking data. Systematic surveys from ships or planes began in earnest in the 1970s (Ainley et al. 2012). They provide coarse-scale (1–10 km) Eulerian data (i.e., observations at fixed points in space) but cannot reliably ascribe provenance or, in many cases, life history stage. Hence, colony-level distributions cannot be estimated using this technique. Since the 1990s, it has also been feasible to track the movements of seabirds using bird-borne devices, which are now becoming sufficiently small and cost-effective to obtain statistically robust sample-sizes for a wider range of species (Burger and Shaffer 2008). Devices are usually deployed at colonies so the origin and status of tracked birds are known. However, while GPS tracking is now providing a wealth of fine scale (10^{-2} km) Lagrangian data (i.e., observations following the animal in space) on distribution, these tend to be from a relatively small proportion of colonies.

Comparatively few studies have so far tracked birds from multiple colonies within metapopulations (Frederiksen et al. 2011, Wakefield et al. 2013, Dean et al. 2015) or across species (Block et al. 2011, Raymond et al. 2015) and fewer still have attempted to estimate the distribution of birds from unsampled colonies using such data (Wakefield et al. 2011, Raymond et al. 2015, Torres et al. 2015). Hence, there is little information on the relative distributions of seabirds from most colonies. This is important both because it hampers conservation (Lewison et al. 2012) and because such information can reveal aspects of the ecology of colonial central-place foragers that have important wildlife management implications (Wakefield et al. 2009). For example, theory predicts that density-dependent competition among seabirds breeding in the same colony (sympatric competition), mediated either through prey depletion or disturbance, leads to a positive relationship between colony size and foraging range, ultimately regulating colony growth (Ashmole 1963, Lewis et al. 2001). Similarly, density-dependent competition between colonies (parapatric competition) may lead to spatial segregation of the utilization distributions (UDs) of adjacent colonies (Wakefield et al. 2013; a UD is defined as a population's spatial probability distribution; Fieberg and Kochanny 2005). Evidence has been advanced in support of both hypotheses (Lewis et al. 2001, Masello et al. 2010, Catry et al. 2013) yet it remains uncertain how sympatric and parapatric intraspecific competition, foraging costs (which increase with distance from the colony) and resource availability interact to shape the UDs of breeding seabirds and other colonial central-place foragers (Wakefield et al. 2013). For example, the size and shape of colony UDs depend on the density of conspecifics but this is a function not only of colony size and resource availability but also the area of sea accessible from that colony, which in turn varies with coastal morphology (Birkhead and Furness 1985). Hence, we might predict birds breeding at colonies with restricted access to the sea travel further than those breeding on isolated islands. Moreover, although it is clear that seabirds breed in hierarchically nested aggregations (i.e., with increasing scale, nests within sub-colonies, within colonies, within islands, archipelagos, etc.) it is not clear how these aggregations function as groups or independently at different scales (Wakefield et al. 2014). Colonies, defined subjectively during censuses, may not therefore correspond to functional units.

Despite these uncertainties, it is clear that while some threats to seabirds are widespread (e.g., climate change) others, such as offshore windfarms, episodic pollution incidents, fisheries bycatch, and the depletion of fish stocks, may be localized, impacting colonies within wider metapopulations unequally (Furness and Tasker 2000, Inchausti and Weimerskirch 2002, Montevecchi et al. 2012). Hence, colony-level distribution estimates may be required in order to target and monitor conservation measures, such as Marine Protected Areas (MPAs) or fisheries closures, effectively (Lascelles et al. 2012, Russell et al. 2013).

Current barriers to estimating colony-level distributions via individual tracking are both logistical and analytical: for most species, it would be impractical to track birds from all colonies. In theory, distribution could be predicted from tracked birds from a sample of colonies by modelling space use as a function of habitat, foraging costs, competition, etc. (Aarts et al. 2008, Wakefield et al. 2009, 2011, Catry et al. 2013). However, statistical techniques for producing unbiased estimates of distribution using tracking data are still in development (Aarts et al. 2008, Patterson et al. 2008, Illian et al. 2012). This is partly because tracking data violate many of the assumptions inherent to conventional parametric models (reviewed by Turchin 1998, Aarts et al. 2008, Wakefield et al. 2009). Repeat observations on individuals (typically 10^2–10^4 locations/individual in seabird studies) tend to be spatiotemporally autocorrelated and the movements of individuals drawn from the same colony may be dependent on one another due to public information transfer and cultural and genetic divergence (Wakefield et al. 2013, Paredes et al. 2015). Furthermore, tracking data record the presence of animals but not their absence (Aarts et al. 2012). In order to account for these attributes, habitat use by tracked animals has been modelled using logistic mixed-effects models (Aarts et al. 2008, Wakefield et al. 2011). This entails the construction of a binary response variable, which comprises animal locations and randomly generated pseudo-absence points. However, the logistic model approximates an inhomogeneous Poisson point process (IPP) model (Cressie 1993, Aarts et al. 2012), which may be fitted more directly and efficiently by using numerical quadrature to approximate the model's pseudo-likelihood (Berman and Turner 1992, Baddeley and Turner 2000, Warton and Shepherd 2010; (see *Methods* for details). This approach may therefore be more tractable for GPS tracking data sets, which typically comprise 10^3–10^4 locations per individual. A further substantial problem is that habitat selection may vary between colonies due to differences in the relative availability of prey and habitats among those colonies (e.g., Chivers et al. 2012, Paredes et al. 2012), a phenomenon termed functional response in resource selection (Mysterud and Ims 1998). As such, habitat selection models fitted to data from one site may predict poorly for others (Torres et al. 2015). Matthiopoulos et al. (2011) show that Generalized Functional Response (GFR) models can interpolate usage to unsampled sites more accurately than conventional habitat selection models. GFR models require that usage is sampled under a range of availability regimes allowing the response to environmental covariates to be conditioned on the expected site-level availability of all environmental covariates in the model.

Britain and Ireland are home to internationally important populations of breeding seabirds (Fig. 1). These include 34% of the world's European Shags *Phalacrocorax aristotelis* (26,600 pairs), 20% of its Razorbills *Alca torda* (93,600 pairs), 13% of its Common Murres *Uria aalge* (708,200 pairs), and 8% of its Black-legged Kittiwakes *Rissa tridactyla* (378,800 pairs) (Mitchell et al. 2004). Our study focuses on these species, referred to hereafter as Shags, Razorbills, Murres, and Kittiwakes. Although the foraging niches of these species partially overlap, they are differentiated along several axes. In Britain and Ireland, all are almost exclusively neritic while breeding, feeding primarily on sandeels (*Ammodytes* spp.) and other small fish and crustaceans (Grémillet et al. 1998, Watanuki et al. 2008, Thaxter et al. 2010). Shags forage either benthically or pelagically (maximum dive depth ~60 m) in coastal waters, relatively close (≤~30 km) to their colonies (Grémillet et al. 1998, Watanuki et al. 2008, Bogdanova et al. 2014). Kittiwakes, Murres, and Razorbills are more wide ranging, foraging tens to hundreds of kilometers from their colonies. Kittiwakes are surface feeders; Murres make relatively long, deep, foraging dives to the pelagic and demersal zones; while Razorbills make more frequent, shallow, dives to the pelagic zone (Thaxter et al. 2010, Linnebjerg et al. 2013). There is some evidence that Kittiwakes from adjacent colonies segregate in space while foraging (Ainley et al. 2003, Paredes et al. 2012) but nothing is known about this phenomenon in the other species.

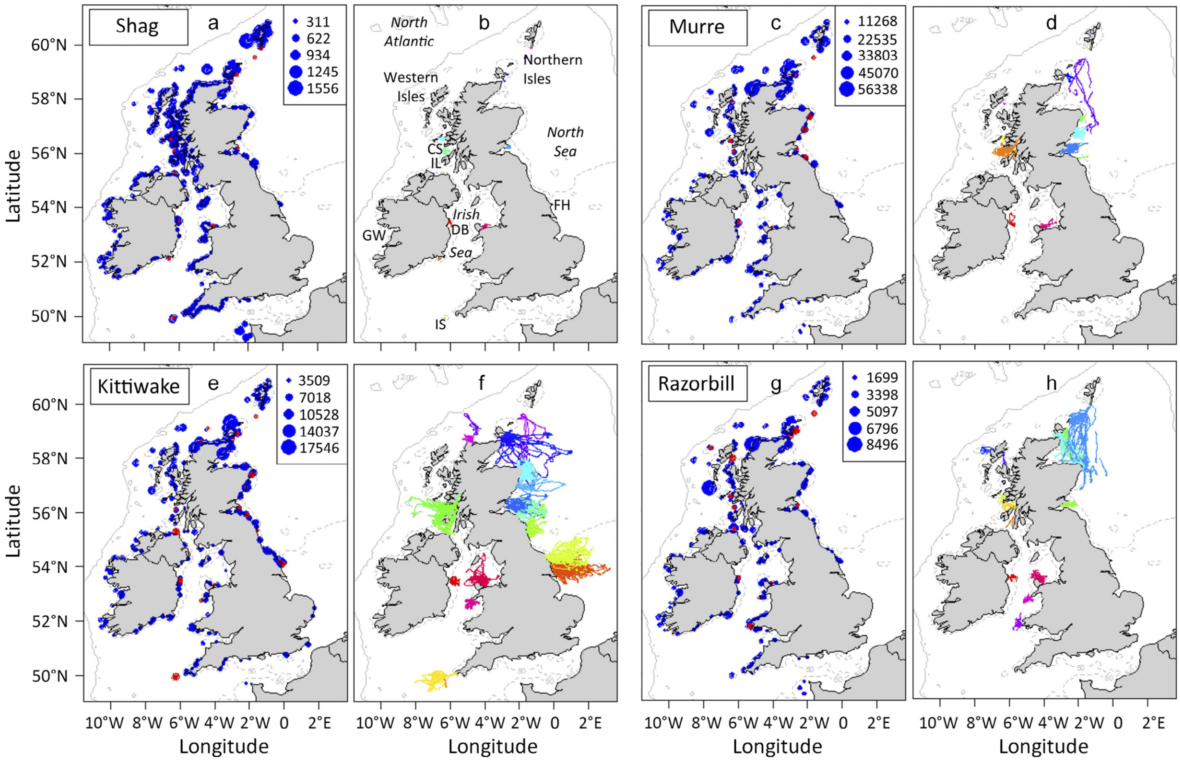

FIG. 1. Breeding distribution and individual movement data used to estimate the distribution at sea of seabirds foraging from UK colonies. Panels a, c, e, and g show numbers of apparently occupied nests recorded during the Seabird 2000 census (Mitchell et al. 2004; red indicates study colonies). Panels b, d, f, and h show tracks of individual birds (colors correspond to colonies). Places mentioned in the text are shown in the upper right panel: CS, Colonsay; DB, Dublin Bay; FH, Flamborough Head; GW, Galway Bay; IL, Islay; IS, Isles of Scilly.

In a recent assessment of conservation status in the UK, Shags and Kittiwakes were reclassified from amber to red due to 62% and 71% declines, respectively, over 25 years (Eaton et al. 2015). In the UK, Murres and Razorbills are amber listed due to their restricted range and international importance (Eaton et al. 2015), while internationally Razorbills have recently been reclassified from Least Concern to Globally Near-threatened (BirdLife International 2015). Current declines are thought to be due in part to falls in prey stocks (especially sandeels in the northern North Sea), due to over fishing and climate change (Frederiksen et al. 2007, Cook et al. 2014). Kittiwakes are also regarded to be particularly vulnerable to wind farm developments, which are burgeoning in UK waters (Furness et al. 2013). The diving species face ongoing threats from oil spills (Williams et al. 1995, Votier et al. 2005) and gill nets (Žydelis et al. 2013). Domestic and international legislation and agreements require countries to manage and conserve seabirds (Croxall et al. 2012). Two measures adopted by governments in UK and elsewhere in the European Union that contribute to seabird conservation are the extension of existing colony-based Special Protection Areas (SPAs) for seabirds to adjacent waters that are used for "maintenance activities" (e.g., foraging, courtship, etc.) and secondly, the establishment of marine SPAs around important foraging areas (Garthe et al. 2012, Perrow et al. 2015). However, both marine protected area identification and wider spatial planning at sea are being hampered by a lack of colony-specific distribution estimates (Perrow et al. 2015). In the absence of such information, policy-makers frequently make the unrealistic assumption that seabirds are uniformly distributed out to some threshold distance from their colonies, such as their putative maximum foraging range (Thaxter et al. 2012).

The main aim of our study is to estimate the coarse scale (1–10 km) metapopulation and colony-level utilization distributions of four species of seabirds breeding in Britain and Ireland during the late incubation and early chick-rearing periods. To do so, we tracked birds from a sample of colonies drawn from throughout the geographical, environmental, and colony size range of our study species in Britain and Ireland and modelled their distributions as functions of colony distance, sympatric and parapatric intraspecific competition, coastal morphology, and habitat availability. In so doing, we estimate the distribution of birds from >5,500 breeding sites. Further, we specifically explored the marine distributions of birds from all colonies designated as SPAs, in order to establish the extent, and intensity of usage, of the marine areas required by individuals from these protected breeding locations.

Materials and Methods

Tracking data collection

We carried out fieldwork at seabird colonies around the coast of Britain and Ireland during May-July 2010–2014, when the study species were either approaching the end of the incubation period or raising small chicks. We stratified sampling effort to reflect the northwards bias in the breeding distribution of seabirds in the region (Mitchell et al. 2004). We caught birds while they attended their nests, either by hand or using a wire noose or crook fitted to a pole, and temporarily attached a modified i-GotU GT-120 (Mobile Action Technology, Taipei, Taiwan) GPS logger to their backs (or rarely, in the case of Kittiwakes, to their tails) with Tesa tape (Tesa SE, Norderstedt, Germany). Total instrument mass was ≤3% body mass for all species, except Kittiwakes, for which it was ≤5% body mass and ≤3% if tail attachment was used. We programmed loggers to record one position every 100 s. Handling time during capture/recapture was <6 min. GPS deployments were carried out following the ethical guidelines of the British Trust for Ornithology, under license by Scottish Natural Heritage, Natural England, Natural Resources Wales, the Northern Ireland Environment Agency and the National Parks and Wildlife Service, Ireland.

Data preparation

Diving by tagged seabirds can result in short hiatuses in tracking data. To estimate missing locations, and to standardize sampling effort to exactly 100-s intervals, we resampled GPS tracks data by linear interpolation prior to further analysis. Due to the need to deploy and retrieve loggers at the nest, it is normal practice in tracking studies of breeding seabirds to record and analyze bursts of data from one or more complete foraging trip per individual. However, this usually results in individuals being observed for unequal amounts of time because trip duration typically varies widely among individual seabirds. To reduce this bias we subsampled tracking data by randomly selecting a 24-h burst of locations

Table 1. Summary of tracking data obtained during the study (see Appendix S2: Table S1 for full details).

Species	No. sites	No. birds tracked	No. birds tracked ≥24 h	Median tracking duration (h)	Median trip length (h)
Shag	13	239	230	75 (55–94)	1.7 (1.0–2.6)
Kittiwake	20	583	464	42 (25–51)	4.0 (1.6–8.7)
Murre	12	192	178	54 (45–74)	7.5 (2.0–13.1)
Razorbill	14	299	281	70 (50–86)	6.3 (1.8–12.6)

Note: Numbers in parentheses are Interquartile Range IQR.

from each bird (Table 1). We omitted the small number of individuals that were tracked for <24 h from our analysis. We then selected locations recorded when birds were at sea, categorized according to distance and time from the nest (see Appendix S1 for details). Prior to analysis, we projected all spatial data in Lambert Azimuthal equal area (LAEA) coordinates.

Modelling approach

We modeled habitat use as a function of habitat availability, accessibility and proxies of intraspecific competition. In view of the size of the data set (55,000–210,000 locations per species), we fitted IPP models by numerical quadrature (Berman and Turner 1992, Baddeley and Turner 2000, Warton and Shepherd 2010) rather than approximating them using logistic regression (Aarts et al. 2012). Following Warton and Shepherd (2010), we modeled the intensity of tracking locations $\lambda(y_i)$ at the point i in space as a function of n explanatory variables:

$$\log(\lambda_i) = \beta_0 + \sum_{j=1}^{n} c_{i,j}\beta_j \tag{1}$$

where c is a vector of covariates and $\beta = (\beta_0, \beta_1, \ldots, \beta_n)$ the corresponding parameters. The pseudo likelihood of IPP models can be estimated by numerical quadrature (Berman and Turner 1992) as

$$l_{\text{IPP}}(\beta; y, y_0, \mathrm{w}) \approx \sum_{i=1}^{m} w_i(s_i \log \lambda_i - \lambda_i) \tag{2}$$

where $y_0 = \{y_{n+1}, \ldots, y_m\}$ are quadrature points (i.e., both data and dummy points),

$\mathbf{w} = (w_1, \ldots, w_m)$ is a vector of weights,

$$s_i = z_i/w_i \text{ and } z_i = \begin{cases} 1 & \text{if } y_i \text{ is a data point} \\ 0 & \text{if} y_i \text{is a dummy point} \end{cases}$$

The right-hand side of Eq. 2 is equivalent to the likelihood of a weighted log-linear Poisson model, which can readily be estimated using conventional GLM software (Baddeley and Turner 2000). We assigned the centroids of the cells of a regular LAEA grid as dummy points, a quadrature scheme that ensures even distribution across the study area (Warton and Shepherd 2010). We then assigned weights $w_i = a/n_i$ to each quadrature point, where n_i is the number of points (data or dummy) in the same cell as the ith point and a is the area of that cell (Baddeley and Turner 2000). Note that dummy points are not equivalent to the "pseudo-absence" points used in some case-control models fitted to tracking data (see Aarts et al. [2012]).

In order to account for the highest level of grouping in the tracking data (i.e., breeding colony) we structured models as mixed-effects GLMs

$$\begin{aligned} \lambda_{k,i} &\sim \text{Poisson}(\mu_{k,i}) \Rightarrow E(\lambda_{k,i}) \sim \mu_{k,i} \\ \log(\mu_{k,i}) &= \text{offset}(n_k) + \beta_0 + \sum_{j=1}^{m} x_{i,j}\beta_j + u_k \end{aligned} \tag{3}$$

where $\lambda_{k,i}$ is the intensity of locations of birds from the kth colony and u_k is a random, colony-level, intercept. The offset term is included to standardize model predictions because the number of birds tracked n_k varied across colonies. Each bird was tracked for a period of 24 h so the response $\mu_{k,i}$ is the expected number of tracking locations at sea per bird per day per unit area from the kth colony. Normalized to sum to unity over all grid cells this approximates the colony-level utilization distribution UD_k. The inclusion of the colony-level random intercept necessitated a separate set of dummy points for each colony: for the kth colony, we therefore generated dummy points and weights within the sea area accessible from each colony, which we define as that lying $<d_{\max}$ from that colony, where $d_{\max}$ is 1.1× the maximum foraging range observed across colonies in our study (Shags 35 km, Kittiwakes 300 km, Murres 340 km, Razorbills 305 km). In the absence of theoretical estimates of the maximum foraging ranges for our study species, we used the maximum observed foraging range. We apply the multiplier 1.1 to ensure that the quadrature grid encompasses the areas bounded by the putative maximum foraging range. Models were fitted using the R package lme4 (Bates et al. 2015).

