From:	James Dawkins
To:	Hornsea Project Three
Subject:	RSPB submissions for Deadline 4
Date:	15 January 2019 21:17:01
Attachments:	image001.png image002.png RSPB submission - Deadline 4.pdf Appendix_A_Aitken_et_al_2017.pdf Appendix_B_Bowgen_&_Cook_2018.pdf Appendix_C_Cook_&_Robinson_2017.pdf Appendix_D_Ferrer_et_al_2012.pdf Appendix_E_Garthe_et_al_2004.pdf Appendix_F_Green_et_al_2016.pdf

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I attach the RSPB's submission plus appendices A to F.

Appendices G to N will follow via a separate email.

Please could you acknowledge safe receipt?

Kind regards,
James

James Dawkins
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Flamborough and Filey Coast pSPA Seabird Monitoring Programme

2017 Report

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

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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

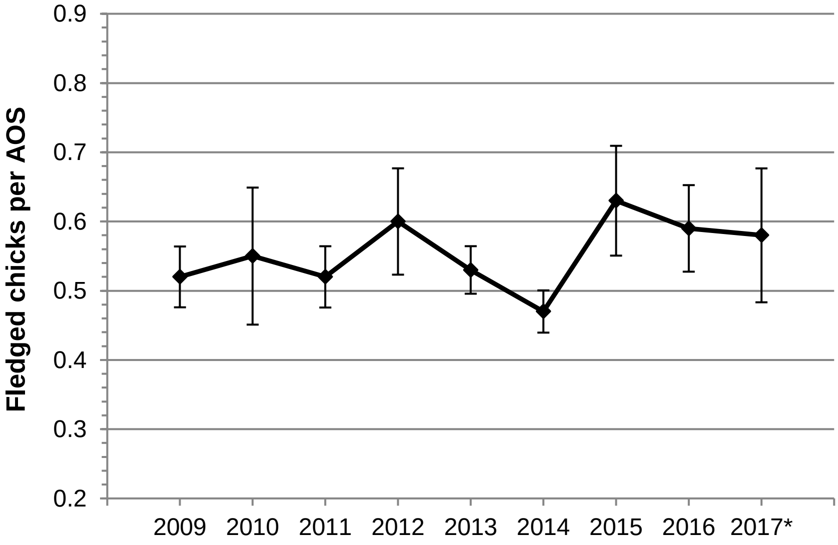

Fig. 1: Northern Fulmar productivity 2009 – 2017, mean of plot productivity results plus/minus SE. *Note: two new plots added in 2017.

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**

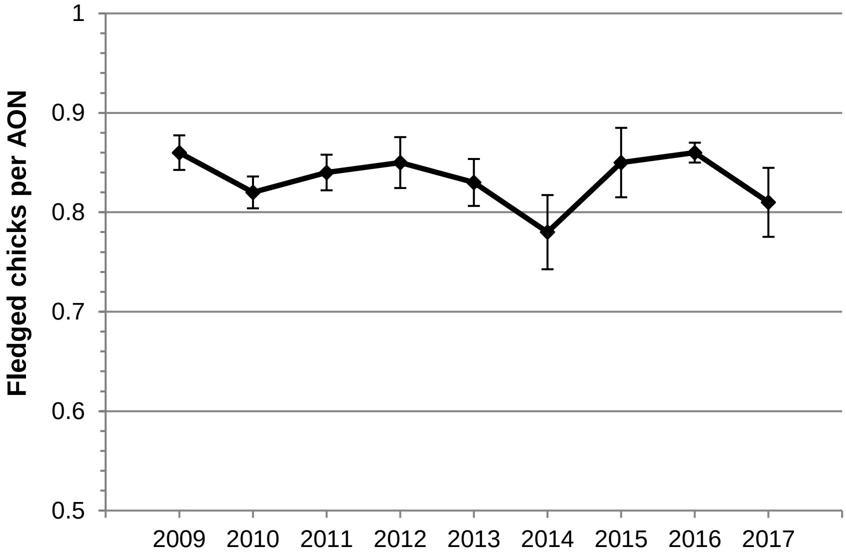

Fig. 2: Northern Gannet productivity 2009 – 2017, mean of plot productivity results plus/minus SE.

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**

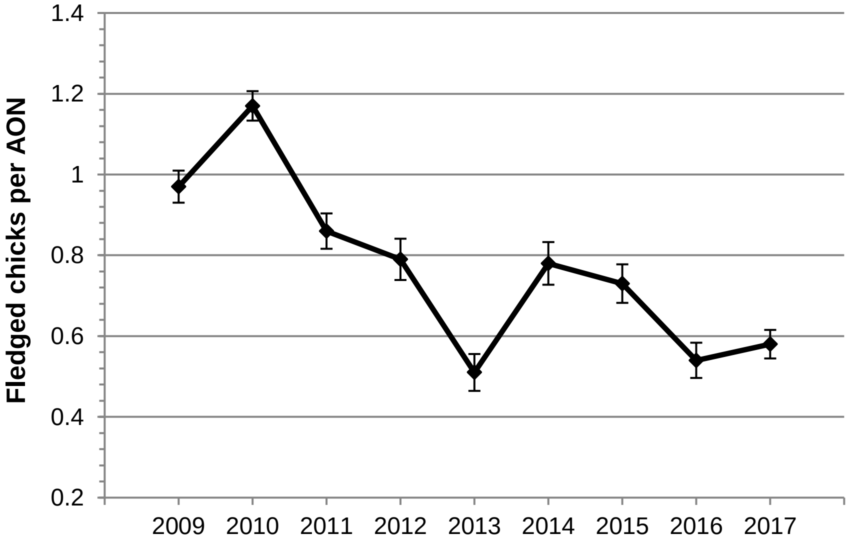

Fig. 3: Flamborough/Bempton Black-legged Kittiwake productivity 2009 – 2017, mean of plot results plus/minus SE.

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**

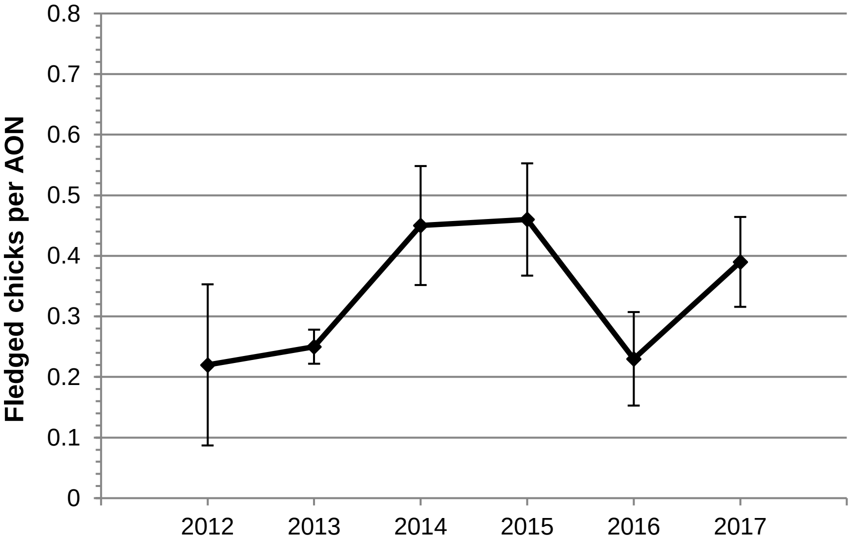

Fig. 4: Filey Black-legged Kittiwake productivity 2012– 2017, mean of plot results plus/minus SE. In 2017 only three plots were monitored with a total sample size of 150 AON. Note that in 2012 productivity on 3 plots was 0.0 due to landslip.

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**

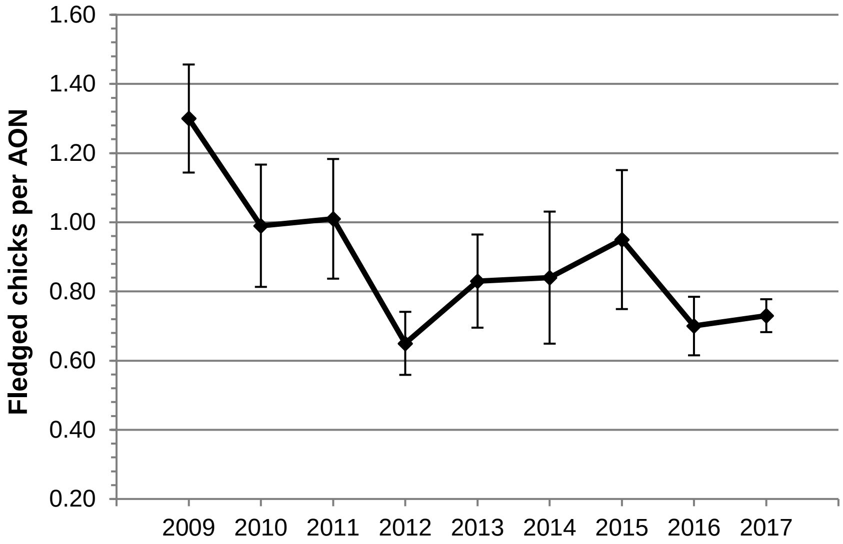

Fig. 5: European Herring Gull productivity 2009 – 2017, mean of plot results plus/minus SE.

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

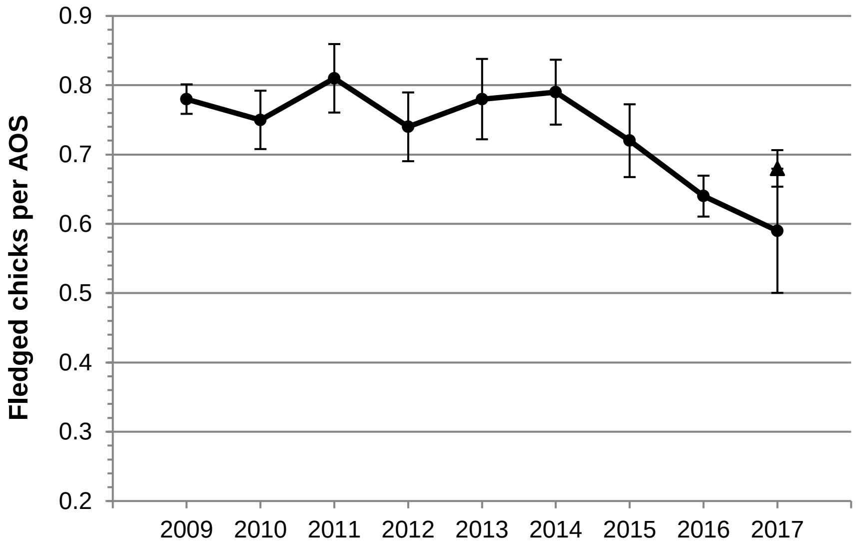

Fig. 6: Common Guillemot productivity 2009 – 2017, mean of plot results plus/minus SE. ▲ = mean of plot results excluding Grandstand North plus/minus SE.

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

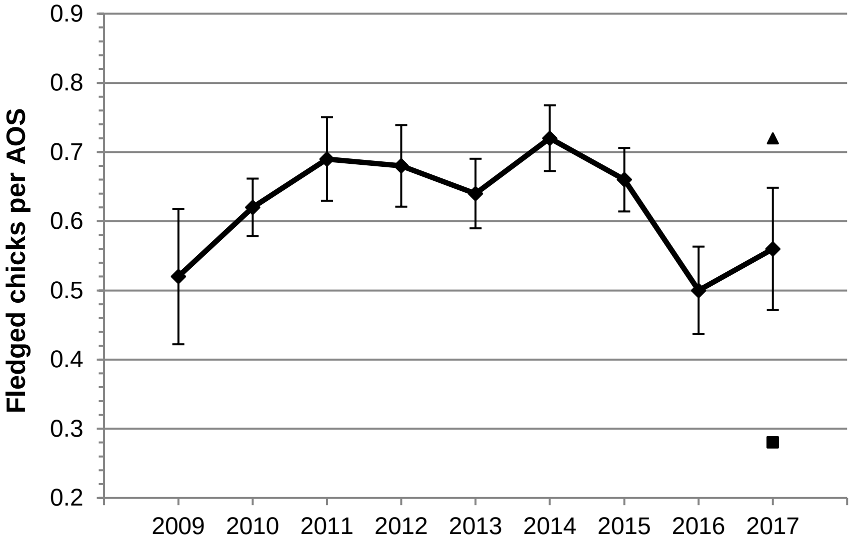

Fig. 7: Razorbill productivity 2009 – 2017, mean of plot results plus/minus SE. ▪ = mean of Grandstand plot results; ▲ = mean of Flamborough plot results.

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.

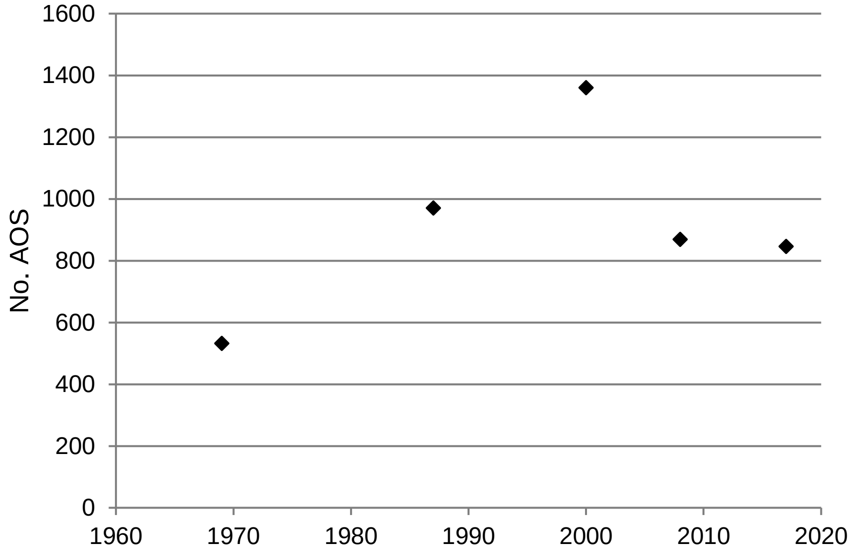

Fig. 8: Trend in Northern Fulmar *Fulmaris glacialis* breeding population within the Flamborough Head and Bempton Cliffs SPA (1969-2017).

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.

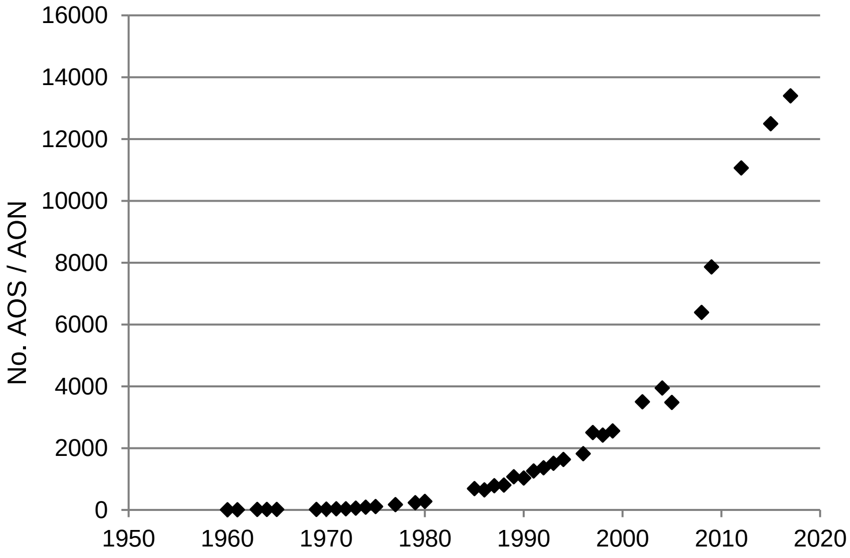

Fig. 9: Trend in Northern Gannet *Morus bassanus* breeding population within the Flamborough Head and Bempton Cliffs SPA (1960-2017).

European Shag *Phalacocorax aristotelis*

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

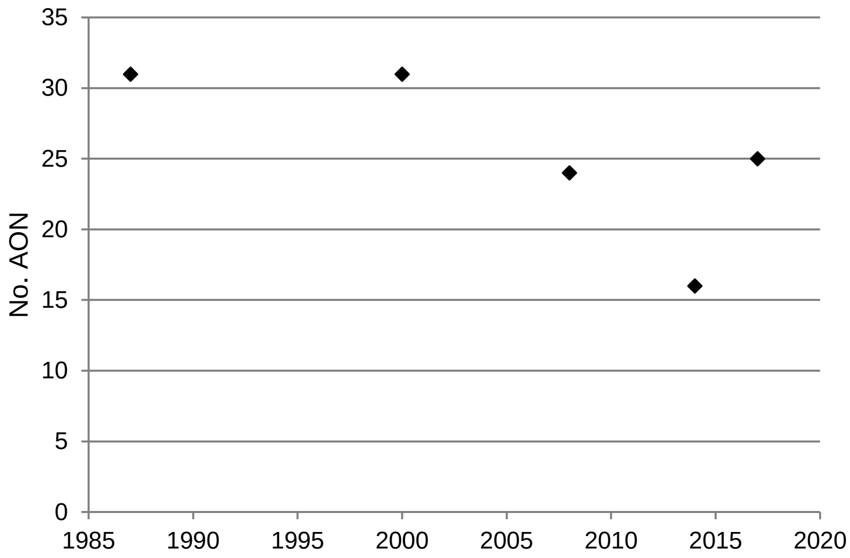

Fig. 10: Trend in European Shag *Phalacrocorax aristotelis* breeding population within the Flamborough Head and Bempton Cliffs SPA (1987-2017).

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.

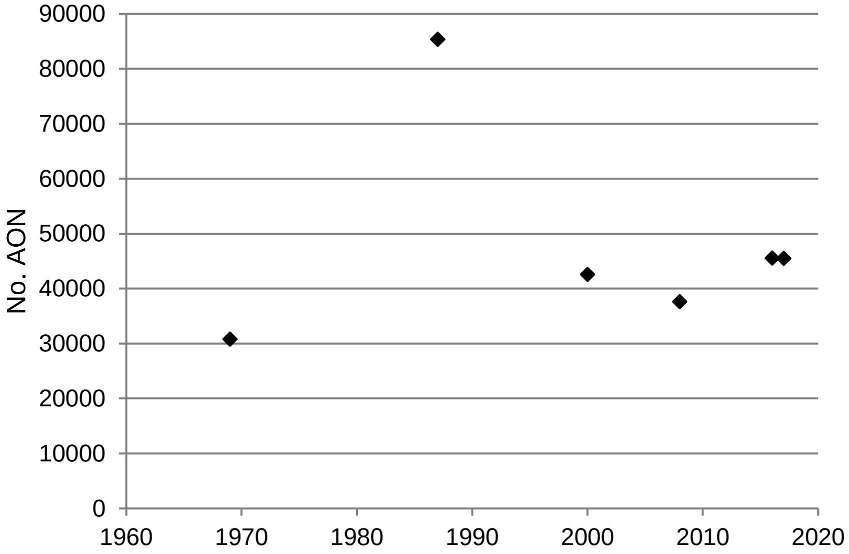

Fig. 11: Trend in Black-legged Kittiwake *Rissa tridactyla* breeding population within the Flamborough Head and Bempton Cliffs SPA (1969-2017).

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.

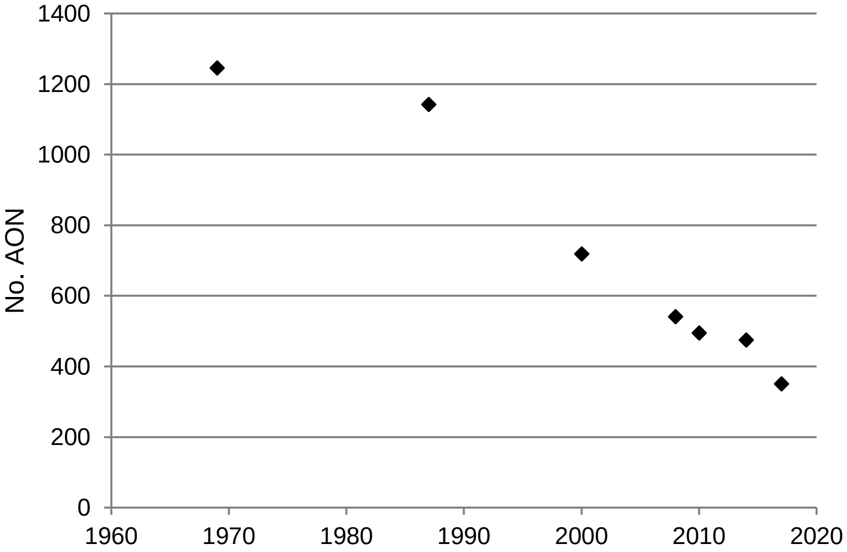

Fig. 12: Trend in European Herring Gull *Larus argentatus* breeding population within the Flamborough Head and Bempton Cliffs SPA (1969-2017).

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.

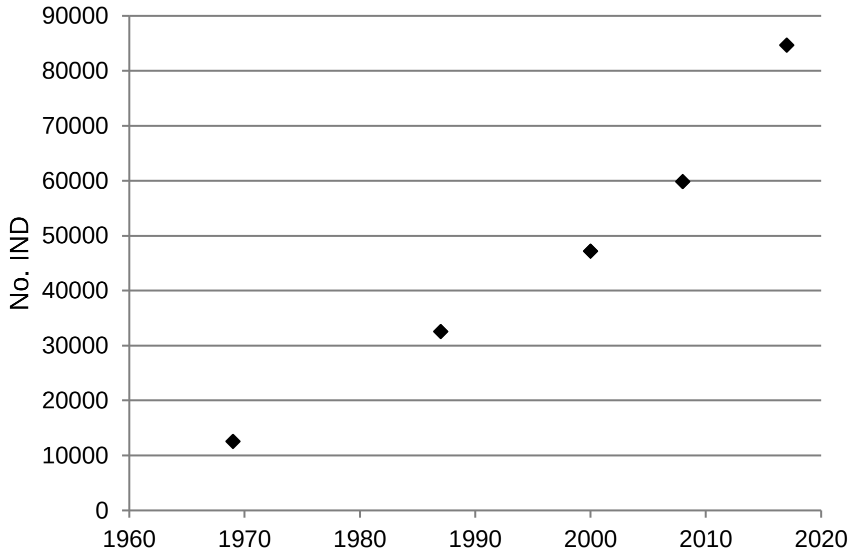

Fig. 13: Trend in Common Guillemot *Uria aalge* breeding population within the Flamborough Head and Bempton Cliffs SPA (1969-2017).

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.