Warton and Shepherd (2010) show that the accuracy of the quadrature approximation method described above increases as the ratio of dummy points to data increases. During model development we investigated this effect by fitting single covariate models to data sets generated using quadrature grids of varying resolutions. We found that, within the computationally practicable range of scales, parameter estimates did not converge with increasingly finer scale (Appendix S1: Fig. S1). Hence, following Warton and Shepherd (2010), we conducted our analysis at the finest resolution practicable. This was 0.5 km for Shags (55,436 tracking locations; 150,557 dummy points) and 2 km for the other species (range 82,741–206,413 tracking locations; 417,578–806,384 dummy points.

Model selection

Eq. 3 assumes independence among data (Baddeley and Turner 2000) yet animal tracking locations are repeated measures on individuals and tend to be serially and spatially autocorrelated (Aarts et al. 2008). Hence, the standard errors provided should be treated as relative rather than absolute. The full likelihood of Eq. 3 is unknown, precluding the provision of P values or model selection using conventional information criteria. Rather, we used k-folds cross-validation to compare the out-of-sample predictive performance of models based on the similarity between the observed and predicted utilization distributions (Fewster and Buckland 2001). To do so, we calculated the observed UD of tracked birds from the *k*th colony (i.e., the proportion of all locations of birds tracked from that colony falling in each cell in the regular grids mentioned above). We then fitted the model under consideration to data from the remaining colonies, predicted the UD of the *k*th colony from this model and calculated the Bhattacharyya affinity between the observed and predicted UDs

$$\mathrm{BA}_k = \sum_{x,y} \sqrt{\mathrm{UD}_{\mathrm{obs},k}(x,y)\mathrm{UD}_{\mathrm{pred},k}(x,y)} \tag{4}$$

BA has previously been used in the contexts of UD comparison and model selection (Thacker et al. 1997, Fieberg and Kochanny 2005). It ranges from 0 (no similarity) to 1 (identical UDs). We calculated the weighted mean similarity across colonies

$$\overline{\mathrm{BA}} = \frac{\sum_{\mathrm{All}k} n_k \mathrm{BA}_k}{\sum_{\mathrm{All}k} n_k} \tag{5}$$

where n_k is the number of birds tracked from the *k*th colony. The contribution to $\overline{\mathrm{BA}}$ of colonies from which larger numbers of birds were tracked is upweighted because the UDs of colonies with small samples of tracked birds are likely to underestimate the area used by the entire colony (Soanes et al. 2013, Bogdanova et al. 2014).

In order to estimate space use from all colonies in the study area, we aimed to select the best model from a field of biologically plausible alternatives. Previous studies suggest that seabird space use may depend on numerous covariates, including colony distance, density-dependent competition, and habitat (Wakefield et al. 2009, 2011). The number of plausible alternative models is therefore large. This, combined with the time taken for models to converge, precluded backward model selection. Rather, we built usage models using a stepwise forward selection procedure, adding candidate explanatory covariates to the intercept-only model in order of their expected effects sizes. We retained covariates if $\Delta\ \overline{\mathrm{BA}}$ was positive, selecting the most parsimonious model if $\Delta\ \overline{\mathrm{BA}}$ was tied. In order to compare effect sizes using standardized partial regression coefficients we standardized covariates prior to analysis (Schielzeth 2010).

Candidate explanatory covariates

In the absence of other factors, central-place foraging theory suggests that breeding seabirds should seek prey as close to their nest sites as possible (Orians and Pearson 1979). First, therefore, we added distance to colony *d* to the model, with the expectation that usage would decline with distance (Dukas and Edelstein-Keshet 1998, Matthiopoulos 2003). Our study species generally avoid crossing extensive land barriers when commuting (Fig. 1) so we defined $d_{k,i}$ to be the minimum distance by sea between the *k*th breeding site and the *i*th location, which we calculated on a 0.5 km (Shags) or 1 km (Kittiwakes, Murres, and Razorbills) LAEA grid using the R package gdistance (van Etten 2012, Wakefield et al. 2013). Space use by breeding seabirds is further modulated by density-dependent competition among sympatrically breeding conspecifics (Wakefield et al. 2013, Jovani et al. 2015). Given that competition is proportional to the density of animals we next considered whether the area of sea available to birds from each breeding site, which varies with coastal geometry, affects usage (Birkhead and Furness 1985). We hypothesize that density-dependent competition would be higher among birds foraging from colonies with restricted access to the open sea such that they would forage further from their colonies than birds from colonies surrounded by open water. To model this effect, we considered the addition of $A_{k,i}$ the cumulative area at the *i*th location relative to the *k*th breeding site, to our models, where

$$A_{k,i} = \sum_{\mathrm{All}x} a_x \delta_{k,x},\ \delta_{k,x} = \begin{cases} 1 & \text{if } d_{k,x} \le d_{k,i} \\ 0 & \text{otherwise} \end{cases} \tag{6}$$

and a_x is the area of the *x*th cell of the LAEA grids mentioned above. Exploratory analysis indicated that log-transforming A_k reduced colinearity with d_k, improving model stability.

We next considered the number of sympatric breeders, the other determinant of density at sea. We extracted numbers of apparently occupied nests (AON) recorded during the most recent complete census of seabird colonies in Britain and Ireland (Seabird 2000, carried out between 1998 and 2002; see Mitchell et al. [2004] for methods) from the Seabird Monitoring Programme (SMP) Database.[13] Defining seabird colonies objectively can be problematic because the degree to which breeding seabird nests are clustered in space varies with scale (Wakefield et al. 2014). During the Seabird 2000 census, AON were recorded by “subsite” (for clarity, simply referred to as “sites” hereafter). These Mitchell et al. (2004) nominally defined as segments of coastline <1 km long, containing clusters of breeding seabirds. However, for practical reasons fieldworkers were allowed some scope to deviate from this definition. In practice, sites sometimes therefore comprise isolated islands or segments of coastline >1 km long. In the latter cases, we reassigned sites by splitting the coastline into the minimum possible number of segments ≤1 km long, dividing AON equally between each. During model selection we considered several potential proxies of competition from sympatric breeders. First, the number, *N*, of conspecific AONs at the home site. Second, because arbitrary census divisions may not correspond to ecologically functional units (Wakefield et al. 2014) we considered proxies that include conspecifics breeding in the vicinity of the home

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[13] www.jncc.gov.uk/smp

site of tracked birds. These were the inverse-distance weighted number of breeding conspecifics

$$\theta = \sum_{\mathrm{All}k} \frac{\mathrm{N}_k}{d_{h,k} + 1} \tag{7}$$

where N_k is the number of conspecific AON at the kth site of the set of all breeding sites (including the home site) within the species' maximum foraging range, and $d_{h,k}$ is the distance from the home breeding site to the ith breeding site. Finally, based on exploratory analyses, we also considered the square-roots of these indices, as well as Eq. 3 the inverse-distance weighted square-root number of conspecific breeders

$$\theta' = \sum_{\mathrm{All}k} \frac{\sqrt{\mathrm{N}_k}}{d_{k,i} + 1} \tag{8}$$

We considered each of the indices of sympatric competition as a main effect and interaction with A, selecting that which resulted in the best improvement in model performance (step 3).

In addition to sympatric competition, breeding seabirds may be subject to competition from conspecifics breeding at neighboring colonies (Furness and Birkhead 1984; hereafter, parapatric competition [Wakefield et al. 2011]). As with sympatric competition, this is thought to be density dependent (Wakefield et al. 2013). Our expectation is therefore that birds avoid locations at which the null density of conspecifics from other colonies is high (Wakefield et al. 2011, 2013, Catry et al. 2013). In some systems, this leads to striking patterns of among-colony spatial segregation (Masello et al. 2010, Wakefield et al. 2013). It has been hypothesized that these are mediated by individual-level information transfer and cultural divergence during colony growth (Wakefield et al. 2013). Current uncertainly about these mechanisms makes this phenomenon difficult to model satisfactorily but as a first approximation we considered whether birds avoided areas in which the null density of conspecifics from other colonies was greater than that from their own (Catry et al. 2013). Taking the best models from previous steps (hereafter models I–IV for Shags, Kittiwakes, Murres, and Razorbills, respectively), we predicted $\rho_{h,i}$, the ratio of the expected intensity of locations $\lambda_{h,i}$ from the focal breeding site h to the sum of those from all other sites in the region

$$\rho_{h,i} = \frac{\lambda_{h,i}\mathrm{N}_h}{\sum\limits_{k \neq h} \lambda_{k,i}\mathrm{N}_k} \tag{9}$$

We then determined whether adding this covariate to the usage models improved their performance.

We next considered whether the addition of environmental indices describing habitat improved model performance (Wakefield et al. 2009). We identified candidate biophysical covariates meeting two criteria: first, that data coverage was sufficient to allow seabird distributions to be estimated throughout the waters of Britain and Ireland, and second, that previous studies had established links between the covariate (or the phenomenon it quantifies) and the foraging behavior or distribution of the study species or their prey. As noted above, each model level requires a separate set of quadrature points. Hence, although we considered both static and dynamic covariates, we averaged monthly mean dynamic covariates over the study period (May–July 2010–2014; Appendix S1: Fig. S2) to maintain the number of data, and thereby computing time, within tractable limits. We then determined the value of environmental covariates at each quadrature point. We considered (1) depth (ETOPO2 Global Relief 2v2, provided by the U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Geophysical Data Center, 2006); (2) seabed slope, calculated from the latter in ArcGIS 10 (ArcGIS10 manufactured by ESRI based in Redlands, California, USA); (3) minimum distance to the coast, calculated in ArcGIS 10; (4) the proportion of gravel; and (5) the ratio of sand to mud in seabed sediments, derived from British Geological Survey 1:250,000 maps (*available online*; see Appendix S1);[14] (6) the potential energy anomaly (PEA), which quantifies the intensity of thermohaline stratification; and (7) the proportion of time during which the water column was stratified, both estimated using UK Met Office FOAM AMM reanalysis data (*available online*, see Appendix S1);[15] (8) AVHRR sea surface temperature (SST), supplied by the Natural Environment Research Council Earth Observation Data Acquisition and Analysis Service (NEODAAS); (9) standardized sea surface temperature (sSST), calculated on a monthly basis by subtracting the mean SST in the study area and dividing by its standard deviation, which is an alternative index of stratification (Wakefield et al. 2015); (10) thermal front gradient density (TFGD), estimated following (Scales et al. 2014) using AVHRR SST to provide an index of the mean intensity and persistence of thermal fronts (Miller and Christodoulou 2014); and (11) net primary production (NPP) estimated and supplied by NEODAAS using MODIS chlorophyll and photosynthetically available radiation data. For further details of candidate covariates and our rationale for their consideration, see Appendix S1: Table S2 and reviews by Hunt (1997), Mann and Lazier (2006), and Wakefield et al. (2009). In brief, the phenomena described by these covariates may affect our study species' distributions either by modulating lower trophic level production (depth, seabed slope, indices of stratification, SST, TFGD, NPP [Begg and Reid 1997, Mann and Lazier 2006, Scott et al. 2010, Carroll et al. 2015]); by physically aggregating prey (indices of stratification, TFGD, and indirectly SST and depth; Lefevre 1986, Begg and Reid 1997, Mann and Lazier 2006, Embling et al. 2012); or

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[14] http://digimap.edina.ac.uk

[15] http://marine.copernicus.eu/

due to the habitat preference of prey species, especially Ammodytidae and Clupeidae (depth, coast distance, sediment, indices of stratification [Whitehead 1986, Holland et al. 2005, van der Kooij et al. 2008]).

In order to establish in what order to add environmental covariates to models, we first determined the improvement in performance afforded by adding each singly to the best model resulting from the previous steps. Based on previous work (Wakefield et al. 2011, 2015) and exploratory analyses we considered log and square-root transformations of some covariates (Appendix S1: Table S2). In order to model potential variation in habitat selection in response to among-colony variability in habitat availability, we also considered interactions between each covariate and its expected value at each colony. This we define as the covariate's mean (hereafter denoted by an overbar) in waters accessible from that colony (i.e., the sea area within d_{max}). This partially implements the GFR model proposed by Matthiopoulos et al. (2011). The full GFR model, in which variables interact not only with their own colony-level expectations but those of all other environmental covariates, proved computationally unfeasible with our data set (see Appendix S1). We ranked environmental covariates in order of $\Delta\ \overline{\mathrm{BA}}$ afforded by the addition of each covariate (transformed or otherwise) and its GFR equivalent. We then added these terms sequentially to the model, retaining them if $\Delta\ \overline{\mathrm{BA}}$ was positive (step 5). If two covariates were considered proxies of the same phenomenon (e.g., stratification) or were otherwise colinear, we considered only that ranked highest. Finally, because relationships between space use and environmental covariates may be nonlinear, we also considered their second degree polynomials, retaining them if their addition resulted in an increase in $\Delta\ \overline{\mathrm{BA}}$ (step 6).

Estimating usage

For each species, we estimated λ and thereby the UD for birds from each Seabird 2000 site s using the fixed-effects part of the best models (hereafter models V–VIII for Shags, Kittiwakes, Murres, and Razorbills, respectively). We then calculated the population-level UD across the study area

$$\mathrm{UD}_{P,i} = \sum_{\mathrm{All}x} \mathrm{UD}_{s,x}\mathrm{N}_s \qquad (10)$$

where N_s is the number of AON at the sth site. Notwithstanding the comments on standard errors above, we quantified spatial variation in the relative uncertainty of our model estimates by plotting the coefficient of variation (CV) of UD_P, which we calculated using parametric resampling adapted from Bolker (2008) and Lande et al. (2003). Assuming that the sampling distribution of $\hat{\beta}$ is multivariate normal, we generated 100 random sets of fixed-effects parameters for each model, predicted the UD_P using each set of parameters, and then calculated its CV.

In order to illustrate how one might use these UDs to identify marine areas whose statutory protection would facilitate the functional protection of the existing suite of colony SPAs, following Eq. 10, we also calculated the mean UD of birds breeding at sites within each UK SPA. We identified breeding sites falling within existing colony SPAs using boundaries downloaded from the Joint Nature Conservation Committee (*available online*).[16] For each SPA, we then determined polygons encompassing the core 50%, 75%, and 90% of estimated usage as well as the maximum curvature boundaries (MCBs, see Appendix S1). While MCBs have no ecological basis (Perrow et al. 2015), it has been suggested that they balance the proportion of a population protected against the extent of the protected area and have been used by statutory bodies to define boundaries for delimiting avian marine protected areas in the UK (O'Brien et al. 2012). We then overlaid percentage UDs and MCBs of all species in order to estimate the overall extent of sea area thus encompassed.

RESULTS

Seabird tracking

We tracked 1,313 birds from 29 colonies for a median of 2–3 d/bird (Fig. 1, Table 1; Appendix S2: Table S1). Following resampling to standardize the observation period to 24 h/bird, data from 80% of Kittiwakes and 93–96% of the remaining species were retained for analysis, totaling 1,153 individuals. Full data are available for download from the BirdLife Seabird Tracking Database (*available online*).[17] The duration of deployment was set by the need to recapture birds before tags became detached from feathers. Recapture was attempted after 24 h (Kittiwake, where the mantle feathers are relatively weak) to 48 h (other species). Median foraging trip length was <24 h in all species (Table 1) so the 24-h observation window generally spanned >1 trip/individual. Differences in foraging ranges were marked among species (Fig. 1): Shags remained relatively close to their nest sites (median 3.4 km, IQR 1.6–7.5), whereas Kittiwakes (11.9 km, IQR 4.2–30.9), Murres (10.5 km, IQR 3.2–19.1), and Razorbills (13.2 km, IQR 5.1–26.2) travelled further from their colonies during foraging trips.

Explanatory covariates

The addition of distance to colony d improved the performance of usage models for all species (Appendix S2: Tables S2 and S3) and its effect, which was negative, was relatively large (Table 2). The addition of A (the cumulative area at d), interacting with indices of sympatric

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[16] http://jncc.defra.gov.uk/ProtectedSites/

[17] http://seabirdtracking.org/mapper/contributor.php?contributor_id=950

TABLE 2. Summary of fixed effects in inhomogeneous Poisson point process models of the density of seabird tracking locations as functions of colony distance, coastal geometry, intra-specific competition, and habitat.

Model and covariate†	Estimate	SE‡	z
V. Shag			
Intercept	−6.092	0.240	−25.43
d	−1.254	0.018	−71.52
$\log(A)$	−1.239	0.010	−128.17
θ'	0.353	0.250	1.41
Gravel	0.512	0.012	41.92
$\overline{\text{gravel}}$	−0.112	0.355	−0.32
$\sqrt{\text{PEA}}$	−1.613	0.028	−58.64
NPP	0.048	0.011	4.22
Coast distance	−1.187	0.034	−35.28
SST	0.797	0.046	17.37
SST^2	0.474	0.026	18.14
$\log(A) \times \theta'$	0.110	0.005	23.02
$\text{gravel} \times \overline{\text{gravel}}$	−0.627	0.020	−30.78
VI. Kittiwake			
Intercept	−6.375	0.175	−36.39
d	−1.338	0.010	−140.65
$\log(A)$	−0.486	0.005	−91.12
θ	−0.388	0.189	−2.06
$\log(\rho)$	1.669	0.014	118.75
log(seabed slope)	−0.019	0.005	−4.15
$\overline{\text{log(seabed slope)}}$	−1.381	0.261	−5.29
$(\text{log(seabed slope)})^2$	−0.161	0.003	−57.53
sSST	−1.006	0.007	−143.32
stratification	0.033	0.004	9.21
$\overline{\text{stratification}}$	0.969	0.308	3.15
$\log(A) \times \theta$	0.167	0.004	46.31
$\text{log(seabed slope)} \times \overline{\text{log(seabed slope)}}$	0.979	0.009	104.99
$\text{stratification} \times \overline{\text{stratification}}$	0.942	0.011	87.33
VII. Murre			
Intercept	−7.294	0.177	−41.19
d	−1.627	0.028	−57.56
$\log(A)$	−0.862	0.007	−124.54
$\sqrt{\theta}$	0.206	0.171	1.21
$\log(\rho)$	0.929	0.029	32.07
Gravel	−0.223	0.005	−46.71
$\sqrt{\text{sand:mud}}$	−0.184	0.011	−16.42
$(\sqrt{\text{sand:mud}})^2$	−0.196	0.010	−18.80
$\overline{\sqrt{\text{sand:mud}}}$	−2.037	0.543	−3.75
TFGD	0.331	0.004	77.63
Coast distance	−1.709	0.032	−53.81
$\overline{\text{coast distance}}$	3.098	0.370	8.38
$\log(A) \times \sqrt{\theta}$	0.273	0.005	54.23
$\sqrt{\text{sand:mud}} \times \overline{\sqrt{\text{sand:mud}}}$	−0.481	0.034	−14.16
$\text{coast distance} \times \overline{\text{coast distance}}$	1.760	0.057	30.68
VIII. Razorbill			
Intercept	−4.623	0.105	−43.84
d	−1.066	0.009	−119.85
$\log(A)$	−1.106	0.004	−255.08
$\sqrt{\text{N}}$	0.552	0.106	5.23
SST	−0.083	0.008	−10.60
$\overline{\text{SST}}$	0.336	0.130	2.58
sand : mud	−0.290	0.006	−47.53
$(\text{sand : mud})^2$	−0.266	0.005	−53.46
log(seabed slope)	0.027	0.005	5.30
$\overline{\text{log(seabed slope)}}$	−0.306	0.221	−1.38

TABLE 2. (Continued)

Model and covariate†	Estimate	SE‡	z
$\log(A) \times \sqrt{\mathrm{N}}$	0.331	0.003	123.37
$\mathrm{SST} \times \overline{\mathrm{SST}}$	−0.882	0.010	−90.40
$\log(\text{seabed slope}) \times \overline{\log(\text{seabed slope})}$	−0.525	0.015	−34.22

Notes: Numbers in parentheses after model name are the numbers of sites and birds).