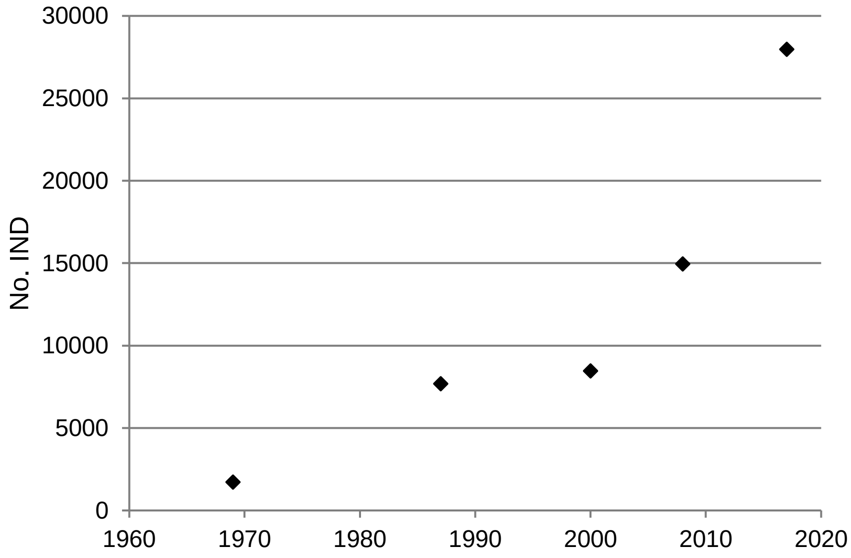

Fig. 14: Trend in Razorbill *Alca torda* breeding population within the Flamborough Head and Bempton Cliffs SPA (1969-2017).

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.

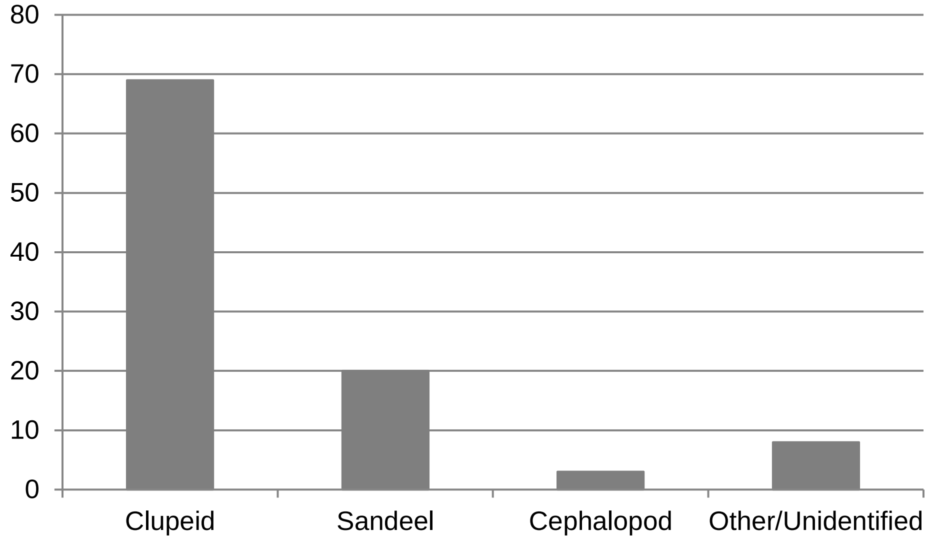

Fig. 15: 2016 Common Guillemot diet composition – percentage of observed prey items at Bempton Cliffs/Flamborough Head. n = 284 prey items.

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.

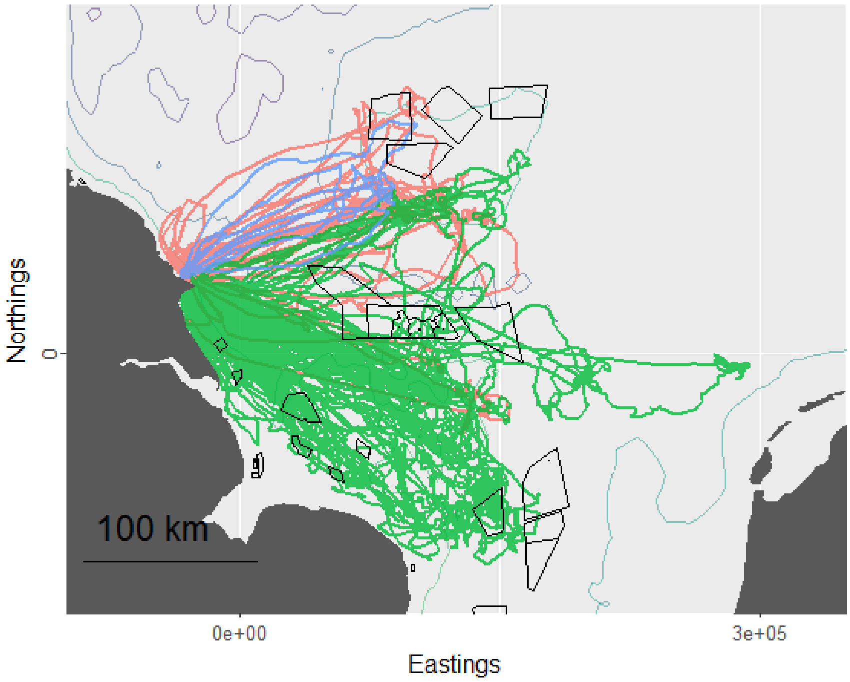

Fig. 16: Kittiwake GPS trips collected during the 2017 breeding season at the Flamborough and Filey Coast pSPA. All trips are shown from Flamborough (Green, N=133 trips from 13 birds), Filey (Red, N=29 trips from 4 birds) and Speeton (Blue, N=6 trips from 1 bird). Bathymetric contours and scale bars 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

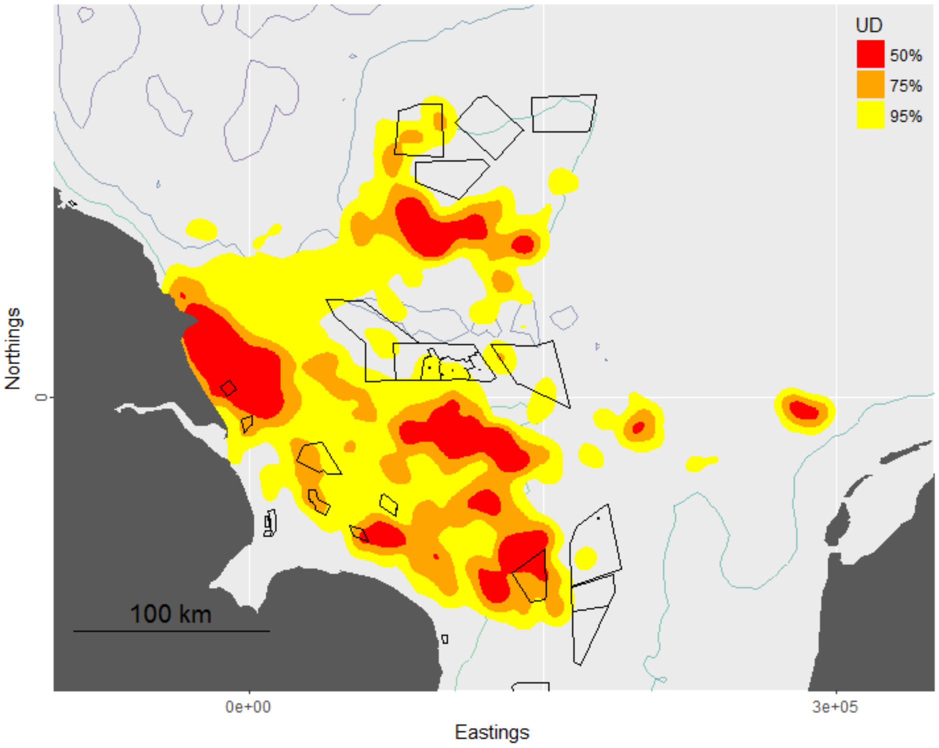

Fig. 17: Utilization distributions of all Kittiwakes tracked at Flamborough and Filey Coast pSPA during the 2017 breeding season. N=168 trips from 18 birds. 50, 75 and 95% contours are shown. Bathymetric contours, scale bar and outlines of all proposed, planned or active windfarm zones are shown, with 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.

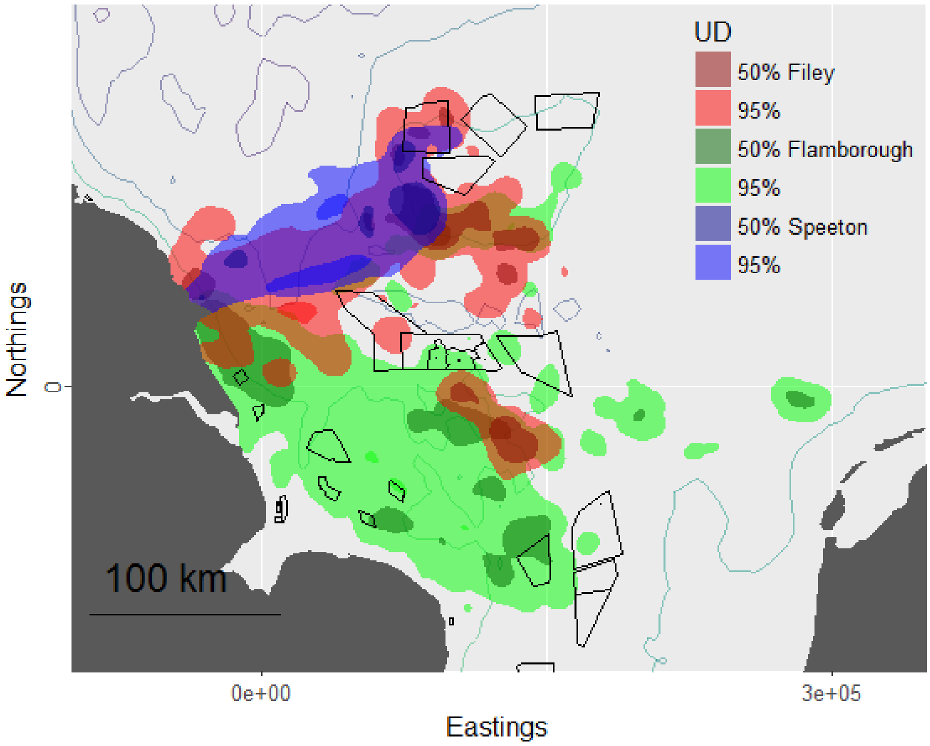

Fig. 18: Utilization distributions of all Kittiwakes tracked at Flamborough and Filey Coast during the 2017 breeding season by study site. 50 and 95% contours are shown. Green tones refer to Flamborough (N=133

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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We would also like to thank:

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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.

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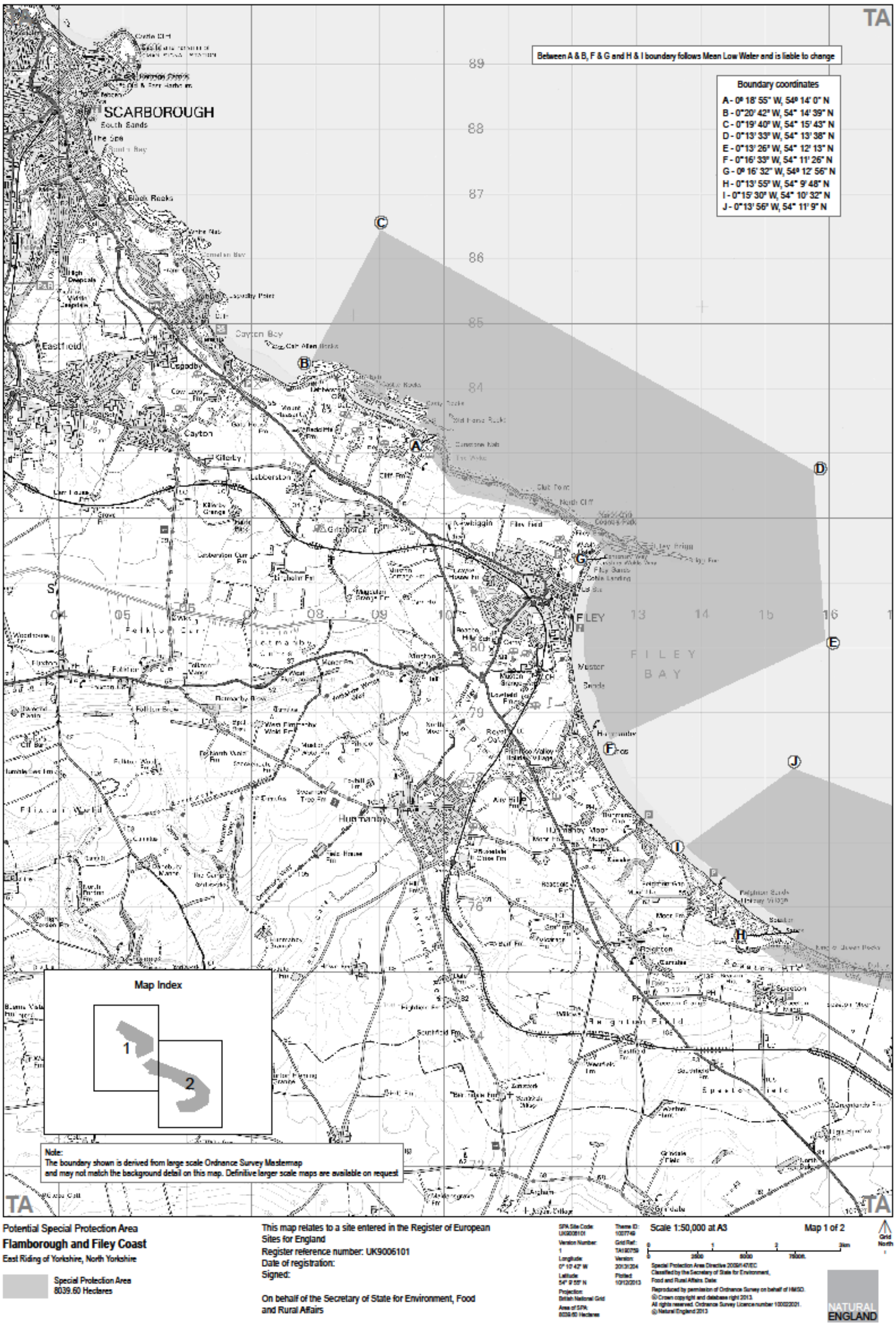
Blue Dolphin Holiday Park at Filey for allowing access to reach important sections of the colony for essential monitoring works.

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


TA
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
Bempton
FLAMBOROUGH HEAD
Flamborough
BRIDLINGTON
Hilderthorpe
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
SPA Site Code: UK9006101
Version Number: 1
Longitude: 0° 10' 42" W
Latitude: 54° 9' 55" N
Projection: British National Grid
Area of SPA: 8039.60 Hectares
Theme ID: 1007749
Grid Ref: TA190759
Version: 20131204
Plotted: 10/12/2013
Scale 1:50,000 at A3
Map 2 of 2
Grid North
Special Protection Area Directive 2009/147/EC
Classified by the Secretary of State for Environment, Food and Rural Affairs. Date:
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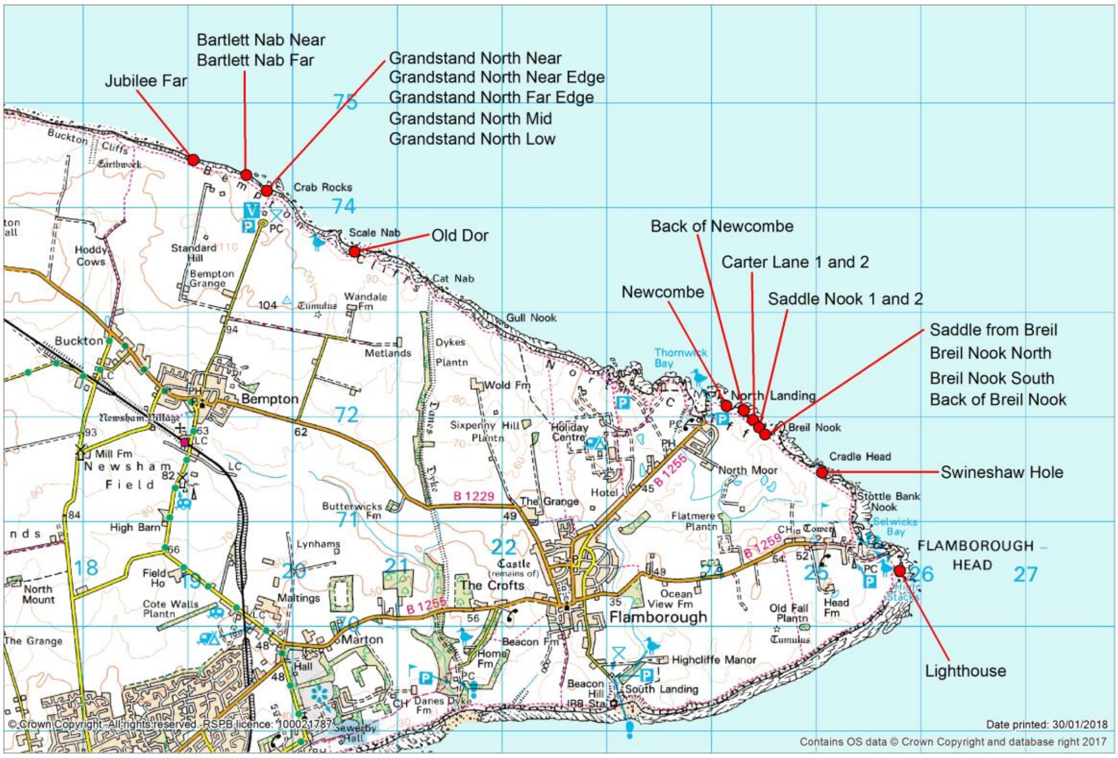
Appendix 2 - Productivity Plot Locations

Northern Fulmar productivity plots

Northern Gannet productivity plots

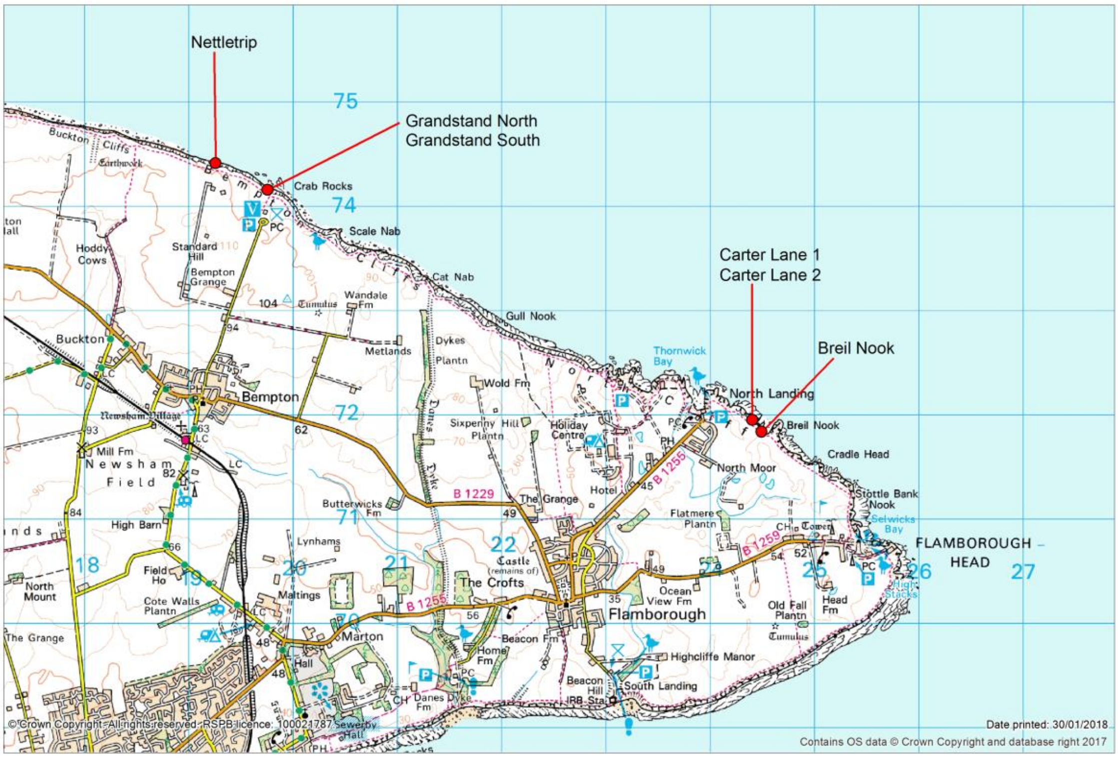
European Herring Gull productivity plots

Black-legged Kittiwake productivity plots – Flamborough and Bempton



Black-legged Kittiwake productivity plots – Filey

Common Guillemot productivity plots



Razorbill productivity plots

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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No: 614

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Bowgen, K. & Cook, A.

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Summary

In response to concerns about the risk of collision between seabirds and offshore wind farms, the Offshore Renewables Joint Industry Programme (ORJIP) funded a study to collect data on seabird collision and avoidance rates at an operational wind farm, referred to as the Bird Collision Avoidance (BCA) study. Over the course of this study, it became clear that the data collected in relation to avoidance behaviour, termed empirical avoidance rates, may not be directly comparable to the avoidance rates as presently used by collision risk models, such as the Band model. The aim of this work is to consider how best to use the data collected as part of the ORJIP BCA study in order to inform pre-construction assessments of collision risk at offshore wind farms.

Our analyses demonstrate how assumptions, both in relation to the model itself and, the data used in the model, can affect predicted collision rates. In particular, assumptions about seabird flight height and speed can have important implications for predicted collision rates. Of concern is the fact that reported seabird flight speeds were significantly lower than those typically used in existing guidance. This is important as flight speed is used by the Band model twice. Firstly, in the calculation of the total number of birds that may pass through a wind farm over a given time period and, secondly to estimate the probability that any individual bird may collide with the turbine blades. Flight speed may be estimated from the data collected as part of the ORJIP BCA study in two ways, either as a point estimate or, as an average of the speed at which the birds move through the wind farm. In order to be consistent with how the Band model is implemented, the point estimate of bird speed should be used to calculate the probability of a bird colliding and the average rate at which it moves through the wind farm should be used to estimate the total number of birds likely to move through the wind farm over a given time period.

As suggested by previous studies, meso-avoidance appears to be a key component of overall avoidance behaviour, with most birds within a wind farm taking avoidance action well away from turbines. Recorded micro-avoidance rates were also high, although based on limited data and future studies should consider how best to maximise records of micro-avoidance behaviour. Significantly, the number of birds crossing the turbine rotor-swept area and colliding appeared higher than the predictions made by the Band collision risk model, although this was based on limited data. Given evidence collected by the ORJIP BCA about birds flying in parallel to turbine blades, consideration should be given to taking this into account as part of calculations for the probability of collision.

As may be expected, the empirical avoidance rates recorded as part of the ORJIP BCA study were higher than those collected previously. In part, this is because the avoidance rates used by the Band collision risk model incorporate elements of error, both in relation to the model itself and, in relation to the input parameters. However, by comparing collision rates recorded by the ORJIP BCA study to those that would have been predicted by the Band model in the absence of avoidance behaviour, we are able to recommend avoidance rates for use in the deterministic Band model of 0.995 for northern gannets and large gulls and 0.990 for black-legged kittiwake in relation to option 1 of the Band model and 0.993 for large gulls and 0.980 for black-legged kittiwake in relation to option 3 of the Band model. We were able to undertake further analyses in order to derive avoidance rates suitable for use in the stochastic collision risk model for black-legged kittiwake of 0.994 (95% CIs 0.976 - 0.998) for option 1 and 0.970 (95% CIs 0.871-0.989) for option 3 and, for large gulls 0.997 (95% CIs 0.992 - 0.999) for option 1 and 0.990 (95% CIs 0.974 - 0.995) for option 3. Note that the median values recommended for use in the stochastic collision risk models differ from the values recommended for use in the deterministic model, this relates to differences in the way in which flight height distributions are incorporated into the models. It should be

noted however that the values recommended for use in the deterministic model are within the 95% confidence intervals of those recommended for use in the stochastic model.