†Covariates standardized prior to model fitting; *d*, distance by sea from the colony; *A*, cumulative area at distance *d*; θ, inverse-distance-weighted number of conspecific breeders; θ′, inverse-distance-weighted square-root number of conspecific breeders; *N*, number of conspecific breeders at the home site; ρ, density of birds from the home site relative to those from all other sites; NPP, net primary production; PEA, mean potential energy anomaly; SST, mean sea surface temperature; sSST, mean standardized SST; TFGD, thermal front gradient density. Overbars indicate the mean of the covariate in water accessible from each colony.

‡Relative standard errors.

competition, further improved model performance (Appendix S2: Table S3). In the case of Razorbills, the square-root of the number of breeding pairs in the home site was the best index of sympatric competition. For the other species, competition indices based on the summation of numbers of breeders inversely weighted by distance from the focal breeding site best improved model performance (Appendix S2: Table S2). Models I–IV suggest that, in all species, the rate of decline in usage with *A* lessens with increasing sympatric competition (Table 2; Appendix S2: Tables S2 and S3, Fig. S1). The inclusion of relative parapatric competition improved the performance of Kittiwake and Murre usage models but not those of Shags and Razorbills (Appendix S2: Table S2). The former species tended to avoid areas in which the potential density of conspecifics from other colonies was higher than that from their own colony. The addition of environmental covariates improved the performance of all models (cf. Fig. 2; Appendix S2: Fig. S3) and conditioning some but not all covariates on their regional means improved performance further (Appendix S2: Table S4). Cross-validation shows that the final models for Shags, Kittiwakes, and Murres all performed similarly well ($\overline{\mathrm{BA}} \pm \mathrm{SD} = 0.52 \pm 0.13$, 0.53 ± 0.13, and 0.53 ± 0.22, respectively) but the performance of the Razorbill model was somewhat poorer ($\overline{\mathrm{BA}} \pm \mathrm{SD} = 0.34 \pm 0.11$). Spatial plots confirm our expectation that the similarity between observed and predicted utilization distributions was greatest for colonies from where more birds were tracked (Appendix S2: Fig. S3).

The effects of many environmental covariates were comparable in magnitude to those of colony distance, cumulative area, and competition (Table 2). Taking the environmental covariates retained during model selection in order of their effect sizes, these suggest that Shags tend to use relatively mixed waters (i.e., low PEA) close to the coast. In areas where gravel is scarce, they use relatively gravelly substrates but this is reversed in more gravelly areas (Appendix S2: Fig. S5). Shags' usage with respect to SST was quadratic, with a tendency to visit areas where SST was either warmer or cooler than the average (Table 2; Appendix S2: Fig. S5). Shags also manifested a weak preference for areas of high NPP. Usage by Kittiwakes with respect to seabed slope and stratification was complex: in areas where the mean seabed slope was low, they tended to avoid steep bathymetric relief but this preference was reversed somewhat in areas where the mean slope was high (Appendix S2: Fig. S5). Similarly, in areas where the mean occurrence of stratification was low, Kittiwakes avoided stratified

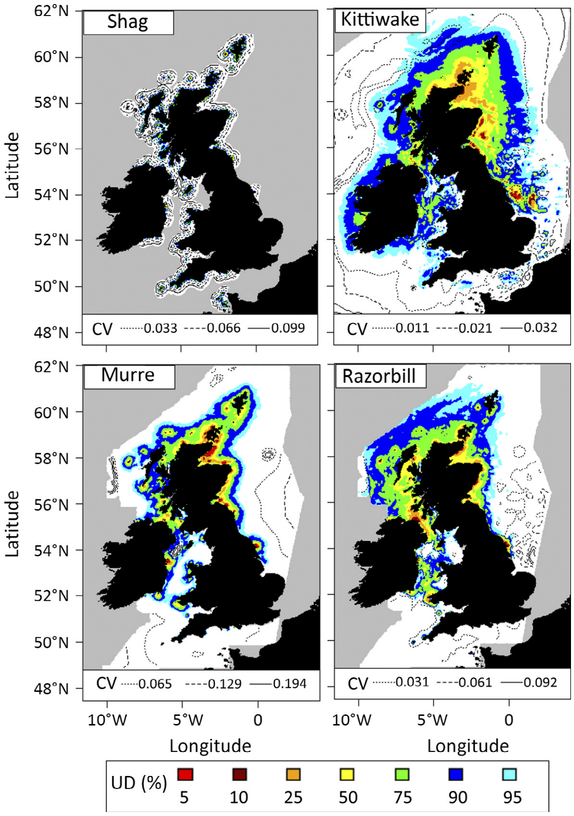

FIG. 2. Percentage at-sea utilization distribution (UD) of seabirds breeding within Britain and Ireland during late incubation/early chick-rearing estimated as functions of colony distance, coast geometry, intra-specific competition, and habitat (models V–VIII). Warmer colors indicate higher usage. Isopleths indicate relative coefficient of variation (CV) of the estimated probability density (gray, no environmental data).

waters, whereas in more frequently stratified areas, they tended to avoid mixed waters.

In areas with low regional mean coastal distance (i.e., archipelagos) Murres used areas close to the coast, whereas in areas with less complex coastlines they tended to forage further from land (Table 2; Appendix S2: Fig. S5). In regions with a relatively high proportion of sand in the substrate, Murres preferred sandy areas but this preference reversed in less sandy regions. Murres also showed a weak preference for frontal regions and substrates containing a relatively low proportion of gravel (Table 2). Razorbills used areas with higher SSTs in regions with relatively cool surface waters, whereas in warmer regions the opposite was true (Appendix S2: Fig. S5). In regions with relatively low seabed relief they tended to select areas with steep relief and vice versa. Razorbills' habitat preference with respect to the sand: mud ratio of the substrate was quadratic, peaking just below intermediate values (Appendix S2: Fig. S5).

Estimated population-level distributions

Raster files of space use during late incubation and early chick-rearing from all of the region's colonies estimated using models V–VIII are available for download from the Data Archive for Marine Species and Habitats (DASSH; *available online*).[18] Composite usage maps predict that breeding Shags, Kittiwakes, Murres, and Razorbills forage mainly within 100 km of the coast of Scotland, primarily to the north and east of the mainland in the North Sea, and around the Northern Isles (Fig. 3a; Appendix S2: Fig. S7). For all species, 90% of the UK regional population's UDs also included waters in the southern North Sea; Dublin Bay and the North Channel of the Irish Sea; as well as waters surrounding Islay; the northern Minch; and isolated islands northwest of Scotland (Appendix S2: Fig. S7). The estimated distributions of Shags, which is the least wide-ranging of the study species, largely reflects that of its colonies (cf. Figs. 1 and 2). In contrast, that of Kittiwakes is more pelagic, with activity more patchily distributed offshore (Fig. 2). In addition to core areas mentioned above, usage hotspots included a large area southeast of Flamborough Head and the northern Norfolk Banks; the central Irish Sea; and Galway Bay, west of Ireland. Of the two Auks, our models suggest that Murres forage closer on average to their colonies (Fig. 2), outnumbering Razorbills in many coastal areas and in the vicinity of the Celtic Sea front. In contrast, Razorbills predominate in the North Channel and much of the Minch (Appendix S2: Fig. S8).

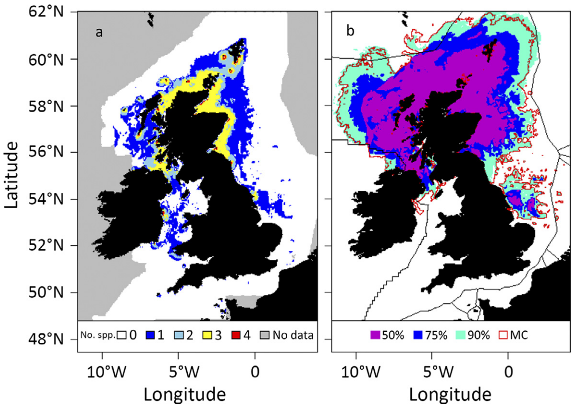

FIG. 3. Predicted multi-species hotspots. (a) Overlap between estimated core areas used by the four study species during late incubation/early chick-rearing. Colors indicate number of overlapping species' core areas (75% of the species' utilization distribution, UD; see Appendix S2: Fig. S7 for equivalent plots using the 50% and 90% UDs). (b) Combined usage by all four study species breeding at Special Protected Area (SPA) colonies. Colors indicate areas supporting 50%, 75%, and 90% of the estimated marine utilization distribution of one or more species breeding within existing colony-based SPAs. Red lines indicate areas contained within maximum curvature (MC) boundaries (O'Brien et al. 2012) for one or more species and black lines boundaries between national Exclusive Economic Zones.

DISCUSSION

Several recent studies have assimilated tracking data from multiple colonies in order to map and understand seabirds distributions (BirdLife International 2004, Block et al. 2011, Wakefield et al. 2011, 2013, Ramos et al. 2013). However, this is the first to model how colony-level distributions vary due to the combined effects of sympatric and parapatric conspecific interactions, coastal geomorphology, and regional habitat availability. By tracking and modelling the space use of Shags, Kittiwakes, Murres, and Razorbills from a sample of colonies around Britain and Ireland, we estimated the coarse-scale (tens of kilometers) distribution of these species at sea from all of colonies in the region. Moreover, by combining these results, we were able to map the at-sea distribution of each species' breeding population across a study area extending over ~1.5 million km^2 (Fig. 2). Until recently, it was only practicable to attempt to estimate the distributions of seabirds over such wide areas at comparable resolutions by surveying birds from boats or planes (Stone et al. 1995, Bradbury et al. 2014). However, these methods generally fail to discriminate among birds from different colonies or life history stages (e.g., breeders vs. non-breeders). Our results therefore provide unprecedented insights into marine distributions of breeding seabirds.

We modelled the occurrence of tracking locations as an inhomogeneous Poisson point (IPP) process (Cressie 1993), which is a computationally efficient and, it has been argued, natural method of treating presence-only data (Warton and Shepherd 2010, Aarts et al. 2012, Renner et al. 2015). We discuss our approach in more detail in Appendices S1 and S3. However, it is pertinent to highlight two caveats to our results. First, due to the

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[18] www.dassh.ac.uk

large volume of data involved in our analysis, only relatively simple models were computationally tractable and therefore not all correlation structures inherent to the data (e.g., serial autocorrelation within individuals; Aarts et al. 2008) were modelled. Hence, although we presume that our parameter and usage estimates are unbiased their associated uncertainty is likely to be underestimated. Second, the likelihood estimation technique we used is approximate (Berman and Turner 1992). We therefore opted to select among models by *k*-folds cross validation, rather than using penalized information criteria, such as AIC. The *k*-folds cross validation technique is robust to over-fitting when the number of data is large, and the field of candidate models relatively small (Arlot 2010). However, our models are optimized for prediction, rather than parsimony, so the biological inferences drawn from them here are tentative.

Distribution with respect to colony distance and competition

Space use by all four study species declined with distance from the colony (Table 2), supporting the hypothesis that central-place foragers minimize distance-dependent travel costs (Orians and Pearson 1979). Our results also support the hypothesis that colonial central-place foragers seek to minimize density-dependent intraspecific competition (Ashmole 1963, Lewis et al. 2001, Wakefield et al. 2013): in all species, the rate of decline in usage with cumulative area at distance decreased as the number of sympatrically breeding conspecifics increased (Appendix S2: Table S3, Fig. S1). Although this echoes the observation that foraging range is positively dependent on colony size in many seabirds (Lewis et al. 2001, Wakefield et al. 2013), it also demonstrates that conspecific density is dependent not only upon numbers of birds but the availability of suitable habitat (most simply, open sea). In short, models V–VIII show that birds foraging from a colony with limited access to the sea (e.g., those located in inlets) travel further on average than those from a colonies of the same size surrounded by open water (i.e., on isolated islands; Appendix S2: Figs. S4 and S5). For the purposes of our analysis, we recognized that colonies as defined in the Seabird 2000 census (Mitchell et al. 2004) might not correspond to functional units. Our results suggest that, in all species except Razorbills, this is indeed the case (Appendix S2: Table S2). For the other three species, we found that sympatric competition was better quantified by the sum of the inverse distance-weighted number of conspecifics breeding in the area. We hypothesize that this is because the intensity of potential competition from any one conspecific declines as a function of distance to its nest.

It has been hypothesized that seabirds foraging from adjacent colonies segregate in space if potential density of competing conspecifics is high (Wakefield et al. 2013). Segregation among the UDs of colonies has been observed in several species (Masello et al. 2010, Wakefield et al. 2013) but evidence for this phenomenon in our system was mixed: in accordance with the density-dependence hypothesis (Wakefield et al. 2013), Kittiwakes and Murres avoided the areas at which the null ratio of the density of birds from the home colony to those from other colonies was low but Shags and Razorbills did not. Among-colony segregation is also evident in Kittiwake populations geographically disparate from the UK (Ainley et al. 2003, Paredes et al. 2012) and may therefore be widespread in this species but this is the first time that the phenomenon has been reported in Murres. Given the close taxonomic and functional affinities between Razorbills and the latter species it is perhaps surprising that terms describing among-colony segregation were not retained during model selection for Razorbills. This may be because a relatively large proportion (48%) of the Razorbills in our study were tracked from the Northern Isles (Fig. 1; Appendix S2: Table S1), where populations of this and other seabird species have been in decline for the past decade (JNCC 2014) due to declines in forage fish availability (Cook et al. 2014). Razorbills from this region travelled much further (median range 62.7 km, IQR 39–87) than those from other areas (median 20 km, IQR 11–28), possibly due to local food shortages. It is hypothesized that patterns of spatial segregation are, in part, culturally perpetuated via information transfer among conspecifics (Wakefield et al. 2013). If so they may become unstable in a declining population. The apparent lack of spatial segregation among Shags from different breeding sites is notable given that this phenomenon occurs in several other members of the Phalacrocoracidae, such as *Phalacrocorax atriceps*, *P. magellanicus*, and *P. georgianus* (Wanless and Harris 1993, Sapoznikow and Quintana 2003). However, in comparison to these species, European Shags breed in relatively dispersed colonies throughout much of their range in Britain and Ireland so density-dependent competition among breeding aggregations may be insufficient to cause segregation of foraging areas. This could be viewed as an extreme form of segregation, where inter-colony spacing generally exceeds the species' maximum foraging range. Additionally, in Britain and Ireland, Shag colonies tend to be small, further reducing inter-colony competition. For example, in the Isles of Scilly, where Shags breed at very low densities, birds from different breeding sites forage in common areas (Evans et al. 2015), as suggested by model V (Appendix S2: Fig. S3). Notwithstanding these comments it is possible that our analysis could not detect among-colony foraging segregation in Razorbills and Shags, for two reasons. First, we were unable to track these species from multiple large and closely adjacent breeding sites, where theory suggests segregation is most likely to occur (Wakefield et al. 2013). Second, the census data we used to estimate intraspecific competition was collected 8–16 years before our tracking campaign. Populations of all species in our study are in a state of flux: over the past 15 years, Shags have declined by ~30% throughout the region, while Razorbill have declined in the Northern Isles (JNCC 2015). Further tracking from pairs of large, closely adjacent and recently censused colonies would be required to conclusively establish the degree to which spatial segregation occurs among colonies of Shags and Razorbills.

In modelling competition, we made the assumption that seabirds avoid areas of high conspecific density. This is consistent with established foraging theories (the ideal free distribution, optimal foraging, etc.) and is supported by empirical evidence at scales of tens of kilometers and above (Ford et al. 2007, Wakefield et al. 2013). However, at finer scales, local enhancement (when individuals searching for prey are attracted to feeding conspecifics) may cause seabirds to cluster (Fauchald 2009). In our modelling framework, this would manifest as unexplained spatial autocorrelation. Similarly, memory-based foraging or site fidelity, which cause individuals to return repeatedly to the same area (Irons 1998, Wakefield et al. 2015), would result in unexplained temporal, as well as spatial, autocorrelation within individuals. Techniques have been developed for modelling some of these sources of autocorrelation (Marzluff et al. 2004, Aarts et al. 2008, Johnson et al. 2013) but as far as we are aware, no study on a colonial central-place forager to date has been able to model all of these correlation structures simultaneously. This is not only because of the complexity of the task but because the underlying mechanisms are still poorly understood. Conversely however, modelling these dependencies in a hierarchical framework would provide important insights into the foraging strategies employed by seabirds and similar taxa. Recent methodological advances, especially in Integrated Nested Laplace Approximation, may soon make this possible and we look forward to further development of these techniques (Blangiardo et al. 2013).

Distribution with respect to habitat

Our principal aim was to estimate usage at sea, irrespective of behavior. Had we modelled foraging locations only, stronger associations than we report might be expected between habitat and distribution (Wakefield et al. 2009). Similarly, considering time-averaged environmental covariates, though expedient, may have reduced the ability of our models to resolve dynamic environmental drivers of distribution if seabirds closely track spatiotemporally unpredictable prey. However, there is increasing evidence that, at the coarse scale, temperate neritic seabirds forage in individually consistent locations, both within and across breeding years (Irons 1998, Weimerskirch 2007, Woo et al. 2008, Wakefield et al. 2015). This may be because shelf sea oceanography is predictably structured by seasonal insolation and tidal stirring (Simpson et al. 1978), suggesting that time-averaged environmental covariates may be reasonable proxies for prey distribution.

The effects of habitat on spatial usage in our models were comparable in magnitude to those of foraging costs and competition (Table 2). Moreover, the habitat preferences indicated by models V–VIII accord with current understanding of the foraging ecology of the study species. For example, covariates describing substrate were retained only in models of habitat use for the three diving species (Shags, Murres, and Razorbills). Shags and Murres forage both at or near the seabed and in the water column so substrate type may affect prey availability directly (Watanuki et al. 2008, Thaxter et al. 2010). Razorbills forage at shallower depths but in common with all species in the study, prey primarily on sandeels, whose distribution varies with sediment coarseness and silt content (Wright et al. 2000, Holland et al. 2005). Previous studies suggest that sympatrically breeding Razorbills and Murres, which are closely related, do not segregate in space (Thaxter et al. 2010, Linnebjerg et al. 2013). However, our results suggest some landscape-scale niche partitioning: Murres outnumber Razorbills in inshore waters of the North Sea, the Northern Isles, and the Irish Sea, whereas Razorbills predominate in the Western Isles. Notably, our models also suggest a preponderance of Murres in the vicinity of the Celtic Sea front, which may reflect divergent foraging adaptations in these species (Appendix S2: Fig. S8).