Contents

1 Introduction

Offshore wind farms are seen as a key part of efforts to combat climate change (Snyder & Kaiser 2009). However, there are a number of significant concerns about the potential of these wind farms to have a negative impact on wildlife and biodiversity, particularly in relation to birds (Drewitt & Langston 2006; Gibson *et al.* 2017). Of particular concern is the potential for birds to collide with turbines (Thaxter *et al.* 2017; Furness *et al.* 2013; Garthe & Huppop 2004).

To inform the planning process of the potential impacts of the effects associated with wind farms, detailed Environmental Impact Assessments (EIAs) are required. With respect to birds, a key component of these EIAs is a Collision Risk Model (CRM) which is used to predict the number of individuals of any given species at risk of collision. A variety of different CRMs are available (Tucker 1996; Band 2012) but, at their core, most combine an estimate of the number of birds within a collision window with an estimate of the probability of any individual bird colliding in order to forecast the number of likely collision events (Masden & Cook 2016). These models also require an understanding of bird avoidance behaviour, often referred to as the avoidance rate (Cook *et al.* 2014). Whilst some attempts have been made to measure avoidance behaviour empirically (Krijgsveld *et al.* 2011), more commonly, they have been estimated by comparing the number of recorded collisions with the number predicted prior to construction, in the absence of any avoidance behaviour (Cook *et al.* 2014). Consequently, whilst the avoidance rate is often thought to solely reflect the proportion of birds taking action to avoid collision, in reality it also accounts for uncertainty arising as a result of other factors including weather conditions and model error (Band 2012; Cook *et al.* 2014; Masden 2015). This is of concern as the CRM predictions themselves are known to be highly sensitive to assumptions about avoidance behaviour (Chamberlain *et al.* 2006; Masden 2015). This sensitivity may contribute significant uncertainty into the decision-making process, at significant cost to developers, decision-makers and other stakeholders (Masden *et al.* 2015). Furthermore, whilst no detailed comparisons have been made between predictions from CRMs and observed collision rates, some initial studies suggest that key assumptions, such as a linear relationship between abundance and collision risk, may not be realistic (de Lucas *et al.* 2008; Ferrer *et al.* 2012).

As the size and number of offshore wind farms increases, the probability of estimated collision rates which are of a magnitude likely to have significant population level effects also increases. This poses a challenge for decision-makers who must balance the need to invest in renewable energy, in order to mitigate the impacts of climate change, with the need to minimise deleterious impacts on the environment (Green *et al.* 2016; Gibson *et al.* 2017). Consequently, there is a growing interest in exploring how well estimates from CRMs reflect true collision risk and, the extent of collision avoidance behaviour in vulnerable species. This interest culminated in an Offshore Renewables Joint Industry Programme (ORJIP) funded project on bird collision avoidance at an operational wind farm (Davies *et al.* 2013; Skov *et al.* 2018).

In contrast to previous efforts to estimate avoidance behaviour, the ORJIP Bird Collision Avoidance (BCA) project collected data on empirical estimates of bird behaviour (Skov *et al.* 2018). These estimates of bird behaviour can be used to describe the proportion of birds taking action to avoid collision with turbines. However, as they do not incorporate data describing model error or how birds respond in relation to other factors, for example weather conditions, these behaviour-based avoidance rates will not be directly comparable to those used to date. Consequently, it is important to understand how transferable these rates, termed empirical avoidance rates, are to the existing models.

We aim to assess how these empirical avoidance rates can be used to inform renewable energy development impact assessments and support decision making. We aim to achieve this by taking advantage of the data which have been collected by the ORJIP BCA project describing bird movements within an operational wind farm in fine detail. These data included records of both birds that did not collide and those which did. As we have an estimate of the number of birds which have collided over a given time period, we can use these data both to test how well a CRM performs and to understand how much uncertainty remains in collision estimates once empirical avoidance rates have been accounted for.

The key aims of this project were:

- To consider how best to use the information and outputs from the ORJIP funded BCA project to best assess collision risk at offshore wind farms.
- Consideration of how the flux rate estimated as part of Options 1 & 3 of the Band (2012) model relate to the empirical avoidance rates estimated by the ORJIP BCA study.
- Consideration of error introduced into the avoidance rates used by the Band (2012) model and the extent to which this is unaccounted for once empirical avoidance rates are applied.
- To consider how the information collected as part of the ORJIP BCA study could be used to derive avoidance rates suitable for use in the Band (2012) model.

2 Methodology

Uncertainty is introduced into the collision risk modelling process through the use of summarised data, often collected from unconnected sites, and through simplifications and assumptions in the modelling process. At present, this uncertainty is captured by a correction factor, often referred to as the avoidance rate. However, the relative importance of each of the sources of uncertainty which contribute to the avoidance rate is unclear. In order to determine how applicable avoidance rates, such as that derived from the ORJIP BCA study, are to CRMs, it is important to understand the magnitude of the uncertainty remaining once behaviour, and other measurable factors, have been accounted for.

In this study we assess the results of the ORJIP BCA project, using data both from that project and from other surveys of the Thanet Offshore Wind Farm study site, and consider their application to the Band CRM (Band 2012). Specifically, we aim to compare estimates of the number of collisions expected in the absence of avoidance behaviour, based on pre-construction density estimates of bird abundance and generic data describing bird behaviour, to estimates refined through introduction of site-specific data collected as part of the ORJIP BCA project. We use data describing bird density presented in the post-consent monitoring report for Thanet Offshore Wind Farm (Royal Haskoning 2013), data describing bird behaviour collected by observers using laser rangefinders on turbines G01 and G02 in Thanet Offshore Wind Farm and collisions recorded by cameras mounted on turbines D05 and F04 within the Thanet Offshore Wind Farm (Skov *et al.* 2018).

For the purposes of this analysis, we split the Band CRM into its component parts, as follows, to:

1. identify the area in which to estimate collision risk (study area);
2. estimate the flux rate, i.e. the total number of birds which may pass through the study area over the period of interest (study period);
3. estimate the probability of a bird colliding with a turbine (P_{coll} or $Coll_{int}$);
4. estimate the proportion of birds flying at collision risk height (PCH);
5. combine the data above in order to estimate the total number of expected collisions.

We focus analyses on the five, key species covered by the ORJIP BCA study – northern gannet *Morus bassanus*, black-legged kittiwake *Rissa tridactyla*, lesser black-backed gull *Larus fuscus*, herring gull *Larus argentatus* and great black-backed gull *Larus marinus*.

2.1 Defining area in which to estimate collision risk

Data for this project were collected at the Thanet Offshore Wind Farm. Thanet is located on the east coast of the United Kingdom, in the Southern North Sea. It consists of 100 3 MW turbines (Table 1), covering an area of 35 km^2. However, the data describing collisions were collected from cameras located on the northern edge of the wind farm. Collectively, these cameras were able to observe interactions between birds and eight other turbines (Figure 1). Consequently, we restricted our analyses to the area covered by these cameras (Figure 1).

Table 1. Specification of turbines at Thanet Offshore Wind Farm.

Parameter	Value
Capacity	3 MW
Number of Blades	3
Blade Width	3.5 m
Rotor Diameter	90 m
Rotor Speed	16.1 rpm
Pitch	15˚
Hub Height	70

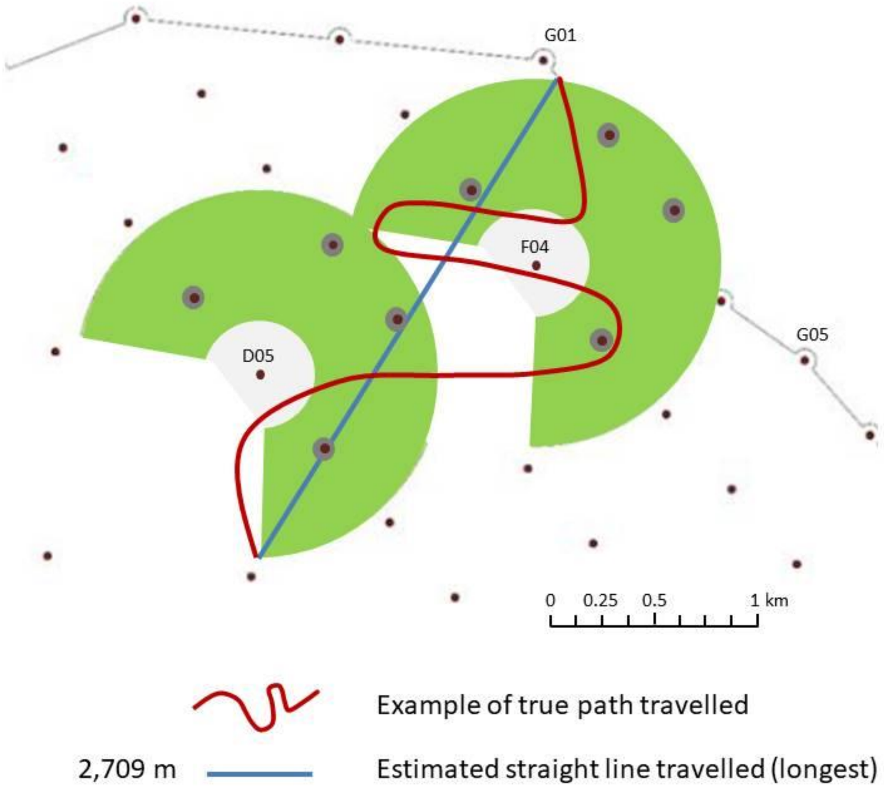


Figure 1. Area covered by two cameras mounted on turbines within the Thanet Offshore Wind Farm (2.983 km^2). Analyses were restricted to the area covered by these cameras, shown in green. Adapted from Figure 4.6 in Skov *et al.* (2018).

2.2 Estimating flux rate

The first step in a CRM is to estimate the flux rate, the total number of birds passing through the study area (figure 1) over the time period of interest. Post-construction density estimates were available only for the period from October – March (Royal Haskoning 2013), consequently, we restricted our analyses to data collected by the ORJIP BCA study in the October-March period. For the purposes of estimating flux, we used the mean of the values for the three post-construction years. The apparent increases in density recorded for lesser black-backed gull, great black-backed gull, black-legged kittiwake and northern gannet between the pre- and post-construction periods must be treated with caution. Pre-construction density estimates are based on a single years' worth of data. The post-construction density data show that there may be substantial annual variation in the estimated density. Monthly surveys from a single year are insufficient to characterise the true usage of a site by the species concerned (Maclean *et al.* 2013) and a recent review has demonstrated that northern gannet in particular shows a strong displacement effect in response to the presence of an offshore wind farm (Dierschke *et al.* 2016). Ideally, density data would have been collected concurrently in relation to the ORJIP BCA study. However, unfortunately this was not possible and the best available density data are those published in the post-construction monitoring report (Royal Haskoning 2013).

Table 2. Density estimates (birds km^{-2}) from within Thanet Offshore Wind Farm (excluding buffer area) used to estimate flux rate for the collision risk model. Taken from table 6 in Royal Haskoning (2013).

	Pre-construction (2004-05)	**Post-construction YR1 (2010-11)**	**Post-construction YR2 (2011-12)**	**Post-construction YR3 (2012-13)**	**Post-construction Mean**
Herring gull	1.95	0.90	0.87	2.30	1.36
Lesser black-backed gull	0.33	0.41	0.62	0.08	0.37
Great black-backed gull	0.02	0.39	1.16	1.53	1.03
Black-legged kittiwake	0.20	1.56	0.92	0.81	1.10
Northern gannet	0.05	0.05	0.17	0.96	0.39

To estimate flux rate, we calculated the total number of birds that would pass through the study area outlined in figure 1 between October and March each year. This followed the methodology set out in Band (2012) combining estimates of bird density with estimates of flight speed, both from generic sources and those recorded as part of the ORJIP BCA and the total duration of the observation period. As only the data collected from the cameras during daylight hours were fully processed, we based our analysis on the number of birds expected to pass through during daylight. We estimated daylight hours between October and March using the suncalc function in the R library RAtmosphere (Gionata *et al.* 2015) to be 1733.55 hours taking an average during the post-construction years (2010/11 to 2012/13 to avoid a leap year).

Within the data collected using the laser rangefinders two possible distances were measured for each bird – a straight line between the first and last encounters and the true distance travelled between these two. The differences between the values of speed derived from these two measures have potential implications for the final collision rates given the differences in the numbers of birds that may pass through the areas if they take more meandering paths. Table 3 details these differences for each species.

Table 3. Average distances travelled (m) and speed (ms^{-1}) of birds depending on distance measured as part of the ORJIP BCA study (Skov *et al.* 2018) and the generic speed estimate taken from Alerstam *et al.* (2007) and Pennycuick (1997). Note that the values presented here differ to those presented in table 5.13 of the ORJIP BCA final report as we restrict our analyses to the data collected between October and March.

	Average distance (m) straight line	Average distance (m) true length	Average speed - straight line (ms^{-1})	Average speed - true length (ms^{-1})	Generic Speed (ms^{-1})
Herring gull	869.23	1213.24	8.0	9.8	12.8
Lesser black-backed gull	715.70	1012.22	8.4	10.4	13.1
Great black-backed gull	760.85	1053.04	8.5	10.0	13.7
Black-legged kittiwake	614.32	923.60	6.7	8.6	13.1
Northern gannet	1045.45	1251.85	11.7	13.1	14.9

2.3 Probability of collision/collision integral

To estimate the number of expected collisions, the flux of birds passing through the rotor swept area over a given period is multiplied by the probability of an individual bird passing through the rotor and colliding. The ‘probability of collision’ is based on the probability of the bird and the turbine being in the same place at the same time. For Option 1 of the Band CRM – the ‘basic’ model – this is estimated based on the size (Table 1) and speed of the turbine blades and the size (Table 4) and speed (Table 3) of the birds, assuming that the birds have a cruciform shape (Band 2012; Masden & Cook 2016). Option 3 of the Band CRM – the ‘extended’ model –, also considers the flight height distribution of the species concerned, accounting for the fact that birds are less likely to collide further away from the centre of the rotor swept area (Band 2012), in order to estimate the ‘collision integral’.

2.4 Flight height models

In order to determine the proportion of birds at collision risk height, species and site-specific flight height distributions were derived from the data collected using laser rangefinders. Data reflect a sample of the birds present in the study region. Consequently, in deriving distributions of seabird flight heights similar to those of Johnston *et al.* (2014), it was necessary to use a modelling approach that was sufficiently flexible that it could fit to a variety of forms, but not so flexible that it would over-fit to the data. We considered a number of different distributional forms for each species using the fitdistr function in MASS (Venables & Ripley 2002) and the normalmixEM function in Benaglia *et al.* (2009). For each species, we then consider which distribution best fitted the observed data.

2.5 Collision models

Using the information derived from the steps above, we are able to work through the Band CRM (Band 2012), introducing site-specific information at each step in order to understand how estimates of collision change as the parameters used by the model are refined. Initially, we replicate the collision risk model as it would be carried out ‘pre-construction’ as part of an Environmental Impact Assessment (EIA), using pre-construction density estimates, generic bird data and parameters based on the turbines installed (Tables 1-4), but, in contrast to the CRMs carried out as part of EIAs, we assume no avoidance behaviour.

We then refine the predictions by introducing: (i) post-construction density data, (ii) site-specific information on flight speed and (iii), finally, site-specific information on avoidance behaviour (Figure 2). Following this approach, we have eight different pathways leading to estimated collision rates based on the assumptions and data used (Figure 2). As the study area was wholly within the area of the Thanet Offshore Wind Farm, we consider only meso- and micro-avoidance and not, macro-avoidance.

Table 4. Seabird morphometric data, taken from Robinson (2017), flight mode (flapping or gliding flight) and, avoidance rates taken from Skov *et al.* (2018).

	Length	**Wingspan**	**Flight mode**	**Macro-avoidance**	**Meso-avoidance**	**Micro-avoidance**	**Overall avoidance**
Herring gull	0.61	1.44	flap	0.442	0.9614	0.9565	0.999
Lesser Black-backed Gull	0.59	1.45	flap	0.639	0.8937	0.9565	0.998
Great-black-backed Gull	0.71	1.575	flap	0.469	0.8423	0.9565	0.996
Black-legged kittiwake	0.39	1.075	flap	0.575	0.9160	0.9500	0.998
Northern gannet	0.935	1.725	glide	0.816	0.9205	0.9500	0.999

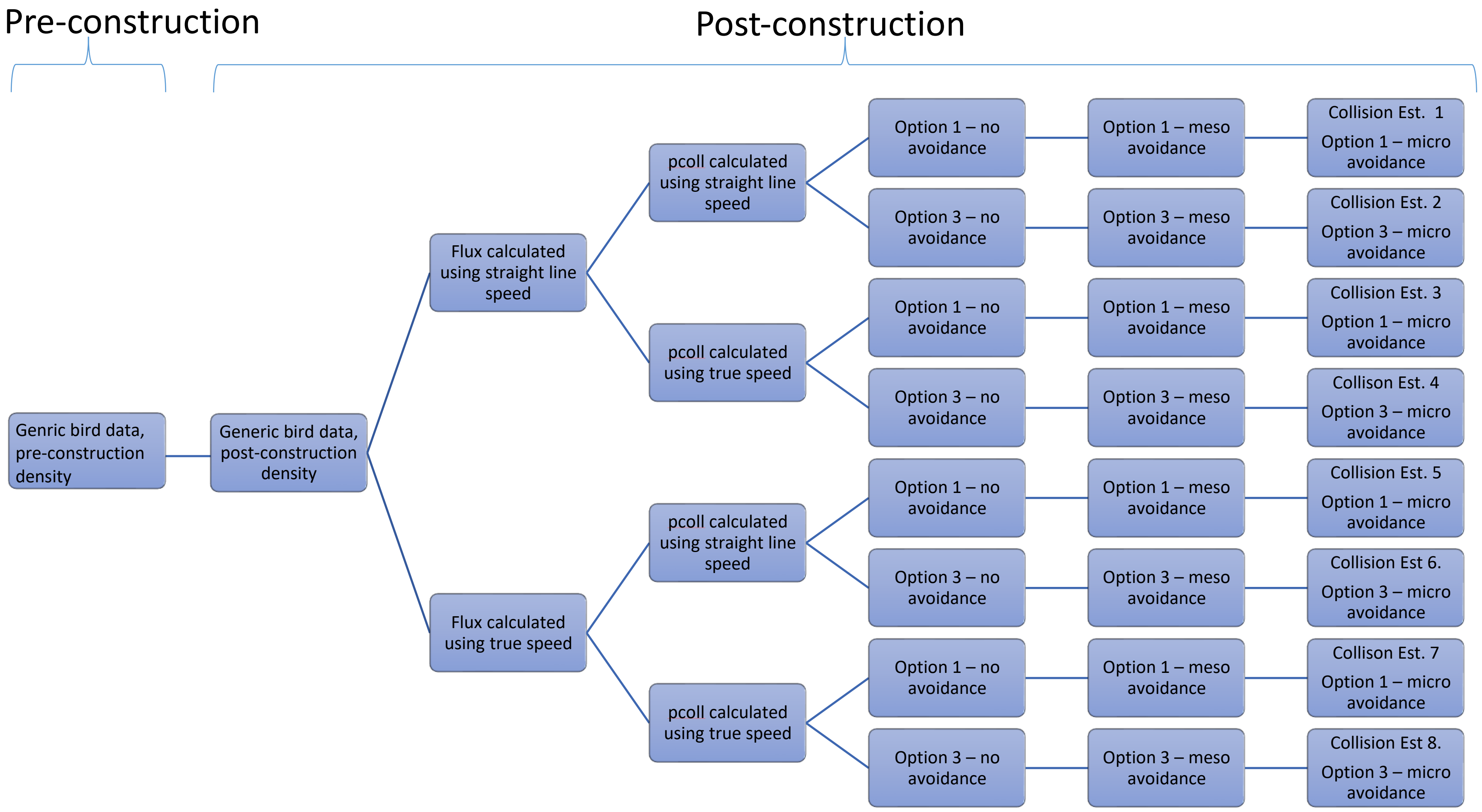


Figure 2. Schematic for producing estimates of collision at the eight turbines monitored during the ORJIP BCA project comparing the generic estimate that might be produced ‘pre-construction’ with more refined estimates produced using ‘post-construction‘ data. At each step, collision rates are refined by introducing more site-specific data. Different pathways reflect the different ways in which flight speed and flight height may be incorporated into the model.

2.6 Recorded collisions

Over the course of the ORJIP study, six birds were recorded colliding with turbines (Table 5). As density data were only available for the period October-March, the collision involving a large gull recorded on 21st August 2015 was not included in our analyses. As, with the exception of the black-legged kittiwake, birds involved in the collisions were not identified to species level, we group them as large gulls and, for the purposes of the analysis, compare these collision rates to the sum of those estimated for herring, lesser black-backed and great black-backed gulls.

Table 5. Birds recorded colliding with turbines during the ORJIP Bird Collision Avoidance Project (Skov *et al.* 2018).

Species/Group	Date
Black-legged kittiwake	1st November 2014
Lesser/Great Black-backed Gull	24th November 2014
Unidentified gull	28th November 2014
Large gull	21st August 2015
Large gull	12th December 2015
Unidentified gull	10th February 2016

2.7 Comparison of avoidance rates derived from ORJIP BCA study with those estimated using traditional approach

The above steps consider only birds within the wind farm and, therefore, do not account for macro-avoidance behaviour or, the avoidance rate as used in the Band CRM at present. The ORJIP BCA estimated an overall empirical avoidance rate, combining macro-, meso- and micro-avoidance. These values (Table 4) were well above those presently recommended (Cook *et al.* 2014). However, the values from the ORJIP BCA study and existing guidance may not be strictly comparable as they were derived in different ways.

The avoidance rates recommended in existing guidance are derived by comparing observed and predicted collision rates (Cook *et al.* 2014). As the predicted collision rates are based on estimates from the Band model, they incorporate elements of model error arising as a result of the assumptions made (Band 2012). The empirical avoidance rates derived from the ORJIP BCA project do not incorporate this model error and, consequently, are likely to be higher than those used at present. Furthermore, macro-avoidance incorporates both barrier effects and displacement (Cook *et al.* 2014). The data collected by the ORJIP BCA project at the macro scale covers birds in flight approaching the operational wind farm but, is not able to compare pre- and post-construction bird densities within the wind farm. Consequently, the ORJIP BCA data only incorporates the barrier effects element of macro-avoidance and not the displacement element. How these elements interact is unclear, however, in the absence of such information, the macro-avoidance rates derived as part of the ORJIP BCA project are not consistent with the assumptions about avoidance behaviour made by the Band model.

In order to facilitate a comparison between the existing guidance and the values obtained from the ORJIP BCA study, we recalculate avoidance rates by the ‘traditional’ approach of comparing the number of observed collisions to those predicted in the absence of avoidance behaviour (Eq. 1). We do this for each of the pathways set out in Figure 2. As avoidance rates will typically be applied in a pre-construction context, we also estimate a predicted collision rate based on the pre-construction estimates of bird density data and site-specific estimates of flight speed and height measured as part of the ORJIP BCA project. To investigate the impact of site-specific data in this calculation, we also estimate avoidance rates based on pre- and post-construction density data using generic bird data.

$$Avoidance\ Rate\ = 1 - \left(\frac{observed\ collision\ rate}{predicted\ collision\ rate}\right)$$ Equation 1.

3 Results

3.1 Flux calculations under two flight path measurements

Following the protocol described above, pre- and post-construction flux values were calculated for each of the five species under investigation using generic, ‘straight line’ and ‘true length’ estimates of speed (Table 6). The difference between the estimates of flux based on ‘straight line’ and ‘true length’ estimates of speed ranged from 203 great black-backed gulls to 22,892 herring gulls based on pre-construction densities and from 3,781 northern gannets to 15,926 herring gulls based on post-construction densities. Figure 3 visually represents the differences between the measurements based on straight line’ and ‘true length’ estimates of speed, those based on the latter resulting in increases in the numbers of bird likely to pass through the area surrounding the two turbines (Figure 1). Changes in the density of the species between the pre- and post-construction periods (Table 2) also result in changes in estimated flux rates.

Table 6. Values of flux for five seabird species using generic and site-specific estimates of speed and pre- and post-construction density data.

Species/Group	Generic Flux		Straight-line flux		True length flux	
	Pre-construction	**Post-construction**	**Pre-construction**	**Post-construction**	**Pre-construction**	**Post-construction**
Herring gull	171525	119334	107823	75015	130715	90942
Lesser black-backed Gull	29707	33308	18954	21251	23682	26553
Great black-backed Gull	1882	96656	1172	60175	1374	70577
Black-legged kittiwake	18004	98725	9184	50363	11779	64592
Northern gannet	5119	40274	4020	31628	4501	35409

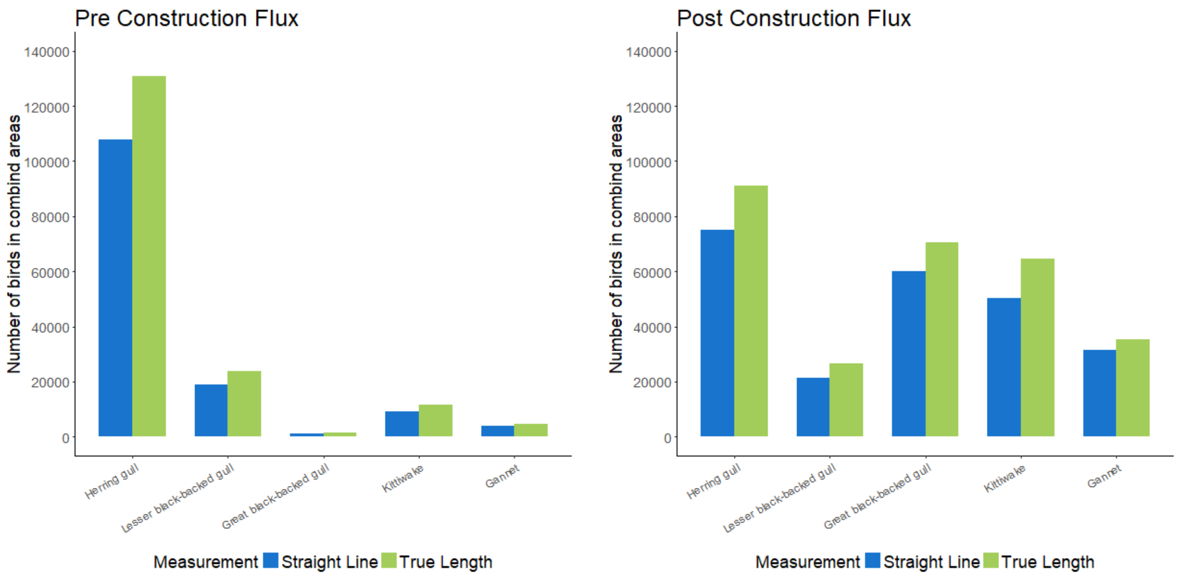


Figure 3. Pre- and post-construction flux values for five species using ‘straight line’ or ‘true length’ estimates of speed.

3.2 Probability of collision/collision integral

By refining the data used for the Band CRM, for each species, we obtained three estimates (based on generic, 'straight line' or 'true length' estimates of speed) for the probability of collision and two estimates (based on 'straight line' or 'true length' estimates of speed) for the collision integral (Table 7).

The probability of a bird colliding with a turbine is based on the length of time it takes for the bird to cross the rotor-swept area (Band 2012). Consequently, utilising the slower site-specific flight speeds obtained using the laser rangefinders results in an increased probability of collision. These differences are most noticeable for species such as black-legged kittiwake and herring gull, for which there is the greatest difference between the generic and straight line or true speeds. Similarly, as the straight-line speeds are slower than the true speeds, both the probability of collision and collision integral are higher when estimated using the straight-line speed.

Table 7. Estimates of probability of collision and collision integral obtained using generic and site-specific estimates of speed.

	Probability of Collision			**Collision Integral**	
	Generic Speed	**Straight line Speed**	**True Speed**	**Straight line Speed**	**True Speed**
Herring gull	0.092286	0.123504	0.107849	0.080257	0.069501
Lesser black-backed gull	0.090344	0.118373	0.101968	0.065471	0.056400
Great black-backed gull	0.095414	0.127808	0.114204	0.092973	0.080208
Black-legged kittiwake	0.077145	0.116359	0.096935	0.045394	0.038025
Northern gannet	0.103378	0.118540	0.110711	0.021401	0.017663

3.3 Flight heights

For lesser black-backed gull, great black-backed gull and herring gull, flight height data most closely fitted a gamma distribution (Figure 4). For black-legged kittiwake, flight height data most closely matched a normal distribution while flight height data for northern gannet most closely matched a normal-mixture distribution. It should be noted that these data indicated a higher proportion of birds at collision risk height than was observed in the generic flight height distributions (Johnston *et al.* 2014). There are several potential explanations for differences between the observed flight height distributions and the generic data:

1. The laser rangefinder data may be biased against birds flying closer to the sea surface. Birds close to the sea surface may be harder for observers to detect if flying between the troughs of waves and/or less conspicuous against the background. A previous study using laser rangefinders (Borkenhagen *et al.* 2018) suggested that birds at lower altitudes may be under-represented in estimates of flight height.

2. There is also the possibility that the generic data may be biased as a result of birds being attracted to survey vessels or due to observers detecting birds as they were flushed from the sea surface by the survey vessels (Johnston *et al.* 2014; Camphuysen *et al.* 2004).

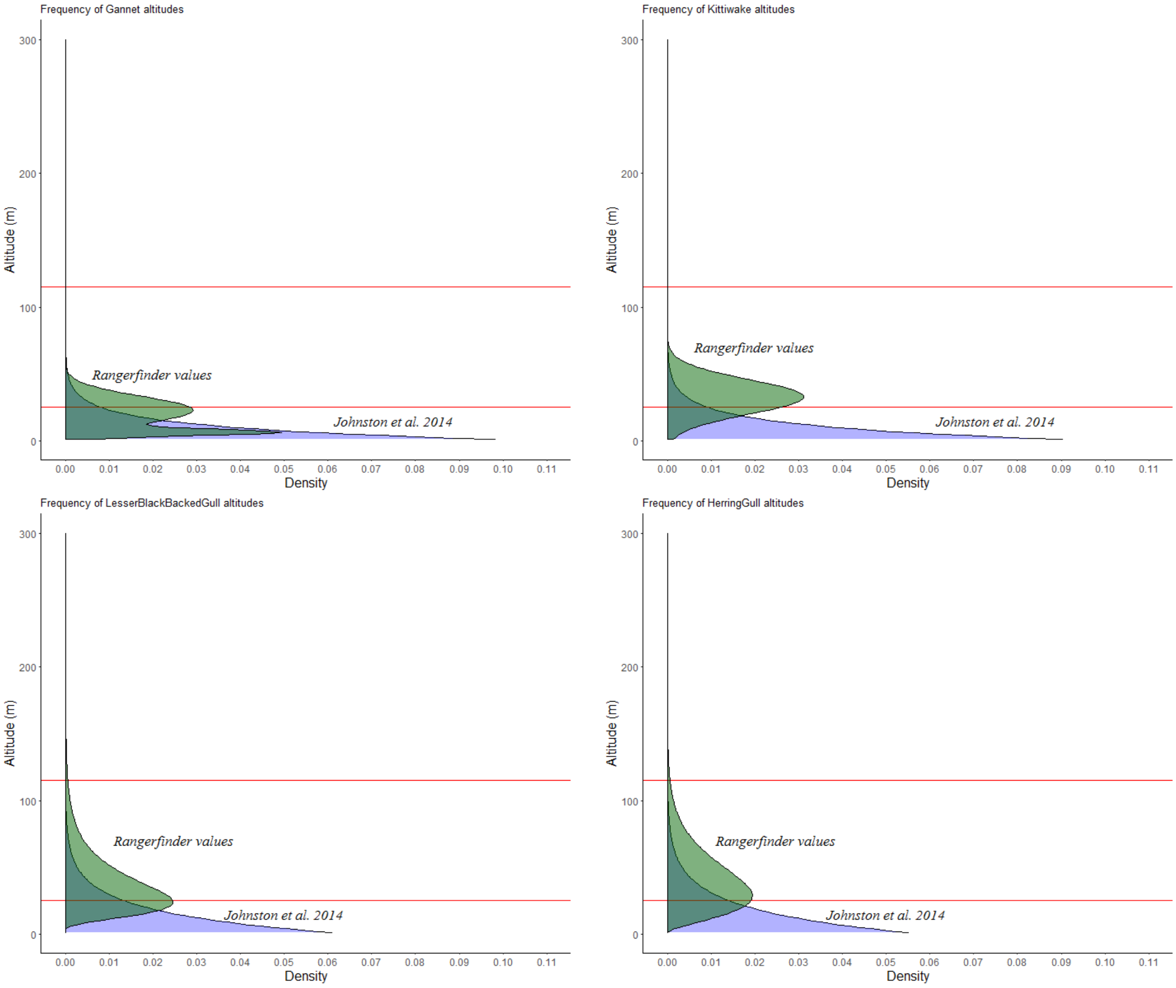
3. The flight heights of birds differed inside and outside the wind farm. There is some evidence that gulls may fly higher inside a wind farm than outside from both the

ORJIP BCA study and previous studies (Cook *et al.* 2014; Thaxter *et al.* 2017; Skov *et al.* 2012), although this difference may potentially reflect the locations of wind farm sites relative to the coast (see below). The data underpinning the generic distributions in Johnston *et al.* (2014) were all derived from pre-construction estimates of seabird flight height.

4. There are site-specific differences in seabird flight heights. Previous studies have shown that seabird flight heights may vary on a site-specific basis (Johnston & Cook 2016; Ross-Smith *et al.* 2016). Such differences may relate to behavioural characteristics such as whether birds are using an area for foraging or commuting flights. In contrast, data from Johnston *et al.* (2014) averaged flight heights across a broad range of habitats.

5. Wind speed and direction are likely to influence seabird flight altitudes. The laser rangefinder data available to the ORJIP BCA study analyses were constrained by the limited range of weather conditions during which observers were able to safely access turbines to collect these data, i.e. during relatively calm weather conditions. Consequently, the laser rangefinder data may be biased towards behavioural flight height responses to calm weather.

With the data available, it is not possible to determine which, if any, of these explanations is the key reason for the differences between the distributions reported here and those reported by Johnston *et al.* (2014). In practice, all five are likely to have had some impact.



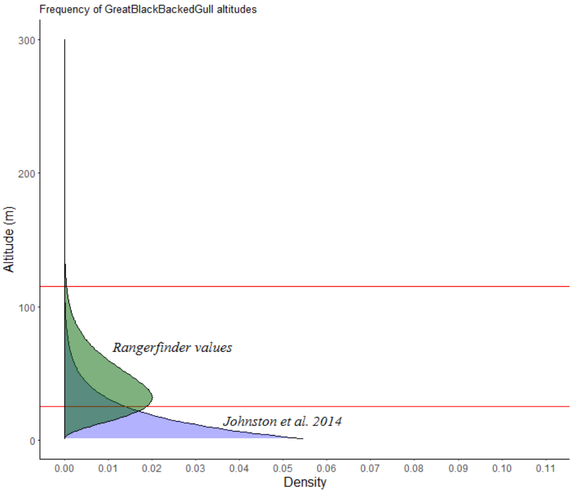


Figure 4. Comparison of flight height distributions derived from laser rangefinder data (green) collected as part of the ORJIP BCA project and generic flight height distributions (purple) derived from data collected as part of boat surveys and presented in Johnston *et al.* (2014). Red lines indicate the upper and lower limits of the turbine rotor swept areas of turbines installed at Thanet.

Table 8. Proportion of birds at collision risk height in relation to turbines installed at Thanet (25-115m) recorded using laser rangefinders as part of the ORJIP BCA project and predicted from generic data (Johnston *et al.* 2014).

	ORJIP BCA	(Johnston *et al.* 2014)
Herring gull	0.768	0.239
Lesser black-backed Gull	0.725	0.205
Great black-backed Gull	0.826	0.245
Black-legged Kittiwake	0.744	0.090
Northern gannet	0.285	0.075

3.4 Collision models

We combine the revised estimates of flux, the probability of collision and flight heights presented above in order to investigate how estimated collision risk varies in relation to the assumptions made during the modelling process and the incorporation of site-specific data. For each species, we are able to estimate a collision rate at each point along the eight pathways identified in Figure 2. Full details of the calculations underpinning the following table are available in Appendix 1.

Table 9. Change in predicted collision rates for the non-breeding season from Option 1 and Option 3 of the Band CRM (Band 2012) as density data are changed from pre-construction (pre) to post-construction (post) estimates, generic bird data (gen) are replaced with site-specific bird data (SSp), flux rates and probability of collision are calculated using either straight line (SL) or true (TD) speed and meso- (Me) and micro- (Mi) avoidance are introduced.

Density Estimate			Pre	Post														
Flight height			Gen	Gen	SSp	SSp		SSp		SSp		SSp		SSp		SSp	
Flight speed			Gen	Gen	Gen	SSp		SSp		SSp		SSp		SSp		SSp	
Distance measure			Gen	Gen	Gen	SL	TD	SL	TD	SL	TD	SL	TD	SL	TD	SL	TD
Pcoll/CollInt			Gen	Gen	Gen	SL	TD	TD	SL	SL	TD	TD	SL	SL	TD	TD	SL
Avoidance			No	No	No	No		No		Me		Me		Me/Mi		Me/Mi	
Band CRM Option	1	Herring gull	618.94	430.61	1381.01	1161.79	1229.92	1014.53	1408.45	44.85	47.47	39.16	54.37	1.95	2.07	1.70	2.36
	3		408.74	284.37	284.37	983.54	1032.55	851.73	1192.36	37.96	39.86	32.88	46.03	1.65	1.73	1.43	2.00
	1	Lesser black-backed gull	89.86	100.75	356.41	297.95	320.69	256.66	372.28	31.67	34.09	27.28	39.57	1.38	1.48	1.19	1.72
	3		56.01	62.80	62.80	227.30	244.66	195.81	284.01	24.16	26.01	20.81	30.19	1.05	1.13	0.91	1.31
	1	Great black-backed gull	7.19	368.95	1244.60	1037.92	1087.75	927.44	1217.33	163.68	171.54	146.26	191.97	7.12	7.46	6.36	8.35
	3		5.06	259.95	259.95	913.98	924.79	788.49	1071.97	144.14	145.84	124.35	169.05	6.27	6.34	5.41	7.35
	1	Kittiwake	21.07	115.52	926.04	712.54	761.30	593.60	913.85	59.85	63.95	49.86	76.76	2.99	3.20	2.49	3.84
	3		9.17	50.28	50.28	373.49	401.25	312.86	479.00	31.37	33.70	26.28	40.24	1.57	1.69	1.31	2.01
	1	Northern gannet	6.46	50.84	193.52	174.27	182.21	162.76	195.10	13.85	14.49	12.94	15.51	0.69	0.72	0.65	0.78
	3		3.26	25.65	25.65	110.58	102.17	91.26	123.80	8.79	8.12	7.26	9.84	0.44	0.41	0.36	0.49

3.4.1 Herring gull

Based on the pre-construction density data and generic bird data, 618 herring gulls were predicted to collide during daylight hours between October and March each year (Figure 5). Following construction of the wind farm, the density of birds, and therefore number of expected collisions, decreased. However, site-specific flight height data suggests a far higher proportion of birds at risk height than is assumed by the generic data, reflected in an increase in the predicted collision rate at the third step of the analysis. Introducing site-specific flight speed information results in further changes to the predicted collision rates, although the extent of changes is dependent on whether these estimates are based on straight line or true speed. Incorporating different measures of speed affects both the estimated flux rate and estimations of the probability of birds colliding. However, in relation to the predicted collision rate, the selection of the appropriate measure of speed appears to be most important when calculating the probability of collision (Table 9). As may be expected, the selection of Option 1 or Option 3 of the Band CRM (Band 2012) also results in a significant change in the predicted collision rate. However, as avoidance behaviour is incorporated, predicted collision rates begin to coalesce. When only meso-avoidance is incorporated, differences are still evident and, the lowest collision rates are observed when flux rate is estimated using straight line speed and the probability of collision is estimated using true speed. When micro-avoidance is incorporated, collision estimates following each of the eight pathways all fall to around 1-2 birds per winter. The most noticeable changes in the number of predicted collisions occur in relation to the introduction of site-specific flight height data and the introduction of micro-avoidance behaviour.

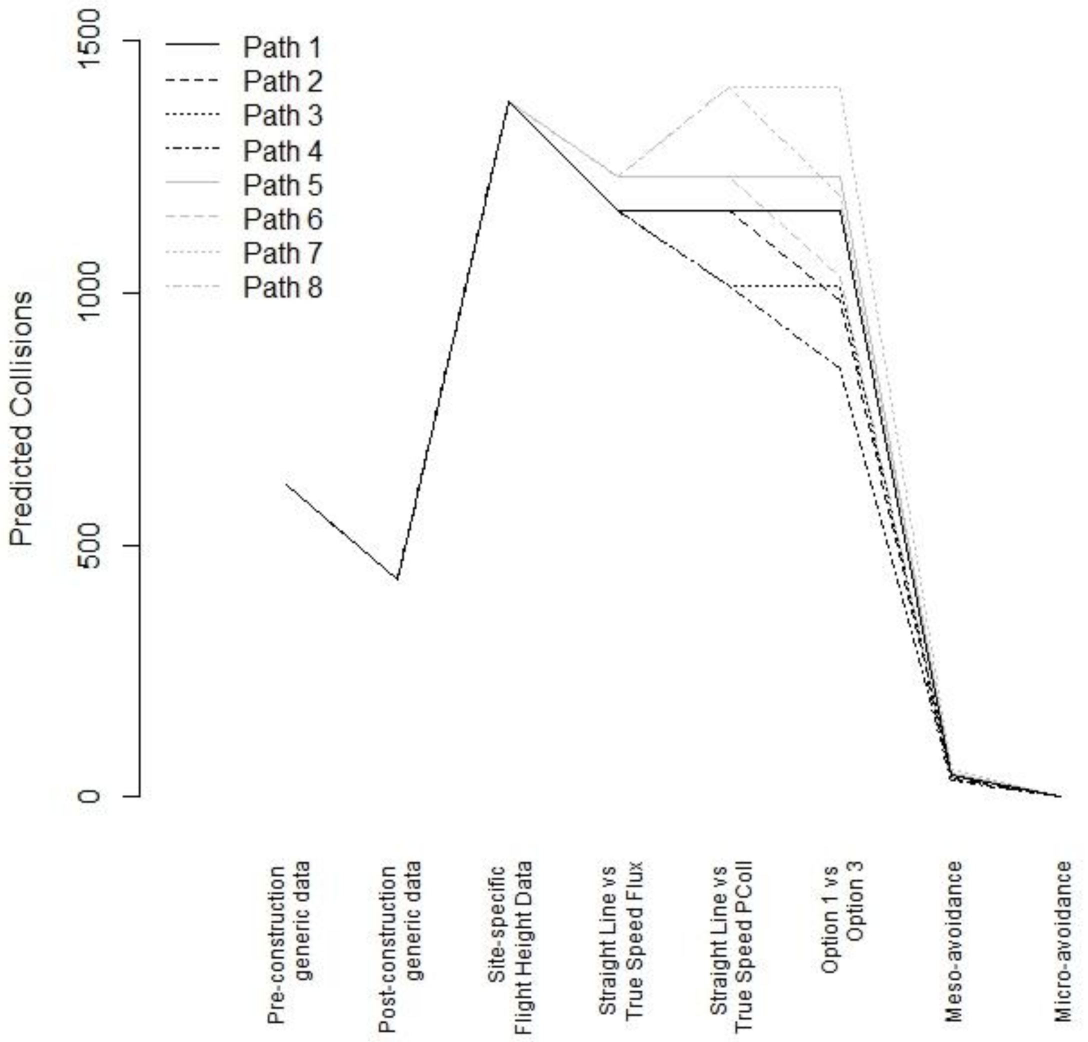


Figure 5. Change in predicted collision rate for herring gull as model assumptions and parameters are refined.

3.4.2 Lesser black-backed gull

The density of lesser black-backed gulls in the study area over winter was much lower than for herring gulls. In contrast to herring gull, there was a slight increase in the density of lesser black-backed gulls recorded during the post-construction monitoring. Aside from this difference, the changes in the predicted collision rates of lesser black-backed gulls as model assumptions and parameters were refined were broadly similar to those recorded for herring gulls (Figure 6).