Covariates best describing the distribution of Kittiwakes, which are obligate surface feeders, either described properties of the water column (stratification and relative sSST) or the morphology of the seabed (slope), which affects turbulent mixing. Presumably, these covariates were retained because they describe physical mechanisms that affect prey availability indirectly, either by enhancing production at lower trophic levels (e.g., tidal stirring resupplies nutrients to the photic zone; Scott et al. 2010, Carroll et al. 2015) or by advecting prey to the surface (Embling et al. 2012, Cox et al. 2013).

Species distribution models fitted to data collected in one area may predict usage poorly in another where habitat availability differs. To account for this effect we considered models in which the response of birds to candidate environmental covariates was conditioned on their regional means (i.e., a partial implementation of a Generalized Functional Response [GFR] to resource availability; Matthiopoulos et al. 2011). GFRs with respect to some but not all covariates improved model performance, indicating that seabirds responded non-linearly to changes in the availability of some environmental covariates. This is perhaps unsurprising, given the oceanographically complex nature of the study area (Appendix S1: Fig. S2). For example, Murres tend to forage far from the coast in areas where the mean distance to the coast was high, such as the North Sea, which has a relatively simple geometry. In areas where the mean distance to the coast was low, such as the geometrically complex Northern and Western Isles, this relationship was reversed (Table 2; Appendix S2: Fig. S5). Presumably, this reflects differences in the dominant physical drivers of prey distribution or the type of prey available to Murres in these areas.

Conservation implications

For conservation measures to be effective they must be evidence-based so there is an urgent need to map the distributions of seabirds at sea and to understand how these are shaped by intrinsic and extrinsic factors (Lewison et al. 2012). We estimated seabird distribution using data on the size and location of all known colonies in Britain and Ireland. However, missing substrate data meant that we did not estimate usage by Shags, Murres, and Razorbills outside the UK Exclusive Economic Zone (EEZ) or for parts of the Northern and Western Isles (Fig. 2). Moreover, we did not have access to contemporaneous data on conspecific colonies in countries bordering the study area. Although these may interact with colonies in Britain and Ireland, their relatively small size and large distance from Britain and Ireland suggest that any density-dependent competition from these colonies is likely to be negligible. Notwithstanding these caveats, the performance of our time-invariant models suggest that the factors determining the marine distribution of breeding seabirds in Britain and Ireland are sufficiently consistent across time to permit reliable estimation of area usage from biotelemetry, environmental covariates, and central-place foraging theory, which has important consequences for identification of priority areas for conservation measures. To date, potential offshore SPAs for European seabirds have been identified largely using at-sea transect survey data (Skov et al. 1995, Kober et al. 2012) and progress to designate offshore protected areas has been slow (BirdLife International 2010). Moreover, because it is impossible to derive colony-specific distribution estimates from at-sea observations, tracking is increasingly used to obtain the colony-level seabird distributions (Wakefield et al. 2011, Raymond et al. 2015) that are required for the assessment of impacts of marine industries on protected breeding colonies. Unfortunately, it is neither practicable to track widespread species from all their colonies, nor clear how usage can be interpolated from surveyed to unsurveyed colonies (Aarts et al. 2008, Matthiopoulos et al. 2011, Torres et al. 2015). Thaxter et al. (2012) suggested that, until better information becomes available, a pragmatic approach (the "radius" method) is to assume that seabirds are distributed uniformly out to some putative maximum range from their colonies. However, as our analysis and others confirm (e.g., Wakefield et al. 2011, 2013, Catry et al. 2013, Dean et al. 2015), seabird density declines with distance from the colony. Moreover, density-dependent competition, coastal morphology, and habitat preference result in highly non-uniform distributions. We show that these effects can be estimated by tracking birds from a sample of colonies and fitting IPP models, structured as partial GFRs (Matthiopoulos et al. 2011), to the resulting data. The ability of these models to estimate seabird distributions at unsampled colonies is a major innovation. Moreover, an advantage of IPP models over the logistic presence/pseudo-absence models latterly applied to tracking data is their interpretability (Aarts et al. 2012, Renner et al. 2015). Our models predict "occurrences at sea per day per individual" (i.e., incorporating information on both activity budget and space use), which is directly proportional to the average amount of time birds are expected to spend at a location and therefore of direct utility to conservation managers. The areas of intensive usage we identified, especially those used by birds from SPA breeding colonies, may warrant consideration for statutory protection following the principles recently outlined by Wilson et al. (2014). Moreover, the provision of colony-level predictions allows the potential impacts of anthropogenic and natural processes to be apportioned to specific colonies much more accurately than is possible using the radius method. This will be of particular importance in assessing potential impacts from offshore windfarms, which are projected to increase 10-fold in European shelf seas in the next decade, with the majority being constructed in UK waters (Infield 2013). Current assessments of the potential barrier, displacement, and collision impacts, both at the individual windfarm level and the region-wide level, rely either on data from boat or aerial surveys (Furness et al. 2013, Maclean et al. 2013), tracking from very few colonies (Perrow et al. 2006) or the radius method (Thaxter et al. 2012, Bradbury et al. 2014). As such, potential impacts cannot be reliably apportioned to breeding colonies, hampering attempts to predict their demographic consequences (Bailey et al. 2014). Similarly, the impacts of oil pollution and bycatch may be highly localized (Williams et al. 1995, Žydelis et al. 2013) so colony-level distribution estimates will facilitate spatial planning decisions that more effectively balance seabird conservation with competing interests, by linking marine aggregations of seabirds to specific colonies. The methods presented here demonstrate the utility of tracking data to estimate seabird distribution at national scales and further data are now required to allow the application of this modelling approach to other breeding seabird species. Moreover, by combining our results across species, potential areas of high conservation priority are revealed (Fig. 3; Appendix S2: Fig. S7). It is clear that, within Britain and Ireland, the core areas of usage of all four study species overlap within most of the coastal waters in Scotland. Areas of high multi-species usage may warrant particular attention, since both the vulnerability to threats and the potential benefits of conservation measures, are likely to be highest there. The regions identified as supporting the core 90% UD of at least three of the species considered here (Appendix S2: Fig. S7b) correspond well to those areas identified as of greatest international importance for 30 seabird species in the North Sea across all seasons (Skov et al. 1995), indicating the likely importance of these areas for a broad range of avian taxa.

Inclusion of density-dependent competition in our models increased their predictive performance. However, this improvement over previous similar analyses (Wakefield et al. 2011, Raymond et al. 2015) was only possible because the sizes of most seabird colonies in Britain and Ireland are known (Mitchell et al. 2004). In contrast, seabird colonies in many regions have not been censused (Croxall et al. 2012). Obtaining accurate estimates of colony size should be a priority for wildlife managers intending to use tracking data to estimate the distribution of seabirds from unsampled colonies. Moreover, our results suggest that distribution will change if colony sizes alter. Updating colony counts periodically would allow model-based distribution estimates to be revised without necessarily needing to collect more tracking data.

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Supporting Information

Additional supporting information may be found online at: http://onlinelibrary.wiley.com/doi/10.1002/eap.1591/full

Data Availability

Data available from the Birdlife International Seabird Tracking Database: http://seabirdtracking.org/mapper/contributor.php?contributor_id=950.

Seabird tracking at the Flamborough & Filey Coast:
Assessing the impacts of offshore wind turbines

Pilot study 2017
Fieldwork report & recommendations

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RSPB Centre for Conservation Science Report to Ørsted

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Cover pictures courtesy of: David Aitken (top left) and Saskia Wischnewski

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Executive summary

Hornsea Project One agreed to contribute to ornithological work at the Flamborough and Filey Coast potential Special Protection Area (pSPA), specifically a colony monitoring campaign and tracking work as part of a wider strategic study. It was agreed that the data would be made available for wider strategic research. The implementation of a strategic monitoring programme for the Hornsea Zone is to improve the understanding of the populations of key species (kittiwake, auks and gannet) at the pSPA and any possible dependence on the Hornsea Zone. The objectives of the strategic programme will be to improve understanding, through the collection and analysis of robust data, of:

1) abundance and trends in abundance of the species populations (and assemblage) of interest features of the pSPA. This would be achieved through contributions to whole colony counts that are more comprehensive and regularly undertaken than is currently the case.

2) population processes, which would include the collection of more data on the productivity and survivorship of key species breeding at the pSPA.

3) connectivity of the pSPA populations and the Hornsea Zone, through tagging which can reveal the movements of individuals between the colony and the zone and wind farm project areas within it.

As part of this package of monitoring, 20 adult kittiwakes breeding within the pSPA were fitted with lightweight, remote-download GPS-accelerometer tracking devices during the 2017 breeding season, just after their chicks hatched. Nest monitoring, for tracked and control birds, was also conducted in order to determine whether the tracking devices had any effect on bird welfare or breeding success. The three principal aims of the 2017 work presented in this report were:

(a) to collect baseline tracking data before turbines are in place (so that, in future, the behaviour and area-use of tracked birds can be compared before, during and after turbine construction);

(b) to pilot the use of this tracking system on this species and at the Flamborough and Filey Coast, and methods of data analysis, in order to refine methods for future strategic or post-consent tracking work; and

(c) to collect information on some of the parameters that feed into collision risk models in order to test the assumptions of these models and so that the models can be based on improved data in future.

All 20 tracking devices were deployed successfully at Flamborough, Filey and a new catching site at Speeton, and data were obtained from 18 of these. The new (for this species) medium-term tag attachment method we used worked very well and allowed tracking data to be collected for several weeks, and much later into the breeding season than has been achieved previously for kittiwakes (which had previously only been tracked for a few days at a time early in the chick-rearing period). The remote-download system also worked well in our study area, once we had determined suitable locations for the receivers. We were able to use an additional catching site at Speeton due to the system's ability to bounce data from a receiver near the nests at Speeton, to another on Filey Brigg 9km away, and then to a base station (connected to the internet) in a house 7km away from that. The two trackers that did not produce data were attributed to tag failure (potentially due to birds damaging the devices after attachment) rather than bird death, as

both birds were seen attending their nests in the colony (and within range of the receivers) during the weeks following tag attachment.

Stormy weather partway through the chick-rearing period had catastrophic effects on nesting success throughout the colony (tagged and control nests), with many nests failing at this time. However, many of the birds whose nests failed continued to visit the colony occasionally for some time after nest failure. This meant that we have collected world-first tracking data on the movements of kittiwakes after nest failure, which will be extremely valuable in understanding how birds at this life-stage behave at sea. The trackers did not have any obvious effects on adult welfare, but we are less certain about whether there are effects on nesting success of tracked birds. An initial comparison of tracked and control nests showed that nests of tracked birds were significantly more likely to have failed. However, once nest height was taken into account, this effect disappeared; it seems that lower nests (where we can catch birds) are more likely to fail, particularly at Flamborough, than those higher up the cliff, perhaps because of wave action from below compounding the effects of wind and rain on chicks.

Whilst the data collected were enormously valuable (e.g. ~30,000 GPS locations collected from 18 birds), we were unable to get as frequent GPS fixes as we had expected from the tags, due to issues with battery life caused primarily by the cliffs shading the tag solar panels for part of the day when birds were in the colony. Accelerometer data collection was very limited for this reason. We have spoken with the tag developers who are already working on modifications to the tracking system that should enable us to collect better data next year.

Data analysis is in progress and will continue until early 2018, but some initial results are included here. These results must be interpreted with caution since they represent only one year of study on 18 birds from a selection of nest sites within the large pSPA colony, and previous studies have shown that there is variation in the offshore areas used by kittiwakes both between individuals and years. Tagged birds from both Filey and Flamborough showed some use of the Hornsea wind farm zones, particularly Hornsea 1 and 2, largely for commuting, with the key foraging areas being to the north and south of the wind farm footprints. There was less overlap with the proposed wind farms of birds from Filey, which tended to forage to the north of the Hornsea developments, than those from Flamborough, which foraged further south. Given that birds at the north and south of the colony have different foraging distributions, it is important to understand the at-sea distribution and movements of birds at the centre of the colony, which represents the highest proportion of individuals. For this reason, one of our key recommendations for future years is to attempt to target more birds for tagging studies close to the centre of the colony within the Bempton cliffs RSPB reserve and at the new study site at Speeton.

We include recommendations for future work split into three sections: (i) data analysis/write up we intend to complete as part of the current contract; (ii) additional data analysis that would require new funding for staff time (but no additional data collection), largely due to potential collaborators offering to share data with us in recent weeks; and (iii) recommendations for further data collection in coming years. We recommend that in 2018 both kittiwakes and gannets are tracked using the current system, with the addition of barometric altimeters to allow more accurate flight height data to be collected.

Introduction

Background

The Flamborough and Filey Coast potential Special Protection Area (pSPA) is of international importance for breeding seabirds and contains the UK's largest mainland kittiwake and gannet colonies as well as important numbers of guillemot, razorbill, puffin and herring gull.

The Hornsea offshore wind farm development zones are situated in the North Sea, 63 km from the Flamborough and Filey Coast pSPA. The first two wind farm projects within the Hornsea zone have been granted planning consent and a package of strategic monitoring is being developed.

The potential impacts of the Hornsea wind farms on seabirds breeding in the pSPA include collision risk (particularly for kittiwake and gannet) and displacement (particularly for guillemot, razorbill and puffin), and there are also questions about the origin (breeding sites) of birds present in the Hornsea zone during the winter. However, there is a level of uncertainty associated with the predicted strength of these impacts on the population which, as highlighted in the Environmental Impact Assessment, requires further work. Therefore, the central aim of strategic monitoring is to reduce this uncertainty.

While there is a need to monitor impacts as part of the strategic monitoring for the Hornsea wind farms, there is a corresponding aim to conduct seabird population monitoring and understand potential effects on bird conservation in the area both by Natural England and the Flamborough Head European Marine Site Management Scheme in relation to monitoring the pSPA, and by RSPB as part of the reserve monitoring for the Bempton Cliffs reserve. To ensure a coordinated approach to seabird monitoring and conservation in the area, a voluntary seabird monitoring group has been set up comprising representatives of all the organisations listed above as well as Ørsted and their consultants (currently NIRAS). This group was involved in the concept development for the current strategic monitoring project.

Aims & Objectives

This project, funded as part of the strategic monitoring for Hornsea Project One, primarily addresses questions about kittiwake collision risk by tracking individual birds during the breeding season. However, the project was set up with the aim of collecting additional data in future years as part of a longer-term monitoring package to be developed. Here we report on the pilot / baseline kittiwake tracking work conducted during 2017; as such the aims and objectives have been split into long- and short-term (2017) aims below.

Long-term aims & objectives

Previous GPS and satellite tracking work conducted by RSPB has shown that the Hornsea wind farm zone is within the foraging range of kittiwakes and gannets breeding in the pSPA. Current collision risk modelling predicts that the wind farms will lead to collision mortality in both of these species, with uncertainty surrounding the likely magnitude of this impact and its significance for these populations. More widely, there are significant gaps in understanding of seabird collision risk from offshore wind farms, some of which could potentially be narrowed by this strategic work. As such, this project aims to collect information relevant to the Hornsea development, but which will also have broader relevance to the testing and refinement of collision risk models for offshore wind farms.

A key practical limitation of any project tracking kittiwakes is that we are unlikely to obtain a licence to use a long-term harness attachment of tags to this species, due to welfare considerations. Whilst harnesses have been used successfully (with evidence of no welfare implications) on species such as lesser black-backed and herring gulls, there have also been considerable problems with harness mounting on other species such as great skua (Thaxter *et al*. 2016) that spend the winter at sea. Because kittiwakes spend the winter at sea, we are concerned that they may be similarly adversely affected by a harness. This means that (unless using very small tags that can be attached to leg rings, such as geolocators) we are limited to temporary attachment of tags to feathers using glue or tape, which will last for a few weeks at most.

We use high quality GPS tracking devices developed by the University of Amsterdam (UvA) (http://www.uva-bits.nl/system/) to collect data on birds' locations at a high temporal frequency (between every 3 and 15 minutes). These tags also contain accelerometers, which allow us to detect the individual wing beats of birds and differentiate gliding/soaring from flapping flight. These accelerometers allow us to measure sudden changes in flight behaviour around turbines (i.e. changing from gliding to flapping flight in close proximity to a turbine could indicate micro avoidance). Furthermore, innovative analytical methods developed recently by the British Trust for Ornithology (BTO) have allowed robust modelling of flight height distributions from these tags, taking account of the error in altitude recorded by GPS (Ross-Smith *et al*. 2016[1]). These technical

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[1] This method recognises that GPS tags record altitude with error and treats each recorded flight height as an observation with error. It then uses a state-space model that models the underlying flight height distribution and the error distribution in parallel. The error in the GPS tag altitude estimate is related to the number and position of satellites, captured by the Dilution of Precision (DOP) which is recorded by the tag for each GPS location. The error in altitudinal measurements is assumed to be normally distributed around the true altitude, with the standard deviation of the normal distribution linearly related to the DOP of each

specifications will allow us to address the following questions, which are of relevance to the monitoring of Hornsea wind farms, but also address some of the critical uncertainties in relation to the parameters used in collision risk modelling for offshore wind farms:

1. Provision of site-specific flight height information for kittiwake (a key part of testing whether the assumptions of collision risk modelling are correct)
2. Empirical measurement of flight speed, and variability in flight speed (the flight speeds currently used in collision risk models are based on very limited data and do not account for variability)
3. Empirical measurement of the tortuosity of flight lines (collision risk models currently assume birds fly in straight lines. We know they don't. This will allow us to quantify how much flight lines deviate from a straight line and test how this alters collision risk)
4. Empirical measurement of collision avoidance behaviour – this will be a longer-term output of the project, should tracking be continued in future years, but tracking during 2017 will be important to establish an appropriate baseline. Macro and meso avoidance would be detectable by examining high resolution GPS locations in relation to turbines. Previous tracking is not at sufficient resolution to allow these measurements (points were far enough apart in time that a bird could have flown into a wind farm and out the other side in between location fixes), hence the need for new work to establish a baseline ahead of future monitoring. Micro avoidance could be measured by combining the high-resolution GPS fixes with accelerometer data which allows the detection of individual wing beats. Once turbines are constructed, this would allow us to detect a sudden change to rapid flapping flight (from soaring or slower flapping flight) when birds are in the vicinity of turbines. We propose to use a before-after control-impact (BACI) design to compare the frequency of these types of sudden changes in flight patterns in the vicinity of turbines with control areas away from turbines, and to compare flight patterns before and after construction, to allow a measurement of micro avoidance to be developed, currently a critical gap in knowledge.

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observation. The error distribution varied between species (lesser black-backed gull and great skua), presumably because behavioural differences (more variable vs more constant flight height) change the modelled error and was wider when DOP was higher. The mean DOP was 3.3 (SD 1.6) for lesser black-backed gull, and the standard deviation of the altitude measurement error distribution was 8.9 m for DOP = 1 and 16.9 m for DOP = 10. For great skua the mean DOP was 3.7 (SD 1.6) and the standard deviation of the altitude measurement was much higher at 28.8 m for DOP = 1 and 38.5 m for DOP = 10.

Short-term aims and objectives (2017 work)

The pilot study of 2017 was to fulfil the short term aims of:

i. Collecting baseline data of the same quality as follow-up years of tracking during construction and operation of Hornsea Project One Offshore Wind Farm, and
ii. to pilot the methods of both data collection and analysis to be used in the future.