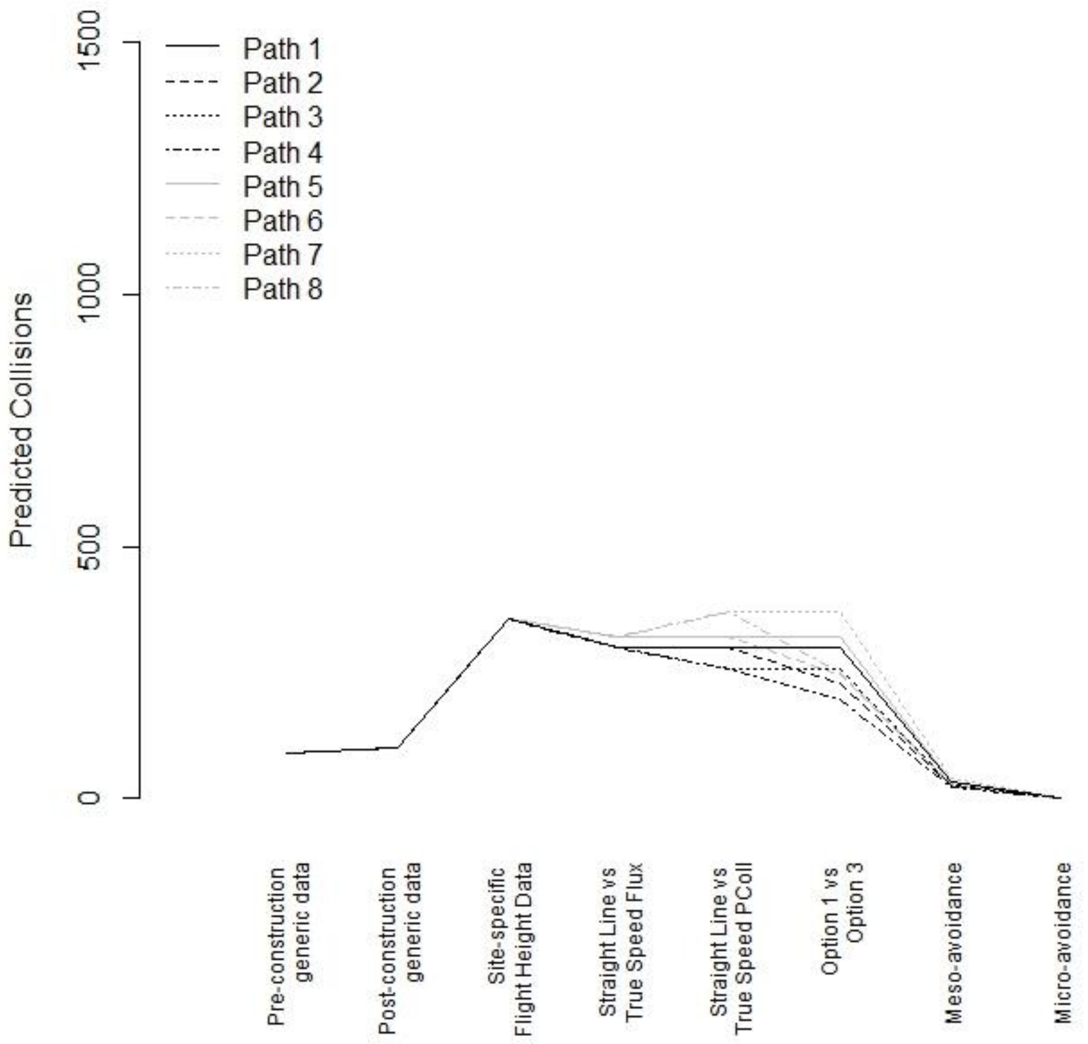


Figure 6. Change in predicted collision rate for lesser black-backed Gull as model assumptions and parameters are refined.

3.4.3 Great black-backed Gull

Great black-backed gulls were the most numerous species recorded in the study area. As with lesser black-backed gull, they increased in density during the post-construction period. Other changes in the predicted collision rates of great black-backed gulls as model assumptions and parameters were refined were broadly similar to those recorded for the other study species.

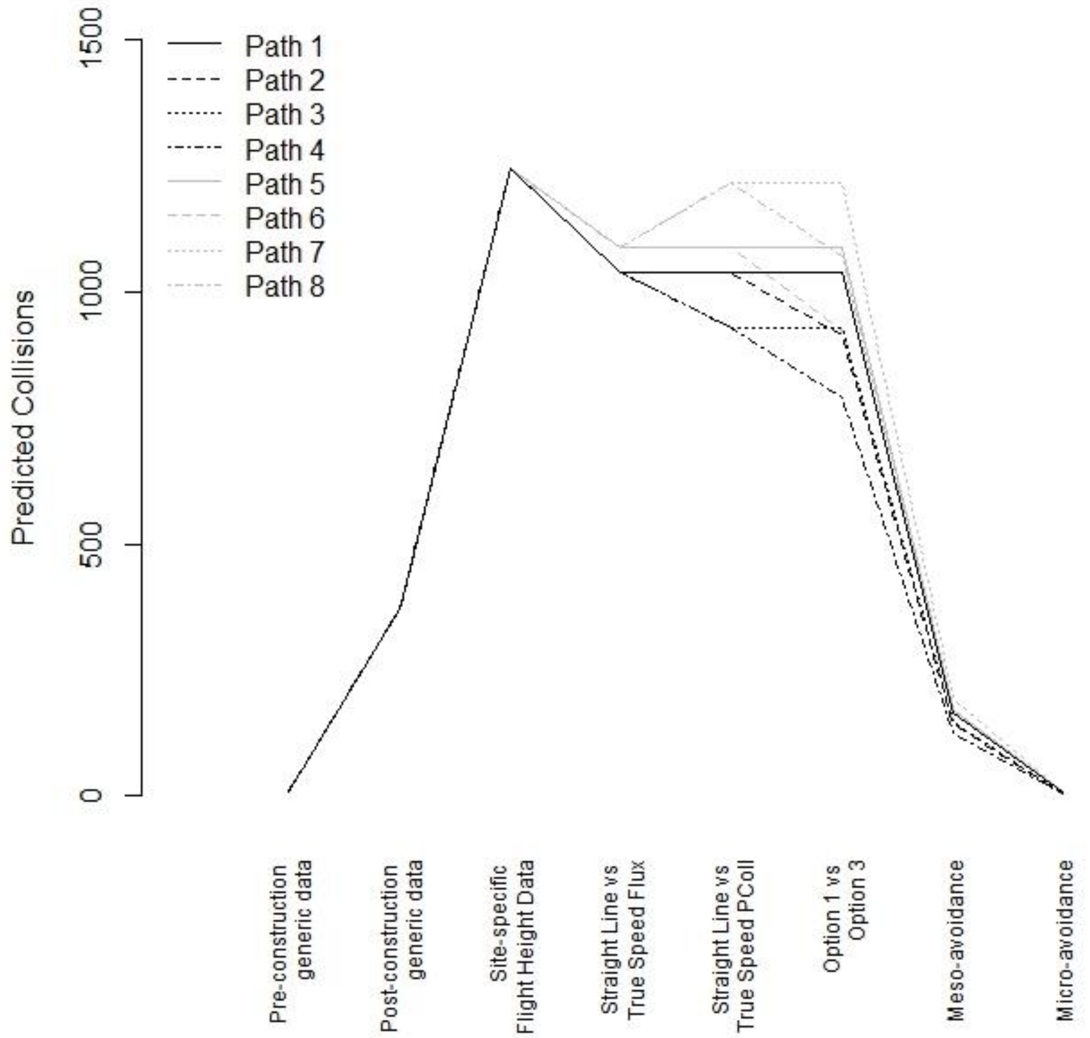


Figure 7. Change in predicted collision rate for great black-backed Gull as model assumptions and parameters are refined.

3.4.4 Black-legged kittiwake

As with the black-backed gull species, black-legged kittiwake increased in density during the post-construction period. Other changes in predicted collision rates as model assumptions and parameters were refined were broadly similar to those recorded for the other study species.

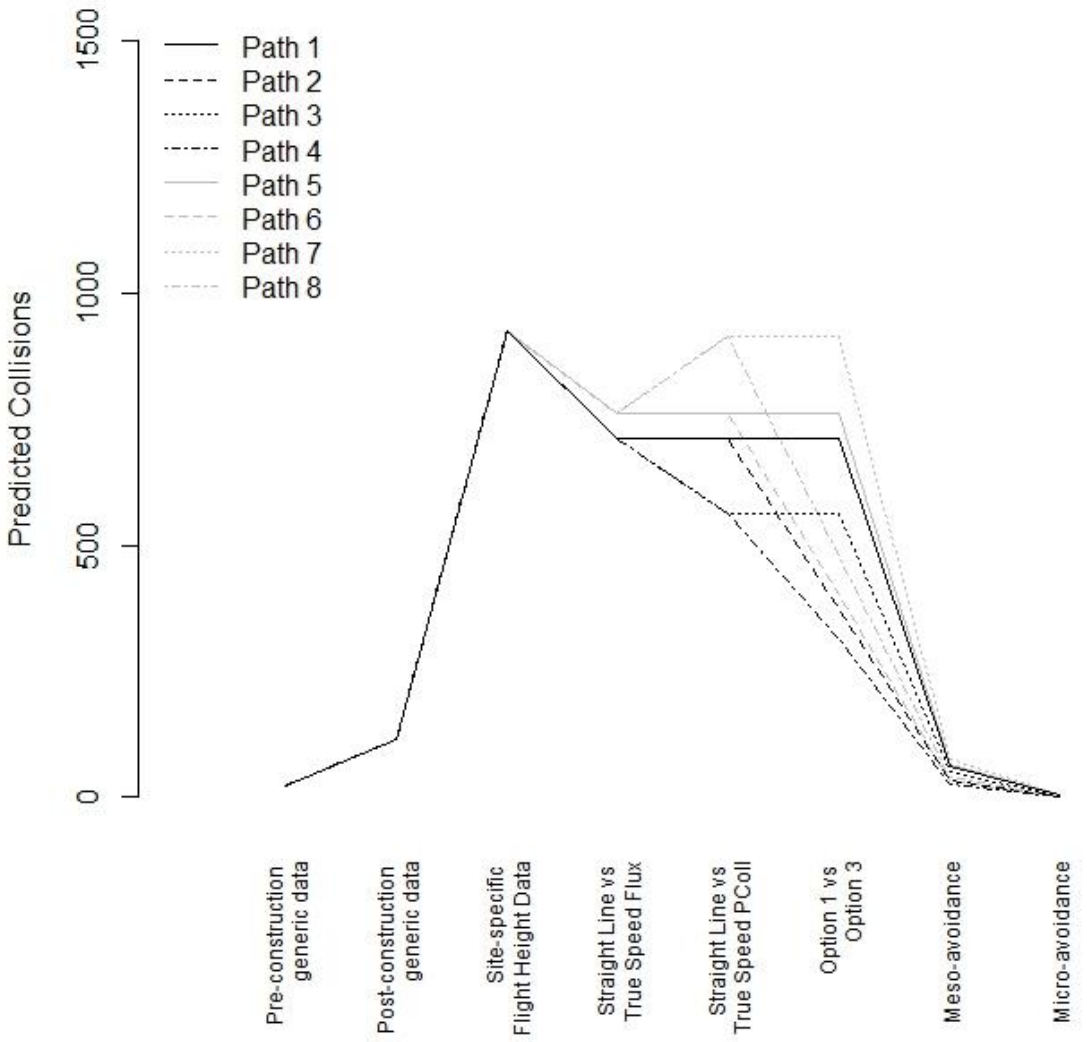


Figure 8. Change in predicted collision rate for Black-legged kittiwake as model assumptions and parameters are refined.

3.4.5 Northern gannet

Northern gannets were the least abundant of the study species in the study area. Densities increased between the pre- and post-construction periods. Other changes in predicted collision rates as model assumptions and parameters were refined were broadly similar to those recorded for the other study species.

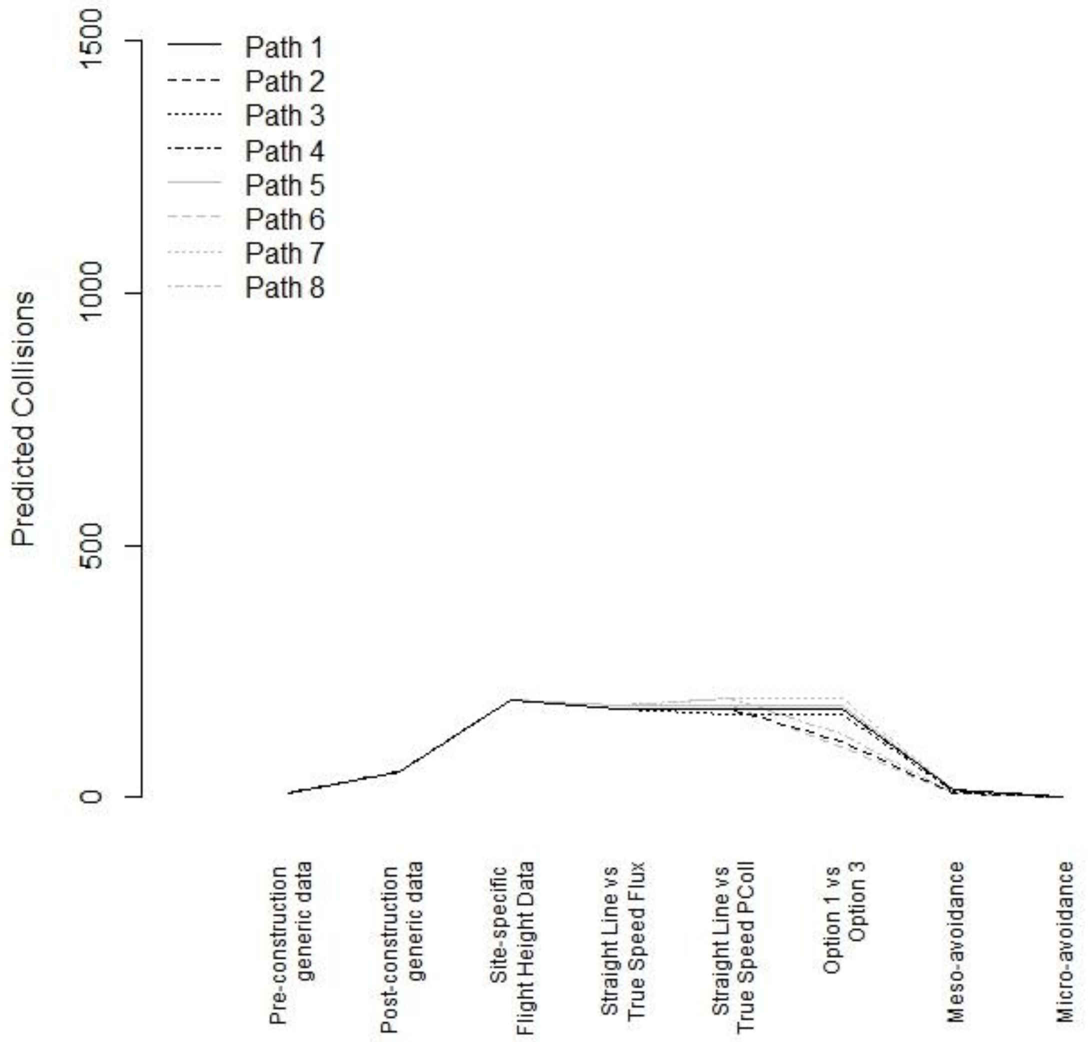


Figure 9. Change in predicted collision rate for northern gannet as model assumptions and parameters are refined.

3.5 Predicted vs. observed collision rates

The final predicted collision rates broadly follow the pattern of species abundance within the study area. Great black-backed gull, the most abundant species, is predicted to have the greatest number of collision while northern gannet, the least abundant species, is predicted to have the fewest (Figure 10a). However, the use of generic or site-specific data and the assumptions made about the data in the Band CRM also have an impact on the final conclusions that are reached (Figure 10b). The relative importance of the use of generic or site-specific data and these assumptions appears to vary by species. For example, the relative change in predicted collision rate according the assumptions made was greatest for black-legged kittiwake (figure 10b). For black-legged kittiwake, collision estimate 5 was 60% greater than the mean collision rate across all eight of the pathways highlighted in figure 2. In contrast, for great black-backed gull, this figure was only 22% (figure 10b).

Having accounted for avoidance behaviour, predicted collision rates were still higher than observed collision rates (Figure 10). For Black-legged kittiwake a single collision was recorded in November 2014, compared to predictions of between 1 and 4 collisions per winter, depending on the data and assumptions used in the model. It was not possible to identify the large gulls that were recorded colliding to species level. Consequently, we compare the observed collision rate for large gulls to the combined predicted collision rate for herring, lesser black-backed and great black-backed gull. Two large gulls were recorded colliding in winter 2014/15 and winter 2015/16. This compares to predicted collision rates of 7-13 birds per year.

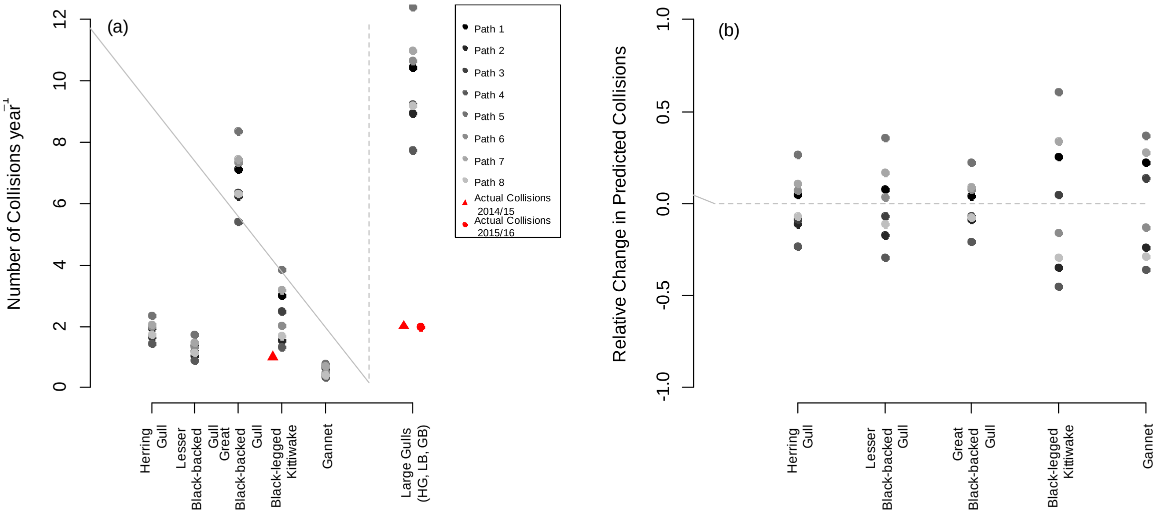


Figure 10. (a) Comparison between predicted and observed collision rates in relation to the data and assumptions incorporated into the Band collision risk model (Band 2012) and whether Option 1 or Option 3 of the Band CRM is used. For actual recorded collisions, it was not possible to distinguish between the large gull species; consequently, the final column includes the total number of predicted collisions for herring, lesser black-backed and great black-backed gulls. (b) Relative change in predicted collision rates in relation to the data and assumptions incorporated into the Band collision risk model (Band 2012) and whether Option 1 or Option 3 of the Band CRM is used.

3.6 Comparison of avoidance rates derived from ORJIP BCA study and those estimated using traditional approach

Avoidance rates estimated using the traditional approach for the Option 3 of the Band CRM were lower than for Option 1 of the Band CRM (Table 10). The reason for this is that avoidance rates estimated in this way incorporate elements of model error. By accounting for the uneven vertical distribution of birds, Option 3 of the Band CRM accounts for some (but not all) of this model error, reducing the predicted collision rate and, following equation 1, the estimated avoidance rate.

For large gulls and black-legged kittiwake it was possible to estimate avoidance rates for the pre- and post-construction periods using generic data. Higher avoidance rates estimated for the post-construction period reflect changes in the density estimates. However, for the reasons explained above (section 2.2), such changes may not reflect macro-responses to the wind farm. Density estimates, and consequently, predicted collision rates for the post-construction period were higher than for the pre-construction period. These differences are most notable for the black-legged kittiwake (Table 10). Avoidance rates estimated in this way based on the post-construction density estimates may be thought of as equivalent to the within-wind farm avoidance rates presented in Cook *et al.* (2014).

Table 10. Avoidance rates calculated from pre- and post-construction density estimates using generic estimates of flight height and flight speed. Avoidance rates based on pre-construction data reflect total avoidance whilst those based on post-construction data reflect within wind farm avoidance only.

	Pre-construction		Post-construction	
	Option 1	Option 3	Option 1	Option 3
Black-legged Kittiwake	0.952	0.891	0.991	0.980
Large gulls (HG, LB, GB)	0.994	0.991	0.995	0.993

The avoidance rates estimated in this way can be further refined by incorporating site-specific data into the calculations of predicted collision rates (Table 11). Incorporating this site-specific information resulted in higher estimated avoidance rates. Again, we see that the change in avoidance rates estimated using pre- and post-construction data was greater for black-legged kittiwakes than it was for large gulls. However, a key reason for the differences in the avoidance rates relates to the substantial differences between the generic and site-specific flight height distributions (Figure 4). The site-specific data includes a far greater proportion of birds at collision risk height. Consequently, following equation 1, this results in a greater predicted collision rate which, when compared to the observed collision rate, results in a higher avoidance rate as the model predicts that a greater number of birds must have taken action to avoid a collision.

Table 11. Avoidance rates calculated from a comparison of predicted and observed collision rates based on **pre-** and *post*-construction density estimates and post-construction site-specific estimates of flight speed and flight height following the collision estimate pathways shown in Figure 2. Avoidance rates based on pre-construction data reflect total avoidance whilst those based on post-construction data reflect within wind farm avoidance only. Clear cells indicate rates calculated based on Option 1 of the Band model, grey cells indicate rates calculated based on Option 3 of the Band model.

	Collision Estimate 1	Collision Estimate 2	Collision Estimate 3	Collision Estimate 4	Collision Estimate 5	Collision Estimate 6	Collision Estimate 7	Collision Estimate 8	ORJIP BCA
Black-legged kittiwake	**0.992** / *0.998*	**0.985** / *0.997*	**0.990** / *0.998*	**0.982** / *0.996*	**0.993** / *0.999*	**0.988** / *0.998*	**0.992** / *0.999*	**0.986** / *0.998*	0.998
Large gulls (HG, LG, GB)	**0.998** / *0.999*	**0.997** / *0.999*	**0.997** / *0.999*	**0.997** / *0.999*	**0.998** / *0.999*	**0.998** / *0.999*	**0.998** / *0.999*	**0.997** / *0.999*	0.998

4 Discussion

4.1 Importance of site-specific data

Much of the focus of uncertainty in relation to collision risk models has focussed on avoidance rates (Chamberlain *et al.* 2006). However, recent analysis suggests that this focus may partly reflect a misunderstanding of how the avoidance rate is used by collision risk models and, that other factors including flight heights and speeds may be similarly important (Masden 2015). The analysis presented above demonstrates the substantial impact that the use of site-specific data can have on estimated collision rates (Figures 5-10). However, it should be emphasised that the estimates of parameters such as flight speed and height presented in Skov *et al.* (2018) come from a single site during the non-breeding season. Given the influence of site-specific data on the estimated collision rates, such data may not be directly transferable to other sites or, to the breeding season.

The estimate of the proportion of birds at collision risk height has a substantial effect on the predicted collision rates (Figures 5-9). This is the case for both the basic and extended models. There are substantial differences between the flight height distribution estimated using laser range finders as part of this study and the generic distributions presented in Johnston *et al.* (2014). Such differences must be treated with caution as it is unclear the extent to which they reflect genuine differences between the two approaches and the extent to which they reflect bias in the data collection methodologies. However, the results do highlight the importance of using a robust and, ideally site-specific, flight height estimate in predicting collision risk.

The Band CRM makes use of bird speed twice: firstly, in order to estimate the flux rate of birds through the wind farm and; secondly, to estimate the probability of a bird colliding with a turbine rotor. Furthermore, flight speed may be estimated at different resolutions, with implications for the model outputs. For the purposes of our analyses, we used two different estimates of flight speeds. The first of these was simply the straight-line distance between the first and last laser range finder points and the time taken to travel between them (referred to above as straight-line speed). However, the birds may not have been travelling in straight lines (as assumed in the calculations of flux rate). Consequently, we estimated a second speed based on the point estimate of speed as measured using the laser range finders (referred to above as True Speed). It is important to note that both of these speeds were markedly lower than the generic speeds typically recommended in guidance (Alerstam *et al.* 2007). Consequently, the flux rates estimated from these data were lower than those estimated using the generic data and the probabilities of collision estimated were greater than those estimated using generic data. There are four possible combinations for how these flight speeds could be incorporated into the collision risk model with respect to their use in estimating the flux rate and probability of collision (straight line-straight line, straight line-true speed, true speed-true speed, true speed-straight line). Which of these combinations is selected has implications for the final estimated collision rate (Figures 5-9). In agreement with Skov *et al.* (2018), we feel that the combination which is most consistent with how the Band CRM is implemented is likely to be the use of the straight line speed estimate of the flux rate and the true speed estimate of the probability of collision. This is because the straight-line speed will reflect the average rate at which birds move through the wind farm while the true speed will be a point estimate of the speed of the bird as it passes the turbine blades.