Data from the baseline year could be used to investigate long-term aims 1-3 above (though the analyses would be stronger once additional years of tracking are incorporated during and after construction), while long-term aim 4 (avoidance rates) could be answered once we have collected data using the similar tracking methods following construction of Hornsea Project One Offshore Wind Farm. In fulfilling the short term aims, the objectives of the pilot study in 2017 were to:

a) Track 20 individual birds from the colonies at Flamborough Head and Filey Brigg.
b) Conduct follow up fieldwork to monitor the 20 tagged birds and at least as many control birds, and to manage the download of data from the tags.
c) Use appropriate analytical methods to analyse the data in order to address the key questions set out above.

This preliminary report summarises the results of the first year of fieldwork and outlines proposals for the following year's work. In the longer term we will report the results of the study in peer-reviewed papers. We believe this is the best way to ensure that the findings are accepted as robust by the wider scientific and industry community, due to the rigorous nature of the peer-review process.

Methods

Field methods

Site selection

Flamborough Head and Filey Brigg were chosen as the main tagging locations within the Flamborough and Filey Coast pSPA as these are the safest places to catch kittiwakes in the area. At these locations, access to nests to catch birds is possible from the bottom of the cliffs without the need for boat access or working at height with ropes, both of which can cause safety issues for birds and people. In addition, both sites have previously been used by RSPB for kittiwake tracking, and hence there are data from other years against which to compare this year's results.

Selection of areas of cliff with groups of nests to monitor and catch birds from (sub-sites), in both Flamborough and Filey, was based on three main criteria:

1. Accessibility for tagging (i.e. numerous nests had to be reachable from ground with a 12 m noose pole)
2. A straight line of view to potential relay positions within 1 km to remotely download data via a network setup
3. Visibility for regular monitoring (i.e. safe and distant positions for observers that minimise disturbance of nests without compromising the ability to see nest contents)

Photographs from the bottom of the cliff as well as potential monitoring and relay positions were overlaid to select sub-sites that fulfilled all three criteria. This was done early in the breeding season when birds were incubating eggs. If there was more than one suitable option for relay positions in either Flamborough or Filey, the one that maximized the number of accessible/observable nests was chosen to account for predictable breeding failure during incubation (approximately 50%). In addition to nests suitable for tagging, at least the same amount of control nests had to be monitored to assess potential impacts of tagging on breeding performance. Therefore, only sub-sites that had at least double the amount of accessible and observable nest than needed for tagging in addition to the same number of control nests were chosen (i.e. If 10 birds are to be tagged at one sub-site, the site must have at least 40 nests).

Nests were further chosen based on their proximity to each other, their height and the direction of the cliff face to ensure comparable exposure to weather, predation etc. and thus the validity of controls. Using the photographs taken from the monitoring positions, all selected nests were numbered. At sites where the nest density was very high (Flamborough) only a subset of nests was selected and labelled to ensure the feasibility of frequent monitoring. These photographs were printed on photo paper and laminated as references. Monitored nests were divided into 6 monitoring plots (6 photographs) in Flamborough and into 2 monitoring plots (2 photographs) in Filey. See Table 1 for an overview of monitored and tagged nests in each of these sites.

As a trial, further two nests were tagged at an accessible site at Speeton cliffs to attempt collecting data from nests closer to the centre of the Flamborough and Filey Coast pSPA. Only limited post-deployment monitoring of tagged nests took place at this site due to more difficult access. No control nests were monitored at Speeton at any time.

All study sites are shown in Figure 1.

Table 1. Summary of sample sizes for monitored and tagged kittiwake nests. The "hatched" column refers to nests that successfully hatched a minimum of one chick. For exact locations of sites and subsites refer to Figure 1.

Site	Nests monitored	Hatched (%)	Tagged	Control
Flamborough	**249**	**226 (91%)**	**13 (+2 dummy)**	**211**
North	45	38 (84%)	2	36
Centre	84	75 (89%)	5	70
South	120	113 (94%)	6 (+2 dummy)	105
Filey	**84**	**78 (93%)**	**5**	**73**
North	72	69 (96%)	3	66
South	12	9 (75%)	2	7

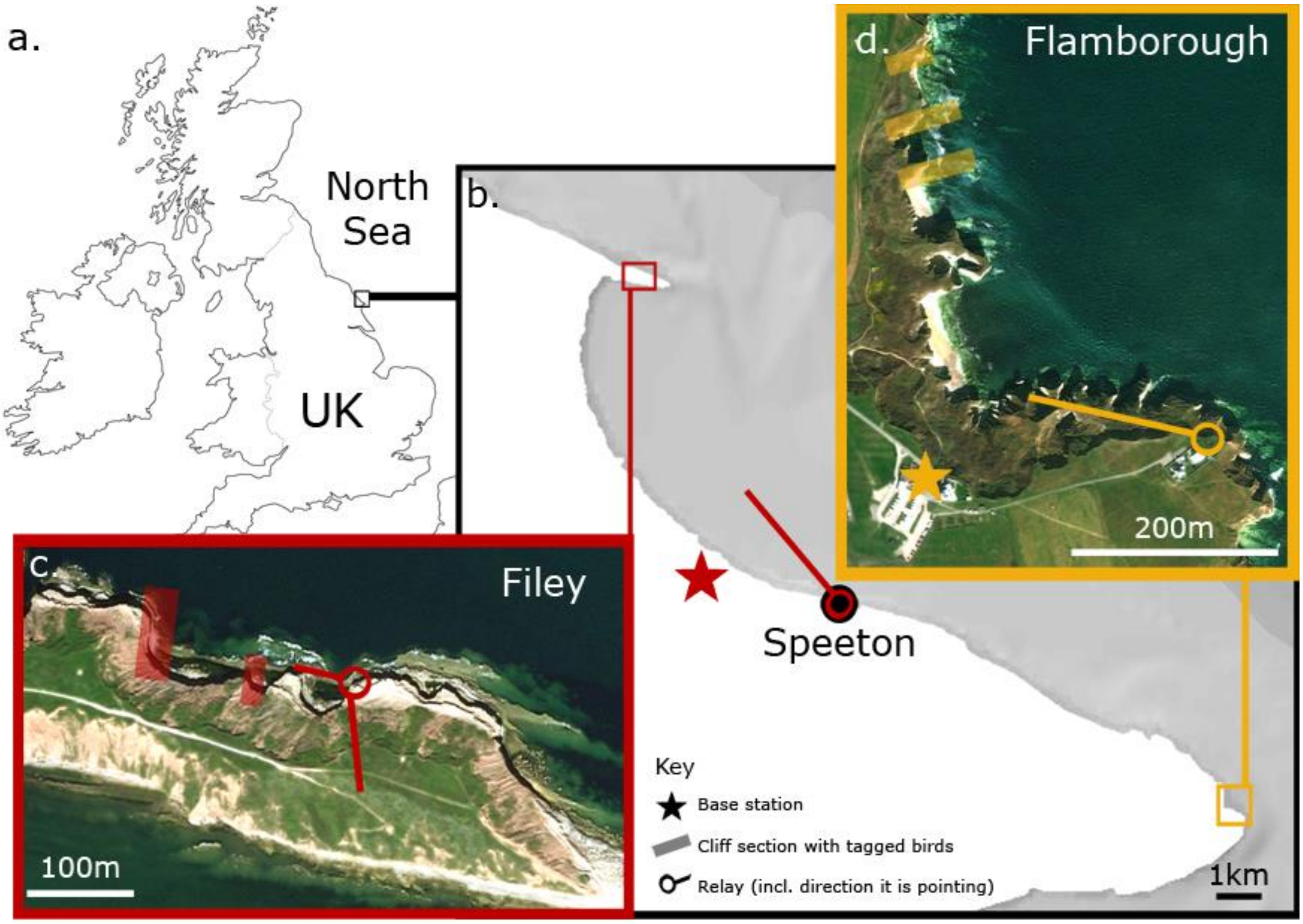

Figure 1. Location of field sites and network set-up. Stars indicate base-station locations. The black circle in map b. indicates the Speeton field site for which no close-up is shown. Circles with lines refer to relay locations; the angle of the line represents the direction the relay pointed (note there were two relays at the same location in Filey pointing in different directions). Shaded rectangles highlight monitored cliff sections (sub-sites) (North, Centre and South for Flamborough; North and South for Filey). Yellow symbols refer to the Flamborough network, Red symbols to the Filey network. Scale bars are shown. Individual maps were exported from ArcGIS 10.5 (ESRI 2016).

Network set-up

Network technology for remote download of data from deployed tags was provided by the University of Amsterdam (UvA) BiTS system and comprised of relay antennas and base stations (a laptop attached to an antenna for data download). Relays were connected to a battery which enabled free deployment in the field, whereas base-stations required access to electricity and, ideally, internet to (a) upload data to the BiTS server and to (b) remotely connect to the base station and change tag settings. One relay needed to face nests with tagged birds from a maximum proximity of 1 km. Additionally another relay at the same position (or the same relay, dependent on angles between cliff and base station) needed to face a base station up to 8km away to connect tags to the base station enabling the remote download of data and adjustment of tag settings. Detailed set-ups for sites are indicated in Figure 1.

Only one relay was needed in Flamborough to connect tags and base station due to a favourable angle between both, whereas in Filey two relays were used to transmit the signal from the tag to the base station. The Speeton relay connected to the Filey relays over approximately 9 km distance and data were subsequently downloaded to the Filey base station. In Flamborough the base station was positioned in very close proximity (~ 250 m) to the study site in a local shop with its antenna directly overlooking the relay. The Filey base station was positioned in the conservatory of a private house with its antenna facing the relay on Filey Brigg approximately 7 km across Filey Bay.

The first tags were deployed on 16th June 2017 and networks were installed before or shortly after the first tag deployment at each site. They were left in the field until mid-September 2017 to allow for potential data downloads from late returning birds.

Tag setup

Birds were tagged with UvA BiTS GPS and accelerometer tags equipped with two solar panels (Figure 2). Because tag data were remotely downloaded via the network, re-trapping of birds was not required. Tag settings could be altered via the network once the tag was on the bird and were adjusted to accommodate changes in battery voltage and maximize data collection. Tags needed to be in the colony or in 1 km proximity to one of the relays to download the updated settings from the base station.

In addition to the overall GPS and accelerometer sampling frequency it was also possible to change settings within geographic areas using a GPS fence (for example around the colony) and across time intervals (e.g. during day and night). Overall, GPS sampling frequency ranged from

3 to 15 minutes and accelerometer samples were taken every 10 minutes for 3 seconds at 20 Hz. Accelerometer sampling was started remotely towards the end of the deployment, after set-up and battery life had been thoroughly tested, to avoid draining batteries and damaging tags. At this stage some of the nests with tagged birds already failed and birds did not return regularly to download data and update tag settings. Consequently, accelerometer data collection was limited to a subset of individuals.

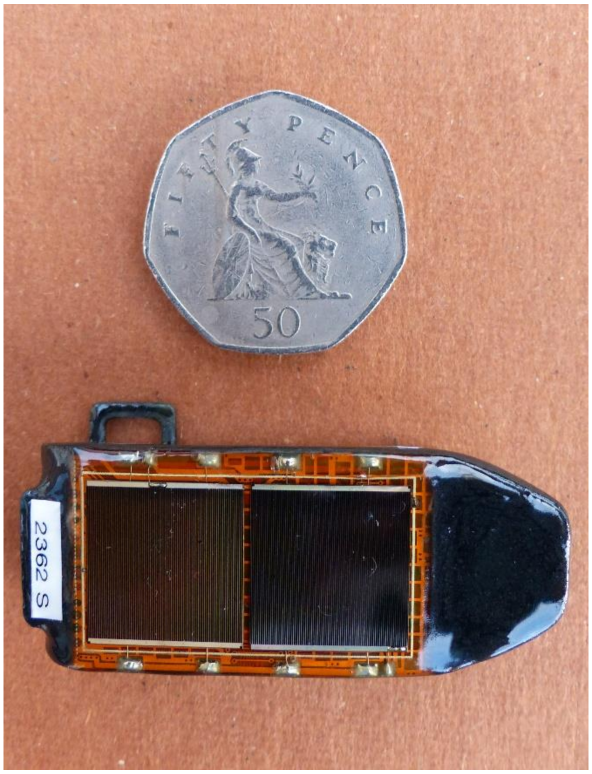

Figure 2. UvA BiTS tags used in this study. A 50 pence piece was used as scale. Picture courtesy of Saskia Wischnewski.

Tagging procedure

Thirteen adults were tagged in Flamborough, five in Filey and two in Speeton.

Tagging commenced on 16th June in Flamborough, just a week after the first eggs started hatching. The last birds were tagged on 8th July at Speeton. To ensure the catchability of adult birds while minimizing the chance of abandonment and nest failure, tagging concentrated on only one adult of a pair during early chick rearing rather than incubation.

Nests for tagging were approximately equally distributed across all monitoring plots at each site. The final selection for tagging was based on accessibility and proximity between each tagged nest to ensure that disturbance to the rest of the colony was equally spread and not all treatments nests were next to each other (Table 1). Adults were caught using a 12 m landing net pole with an attached snare that was moved over the bird's head. Thus, accessibility was mainly defined by nest height (<12 m) noting that increasing height made catching more difficult. However, an effort was made to not only trap birds at lowest nest but get a representative sample across the catchable height spectrum. Overhangs above and below nests as well as other cliff features that blocked the view of the bird from underneath could impede catching attempts and thus affected which nests were targeted.

Trapped birds were quickly moved into a cotton bag to reduce stress levels. Birds were then weighed (body condition), ringed with metal and engraved darvic rings (yellow with black writing; to enable later identification and re-sightings via telescope, without re-catching the bird), measured (wing, head and bill, bill; to be able to sex birds), colour marked with permanent marker on head feathers (to simplify identification of tagged birds from a distance) and tagged (Figure 3 d.).

Tags were glued (Super Glue) to trimmed back feathers of the birds. To increase the tag's surface area for attachment and simplify potential tag removal at a later stage (if necessary, not planned) a piece of muslin was glued to the bottom of the tag in preparation for fieldwork. Before deployment it was cut to extend approximately 0.5-1 cm beyond the tag's base (Figure 3 a.). The bird's body feathers were then trimmed to the shape of the muslin and wiped with acetone to remove any grease (Figure 3 b.). After waiting a few seconds for the acetone to dry the muslin was covered in superglue and the tag was carefully glued to the bird's back (Figure 3 c.).

This attachment method has previously been used in studies of various other bird species but had to our knowledge not yet been tried on this species, which is why detailed tagging impact studies including productivity and survival monitoring were necessary. However, it was chosen since it showed the potential to extend the deployment period compared to other short-term or long-term attachment methods which are either very quickly removed by the bird after a couple of days (Tesa tape) or are known to have negative effects on productivity and survival (harnesses).

To gain a more detailed understanding on the effects of the attachment method on the bird, 6 dummy tags (four in Flamborough and two in Filey) which matched the dimensions and weight of the original tag were deployed. The motivation was to re-trap these birds and assess the wear on the attachment and its interaction with the bird's skin and feathers. Unfortunately, either birds were not in the colony or bad weather jeopardized all re-trap attempts, but two dummy tags that were deployed within the monitored sites were added to the productivity comparison between tagged and control nests.

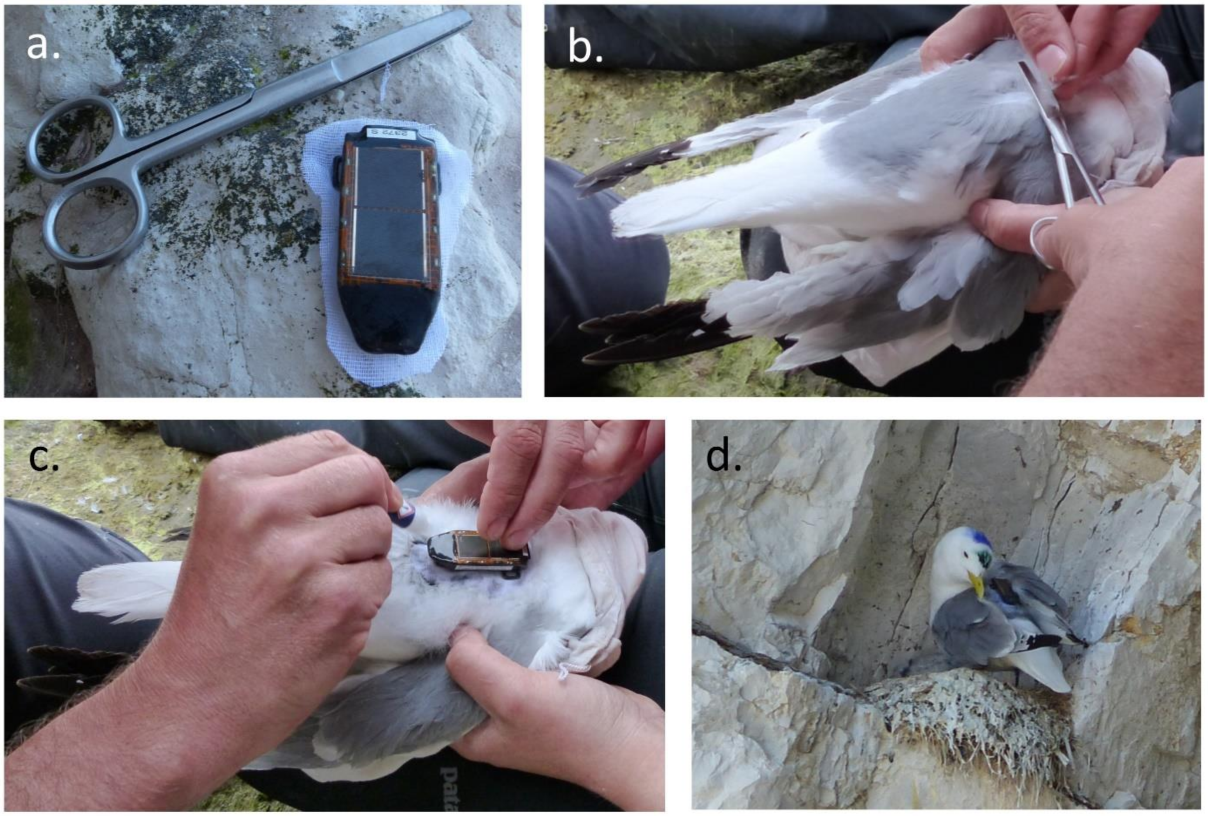

Figure 3. Photographs illustrating the tagging procedure. a. the tag just before deployment, attached to a layer of muslin cut to the shape of the tag. b. trimming the bird's back feathers, leaving short feather stubs that provide a secure base to glue the tag to. c. the tag is glued to the bird after feather trimming. d. the bird back at the nest shortly after release, preening its feathers and inspecting the tag. Note the colour markings on its head. Pictures courtesy of Saskia Wischnewski (a., d.) and David Aitken (b., c.).

Productivity monitoring

The monitoring of nest sites commenced on the 1st of June 2017 and sites were monitored every second day, or as frequently as possible if weather conditions were not favourable. Each monitoring session consisted of checking each nest for contents and present occupancy, and monitoring sessions were randomly distributed across daylight hours. Survey times and durations were recorded. Since adult occupancy was not the focus of this study, no particular time intervals for monitoring were chosen (e.g. to target periods of high nest attendance (such as early in the morning or late at night). Last monitoring took place on 10th August 2017 by which time the majority of nests had fledged and remaining chicks were fully grown and could therefore be considered successfully fledged. Clutch and brood sizes were reported as outlined in the JNCC Seabird Monitoring Handbook (Walsh *et al.* 1995). Chicks were divided into three development categories based on plumage characteristics, where small referred to downy chicks, medium to chicks with fully developed plumage markings, and large to chicks with down free wings with well-developed primary feathers. It was further noted when adults were not present, and nests were unattended. Hatch and fledge dates were determined by the average date between the last egg and first chick observation or the last chick observations and the first time the nest was unoccupied respectively. Empty nests were considered as successful if chicks were above 30 days and thus reached the minimum reported fledge age (Coulson 2011) while being well developed (large for more than the prior 3 observations). If a dead chick could be observed in the nest or these criteria were not fulfilled, and chicks were missing, nests were considered as failed.

Colour ringing

To compare adult survival over winter between tagged and non-tagged nests, a further 30 darvic rings were deployed later during chick rearing across Flamborough and Filey at non-monitored nests. Since kittiwakes are highly philopatric, birds are expected to return to a nest site in close proximity to this year’s. Consequently, next year’s re-sighting rates for tagged and non-tagged birds will be compared to determine whether tagging affected survival rates.

Analytical methods

Tracking data analysis

All data preparation and analysis were performed in R (R Development Core Team 2017), including most data visualization. In some cases, ArcGIS 10.5 (ESRI 2016) was used for mapping, which is highlighted in relevant figure legends. Only key R packages and functions that are central to understanding and ensuring the repeatability of the analytical approach are referenced within this report.

Data preparation

Tracking data were directly accessed via the BiTS server and extracted for each bird from the start of the deployment until the most recent data download. Tracks were visually examined and obvious outliers (likely inaccurate data) were removed; a standard procedure in tracking data analysis. If part of the analysis required projecting the tracking data, a Lambert Azimuthal Equal Area projection centred on the mid-point of all collected tracking data was used. This projection is commonly used for the analysis of tracking data of far ranging marine species to create a data specific projection that maximise the accuracy of distance and area measurements across the whole data set. All data points within a 1 km radius around each of the breeding colony sites were removed from subsequent analyses since they were expected to mainly include non-target behaviours such as rafting on water or resting on the cliff in close proximity to, or within, the pSPA. Individual foraging trips longer than 1 h between the bird's departure from the colony and its return were isolated. Due to varying sampling rates (between 3 and 15 minutes), shorter trips (<1 h) were excluded from the analysis of foraging trips since they may include less than the minimum of 4 data points that are required to calculate representative trip metrics and perform behavioural annotation of the tracking data.

Trip summaries

Trip duration, foraging range, travelled distance and mean speed were calculated for each trip and summarized for all the tracking data combined and by site. Since birds returned immediately to their nest site, spent at least one hour back at the nest or within 1 km of the colony before engaging in the next foraging trip (>1h), and there was no significant difference between the trip metrics (duration, range and distance) for the first and second trip conducted (Appendix I), we assumed that stress from capture and handling did not cause unusual behaviour during the first trip after deployment. Therefore, all foraging trips were included in the data set. To test whether foraging trip characteristics differ between sites, metrics were statistically compared using mixed effect models (*lmer*) with site and day of the year as predictors and bird id as random effect to account for pseudo-replication.

Behavioural annotation

Behavioural annotation is a method to infer different behaviours such as foraging, commuting and resting from simple GPS data when no direct measurements via accelerometers or time-depth recorders are available. Several different methods are widely used in tracking studies, each with their own assumptions and limitations.

Here we used Expectation Maximisation-based Clustering (EMbC) (Garriga *et al*. 2016) on the whole data set in the corresponding R package[2]. Since this annotation method mainly relies on assigning 4 behavioural states to patterns in tortuosity and speed, large gaps in the data or unequal sampling rates can heavily influence annotation. Before running the behavioural annotation algorithm, trips with gaps larger than one hour, which were caused by a break in the sampling regime due to a drop-in battery voltage usually linked to weather conditions or too high sampling frequencies, were therefore split into segments. Afterwards, each of these segment and trips without gaps were equalized to 10-minute intervals using the *redisltraj* function in the adehabitatLT package. Finally, the output of the behavioural annotation run on equalized trips and segments resulted in four output states of low tortuosity and low speed, low tortuosity and high speed, high tortuosity and low speed and high tortuosity and high speed. These were assigned to resting, commuting and two different foraging modes respectively, following Garriga *et al*. (2016). Finally, for each trip the mean minutes per hour a bird engages in commuting, resting and foraging (both modes together) were calculated and summarised for all the tracking data combined and per site. To test whether there are behavioural differences between trips from different sites, hourly rates of behaviour states were compared in a general linear mixed model with site and day of the year as predictors and bird id as a random effect to again account for pseudo-replication.

Kernel analysis

Kernel density analysis is a widely used and simple approach to infer utilization distributions (UDs) from locational data such as provided by GPS tracking using the density of relocations in space.

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[2] This is a new, cutting-edge and highly robust behavioural annotation method that has been ground-proofed on several seabird species with similar foraging behaviour as kittiwakes. Its key benefit to other methods is the binary clustering of input variables (in this case speed and tortuosity), which provides easily interpretable and meaningful results. It further does not require the input of often highly subjective parameters (as for example using First Passage Time) and has the ability to implement uncertainty in the data.

In simple terms, utilization distributions identify on a map the areas used by the birds (essentially their “home ranges”), and within that the areas that are used most frequently. The output is a three-dimensional probability surface. It uses a smoothing parameter to extrapolate the data and account for error, which can be determined using different estimators and methodologies.

In this case we followed the method outlined by Lascelles *et al. (*2016), which uses the mode of optimal search radii of all trips calculated by the *varlogfpt* function in the adehabitatLT package as smoothing parameter (4.1km). Afterwards the *KernelUD* function in the adehabitatHR package was used to calculate 50, 75, 90 and 95 % density contours for the whole data set, 50 and 95% contours for each of the sites and 50, 75 and 95% contours for each of the three behavioural states to a 1 km^2 resolution. Most commonly used contours are 50 and 95%, which for data of full trips are commonly associated with highly used areas (e.g. foraging locations) and the overall distribution/home range respectively. Therefore, the lower the percentage the higher the probability to encounter a bird in the area within the contour. To further examine differences in the spatial overlap between sites, the percentage overlap for each of the contours was calculated. Additionally, the percentage overlap between all determined contours and the Hornsea offshore development zones were calculated.

Accelerometer data

An inventory of all collected accelerometer data was taken, and a sample was plotted. Further analysis of this data will be completed subsequently (see recommended analysis section towards the end of the report).

Productivity analysis

Failure rates of control and treatment (tagged) nests during chick rearing were initially compared using the simple and widely used chi-squared test, which does not allow confounding variables to be included. A likely confounding factor affecting failure rates was nest height. To ensure results reflect tagging impacts, nest height (from the tide line) was measured using photographs of the study sites. A binomial model of failure rate with nest height, site, group (nest vs. control) and an interaction between the site (Flamborough or Filey) and nest height as predictors was fitted to test whether there was a difference in nest failure rates between tagged and untagged birds once nest height and site effects were accounted for.

Results

We successfully deployed 20 tags on adult kittiwakes during the early chick rearing period. Handling time of each bird from capture on the cliff to release was on average 12 (SD=±1) minutes (Range 9 - 16 minutes). Behavioural observations after deployment did not show any unusual behaviour and birds returned to the nest within 6 (SD=±2) minutes.

For 18 of the 20 deployments (all 13 in Flamborough, four in Filey and one in Speeton) we were able to download 168 foraging trips ranging from 3 to 24 trips per deployment/bird (Mean 9 (SD±6)) (Figures 4 and 5). For these birds we collected a total of 29,772 GPS fixes (including altitude estimates) and 755 three second accelerometer measurements (limited to 3 birds) (Figure 6). The first tag was confirmed to have been lost (bird seen in colony without tag) after being deployed for 20 days and the last data download happened after a deployment of 29 days.

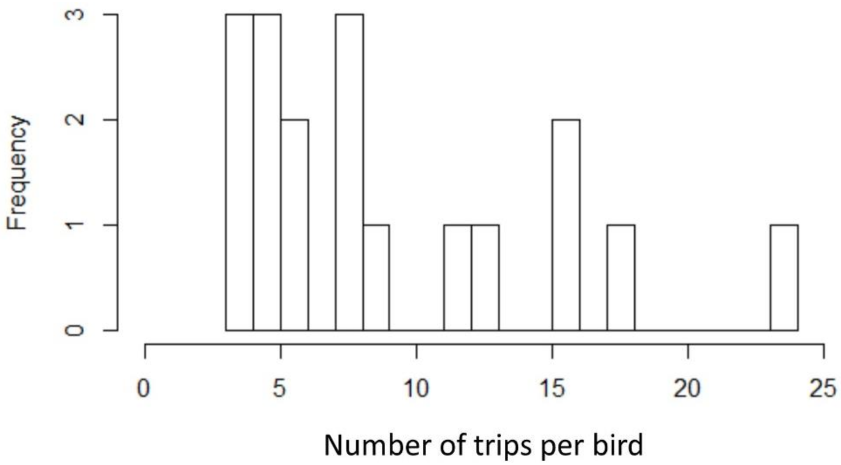

Figure 4. Frequency distribution of the number of trips tracked per tagged bird. Only trips that lead further than 1 km away from the colony and are longer than 1h are included. N=18.

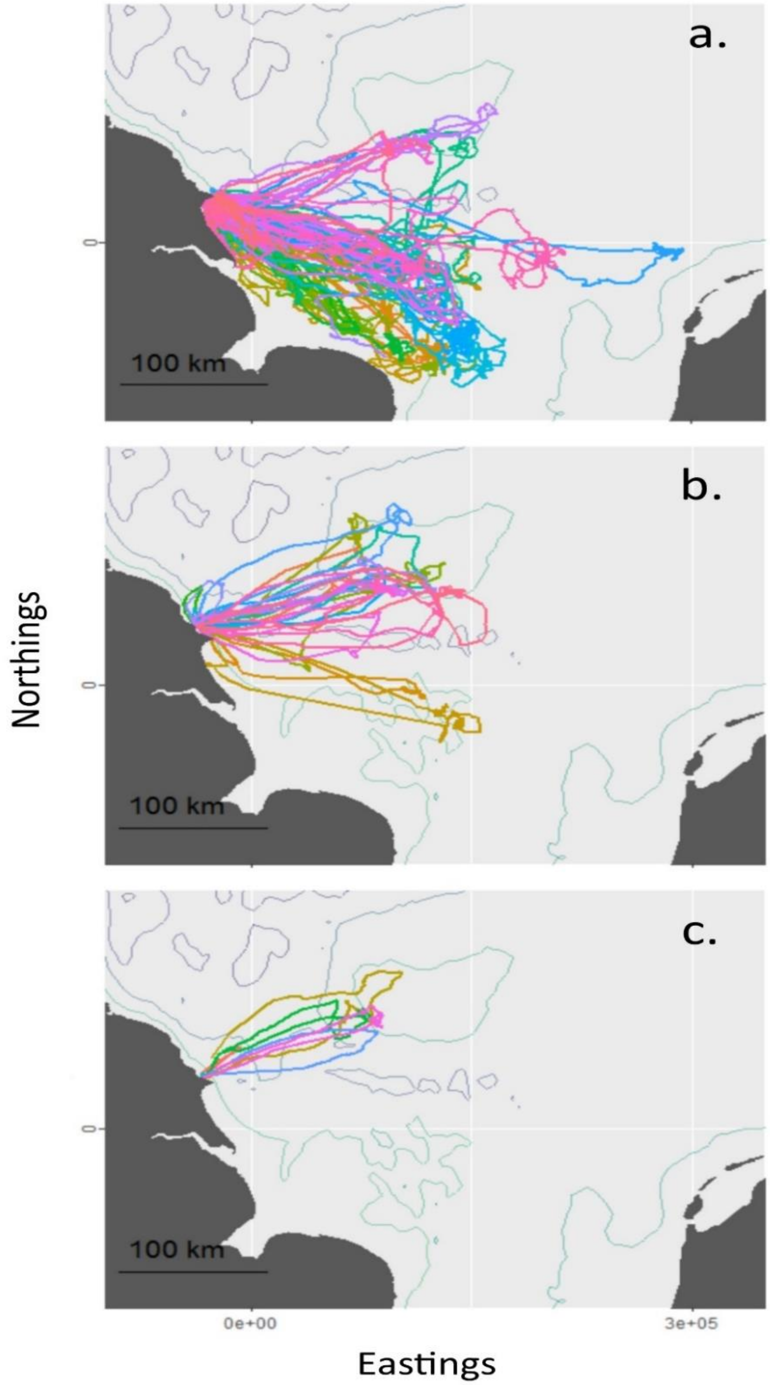

Figure 5. Individual Kittiwake GPS trips collected during the 2017 breeding season at the Flamborough and Filey Coast. All trips are shown from a. Flamborough (N=133 trips from 13 birds), b. Filey (N=29 trips from 4 birds) and c. Speeton (N=6 trips from 1 bird). Different shades represent individual trips. Bathymetric contours and scale bars are shown, with land in dark grey (UK, left; the Netherlands, right). The map is projected to the Azimuthal Equal Area centred on the mid-point of all the tracking data. The number of birds tracked per site is given in Table 1.

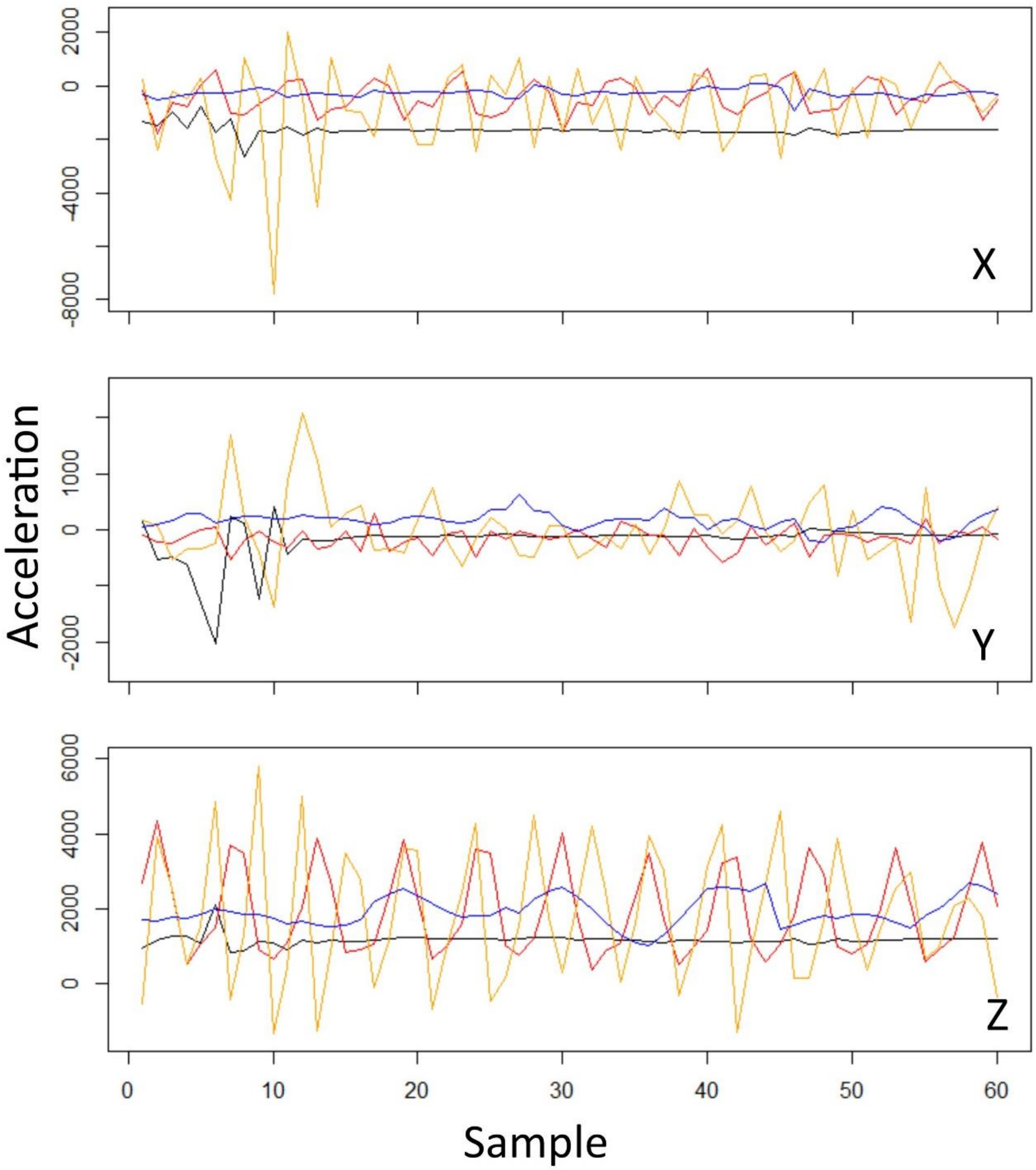

Figure 6. Sample of the acceleration data collected for one tracked individual at Flamborough. Each plot represents one accelerometer axis. Four sets of consecutive samples are shown, and colours across plots correspond to the same consecutive sample set. The 60 measurements (samples) taken reflect a three second consecutive sample set, at 20 Hz. Note the differences in amplitude between different measurements (e.g. the black and the orange line) which will be used to distinguish between behaviours such as flapping flight and resting.

Trip characteristics and behaviour

Trips metrics and hourly rates of the three target behaviours (commuting and two different foraging modes) are summarized in Table 2 and visualized in Figure 7 respectively. Linear mixed models comparing trip characteristics and behavioural composition across all three sites (Filey, Speeton and Flamborough) could not identify any statistically significant differences between any of these sites, but the hourly commuting time seems to decrease very slightly through the duration of the chick rearing period (-0.28 (SE=± 0.13), df=86, t=-2.25, p=0.02).

Table 2. Summary of the three trip metrics calculated for all 168 Kittiwake trips. Trips ranged over a maximum period of 29 days across the chick rearing period and also included trips from failed individuals. Note that the reported large standard deviations are due left skewed distribution of all three trip metrics. N=168 from 18 birds.