4.2 Macro-avoidance

Macro-avoidance relates to the change in bird numbers within a wind farm site arising as a result of processes including, but not limited to, displacement, attraction and barrier effects (Cook *et al.* 2014). These responses may reflect either a functional change in habitat use (i.e. displacement or attraction) or anticipatory evasion of the wind farm due to perceived collision risk (i.e. barrier effects) (May 2015). The analyses described above concerned birds already present within the wind farm and, consequently, did not consider macro-avoidance.

The ORJIP BCA study estimated macro-avoidance by comparing the density of bird tracks within the wind farm to the density of bird tracks in a 3 km buffer around the wind farm (Skov *et al.* 2018). In common with previous findings (Cook *et al.* 2014), Skov *et al.* (2018) suggested significant inter-specific variation in the estimated macro-avoidance rates ranging from 0.797 (SD 0.026) for northern gannet to 0.566 (SD 0.058) for black- legged kittiwake and 0.481 (SD 0.038) for large gulls.

The macro-avoidance rates reported for gulls are much higher than those reported elsewhere (Cook *et al.* 2014; Vanermen *et al.* 2015; Krijgsveld *et al.* 2011). This is likely to be because the ORJIP BCA study focuses on the movements of birds in and around the wind farms and does not account for any displacement or attraction effects. Analysis of post-construction data collected from operational wind farms suggests that large gulls may be attracted to the wind farm and that black-legged kittiwakes may show little or no difference in area usage (Dierschke *et al.* 2016; Vanermen *et al.* 2015). The apparent high rate of macro-avoidance evident in gulls as part of this study may relate to the presence of fishing vessels on the edge of the wind farm. Fishing vessels cannot operate within the wind farm and, a previous study (Krijgsveld *et al.* 2011) noted gulls being attracted to fishing vessels on the edge of a wind farm. Seabird observers noted a similar effect as part of the ORJIP BCA study. In such circumstances, birds will be responding to the fishing vessels rather than the turbines. This may result in the number of birds outside the wind farm being inflated and the number within the wind farm being artificially reduced. This effect may hold for black-legged kittiwakes and large gulls. Consequently, the macro-avoidance rates estimated for large gulls and black-legged kittiwake as part of the ORJIP BCA study should be used with caution in relation to collision risk modelling based on pre-construction bird density estimates.

The results for northern gannet are consistent with previous studies, which suggested high macro-avoidance rates for this species, ranging from 0.64 (Krijgsveld *et al.* 2011) to 0.85 (Vanermen *et al.* 2015) and possibly as high as 0.92 (Welcker & Nehls 2016), although it should be noted the latter study was based on a limited sample size. It should also be noted that the rate estimated as part of the ORJIP BCA project may be an underestimate for two reasons. Firstly, without comparison to pre-construction estimates of bird density, these data do not capture any impact of displacement on the number of birds recorded. Secondly, collecting data on the movements of birds outside the wind farm using radar requires a trade-off between the distance over which a radar system can operate and resolution at which data can be collected. For the systems used in this study, the optimum distance over which to collect data was judged to be 3 km. However, past studies have noted that northern gannets may take action to avoid entering a wind farm at distances far greater than 3 km (Petersen *et al.* 2006). However, like gulls, northern gannets are known to be attracted to fishing vessels (Votier *et al.* 2010). The extent to which displacement from the wind farm, attraction to fishing vessels and the presence of the wind farm as a barrier to flying birds may interact with one another is unclear. Consequently, it is difficult to assess the extent to which the estimate of 0.79 may be precautionary.

It should be noted that comparison of pre- and post-construction density estimates suggested an increase in density post-construction for four of the five species (Table 2).

However, the pre-construction density estimates were based on survey effort from a single year, it is questionable this effort is sufficient to characterise the baseline conditions of the wind farm site (Maclean *et al.* 2013). Consequently, it is difficult to assess the extent to which the reported changes were “genuine”, particularly in the case of northern gannet, a species for which a strong displacement effect has previously been reported (Dierschke *et al.* 2016).

4.3 Meso-avoidance

To our knowledge, the ORJIP BCA study is the first that systematically set out to measure the meso-avoidance rates of seabirds within an operational wind farm. Data presented in a previous review (Cook *et al.* 2014) suggested that meso-avoidance rates were likely to be high, with few birds passing in close proximity to turbines.

In the ORJIP BCA study, meso-avoidance rates appear to be calculated in a logical way, comparing the track length per unit area within the rotor-swept zone and a 10 m buffer (as defined in Cook *et al.* (2014) to a theoretical density assuming birds were spread evenly throughout the wind farm. The resulting rates support previous hypotheses that meso-avoidance rates are likely to be very high and that birds within wind farms show strong avoidance of turbines.

4.4 Micro-avoidance

Micro-avoidance rates collected as part of the ORJIP BCA study were based on extremely limited sample sizes. It is important to highlight that this is likely to reflect the fact that most birds take action to avoid collisions at distances that do not necessitate the “last-second” avoidance behaviour reflected by micro-avoidance, rather than a short-coming in the study design. In total, only 299 birds were recorded approaching turbines closely enough to necessitate “last-second” collision avoidance behaviour. Consequently, it was not possible to consider species-specific micro-avoidance behaviour.

The results from the ORJIP BCA study are consistent with those from past studies that have shown that very few birds approach turbines closely enough to necessitate micro-avoidance behaviour (Krijgsveld *et al.* 2011; Thaxter *et al.* 2017; Mendel *et al.* 2014; Desholm *et al.* 2006). Data from across these studies suggested that micro-avoidance rates were likely to be >0.93, although, it should be noted that there were significant limitations in the derivation of this rate (Cook *et al.* 2014). However, the estimate from the ORJIP BCA study of 0.9500 (SD 0.0128) for all seabirds was consistent with this previous estimate. In terms of the number of records of birds interacting with turbines, the sample size from the ORJIP BCA study is substantially higher than any previous attempt. Consequently, whilst there is clearly a need for additional data collection to support this, the estimate of 0.9500 (SD 0.0128) for micro-avoidance from the ORJIP BCA study is, at this time, the best available data with which to quantify micro-avoidance behaviour in seabirds.

4.5 Use of avoidance rates from ORJIP BCA study

It is important to note that there is a difference between the empirical avoidance rate derived in the ORJIP BCA study and the avoidance rate as used by the Band CRM. The empirical avoidance rate, as derived by the ORJIP BCA study, incorporates detailed information about the distribution and movements of birds within a wind farm and their interactions with turbines. The avoidance rate as used by the Band CRM is based on a comparison of predicted and observed collision rates. The predicted collision rates will incorporate elements of error in relation to both the data used and the model itself (Band 2012). The incorporation of this error is likely to mean that the avoidance rates used by the Band CRM are likely to be lower than those measured empirically.

The total empirical avoidance rates estimated as part of the ORJIP BCA study include avoidance behaviour at the macro-, meso- and micro-scales. However, for the reasons set out above (section 4.2), we feel that the estimates of macro-avoidance from this study are not applicable in the context of how the Band CRM is used in the pre-construction assessment of collision risk. Empirical avoidance rates combining the remaining meso- and micro-avoidance correspond to the within-wind farm avoidance rates presented in Cook *et al.* (2014). The resulting empirical within-wind farm avoidance rates are 0.9956 for large gulls, 0.9958 for black-legged kittiwake and 0.9960 for northern gannet. However, as these rates do not incorporate model error in the same way that those recommended by existing guidance do (Cook *et al.* 2014), they are not directly applicable to the Band collision risk model.

Based on the data collected as part of the ORJIP BCA study and analysed above (section 3.6), we suggest that a total avoidance rate of 0.995 is suitable for use in the basic Band CRM for large gulls (Table 12). Given previous evidence of strong macro-avoidance in the northern gannet (Dierschke *et al.* 2016), we suggest that 0.995 is also a suitable minimum value to use for this species in relation to the basic Band CRM (table 12). Analyses of collision rates presented above (Tables 10 & 11) suggest that black-legged kittiwake may be more prone to collisions than large gulls. Consequently, we suggest that an avoidance rate of 0.990 is suitable for this species (Table 12). It is acknowledged that this is lower than in previous guidance (Cook *et al.* 2014). However, we feel this is justified as, in the previous guidance black-legged kittiwake was grouped with other small gull species (Cook *et al.* 2014). In the density data used in the above analysis (Royal Haskoning 2013) to estimate the predicted collision rate, the number of black-legged kittiwakes not identified to species level is likely to be negligible. As no other small gulls, whether identified to species level or not, were recorded colliding, we feel the estimate of 0.990 for black-legged kittiwake is robust. These avoidance rates are considered to include macro-avoidance (Table 12).

We were able to undertake further analyses (described in Appendix 1) in order to derive avoidance rates suitable for use in the stochastic collision risk model for black-legged kittiwake of 0.994 (95% CIs 0.976 - 0.998) for option 1 and 0.970 (95% CIs 0.871-0.989) for option 3 and, for large gulls 0.997 (95% CIs 0.992 - 0.999) for option 1 and 0.990 (95% CIs 0.974 - 0.995) for option 3. Note that the median values recommended for use in the stochastic collision risk models differ from the values recommended for use in the deterministic model, this relates to differences in the way in which flight height distributions are incorporated into the models. However, it should also be noted that the values recommended for use in the deterministic model are within the 95% confidence intervals of those recommended for use in the stochastic model.

In relation to the extended Band CRM, we note the sizeable difference between the observed and recorded flight height distributions, and the potential for bias associated with the collection of flight height data using laser range finders (Borkenhagen *et al.* 2018) to contribute to this difference. This difference has a noticeable effect on the avoidance rates estimated using generic and site-specific data. Given the precautionary principle in assessing collision risk, we suggest that the estimates of avoidance rate made using generic flight height data (Table 10) should be used for the extended Band CRM. Ideally site-specific estimates of flight height would be used to estimate avoidance rates. However, given uncertainty in the flight height data recorded as part of the ORJIP BCA project and, the discrepancy with previous estimates of seabird flight height (figure 4), we believe this reflects a realistic, precautionary approach. If the number of birds at risk of collision is overestimated, then, following equation 1, the overall avoidance rate is also likely to be overestimated. Consequently, for the extended Band CRM, we recommend using avoidance rates of 0.993 for large gulls and 0.980 for black-legged kittiwake (Table 12). It should be noted that this reflects an increase in the rate recommend for large gulls in previous guidance (Cook *et al.* 2014) and is the first time it has been possible to calculate a total avoidance rate for black-legged kittiwake for Option 3 of the Band CRM based on empirical data. However, based on the data collected as part of the ORJIP BCA project, it has not been possible to calculate an avoidance rate suitable for use in Option 3 of the Band CRM for northern gannet as no collisions were recorded (Table 12).

It is important to highlight some key limitations in how the avoidance rates presented in table 12 were derived. Data were collected from a single site, during the non-breeding season in daylight hours. As the avoidance rates derived from these data are higher than those presented elsewhere (Cook *et al.* 2014), care must be taken before applying them to other sites and to breeding season estimates of collision rates. Consequently, with the exception of black-legged kittiwake, the avoidance rates we recommend are based on generic flight speed and height data as we feel these retain a sufficient level of precaution whilst also being applicable to a broader range of sites and, to the breeding season. In relation to black-legged kittiwake, the recommended rate of 0.990 is derived using site-specific flight height and speed data as this was lower than the rate derived using generic data (tables 10,11 and 12).

Table 12. Recommended avoidance rates for use in the deterministic Band Collision Risk Model, derivation of these avoidance rates and rationale for recommendations.

	Band Model Option	**Recommended Avoidance Rate**	**Derivation of Avoidance Rate**	**Rationale**
Northern gannet	1	0.995	It was not possible to estimate an avoidance rate by comparing predicted and observed collision rates. However, given clear evidence of strong macro-avoidance at Thanet from Skov *et al.* (2018), and at other sites (Dierschke *et al.* 2016), it was felt appropriate to use the same value as recommended for large gulls.	Following the logic of Cook *et al.* (2014), given strong evidence of high macro-avoidance in northern gannets from a variety of sites (Dierschke *et al.* 2016), we feel that it is unlikely that the total avoidance rate for northern gannet would be less than that for large gulls.
	3	NA		As no collisions involving northern gannets were recorded as part of the ORJIP BCA study, it was not possible to compare predicted and expected collision rates. Furthermore, given clear differences in the flight height distributions of northern gannet and large gulls (Johnston *et al.* 2014) it is unlikely to be appropriate to base any value on that for large gulls, as we have done for Option 1 of the Band model. Consequently, in the absence of other data, it is still not possible to recommend a suitable avoidance rate for Option 3 of the Band CRM for northern gannet.
Black-legged kittiwake	1	0.990	Calculated using equation 1 by comparing the observed collision rate to predicted collision estimate 3 (figure 2). The predicted collision rate in the absence of avoidance behaviour was estimated using site-specific estimates of flight height and speed and pre-construction density	Avoidance rates for black-legged kittiwake which were suitable for use with Option 1 of the Band model ranged from 0.952 – 0.998 (tables 10 & 11). Based on the observed collision rate, a rate of 0.952, derived using generic bird flight data and pre-construction density estimates was felt to be overly-precautionary.

			estimates. The flux rate was estimated using straight line speed and the probability of a bird colliding was estimated using the true speed.	The most appropriate approach for deriving avoidance rates using the data collected by the ORJIP BCA project was felt to be the use of straight line speed to estimate flux rate and true speed to estimate the probability of collision. This resulted in estimated avoidance rates of 0.990 based on the pre-construction density estimates and 0.998 based on the post-construction density estimates (table 11). This compared to a rate of 0.991 derived using post-construction density estimates and generic bird data (table 10). Consequently, 0.990 was selected as the most precautionary of the realistic estimated values. Furthermore, as black-legged kittiwake are believed to show little change in numbers in response to the presence of a wind farm (Dierschke *et al.* 2016), this was considered a realistic value for total avoidance.
	3	0.980	Calculated using equation 1 to compare the observed collision rate to the collision rate estimated using generic bird flight data and post-construction density estimates.	Avoidance rates for black-legged kittiwake which were suitable for use with Option 3 of the Band model ranged from 0.891 – 0.998 (tables 10 & 11). Based on the observed collision rate, a rate of 0.891, derived using generic bird flight data and pre-construction density estimates was felt to be overly-precautionary. The most appropriate approach for deriving avoidance rates using the data collected by the ORJIP BCA project was felt to be the use of straight line speed to estimate flux rate and true speed to estimate the probability of collision. This resulted in estimated

				avoidance rates of 0.982 based on the pre-construction density estimates and 0.996 based on the post-construction density estimates (table 11). This compared to a rate of 0.980 derived using post-construction density estimates and generic bird data (table 10). As a consequence of the notable differences between the observed and generic flight height distributions, the rate of 0.980 was felt to be the most precautionary of the realistic values. Whilst this is based on post-construction density estimates, as black-legged kittiwakes do not appear to show a noticeable change in density in response to the presence of an offshore wind farm (Dierschke *et al.* 2016), this is felt to be a realistic value for total avoidance.
Lesser black-backed gull	1	0.995	The large gulls recorded colliding with turbines during the ORJIP BCA project could not be identified to species level. Consequently, a large gull avoidance rates was estimated by comparing the observed large gull collision rate to the sum of the lesser black-backed, herring and great black-backed collision rates predicted in the absence of avoidance behaviour using equation 1. Predicted collision rates were estimated using post-construction density estimates and generic bird flight data for each species.	Avoidance rates for large gulls which were suitable for use with Option 1 of the Band model ranged from 0.994 – 0.999 (tables 10 & 11). Those based on site specific data were felt to be insufficiently precautionary (table 11). This is likely to reflect the high proportion of birds reported at collision risk height by Skov *et al.* (2018) in comparison to previous studies (Johnston *et al.* 2014; Johnston & Cook 2016; Corman & Garthe 2014; Borkenhagen *et al.* 2018; Ross-Smith *et al.* 2016). Given the uncertainty this introduces, it was felt that for large gulls, the avoidance rates derived using generic data were most appropriate (table 10). Using the generic data avoidance rates of 0.994 using the pre-construction density estimates and 0.995 using the post-construction density estimates

				were calculated. Of these, the estimate of 0.995 derived using post construction density estimates was felt to be most appropriate because density estimates were based on multiple years' data and, gulls may be attracted to wind farms following construction (Dierschke *et al.* 2016).
	3	0.993	The large gulls recorded colliding with turbines during the ORJIP BCA project could not be identified to species level. Consequently, a large gull avoidance rates was estimated by comparing the observed large gull collision rate to the sum of the lesser black-backed, herring and great black-backed collision rates predicted in the absence of avoidance behaviour using equation 1. Predicted collision rates were estimated using post-construction density estimates and generic bird flight data for each species.	Avoidance rates for large gulls which were suitable for use with Option 3 of the Band model ranged from 0.991 – 0.999 (tables 10 & 11). Those based on site specific data were felt to be insufficiently precautionary (table 11). This is likely to reflect the high proportion of birds reported at collision risk height by Skov *et al.* (2018) in comparison to previous studies (Johnston *et al.* 2014; Johnston & Cook 2016; Corman & Garthe 2014; Borkenhagen *et al.* 2018; Ross-Smith *et al.* 2016). Given the uncertainty this introduces, it was felt that for large gulls, the avoidance rates derived using generic data were most appropriate (table 10). Using the generic data avoidance rates of 0.991 using the pre-construction density estimates and 0.993 using the post-construction density estimates were calculated. Of these, the estimate of 0.993 derived using post construction density estimates was felt to be most appropriate because density estimates were based on multiple years' data and, gulls may be attracted to wind farms following construction (Dierschke *et al.* 2016).
Herring gull	1	0.995	The large gulls recorded colliding with turbines during the ORJIP BCA project could not be identified to species level.	Avoidance rates for large gulls which were suitable for use with Option 1 of the Band model ranged from 0.994 – 0.999 (tables 10

			Consequently, a large gull avoidance rates was estimated by comparing the observed large gull collision rate to the sum of the lesser black-backed, herring and great black-backed collision rates predicted in the absence of avoidance behaviour using equation 1. Predicted collision rates were estimated using post-construction density estimates and generic bird flight data for each species.	& 11). Those based on site specific data were felt to be insufficiently precautionary (table 11). This is likely to reflect the high proportion of birds reported at collision risk height by Skov *et al.* (2018) in comparison to previous studies (Johnston *et al.* 2014; Johnston & Cook 2016; Corman & Garthe 2014; Borkenhagen *et al.* 2018; Ross-Smith *et al.* 2016). Given the uncertainty this introduces, it was felt that for large gulls, the avoidance rates derived using generic data were most appropriate (table 10). Using the generic data avoidance rates of 0.994 using the pre-construction density estimates and 0.995 using the post-construction density estimates were calculated. Of these, the estimate of 0.995 derived using post construction density estimates was felt to be most appropriate because density estimates were based on multiple years' data and, gulls may be attracted to wind farms following construction (Dierschke *et al.* 2016).
	3	0.993	The large gulls recorded colliding with turbines during the ORJIP BCA project could not be identified to species level. Consequently, a large gull avoidance rates was estimated by comparing the observed large gull collision rate to the sum of the lesser black-backed, herring and great black-backed collision rates predicted in the absence of avoidance behaviour using equation 1.	Avoidance rates for large gulls which were suitable for use with Option 3 of the Band model ranged from 0.991 – 0.999 (tables 10 & 11). Those based on site specific data were felt to be insufficiently precautionary (table 11). This is likely to reflect the high proportion of birds reported at collision risk height by Skov *et al.* (2018) in comparison to previous studies (Johnston *et al.* 2014; Johnston & Cook 2016; Corman & Garthe 2014; Borkenhagen *et al.* 2018; Ross-Smith *et al.* 2016). Given the uncertainty this introduces, it was felt that for large gulls, the avoidance

			Predicted collision rates were estimated using post-construction density estimates and generic bird flight data for each species.	rates derived using generic data were most appropriate (table 10). Using the generic data avoidance rates of 0.991 using the pre-construction density estimates and 0.993 using the post-construction density estimates were calculated. Of these, the estimate of 0.993 derived using post construction density estimates was felt to be most appropriate because density estimates were based on multiple years' data and, gulls may be attracted to wind farms following construction (Dierschke *et al.* 2016).
Great black-backed gull	1	0.995	The large gulls recorded colliding with turbines during the ORJIP BCA project could not be identified to species level. Consequently, a large gull avoidance rates was estimated by comparing the observed large gull collision rate to the sum of the lesser black-backed, herring and great black-backed collision rates predicted in the absence of avoidance behaviour using equation 1. Predicted collision rates were estimated using post-construction density estimates and generic bird flight data for each species.	Avoidance rates for large gulls which were suitable for use with Option 1 of the Band model ranged from 0.994 – 0.999 (tables 10 & 11). Those based on site specific data were felt to be insufficiently precautionary (table 11). This is likely to reflect the high proportion of birds reported at collision risk height by Skov *et al.* (2018) in comparison to previous studies (Johnston *et al.* 2014; Johnston & Cook 2016; Corman & Garthe 2014; Borkenhagen *et al.* 2018; Ross-Smith *et al.* 2016). Given the uncertainty this introduces, it was felt that for large gulls, the avoidance rates derived using generic data were most appropriate (table 10). Using the generic data avoidance rates of 0.994 using the pre-construction density estimates and 0.995 using the post-construction density estimates were calculated. Of these, the estimate of 0.995 derived using post construction density estimates was felt to be most appropriate because density estimates were based on multiple years' data and, gulls may be

				attracted to wind farms following construction (Dierschke *et al.* 2016).
	3	0.993	The large gulls recorded colliding with turbines during the ORJIP BCA project could not be identified to species level. Consequently, a large gull avoidance rates was estimated by comparing the observed large gull collision rate to the sum of the lesser black-backed, herring and great black-backed collision rates predicted in the absence of avoidance behaviour using equation 1. Predicted collision rates were estimated using post-construction density estimates and generic bird flight data for each species.	Avoidance rates for large gulls which were suitable for use with Option 3 of the Band model ranged from 0.991 – 0.999 (tables 10 & 11). Those based on site specific data were felt to be insufficiently precautionary (table 11). This is likely to reflect the high proportion of birds reported at collision risk height by Skov *et al.* (2018) in comparison to previous studies (Johnston *et al.* 2014; Johnston & Cook 2016; Corman & Garthe 2014; Borkenhagen *et al.* 2018; Ross-Smith *et al.* 2016). Given the uncertainty this introduces, it was felt that for large gulls, the avoidance rates derived using generic data were most appropriate (table 10). Using the generic data avoidance rates of 0.991 using the pre-construction density estimates and 0.993 using the post-construction density estimates were calculated. Of these, the estimate of 0.993 derived using post construction density estimates was felt to be most appropriate because density estimates were based on multiple years' data and, gulls may be attracted to wind farms following construction (Dierschke *et al.* 2016).