Trip metric	Mean (±SD)	Range
Trip duration (h)	22.12 (±28.69)	1.00 - 168.67
Foraging range (km)	88.65 (±74.22)	3.20 - 323.85
Travelled distance (km)	256.62 (±261.88)	7.91 - 1249.70

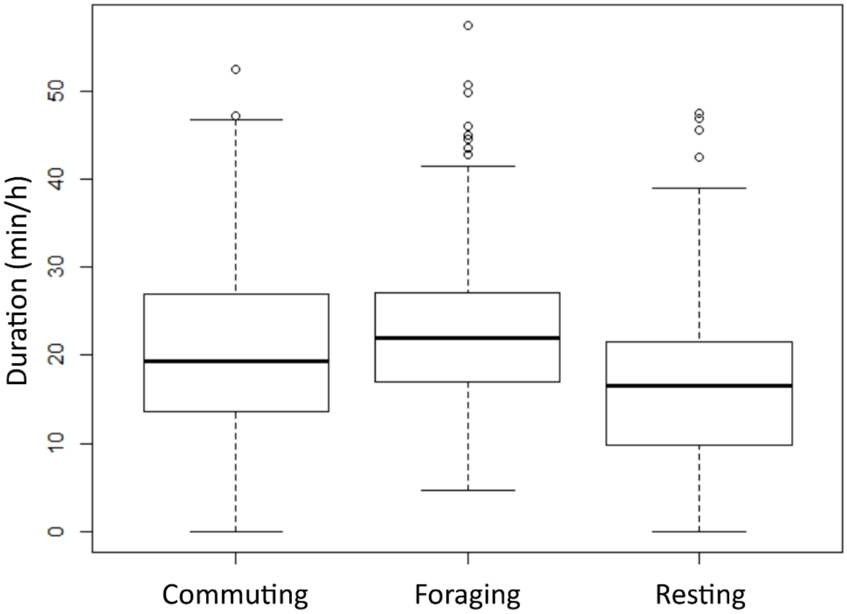

Figure 7. The relative contribution of the three different behaviour states to each foraging trip. Horizontal bars represent median values and whiskers 95% confidence intervals. All values are scaled to the duration of each individual foraging trip such that the mean minutes per hour a bird engages in each of the behaviours is used for each trip. N=168 trips from 18 birds.

Utilization distributions

Tracked birds spent most of the time during their foraging trips either in areas relatively close to the colony or in selected hot spots within 200 km of the colony (Figure 8). The utilization distributions of all three sites (Flamborough, Filey and Speeton) showed a substantial overlap with each other (Figure 9, Table 3). The distribution for foraging and resting show a similar pattern to the UD for all the tracking data (Figure 10 a. and c). However, distribution for commuting behaviour was much more widely spread, highlighting minor and major flight corridors from and to the colony and between hotspots that were not included in the overall or site-specific UDs (Figure 10 b). Note that utilization distributions include all trips collected in this study and thus cover foraging trips of chick rearing adults from just after hatching to when they are caring for chicks that are close to fledging size. They further also include trips collected after nests failed.

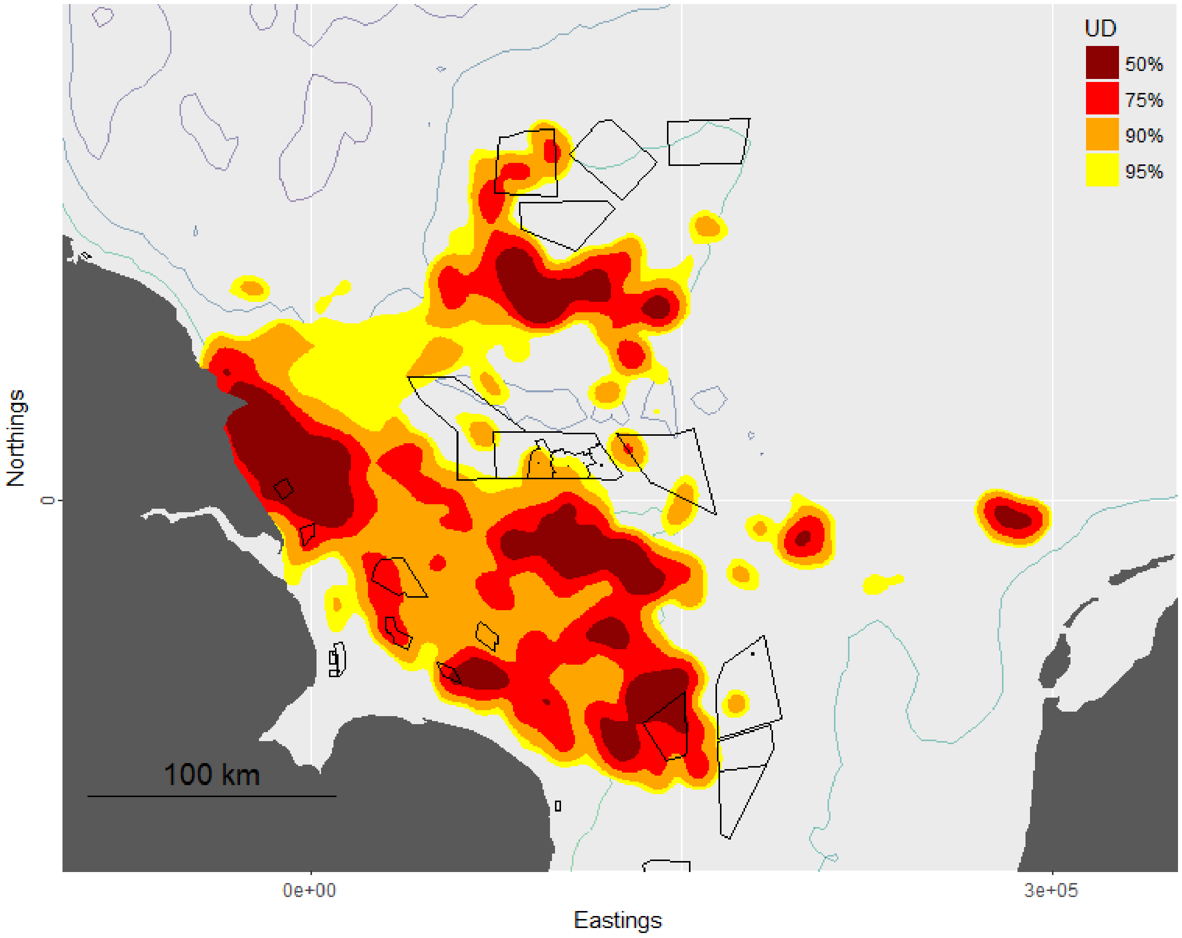

Figure 8. Utilization distributions of all kittiwakes tracked at Flamborough and Filey Coast during the 2017 breeding season. N=168 trips from 18 birds. 50, 75, 90and 95% contours are shown. Bathymetric contours, scale bar and outlines of all proposed, planned or active windfarm

zones are shown, with land in dark grey (UK, left; the Netherlands, right). The map is projected to the Azimuthal Equal Area centred on the mid-point of all the tracking data.

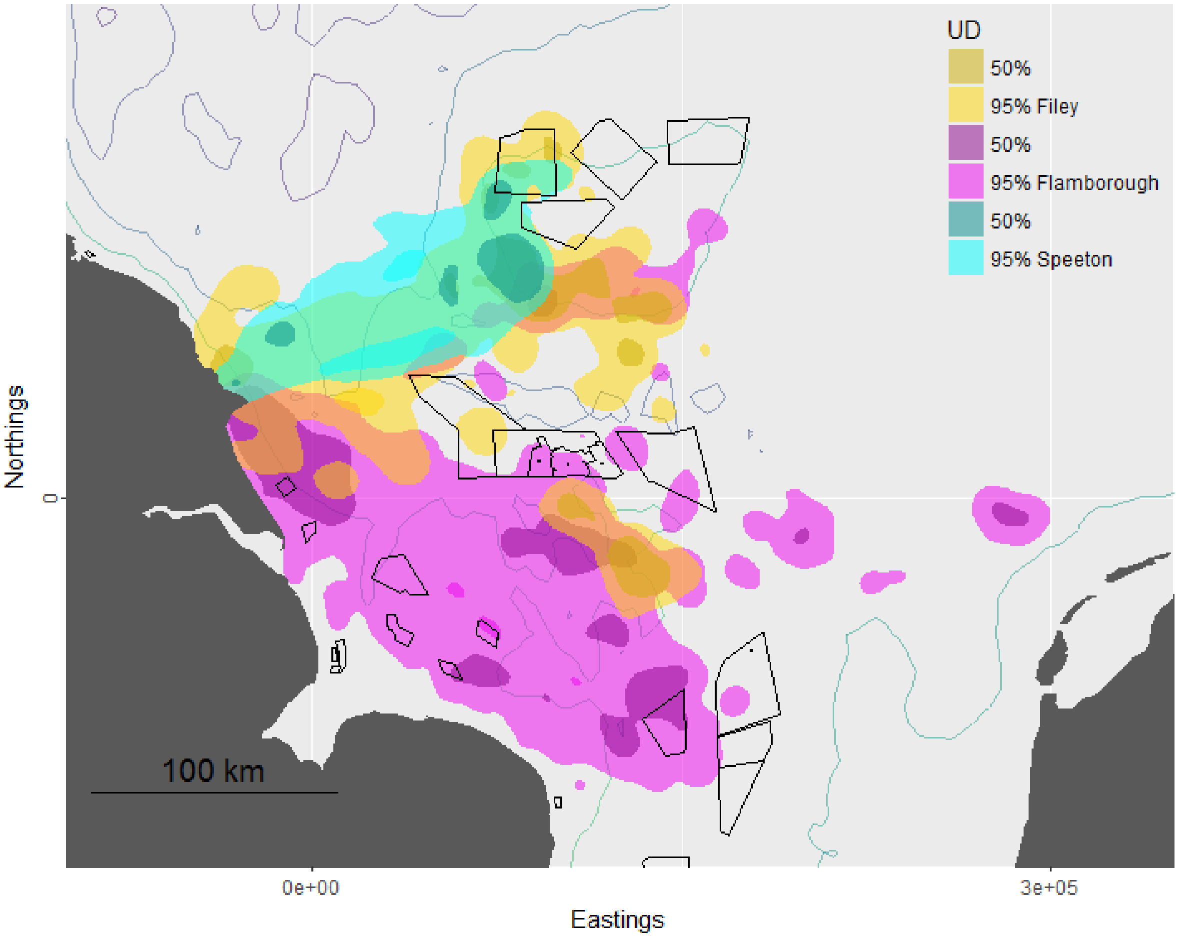

Figure 9. Utilization distributions of all kittiwakes tracked at Flamborough and Filey Coast during the 2017 breeding season by study site. 50 and 95% contours are shown. Pink tones refer to Flamborough (N=133 trips from 13 birds), yellow tones to Filey (N=29 trips from 4 birds) and green tones to Speeton (N= 6 trips from 1 bird). Bathymetric contours, scale bar and outlines of all proposed, planned or active windfarm zones are shown, with land in dark grey (UK, left; the Netherlands, right). The map is projected to the Azimuthal Equal Area centred on the mid-point of all the tracking data.

Table 3. Percentage overlap of Utilisation distributions off kittiwakes tracked at three sites within the Flamborough and Filey coast. Percentage overlap refers to the overlapping section of the total area covered by UDs from both sites. Overlap was calculated for 50% and 95% kernel density contours. Refer to Figure 9 for visualisation.

UD	Flamborough/Filey	Flamborough/Speeton	Filey/Speeton
50%	3.8%	0.2%	15.7%
95%	17.6%	2.9%	25.4%

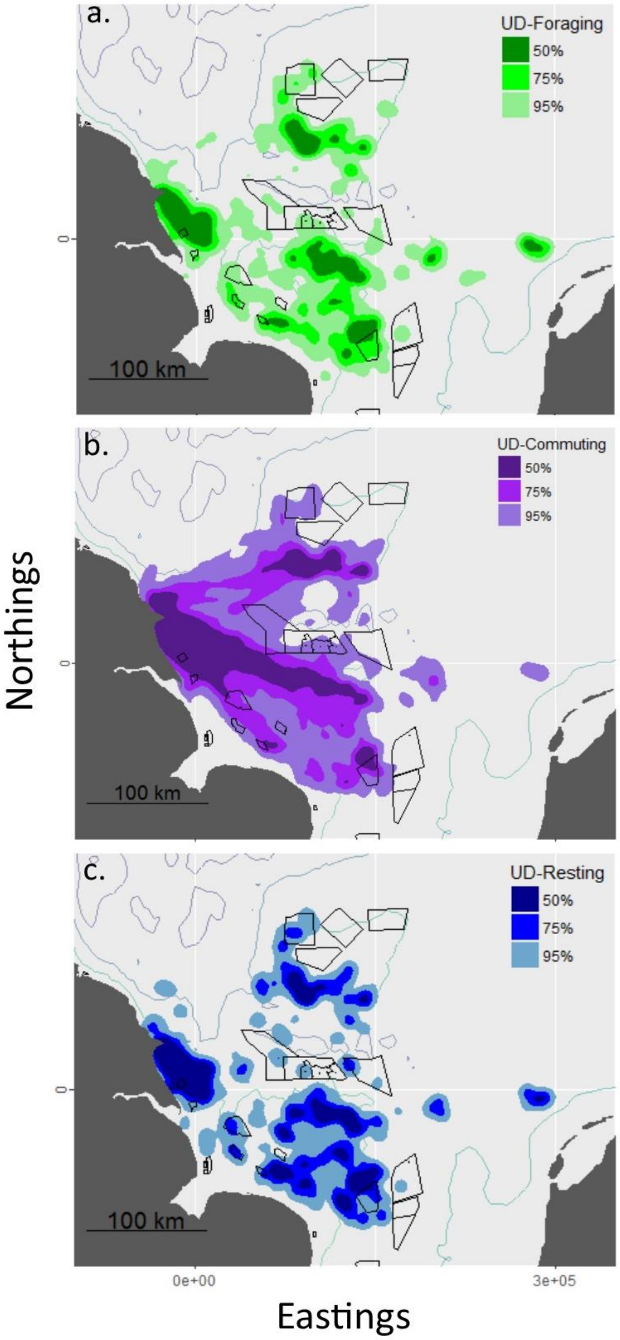

Figure 10. Utilization distributions for each of the behaviour states for all kittiwakes tracked at Flamborough and Filey Coast during the 2017 breeding season. N=168 trips from 18 birds. 50, 75 and 95% utilization distribution contours are shown for a. foraging locations, b. commuting locations and c. resting locations. Behaviour states were annotated using EMbC (see methods). Bathymetric contours, scale bar and outlines of all proposed, planned or active windfarm zones are shown, with land in dark grey (UK, left; the Netherlands, right). The map is projected to the Azimuthal Equal Area centred on the mid-point of all the tracking data. Note the large overlap between the UDs for foraging and resting compared to the commuting UD.

Overlap with Hornsea

The Hornsea project zones showed a certain degree of overlap with the all UDs assessed above, mostly with the 95% contours (Figures 8-10). All Hornsea project footprints overlapped to some degree with the 95% contours of the Flamborough tracking data and all behaviour-specific UDs (Figure 10, Table 4). The Filey UD only overlapped with the footprint of Hornsea projects 2 and 4, and the UD for the single bird successfully tracked at Speeton did not overlap with any of the Hornsea zones (Figure 9, Table 4).

Table 4. Percentage overlap between the Hornsea zones and the 95% utilisation distributions of Kittiwakes tracked on the Flamborough and Filey Coast in 2017. Overlap was assessed for the whole Hornsea zone (full zone footprint without buffer) and individual sub-zones (lease areas) as well as for the overall distribution and UDs per site and behaviour. a. shows the overlap as percentage of the UD, whereas b. shows the overlap as a percentage of the individual Hornsea zone assessed.

a.	**Hornsea I**	**Hornsea II**	**Hornsea**	**Hornsea IV**	**All**
Project area	407	462	696	846	2412
Site					
Flamboroug	1.1%	0.3%	0.9%	0.3%	2.7%
Filey	0%	0.3%	0%	2.3%	2.5%
Speeton	0%	0%	0%	0%	0%
Behaviour					
Foraging	0.8%	0.3%	0.9%	0.8%	2.7%
Commuting	1.1%	0.4%	0.2%	1.9%	3.6%
Resting	0.5%	0.4%	1.0%	0.9%	2.7%
All	0.8%	0.3%	0.7%	1%	2.7%

b.	**Hornsea I**	**Hornsea**	**Hornsea**	**Hornsea IV**	**All**
Project area (km^2)	407	462	696	846	2412
Site					
Flamborough	64.0%	15.5%	31.0%	7.9%	25.5%
Filey	0%	9.7%	0%	39.8%	15.8%
Speeton	0%	0%	0%	0%	0%
Behaviour					
Foraging	44.5%	14.1%	27.8%	20.7%	25.5%
Commuting	81.5%	26.2%	11.0%	71.4%	47%
Resting	24.7%	16.3%	30.1%	22.7%	23.9%
All	56.5%	16.9%	28.3%	33.4%	32.7%

Productivity and tagging effects

Overall 91% of all monitored nests (n=333) hatched at least one chick. The failure rate thereafter for all non-tagged nests was high with 58 % for Filey and 33 % for Flamborough, which were statistically different from each other (χ^2= 13.024, df = 1, p-value < 0.001). Additionally, total failure rates for control nests were significantly lower than for tagged nests (75%, χ^2= 8.5063, df = 1, p-value< 0.001) suggesting an effect of tagging. However, this effect was lost or weakened, dependent on site (Flamborough or Filey), when fitting a more complex binomial model, which also accounts for nest height (Figure 11). The height of tagged nests was approximately 1 meter lower than the average for controls (LM: -1.1672 ± SE 0.4194, t= -2.783, p= 0.006).

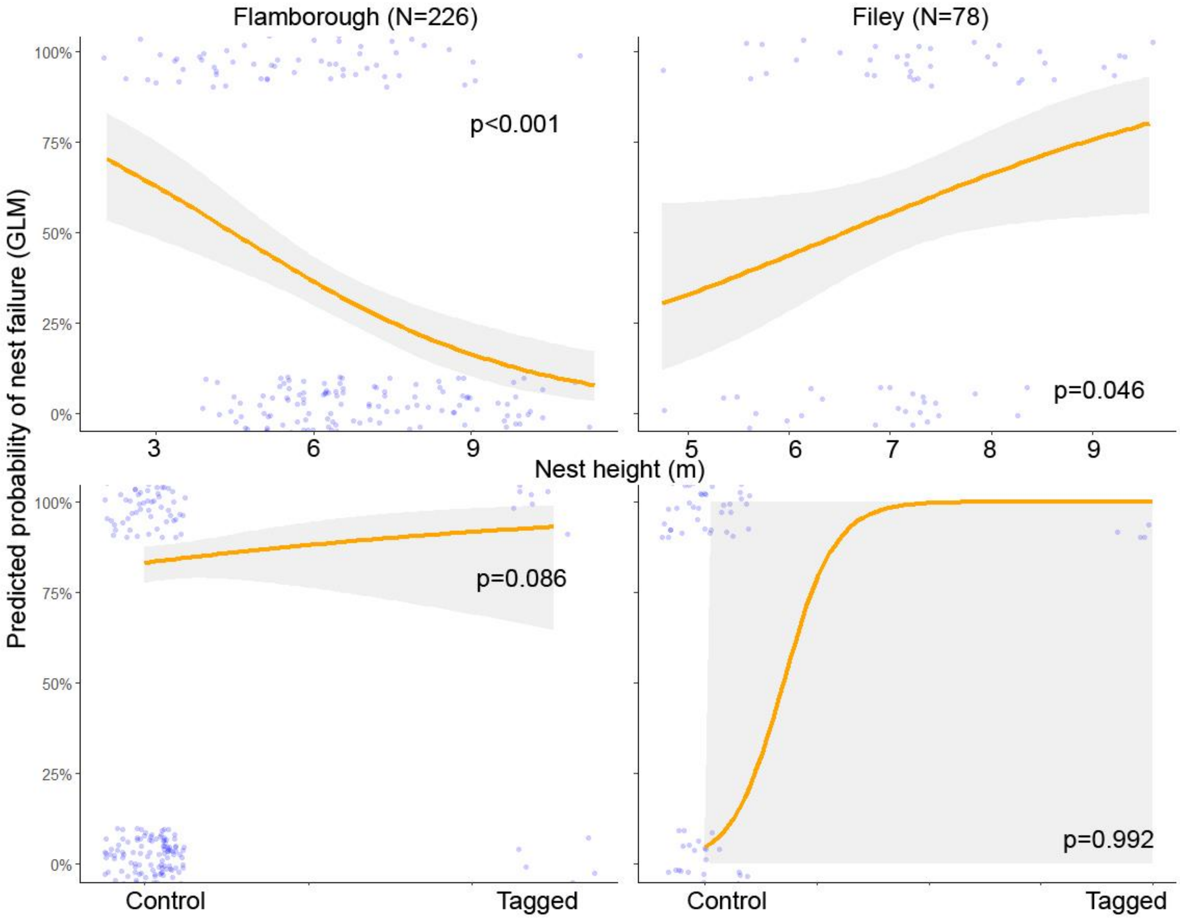

Figure 11. Nest failure probability for each of the monitored sites dependent on nest height and tagging. Raw data (with jitter[3]), model fit, sample sizes and confidence intervals are shown as are p-values for each model parameter.

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[3] In this case "jitter" refers to the introduction of visual spread across the y-axis for the two top graphs and across both axes for the bottom ones. The aim is to improve the visualisation of data by making individual data points visible that would otherwise be plotted on top of each other. Here all the raw data on the y-axis refers to either 0 or 100% and x-axis data on the bottom two plot refers to either control or tagged nests.

Discussion

We successfully tracked 18 Kittiwakes in the Flamborough and Filey Coast pSPA to establish baseline data in reference to the planned Hornsea offshore developments. Two tags, one deployed at Filey and one at Speeton, did not provide any data, presumably due to technical problems with the tags following deployment, and perhaps due to tag damage caused by the birds. Both tags were tested prior to deployment and both birds were seen in the colony and well within range of the relays with their tags attached following deployment, so should have produced data. We piloted the use of the UvA BiTs tracking system at Flamborough, Filey and Speeton, and found that it generally worked well and was suitable for use at these sites. However, since the system is not usually used on cliff nesting seabirds that spend a large amount of time in shaded cliff areas we had initial trouble in adjusting settings so that solar panels were able to re-charge batteries before they ran out of power. Consequently, tags were not able to collect data at the very high temporal resolution originally planned (aimed to record GPS fixes every 3-30 seconds when birds were within the wind farm footprint) and, because of problems with battery life, accelerometer sampling this year was limited. Despite these issues, the data provide a unique and comprehensive insight into the foraging movements of breeding kittiwakes throughout chick-rearing, the period of the breeding cycle when seabirds are most constrained in terms of the foraging areas they can use, and thus most vulnerable to changes in their environment (e.g. Orians & Pearson 1979, Thaxter *et al*. 2012).

Overall trip characteristics and at-sea behaviour mirror previous observations from other studies and therefore support current expectations of foraging range and trip duration of kittiwakes during early chick rearing (Coulson 2011, Kotzerka *et al*. 2010), but the average foraging range and trip duration across the tracking period was larger than has been recorded previously. Based on our knowledge of seabird ecology and an initial examination of the tracks, we expect that this larger average foraging range compared to other studies is because we tracked kittiwakes for a longer part of the breeding season including when adults were provisioning large chicks (that can be left for longer than small chicks) or their nests had failed. An important part of our continuing analysis of this dataset will be to analyse trip characteristics separately for birds at different breeding stages (young chicks, older chicks, failed nests) and test for differences between these groups. Further analysis is also required to determine whether the risk of birds interacting with the Hornsea wind farms changes as the breeding season progresses. Speculatively, we think that if birds are behaving differently later in the season this is likely to change the likelihood of them encountering turbines, though we are not yet sure whether this will make a positive or negative difference to collision risk. We will investigate and quantify this difference as part of the further analytical work conducted during this contract (see below for details). The comparison of foraging characteristics and at-sea behaviour between sites suggests that foraging strategies are similar for birds from all breeding sites. However, comparing the key areas used by birds from different breeding sites showed that birds from Filey birds and the one bird tracked in Speeton tended to forage in similar areas to each other north-east of the colony, while Flamborough birds tended to go south-east. However, some individual birds from Filey and Flamborough did not follow this pattern. Similar foraging segregation has been observed previously in kittiwakes (Ainley *et al*. 2003, Paredes *et al*. 2012, Wakefield *et al*. 2017) and in other seabird species (e.g. Wakefield *et al*. 2013, 2017), but usually on a colony scale. Seeing such a pattern when comparing birds at either end of one colony emphasizes the need to include future tracking work on birds in the centre of the colony (in this case in Speeton or around Bempton) to ensure results are representative, particularly when informing post-consent monitoring or environmental assessments.

The analysis of the overlap of the key areas used by tracked kittiwakes (utilization distributions) with the Hornsea zones shows that all of the individual Hornsea project footprints sit outside the 50% utilization distributions of kittiwakes tracked during 2017. However there is an overlap of the 90% and 95% utilization distribution with all Hornsea zones and of the 75% one with only Hornsea III for all data combined. Whereas for the behaviour specific distributions for which 50, 75 and 95% contours were shown, only the 95% foraging contour overlaps with the overall Hornsea footprint, the 75% distribution of commuting and resting birds overlaps with small areas at the edges of the footprints of Hornsea Project 3 and 4. For the site specific analysis for which the core 50% and 95% percent contours were described, again only the 95% utilization distributions show an overlap for Flamborough and Filey birds. While this overlap for Flamborough birds covers varying degrees of all Hornsea project zones (Table 4), the overlap from Filey birds concentrates on Hornsea Project 2 (~10% coverage) and 3 (~40% coverage). It is important to note three aspects of the overlap of kittiwake area usage with the wind farm footprints:

1. Overlap approximately doubled when looking at the distribution of commuting behaviour compared to other behaviours or all data combined. This implies that although birds might not spend much time foraging or resting in the Hornsea zones they tend to pass through these areas frequently when commuting, which still puts them at risk and underlines the importance of collecting direct behaviour and flight height information from within the wind farm zones, as this project aims to do.

2. The spatial segregation outlined above shows that Filey birds on average tend to go north of the wind farm development zones while Flamborough birds tend to go south of them, which might not be representative for the overall pSPA colony. To exclude the possibility that the largest number of birds directly in the centre of the colony head out straight east into the Hornsea zone, further tracking work at the Speeton site (and potentially directly at Bempton) is needed.
3. The analysis is based on one year of data only and previous seabird tracking studies (including previous tracking conducted on kittiwakes breeding within the pSPA) has shown inter-annual variability in the birds' at-sea distribution and the location of core foraging areas.

The tagging approach successfully extended the deployment period compared to conventional short-term attachment methods. However, we are uncertain whether tags affected the likelihood of nest failure due to an overall low breeding success and confounding variables. A simple comparison of tagged and control nests (using a chi squared test) indicated a significant tag effect on nest failure rates, but more complex models that accounted for confounding factors such as nest height and site did not detect any significant effect of tagging. This highlights the importance of treatment and control nest choice in tagging studies and demonstrates that limited accessibility for tagging studies can lead to biased data. The attachment method therefore needs further testing before being recommended, particularly in direct comparison with more conventional methods such as short-term Tesa tape attachment. However, the trips collected after nest failure and later in the breeding season are extremely valuable since they represent data that cannot be collected using other deployment methods. Further analyses investigating how utilization distributions change during the chick rearing period and after nests fail is therefore warranted.

This year's pilot study successfully provides baseline pre-construction GPS tracking and distribution data from kittiwakes across the chick rearing period, and the first site specific analysis of behaviour budgets derived from behaviour annotation. Although the collection of directly measured behavioural data from accelerometers was limited this year, this first year of tracking enabled us to successfully trial the use of the UvA tracking system on kittiwakes in the Flamborough and Filey Coast pSPA and identified any site/species specific technological limitations. These were already successfully addressed to ensure the collection of GPS and behavioural base line data (derived from accelerometers and altimeters) in upcoming years, to eventually compare all data pre-, during and post-construction.

Recommendations for future work

Recommended data analysis

This report presents a first overview of the data that were collected during 2017 but is not intended to be an in-depth analysis. Numerous further analyses will be conducted during the ongoing contract between RSPB and Ørsted (Saskia is employed to work on this until early February 2018). These will be delivered either as submitted papers/short notes to peer-reviewed journals (Ørsted and the Flamborough and Filey Coast seabird monitoring group will have an opportunity to comment on drafts), or 1-2 page short reports to Ørsted that could be formatted as Annexes to this report (particularly if there is a desire to have all the outputs of this work in one place).

Further analytical ideas have emerged during the course of this work that would be highly beneficial for the strategic monitoring and environmental assessment of the various Hornsea projects. These would require additional staff time beyond that available in this contract, but no additional data collection, and as such would be relatively cheap. The most efficient way to do at least one to two of these additional analyses would be to extend Saskia's current contract by three to four months (all three analyses finalised and submitted as papers will take an additional 9 months), as she is already familiar with the dataset, so would do the work in less time than someone who has not worked with these data. This would also allow us to keep her employed during the few months between February and May 2018, ready to lead next year's fieldwork (assuming further tracking work goes ahead), which would give the project valuable continuity and be more efficient than training a new person to lead future fieldwork. If this is not possible, we recommend that these analyses could be included as part of the future package of strategic monitoring work that follows this contract. Fully costed proposals for the additional suggested analyses can be provided if they are of interest to Ørsted.

We have itemised our recommendations for future analytical work in two lists (one for the work included in the current contract, and one for the work that would require additional resource) below.

Analytical work included in the current contract

1. An investigation of the flight speed distribution of trips using and comparing speed estimates derived from GPS data. The main objective is to develop (spatially explicit) flight speed distributions that can help improve future collision risk models.

2. A paper on changes in the birds' trip characteristics and foraging distribution across the chick-rearing period, and comparison of trip characteristics between birds with active and failed nests, using some of the first tracking data available for kittiwakes whose nests have failed. This will be a crucial addition to the understanding of the temporal variation in at-sea distribution of seabirds in general, and how it is affected by breeding stage. Specifically, it will enable us to understand how kittiwake collision risk varies in relation to breeding stage; this has never previously been measured at any colony, or in relation to any wind farm.
3. A short note submitted to a peer-reviewed journal summarising our findings regarding the impact of nest height on breeding success. This will inform future tagging studies on the risks of introducing a bias by choosing nest sites based on accessibility. This is important to explain the apparent tag effect on nest failure rates observed in our study (which is largely explained by nest height) and will be vital to justify, and get a licence for, future tagging studies using the same methodology.

Analytical work that would require funding for additional staff time

4. A comparison of this year's tracking data with RSPB's tracking data from previous years to investigate annual variability of trip characteristics and at-sea distribution in relation to environmental parameters. Multi-year tracking data for kittiwakes is rare and our extensive dataset would not only allow us to look at the overall distribution across years, but also help us to understand what drives differences in annual distributions. This will help to estimate how kittiwake collision risk might vary between years.
5. An assessment of individual repeatability of kittiwake foraging trip characteristics and sites within one chick rearing period aiming to determine how degrees of site faithfulness affect breeding success. This will be the first study of its kind and will help us to predict the birds' ability to adjust to changes to their environments, such as new windfarms, by adapting their behaviour. This analysis could be extended using data from German collaborators who coincidentally performed longer-term tracking on chick rearing kittiwakes from Helgoland during 2017 and are willing to work with us. Using the additional data from Helgoland would allow us to determine whether site fidelity, and therefore the ability to adapt behaviour in response to new developments, differs between colonies.
6. An investigation of the overlap between the Hornsea development and the flight paths of prospecting juvenile gannets. Juvenile gannets have been tagged for the first time at three major sites in the North and Irish Sea (Grassholm, Bass Rock and Helgoland) by a

potential collaborator at the University of Glasgow who is willing to share data. Birds from all sites visited the Filey and Flamborough pSPA colony on their journey. This provides a unique opportunity to address a major gap in environmental assessments and to go beyond at-sea surveys and tracking data from breeding adults, enabling us to understand risks to juvenile non-breeders for the first time.

Recommended continuation of strategic seabird research

To address remaining gaps in our understanding of the at-sea distribution and behaviour of the two seabird species predicted to be most vulnerable to collision with the planned Hornsea wind farm developments, we recommend a further year of tracking work on both kittiwakes and gannets, as follows:

1. Another 20 kittiwakes should be tracked using an improved version of the tags used this season. These tags will include altimeters, potentially a slightly larger battery and the option to set up three rather than two geographically defined sampling regimes which will address problems with battery life and enable us to choose different sampling regimes within the colony, outside the colony and within the windfarm area. We have discussed these modifications with the tag developer, who will attempt to incorporate all of them alongside the addition of altimeters, subject to the altimeter development work being funded (as itemised in the current contract). Tracking data will therefore provide the most accurate flight altitude measurements for the species to date, more detailed behavioural data via accelerometers, and another opportunity to assess tag effects (which is important given the marginal tagging effect observed this year). We recommend the study should focus on Flamborough Head and the new site at Speeton to collect valuable tracking data of more birds in the centre of the colony, and to exclude Filey from the tracking work due to a high nest failure rate at this site, which limits data collection.
2. Twenty chick-rearing gannets should be tracked during 2018, which will provide the first high temporal resolution tracking data at the Bempton colony (previous gannet tracking at this site has used satellite tags that record positions less frequently). This baseline data collection will be vital if we are to compare gannet behaviour before and after wind farm construction in order to improve understanding of avoidance behaviour. Given construction timescales, 2018 is the only opportunity to collect this important data. This work could be performed using a larger version of the tags described above with more solar panels and a larger battery, due to the high body weight of gannets, which would allow more frequent GPS locations and accelerometer samples than we are able to collect

for kittiwake. Medium-term attachment of tags is easily achieved on gannets compared to kittiwakes by attaching them to the tail feathers, a proven method that has previously enabled tracking data collection over many weeks. As above, tags would feature an altimeter, accelerometer and the same option for sampling regimes, so the proposed tracking work would produce high resolution behavioural and altitude data for gannets across the whole chick-rearing period. This would provide important data on key parameters for collision risk models, including flight speed, flight height, and proportion of time spent (a) in flight, and (b) in the wind farm footprint, and would also provide the baseline against which to compare flight behaviour once turbines are constructed.

3. We must conduct re-sighting work to compare return rates of colour-ringed control and tagged kittiwakes to the colony in early-spring 2018. This will ensure that we can get approval for a licence to use the same tagging method for kittiwakes in future, and is a relatively small cost, particularly if done alongside other work.
4. Set up a project to colour ring kittiwakes to monitor survival rates as already suggested in this year's proposal. Although we did colour ring tracked individuals and a control sample close to the study site to estimate tagging impact on survival rate, we further suggest to set-up a larger scale project next year at an accessible site where no tagging will be conducted. Prospecting for potential sites during this field season highlighted other sites at Flamborough Head but also Flamborough North Landing as suitable choices which are accessible but not ideal for tagging work using relays/remote download tags. By colour ringing an initial sample of 200 birds at those sites and continuing to colour ring and re-sight birds over numerous years we could estimate adult survival rates between years for the colony, which will be an important component of population models assessing colony wide impact of local offshore developments.
5. In the longer-term, getting the best value from the data already collected will require further during- and post-construction tracking and monitoring work. We recommend tracking both species for at least one year during construction, two years immediately post-construction, and then at three-year intervals for the remainder of the post-consent monitoring period to determine whether habituation to the turbines occurs. We recommend that the long-term plan is discussed and agreed with the Flamborough and Filey Coast Seabird Monitoring Group, and that specific methodological details are refined throughout the course of the work, as rapid advances in tracking technology, analytical methods and other developments may bring improved, more efficient or more cost-effective methods to answer key questions in future years.

The recommended kittiwake and gannet tracking work during 2018 would provide more than double the amount of data than in 2017, including one additional species, but costs are likely to be similar, because this year's base-stations, antennas and other field equipment can be reused. Tracking both species would require a maximum of three additional relays, one additional month of salary for a senior research assistant compared to tracking only one species, and tag costs. We are able to provide a detailed, fully-costed research proposal on request.

We highly recommend that the development of a GPS tag with inbuilt barometric altimeter, as itemised in the current contract, is approved by Ørsted as a matter of priority, and by the end of October 2017 at the very latest, so that this work can be completed ahead of the 2018 breeding season. We can also confirm that the previously agreed colony monitoring (already in the current contract and with no further confirmation required) will go ahead next year.

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Appendix I.

Tagging studies sometimes omit the first trip conducted by an individual bird after tag attachment to exclude movements that are likely caused by stress from handling and are not representative of normal foraging behaviour. This is critical in species that are easily stressed and/or temporarily abandon their nest after tagging. In this case, tagged Kittiwakes returned to the nest within minutes and spent an average of 8 hours (N= 18, SD= ±9 hours, Range= 1-20 hours) at the nest or within 1km of the colony before engaging in the first foraging trip (longer than 1 hour). Therefore, we expect that birds returned to their chick rearing routine and direct stress from capture and handling does no longer influence the bird's foraging movements at sea. Consequently, it should not be necessary to exclude the first foraging trip from the sample.

To test this hypothesis, we compared the trip metrics of the first with the second trip conducted by an individual using general linear mixed models also including cofounding predictors that are likely to cause a change in trip metrics. Models included trip number (binomial), a factor indicating if the nest failed before or during the trip (binomial), day after hatching and Julian day as predictors and allowed the intercept to vary between individuals.

Although trip metrics were significantly different when nests failed before or during the second trip (all trip metrics increased), there was no difference in any of the metrics between the first and the second trip (Table), which supports our hypothesis. Therefore, we did not exclude the first foraging trip after deployment from our sample.

Table. Model outputs for the three different general linear mixed model run to assess differences in trip metrics between the first and second trip conducted after tagging. Models further allowed the intercept to vary by individual. Note that “Trip number” and “Failed” were both entered as binomial factors and estimates therefore predict change in intercept from the first to the second trip rather than changes in slope. The one tagged bird from Speeton was not included in the study since the nest was not monitored and co-founding predictors were therefore not available. N=34 from 17 birds.

Model 1:	**Predictor**	**Estimate ±SE**	**DF**	**F**	**Probability**
Duration	Trip number (binomial factor)	6.15±4.30	19	2.039	0.170
	Failed (binomial factor)	72.87±13.05	29	31.200	<0.001***
	Day after hatching	0.04±0.62	15	0.004	0.953
	Julian day	-0.02±1.32	16	0.027	0.873

Model 2:	**Predictor**	**Estimate ±SE**	**DF**	**F**	**Probability**
Range	Trip number (binomial factor)	14.61±23.55	19	0.385	0.543
	Failed (binomial factor)	238.76±71.91	28	11.026	0.002**
	Day after hatching	1.74±3.45	14	0.253	0.623
	Julian day	3.094±7.35	16	0.177	0.679

Model 3:	**Predictor**	**Estimate ±SE**	**DF**	**F**	**Probability**
Distance	Trip number (binomial factor)	65.08±61.41	19	1.123	0.302
	Failed (binomial factor)	725.61±188.74	28	14.780	<0.001***
	Day after hatching	3.22±9.18	14	0.123	0.731
	Julian day	4.10±19.54	16	0.044	0.836