4.6 Recommendations for future work

The ORJIP BCA study has collected detailed data on the movements and behaviour of birds within an operational offshore wind farm at a scale never previously attempted. Whilst it has answered many questions about the movements of birds within a wind farm and how they avoid collisions, it has raised many more. In particular, by attempting to derive seabird avoidance rates based on observed behaviour, it has highlighted the potential consequences for the consenting process of the discrepancy between model assumptions and how birds utilise wind farms. The lack of validation for collision risk models has been a key problem for some time (Masden & Cook 2016), with some evidence that modelled predictions may be a poor match for observed collision rates (Ferrer *et al.* 2012; de Lucas *et al.* 2008).

Recommendations for future work fall into two categories – how lessons learned from the ORJIP BCA study can be incorporated into similar studies in the future and how the data can be used to improve and develop models of collision risk.

4.6.1 Lessons learned

Deriving total macro-avoidance rates from the ORJIP BCA study proved challenging. There are two reasons for this. Firstly, it was not possible to collect seabird density data as part of this project. This meant that it was not possible to assess changes in the numbers of birds present between the pre- and post-construction periods, which may be expected in response to displacement effects. Ideally, future studies should seek to collect information about seabird density in parallel to fine-scale behavioural data in order to better understand any displacement effects. Secondly, whilst the study was well set up to look at the impact of barrier effects, attempts to estimate these may have been confounded by the presence of fishing vessels on the edge of the wind farm. Species like gulls and northern gannets may be attracted to fishing vessels over significant distances (Votier *et al.* 2010). Consequently, there is a risk that some birds may have been responding to the presence of fishing vessels on the edge of the wind farm, as reported by observers in both the ORJIP BCA study (Skov *et al.* 2018) and elsewhere (Krijgsveld *et al.* 2011). In order to fully account for macro-avoidance behaviour, it would be valuable if future studies were able to develop spatial modelling approaches which could account for changes in the density of birds between the pre- and post-construction periods. Ideally, such approaches would also account for the movement of fishing vessels, which may attract birds and, thus, give a misleading impression of the impact of barrier effects. Combining digital aerial survey data with radar data may prove a useful approach for this with digital aerial survey data able to offer information on distribution and radar able to offer information on flight paths and speeds.

In the estimation of macro-avoidance behaviour, Skov *et al.* (2018), highlight the different components of uncertainty that may contribute to the total uncertainty surrounding the final macro-avoidance rates. However, in common with the suggested approach for estimating uncertainty set out by Band (2012), these are largely based on expert judgement. Future studies should consider how the need for expert judgement in relation to the estimation of uncertainty can be overcome. For example, spatial and/or temporal modelling approaches could be used in order to determine the level of uncertainty introduced as a result of factors such as the presence of fishing vessels and weather conditions.

The ORJIP BCA study has supported previous suggestions that a significant proportion of avoidance behaviour may take place at the meso-scale (Cook *et al.* 2014). Gathering additional data at this scale is likely to be extremely valuable. However, as such a high proportion of avoidance behaviour occurs at the meso-scale, collecting data on avoidance behaviour at a micro-scale is much more challenging. Whilst the ORJIP BCA study has collected the most comprehensive dataset on micro-avoidance to date, it is clear that much more data are required in order to fully understand micro-avoidance behaviour, particularly at an inter-specific level. Future studies should consider approaches that will maximise the collection of data at the micro-scale. A key gap relates to our understanding of how avoidance behaviour may differ between day and night. Consequently, future studies should make use of thermal cameras to enable collisions to be recorded during the dark.

Estimates of seabird flight heights were based on measurements collected using laser rangefinders. Such measurements may be biased against low flying birds and, consequently, overestimate the number at risk of collision (Borkenhagen *et al.* 2018). Consequently, it is difficult to ascertain the extent to which the flight height distributions obtained as part of the ORJIP BCA study, which are radically different to generic distributions collected from elsewhere (Figure 4), reflect this bias and/or site-specific factors. Ideally, in order to make the most of these data, future studies should collect flight information both inside and outside wind farms using multiple platforms concurrently in order to better understand any potential biases.

4.6.2 Collision risk model development

The disparity between the number of collisions predicted by the Band CRM relative to those observed (Figure 10) highlights the need to start looking at ways to incorporate realistic assessments of bird behaviour into collision risk models. Ideally, we should be asking how to make the model better fit the data, rather than how to make the data fit the model, e.g. through the use of correction factors. At the same time, there is a need to balance the detailed data collected as part of this study with the more generic data typically available as part of pre-construction impact assessments. Below, we highlight areas where we feel refinements could be made to more accurately assess collision risk.

The Band CRM estimates the number of birds at risk of collision by predicting the number of birds likely to pass through the turbine rotor-swept area per second (Band 2012). This is based on an estimate of bird flight speed, to determine how long it would take a bird to pass through the rotor, and density, in order to estimate the number of birds available to pass through the rotor per unit time. As highlighted above (Table 3 and section 4.1) the generic estimates of flight speed far exceed those measured as part of the ORJIP BCA Study, meaning, the use of generic data results in significantly higher estimated flux rates (Table 6). Furthermore, the Band CRM assumes that birds fly at a constant speed, perpendicular to the rotor. If these assumptions are not met, for example, if the flight path taken by a bird is not perpendicular as it approaches the turbine, this may also have significant implications for the number of birds estimated to be at risk of collision (Table 6). For example, birds engaged in area-restricted search foraging behaviour, may be less likely to be travelling in a straight line than those commuting between foraging areas and breeding colonies (Votier *et al.* 2013). Furthermore, rather than approaching the turbine rotor at a perpendicular angle, as assumed by the Band CRM, the ORJIP BCA study noted a significant number of birds flying in parallel to the turbine blades. Models should be refined in order to account for site specific differences in bird behaviour (e.g. commuting vs. foraging flight) as these differences are likely to have a substantial impact on collision risk.

In contrast, the data collected as part of the ORJIP BCA study suggests that the Band CRM may underestimate the probability of a bird passing through a turbine colliding with the blades. Using site-specific data, the probability of collision was estimated at between 0.07 – 0.12 (Table 7), depending on the species and approach used. However, the data collected as part of the ORJIP BCA showed six of the 15 birds that crossed the rotor swept area colliding, implying a greater probability of collision in the region of 0.4, albeit based on limited data. Again, the ORJIP BCA study is the first, to our knowledge, to offer quantitative data regarding the number of birds crossing the rotor-swept area which collide with the turbine blades. One potential reason for this discrepancy is that birds crossing a rotor swept area at an oblique angle may be more likely to collide than those making a perpendicular approach to the rotor (Band 2012). Band (2012) argues that this effect can be offset by the fact that the elliptical shape of the rotor means that birds are less likely to enter the rotor swept area. Subsequent analyses have shown that accounting for an oblique approach may result in a substantially increased collision risk (Christie & Urquhart 2015). Models should be refined in order to more accurately reflect bird movement patterns and account for an oblique approach to the turbine rotors. This will necessitate data describing bird movement patterns being collected as part of EIAs. This could be achieved either through the use of tracking data or, by examination of images collected by digital aerial surveys.

The analyses presented above suggest that the Band CRM may give a misleading impression of absolute collision risk, with predicted collision rates higher than those observed, even after accounting for avoidance behaviour (Figure 10). However, it should be acknowledged that "*all models are wrong, but some are useful*" (Box *et al.* 2005). In the context of collision risk modelling, at present the Band CRM may reflect our best approach for assessing collision risk. However, ideally the predictions should be treated in relative rather than absolute terms. As more data become available, for example, through radar or tracking studies, these data should be used to refine the models in order to more accurately account for bird movement and behaviour. These model refinements (e.g. accounting for differences in behaviour and oblique approaches to wind turbines) are likely to reduce the error associated model simplifications (Band 2012) meaning that the correction factor referred to as an avoidance rate can be more closely aligned with the empirical avoidance rates calculated by the ORJIP BCA study.

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7 Appendix 1 Estimating Avoidance Rates with Confidence Intervals

In light of the development of stochastic collision risk models (Masden 2015; McGregor *et al.* 2018) the project steering group requested BTO to provide estimates of avoidance rates with associated confidence intervals. Following the logic set out in Table 12, we estimate avoidance rates and their associated confidence intervals for black-legged kittiwake and large gulls using post-construction bird density estimates, generic flight height information and estimates of flux rates derived using straight line speed and estimates of the probability of collision and collision integral derived using true speed.

Following the methodology set out in Masden (2015) and McGregor *et al.* (2018) we use a Monte Carlo simulation approach in order to estimate the number of collisions expected in the absence of avoidance behaviour. Following the sensitivity analyses set out in Masden (2015), bird density, bird flight speed and bird flight height distribution were identified as the parameters most likely to affect estimates of collision in the absence of avoidance behaviour. Accordingly, these were randomly sampled, as set out below, and the model was run for 1000 iterations. For each of these 1000 iterations an avoidance rate was calculated using equation 1. The median avoidance rate and 95% confidence intervals were then calculated from these values for black-legged kittiwake and large gulls.

7.1 Bird Density

Estimates of the number of birds likely to collide in the absence of avoidance behaviour can be very sensitive to the estimate of bird density (Masden 2015). Consequently, it is important that simulated density estimates fall within a distribution which is a realistic representation of the birds present at a site. If this distribution is positively biased (i.e. high densities are over-represented in the data) then the avoidance rates that are derived will be over-estimated because the predicted collision rate in the absence of avoidance will have been over-estimated. Conversely, for the same reason, if the distribution used is negatively biased (i.e. low densities are over-represented in the data) the derived avoidance rates will be under-estimated.

The densities presented in Royal Haskoning (2013) do not include estimates of uncertainty. Consequently, it is necessary to make assumptions about the distributional form of the data. We converted the mean post-construction density estimates in table 2 into estimates of the total number of birds within the area covered by the cameras shown in figure 1. For each iteration of the analysis, we then randomly sampled the number of birds within the area covered by the cameras from a Poisson distribution and converted this back into a density estimate for the remaining steps of the analysis. Distributions of density estimates generated in this way appeared a reasonable approximation for the distributions of density estimates obtained from an adjacent site (APEM 2018).

7.2 Bird Speed

As described above, bird speed can be estimated based on the straight-line distance travelled by a bird or, by based on point estimates of speed. Speed is incorporated in the Band model twice – firstly in order to estimate the flux rate and, secondly to estimate the probability of a bird crossing a turbine rotor swept area and colliding. As described above, the straight line speed is used in order to estimate the flux rate and the point estimates of speed are used to estimate the probability of a bird crossing a rotor swept area and colliding.

Masden (2015) and McGregor *et al.* (2018) both use a normal distribution in order to estimate species flight speeds. However, this may generate estimates of flight speed of less than 0 m/s. Following Ross-Smith *et al.* (2016), we assume that birds in flight are travelling at a speed of 4 km/h, approximately 1.1 m/s. Consequently, we use a truncated normal distribution with a minimum value of 1.1 m/s in order to generate estimates of flight speed in each iteration of our analysis. Mean and standard deviations of straight line and point estimates of species flight speeds are generated from the data collected as part of the ORJIP BCA study and presented in table A1.

Table A1. Mean and standard deviation of species true and straight-line speeds.

	True Flight Speed	**Straight Line Flight Speed**
Black-legged Kittiwake	8.57 (SD 3.47)	6.68 (SD 3.49)
Lesser Black-backed Gull	10.44 (SD 4.25)	8.35 (SD 4.90)
Herring Gull	9.75 (SD 3.44)	8.04 (SD 3.84)
Great Black-backed Gull	10.00 (SD 4.39)	8.52 (SD 5.10)

7.3 Bird Flight Height

Species flight heights are treated differently by the basic and extended Band models (Band 2012). In the case of the basic Band model a single value, the proportion of birds at collision risk height is used. In the case of the extended Band model, a continuous distribution of the proportion of birds at different heights is used. Such distributions can be derived from the survey data collected to support offshore wind farm EIAs (Johnston *et al.* 2014). In order to estimate avoidance rates suitable for use in the basic and extended Band models, we used the generic flight height distributions derived by Johnston *et al.* (2014). The analyses of Johnston *et al.* (2014) used a bootstrapping procedure in order to generate a median flight height distribution and associated confidence intervals for each species. Each bootstrap represented a modelled distribution for random sample of the data for each species. For each iteration of the analysis used to predict the number of birds colliding in the absence of avoidance behaviour we randomly selected one of these bootstrap flight height distributions. In the case of the basic Band model, we used this distribution to estimate the proportion of birds at collision risk height. In the case of the extended Band model this random selection was used as the flight height distribution when calculating the collision integral.

7.4 Avoidance Rates with Confidence Intervals

Following the Monte Carlo simulation exercise described above, the avoidance rates derived were in broad agreement with those outlined in table 12. Whilst there was some discrepancy between the values reported in table 12 and the median values derived using Monte Carlo simulations, the values in table 12 were within the 95% confidence intervals of the new values (Table A2). This discrepancy relates to how the flight height distributions were used when deriving the avoidance rates.

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 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 (Johnston *et al.* 2014; Masden 2015). The values from table 12 are derived using the best fit distribution and the median values in table A2 are derived using bootstrapped values.

We recommend the values in table A2 for use in stochastic collision risk models.

Table A2. Avoidance rates and 95% Confidence intervals derived using a Monte Carlo simulation approach for Black-legged Kittiwake and Large Gulls.

	Basic Band Model	**Extended Band Model**
Black-legged Kittiwake	0.994 (0.976 – 0.998)	0.970 (0.871 – 0.989)
Large Gulls (Lesser Black-backed Gull, Herring Gull, Great Black-backed Gull)	0.997 (0.992 – 0.999)	0.990 (0.974 – 0.995)

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Research article

Towards a framework for quantifying the population-level consequences of anthropogenic pressures on the environment: The case of seabirds and windfarms

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Offshore wind farm

1. Introduction

Human activity is having a substantial impact on the natural environment (Vitousek et al., 1997) and where this affects biodiversity it can have knock-on consequences for ecosystem stability (Hautier et al., 2015). Whilst there is a tendency to focus on rare species, impacts on more common species can also have significant consequences at a population or ecosystem level (Gaston and Fuller, 2008; Inger et al., 2015). Consequently, there is a need to develop methods and approaches which can be used to quantify the impacts on individuals in order to understand their consequences at a population-level, whilst accounting for any uncertainty in the available data. A key challenge is to ensure that any methods can be incorporated into policy decision frameworks, such that they provide clear guidance on the relative risks of realistic management options (Bakker and Doak, 2009). This concept applies equally to interventions that are expected to have a positive effect on populations, such as population translocation (e.g. Canessa et al., 2016) or reserve designation (e.g. Fenberg et al., 2012), as to those with negative impacts. Integrating ecological processes with the social and economic goals associated with resource management represents a significant challenge because of the complexity of the systems concerned (Dale and Beyeler, 2001).

The pace of change has been particularly noticeable in the marine environment (Halpern et al., 2015), with key concerns about the rapid increase in the number of offshore structures and their potential to impact wildlife (e.g. Bailey et al., 2014). The development of large offshore wind farms, often seen as a key part of strategies to reduce reliance on fossil fuels (Toke, 2011), has been particularly significant. These potentially have a number of negative effects on seabird populations including: displacement from foraging areas, collision with turbines, and the wind farm acting as a barrier to migrating or commuting birds (Everaert and Stienen, 2007; Masden et al., 2009; Vanermen et al., 2013). Each impact can be estimated prior to construction by characterising the environment (e.g. Johnston et al., 2015) and applying tools, such as collision risk models (Masden and Cook, 2016). Whilst these approaches can be used to assess the total number of individual birds which may be affected by a development, understanding the implications at a population level is more complex (Maclean et al., 2014).

Tools such as Population Viability Analysis (PVA) are widely used in population management, for example; in order to predict the likely success of different conservation interventions (e.g.Lindenmayer and Possingham, 1996), to determine whether levels of population harvesting are sustainable (e.g. Tufto et al., 1999) and to investigate the efficacy of pest control programmes (e.g. Brook et al., 2003). However, predicting the viability of populations after interventions requires some understanding of the underlying demography of the population, so that a population model can be constructed (e.g. Oro et al., 2004). Our confidence in estimates of the demographic parameters varies considerably by species and population, and must be accounted for if any assessment process is to be effective (Horswill and Robinson, 2015).

In the context of wind farms, for example, a key aim of PVA may be to demonstrate whether or not the level of additional mortality, for example resulting from collisions with turbines, will have an adverse effect at a population level. The predicted impacts can be

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incorporated into PVA models as an additional level of harvesting to understand the population-level consequences associated with the presence of a wind farm. From these models a range of metrics can be calculated, such as population size at time, t (N_t), which may reflect the end of the operational life of the wind farm, the (mean) population growth rate (λ), or probability of extinction (Brook et al., 2000; Ellner et al., 2002) (Fig. 1). These metrics have been applied to assess the impact of management interventions in a variety of different ways. These include, comparing the impact of timber management strategies on the probability of extinction (Lindenmayer and Possingham, 1996), the probability of detecting a given population decline under different conservation strategies (Thompson et al., 2000) and the probability of a population of a reintroduced species being above a given size through time (Wood et al., 2007). When used in a management context, the implications of different (or no) interventions must be presented in a way which is clear so that they can be easily interpreted by decision makers. This requires that these metrics (e.g. population size, population growth rate or probability of a given outcome) be assessed in relation to some criterion, which may be a simple threshold defining a desirable outcome (which may be no effect), perhaps relative to a population not subject to any intervention, or a probabilistic assessment of a range of outcomes.

Recently, Green et al. (2016) summarised three criteria which may be derived from PVAs in order to quantify population level effects arising as a result of the impact of offshore wind farms – Acceptable Biological Change (ABC), Decline Probability Difference (DPD) and Counterfactual of Impacted to Unimpacted Population (CIU) (Table 1). However, interpreting these criteria can be fraught with difficulty and has been the subject of much debate between stakeholders (e.g. Court of Session of Scotland, (2016)), contributing to costly delays in the decision making process (Masden et al., 2015).

The theoretical basis underpinning the models and criteria used to make decisions about the population level effects of management interventions has been criticised (e.g. Coulson et al., 2001; Green et al., 2016). To inform their use in the decision-making process there is a need for an assessment of the strengths and weaknesses of these different approaches. We present a unified framework (Fig. 1) for assessing the impact of developments at a population level using criteria informed by metrics from Leslie matrix models (LMMs) of populations with and without the impact of a management intervention. We test this framework using the example of an offshore wind farm, which may affect survival rates of a seabird population through collision-related mortality or productivity rates through displacement (Drewitt and Langston, 2006). We aim to derive measures within this framework that reflect the impact of the intervention in a clear and unequivocal fashion, rather than differences in model properties (e.g. density dependence, stochasticity) or knowledge of the values of demographic parameters. We discuss how conclusions about the acceptability of any impacts may be influenced by uncertainties either in data availability, or in the modelling process, and what implications this may have for the consenting process.

2. Methods

We considered a generic seabird species (Table 2) with life history traits informed by a recent review of seabird demography (Horswill and Robinson, 2015) and constructed Leslie matrix models with four age-classes, and reproduction confined to the adult age class (≥four years). To simulate the impacts associated with the presence of a wind farm, we modelled a broad range of reductions in productivity of 0–40% and increases in mortality of 0–40% across all age classes, taken as reflective of the range of impacts considered in Environmental Impact Assessments. Models were run for 25 years, taken as the typical life span of an offshore wind farm. All analyses were carried out using R 3.1.1 (R Core Team, 2015).

2.1. Decision criteria

In order to understand how decision criteria behave when applied to the three different metrics we test two variants of each,

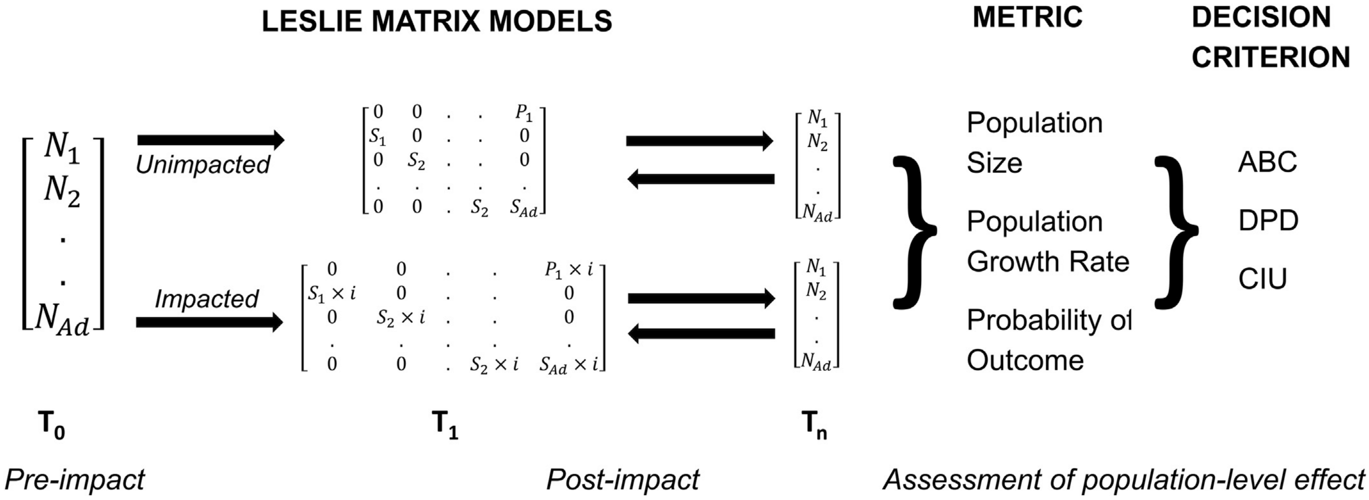

Fig. 1. Framework used to derive decision criteria to assess the population level effects associated with management interventions. At T_0, pre-intervention, there is a population with N individuals in each age class. Using a "matched runs" approach, population changes are projected over the lifetime of the project from T_1 to T_n. Two populations are modelled, the first in which no impact from the management intervention is assumed with demographic parameters S_1 (first year survival), S_2 (sub-adult survival), S_{Ad} (Adult survival) and P_1 (Productivity). The second population is modelled with each parameter modified by an impact associated with the intervention, i. For simplicity, we assume equal impacts for each parameter. In practice, this is unlikely to be the case. Three metrics can be derived from these models, population size, population growth rate and the probability of an outcome (e.g. extinction, or a population decline of a given magnitude). The metrics from the impacted and unimpacted populations can then be compared using three decision criteria – Acceptable Biological Change (ABC), Decline Probability Difference (DPD) and Counterfactual of Impacted to Unimpacted Populations (CIU) –to quantify the population-level effect associated with the management intervention.

Table 1
Description of decision criteria used to assess population responses to impacts of management interventions.

Decision criterion	Description
Acceptable Biological Change (ABC)	ABC was set out as a method for assessing the population impact of an offshore wind farm by Marine Scotland, 2015. Using terminology from the Intergovernmental Panel on Climate Change (IPCC), ABC allows for a change of up to one-third in the probability of a defined target being achieved as a result of the impact of a management intervention, which is classed by the IPCC as representing an outcome that is 'as likely as not' (Mastrandrea et al., 2010). This could be applied to different population model metrics, for example, population size (ABC_n) or the probability of the population growth rate (λ) being less than 1 ($ABC_{\lambda<1}$). These decision criteria are binary with impacts assessed as 1 (acceptable) or 0 (not acceptable). Variants include reduced uncertainty ABC (ruABC), whereby data from a regional population is used to help reduce model prediction uncertainty (JNCC & SNH, 2014).
Decline Probability Difference (DPD)	Using DPD, the probability of a population declining is estimated for models with and without a management intervention. This probability could be expressed in relation to growth rate, e.g. probability that growth rate <1 ($DPD_{\lambda<1}$), or population size, e.g. probability that population declines by 25% ($DPD_{\Delta 25}$). The decision criterion is then assessed in relation to the intervention and non-intervention scenarios. These decision criteria are assessed over a scale from 0 (no impact) to 1 (severe impact).
Counterfactual of Impacted and Unimpacted Populations (CIU)	Population models can be used to estimate the size, or growth rate, of a population through time both with and without the impact of a management intervention. The decision criterion is then assessed in relation to the ratio of the metrics from the two populations for example, population size (CIU_n) or growth rate (CIU_λ). This ratio could be estimated at a fixed point in time or at a series of intervals throughout the life time of a project. These decision criteria are assessed over a scale from 0 (severe impact) to 1 (no impact). CIU_n is equivalent to CPS in Green et al. (2016). However, as described above counterfactuals may be derived from different metrics including population size, growth rate and probability of extinction. Using the acronym CIU serves to emphasise the link between criteria derived from these metrics. We also feel that it serves to remind the user of how the criteria relate to the impacted and unimpacted populations.

one in relation to population size and one in relation to population growth rate. For ABC, we test whether (i) the median impacted population size exceeds the population size that is 66.7% likely to be equalled or exceeded in the unimpacted population (ABC_n) after 25 years, following the IPCC classification of an outcome which is as likely as not (Mastrandrea et al., 2010), and (ii) whether the probability of the population growth rate of the impacted population being less than one exceeds the probability of the population growth rate of the unimpacted population being less than one by more than one third ($ABC_{\lambda<1}$). For DPD, we consider (i) the difference in the probabilities of the growth rate being less than one between the impacted and unimpacted populations ($DPD_{\lambda<1}$) and (ii) the change in probability of a 25% population decline over 25 years ($DPD_{\Delta 25}$) between the impacted and unimpacted populations. For CIU we consider (i) the ratio of the growth rates from the impacted and unimpacted populations (CIU_λ) and (ii) the ratio of the population sizes after 25 years (CIU_n).

2.2. Model properties

It is important to understand how the different properties of the models may influence the conclusions drawn from decision criteria which are derived from them. Consequently, we investigated the influence of three properties which may be applied to population models – density dependence, stochasticity and the use of "matched runs", whereby, prior to any impacts being applied, identical demographic parameters are used for both the impacted and unimpacted populations (Fig. 1). In these initial analyses, we considered our seabird population to be stable.

Table 2
Demographic parameters incorporated in models for a stable, declining or increasing population of a seabird population based on a review of seabird demography (Horswill and Robinson, 2015).

	Increasing	Stable	Decreasing
Initial Population Size	10,000 Breeding adults		
Sex Ratio	0.5		
Age at first breeding	4		
Adult Survival	0.89 (±0.085)	0.89 (±0.085)	0.866 (±0.024)
Immature Survival	0.850 (±0.200)	0.741 (±0.200)	0.741 (±0.206)
First year Survival	0.441 (±0.200)	0.441 (±0.200)	0.358 (±0.219)
Productivity	1.590 (±0.175)	1.030 (±0.175)	0.920 (±0.175)

As a first step, we compare decision criteria derived using deterministic and stochastic models assuming a 2.5% increase in mortality, selected following an assessment of the population level effects of impacts assessed using each decision criteria above. This initial comparison is only meaningful for CIU as the other decision criteria are based on probabilistic distributions outputs of population models. Stochasticity was incorporated into these models by drawing parameter values from a distribution defined by the mean and standard deviation of the values given for a stable population in Table 2. We focussed on environmental, as opposed to demographic, stochasticity as seabirds typically occur in large populations where the impact of demographic stochasticity is believed to be negligible (Sæther et al., 2005). In each iteration, for each year, a productivity rate and a survival rate for each age class was simulated. The survival rate must be bounded by 0 and 1; therefore it was sampled using a logit-link. Similarly, productivity cannot be less than 0, consequently it was sampled using a log-link. Models were run for 1000 iterations using a "matched runs" approach. From a programming standpoint, it may be more straightforward to not use a matched runs approach, i.e. to allow base demographic rates within a stochastic population model to vary between impacted and unimpacted populations. To understand the implications of this, we re-ran the analysis for CIU using both a matched and unmatched runs approach.

Seabird populations are known to be influenced by density dependence (Horswill et al. in review). We investigate the implications of assessing the population level effects of impacts using ABC, DPD and CIU when derived from models incorporating density dependence. To do this, we compare ABC, DPD and CIU calculated using density independent models, models with density dependent regulation of productivity and density dependent regulation of adult survival. A variety of functions can be used to represent density dependent responses, however, the Weibull function (Eq. (1)) has been found to be a realistic representation across a variety of species (Cury et al., 2011).

$$D = \mathrm{maxD} * \exp(-a * N^\wedge b) \qquad (1)$$

Where D is the demographic parameter under consideration (productivity or adult survival), $maxD$ is the biologically plausible maximum value for this parameter, informed by Horswill and Robinson (2015), *N* is the population size, a is a scale parameter and b is a shape parameter.

2.3. Model parameters

There may be uncertainty surrounding the status and trajectory of the population concerned, as is the case for many UK seabird populations (JNCC, 2015), so it is important that the decision criteria reflect the impact of the wind farm as opposed to the status of the population concerned. As a first step, we evaluated how each metric varied in response to current population status by adjusting the population models using biologically plausible values (Table 1) to give populations that were increasing or decreasing. We compare the values for the decision criteria derived using increasing or decreasing populations with those from a stable population.

Estimates of demographic rates are likely to vary between populations (e.g. Breton et al., 2006; Frederiksen et al., 2008) and may not be available for the population of interest. Consequently, it may be necessary to draw estimates for demographic parameters from elsewhere, introducing additional uncertainty into assessments of population level effects. We consider the impact of mis-specifying adult survival, first year survival, immature survival and productivity by 1% in relative terms, to give a measure of the sensitivity of the metric to each parameter. We do this using a stochastic model, assuming a stable population and a 2.5% increase in mortality or reduction in productivity. We vary each parameter in turn by 1% and calculate the percentage change in the metric (regardless of whether it is an increase or decrease).

Non-biological factors, such as sampling variance, may make a significant contribution to the total variance associated with demographic parameters (Gould and Nichols, 1998). In addition, these parameters may not be estimated over a sufficient time period in which to capture the true variability of any population (Lande et al., 2003). It is important to understand how the uncertainty

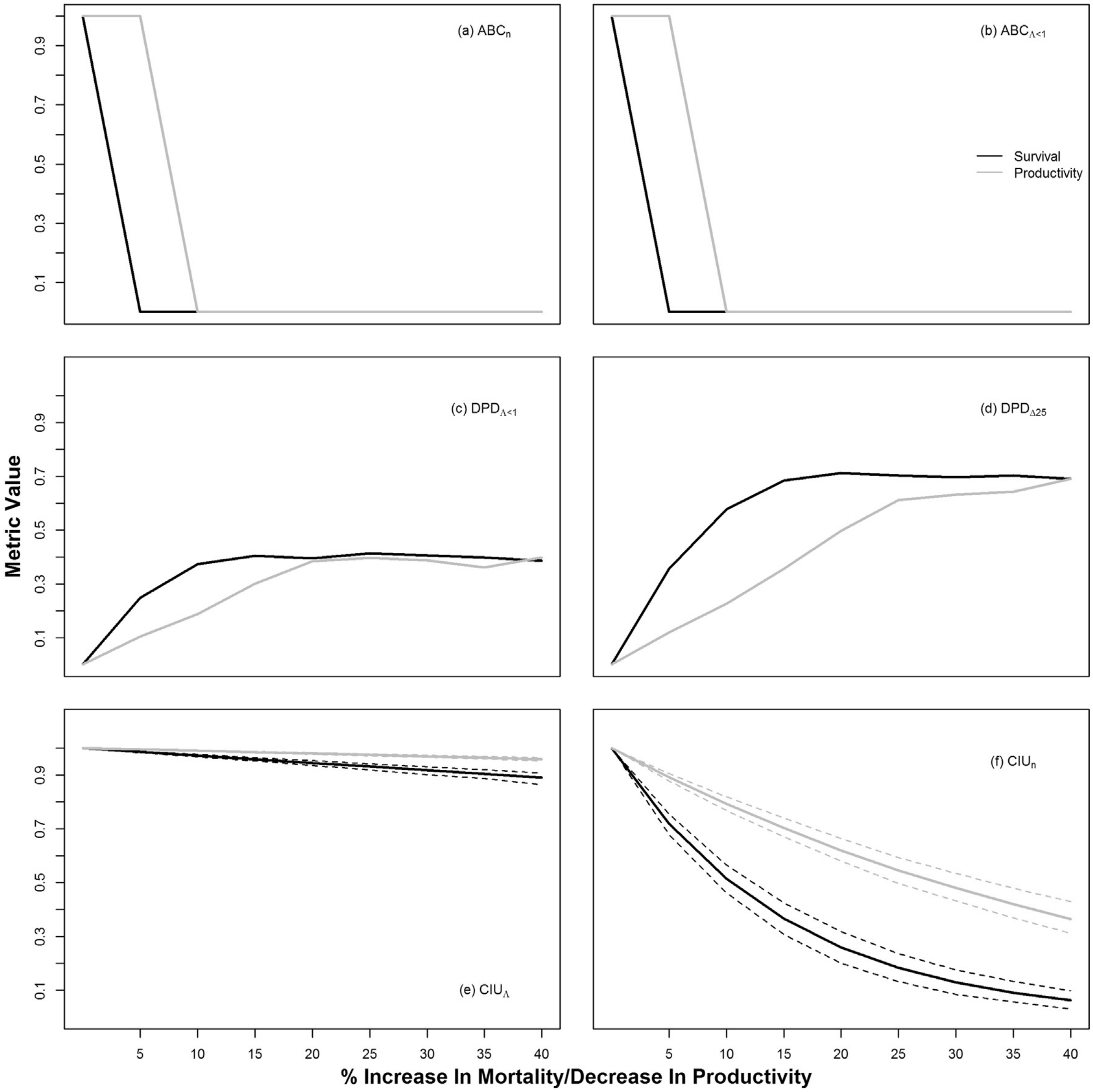

Fig. 2. Relationship between each metric derived from stochastic population models and impacts of increasing magnitude on survival or productivity. 95% Confidence Intervals indicated where appropriate: (a) Acceptable biological change applied to population size after 25 years (ABC_n); (b) Acceptable Biological Change applied to the probability of the growth rate being less than one ($ABC_{\lambda<1}$); (c) Change in probability of population growth rate being less than one ($DPD_{\lambda<1}$); (d) change in probability of a 25% decline over 25 years ($DPD_{\Delta 25}$); (e) Counterfactual of population growth rate (CIU_λ); (f) Counterfactual of population size after 25 years (CIU_n).

introduced by these factors may influence the conclusions drawn using each decision criteria. Again using a stochastic model, assuming a stable population and a 2.5% increase in mortality or reduction in productivity, we investigate the impact of varying the estimate of the standard deviation surrounding the adult survival rate. Using a fixed mean estimate for adult survival (0.89, see Table 2), we compare the values estimated using each decision criteria assuming standard deviations of 0.024, 0.044, 0.064 and 0.085, taken as realistic representations of the error associated with adult survival rates following the review of Horswill and Robinson (2015). Ideally, the decision criteria calculated should have similar values, regardless of the distribution used to estimate adult survival, indicating that populations are responding to the impacts associated with offshore wind farms, rather than uncertainty in demographic rates.

3. Results

The decision criteria considered indicated that where effects of a similar magnitude operated on survival or productivity (e.g. a 20% reduction in productivity or a 20% increase in mortality), effects on survival had a more significant impact at a population level (Fig. 2). The decision criteria showed different relationships with impacts of different magnitude. The nature of the ABC criteria meant that ABC_n (Fig. 2a) and $\mathrm{ABC}_{\lambda<1}$ (Fig. 2b) showed a binary relationship between the assessment of the population level effects and impacts of different magnitude. In contrast, $\mathrm{DPD}_{\lambda<1}$, DPD $_{\Delta 25}$ and CIU_n all showed curved relationships with impacts of increasing magnitude (Fig. 2c, d & f), whilst CIU_λ showed a linear relationship with impacts of increasing magnitude (Fig. 2e).

3.1. Model properties

Where stochastic, rather than deterministic, population models were used to derive the CIU decision criteria, the value of the decision criteria were marginally higher (Fig. 3a). However, this was more noticeable when population size (CIU_n) was used as the metric underpinning the decision criterion than was the case when growth rate (CIU_λ) was used. For both CIU_n and CIU_λ the median values of the decision criteria predicted were found to be greater when a matched runs approach was used (Fig. 3b and c). The reason for such a marked difference between Fig. 3b and c is that CIU_λ operates over a more restricted range than CIU_n (Fig. 2). It is also worth noting that the uncertainty surrounding the decision criteria was much greater when estimated using an unmatched runs approach. Incorporating density dependence into models resulted decision criteria which were suggestive of less severe population level effects (Fig. 3d).

3.2. Model parameters

Pre-construction population trend influenced the decision criteria derived assuming a 2.5% increase in mortality or reduction in productivity; whilst this influence was fairly negligible in the case of the CIU criteria, it was more noticeable in the case of the ABC and DPD criteria (Fig. 4a and b). Assuming impacts on survival or

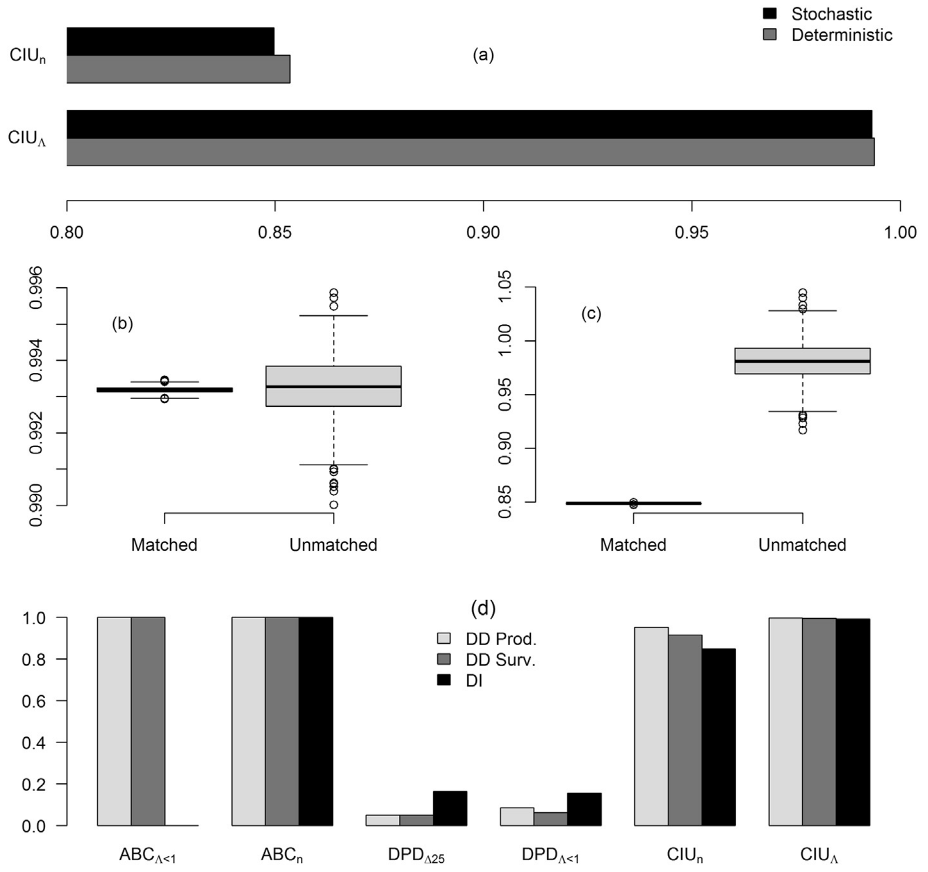

Fig. 3. Influence of model properties on estimates for decision criteria assuming an impact resulting in a 2.5% increase in mortality. (a) Comparison of CIU_n and CIU_λ decision criteria when estimated using stochastic or deterministic models; (b) Comparison of CIU_λ decision criterion estimated using a matched or unmatched runs approach; (c) Comparison of CIU_n decision criterion estimated using a matched or unmatched runs approach; (d) Comparison of $\mathrm{ABC}_{\lambda<1}$, ABC_n, DPD $_{\Delta 25}$, $\mathrm{DPD}_{\lambda<1}$, CIU_n and CIU_λ decision criteria estimated using models assuming no density dependence (DI), Density dependent regulation of survival (DD Surv.) or Density dependent regulation of productivity (DD Prod.).

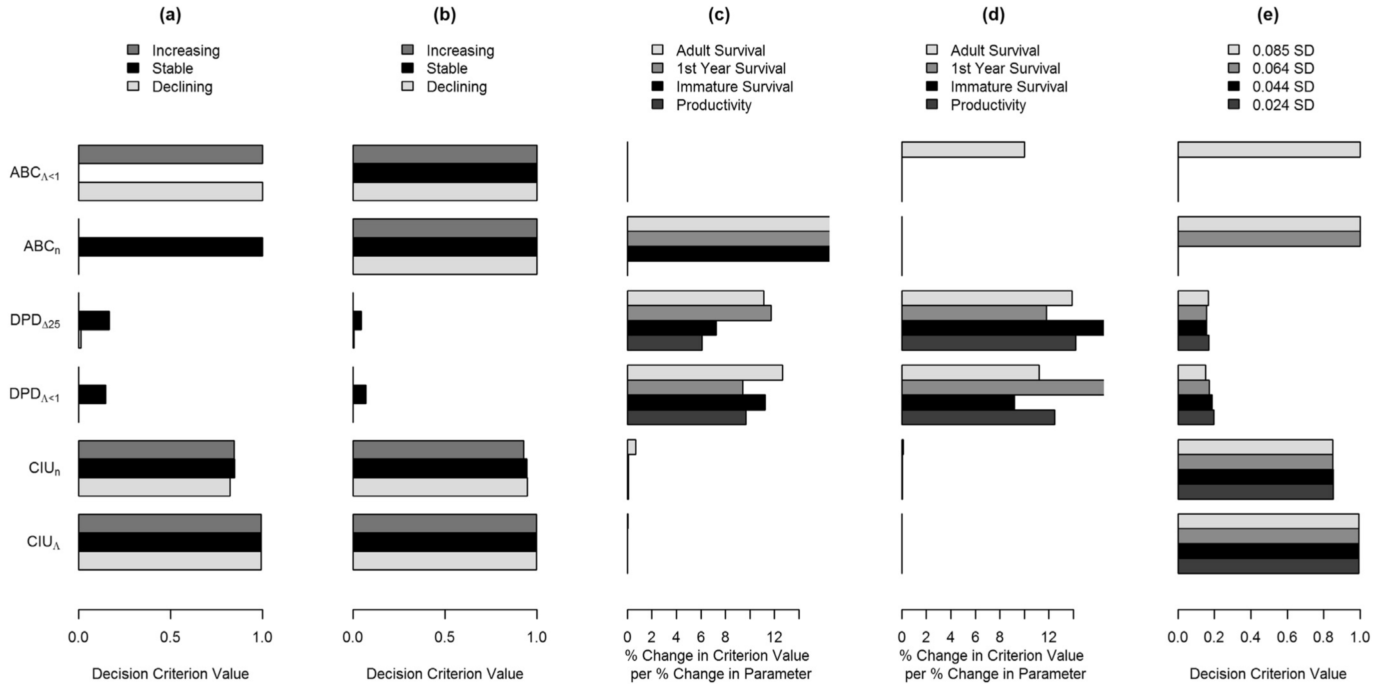

Fig. 4. Influence of model parameters on estimates for decision criteria assuming an impact resulting in a 2.5% increase in mortality or decrease in productivity. (a) Influence of population trend on decision criteria assuming impact on survival; (b) Influence of population trend on decision criteria assuming impact on productivity; (c) Influence of mis-specification of demographic parameters on decision criteria assuming impact on survival; (d) Influence of mis-specification of demographic parameters on decision criteria assuming impact on productivity; (e) Influence of mis-specification of uncertainty surrounding adult survival rate on decision criteria assuming impact on survival.

productivity, both DPD criteria were suggestive of a more significant population-level effect when pre-construction populations were stable as opposed to increasing or declining. For the increasing population, the impacts considered were insufficient to result in an increase in the proportion of simulations showing a decreasing population. For the decreasing population, close to 100% of the simulations from the demographic model will be showing a negative trend. Consequently, there is little scope for the value of these decision criteria to increase in value in response to impacts of increasing magnitude. As a result, using the DPD criteria, there is a risk of drawing the erroneous conclusion that a decreasing population could sustain an impact of greater magnitude than a stable population. Similar results were observed for $ABC_{\lambda<1}$.

Of the demographic parameters considered, each decision criteria was most sensitive to mis-specification of survival, especially of adults (Fig. 4c and d). However, of the criteria considered, CIU_λ and CIU_n were least sensitive to this mis-specification. The ABC and DPD criteria also showed sensitivity to mis-specification of other demographic parameters.

The CIU criteria and $DPD_{\Delta25}$ are relatively unaffected by assumptions about the level of uncertainty associated with adult survival rates (Fig. 4e). However, the ABC criteria and $DPD_{\lambda<1}$ all show similar patterns whereby increasing the estimate of uncertainty surrounding the adult survival rate results in population level impacts that may be deemed less significant.

4. Discussion

A key role for ecologists is to provide decision-makers with the evidence they require in order to reach a conclusion about whether the impact associated with any intervention may be deemed acceptable (e.g. Evans, 2012; Rose, 2014). However, communicating this evidence is often less than straightforward as a consequence of uncertainties surrounding the predicted impacts, and underlying demographic processes affecting the population concerned (Masden et al., 2009). The framework we present (Fig. 1) outlines a process from which metrics derived from a PVA can be used to derive decision criteria which may be used as evidence by decision makers to determine whether an impact is acceptable (in the context of other, competing, interests; see below), or, conversely, whether a conservation intervention is likely to be sufficiently successful. Whilst PVA has been criticised as having low predictive power (Coulson et al., 2001; Ellner et al., 2002), it may be a valuable tool with which to assess the relative risk of different scenarios (Brook et al., 2003; McCarthy et al., 2003). The key issue is how to estimate the relative risk of different scenarios in a robust fashion and present the outputs from the models in a format that is easily interpretable by decision makers.

The models used within the framework may take a variety of forms, each with different levels of complexity (e.g. deterministic vs. stochastic, density dependent vs. density independent). However, generally, the particular population model used is more likely to be driven by data availability than the question under consideration (Radchuk et al., 2016). There is a significant uncertainty present throughout the offshore wind farm consenting process, particularly in relation to the magnitude of predicted effects such as collision and displacement, arising due to the paucity of data from existing wind farms (MMO, 2014; Masden et al., 2015). Consequently, there is a requirement that assessments should be made using a realistic, but precautionary, approach (EC, 2000; Maclean et al., 2014). Our analyses demonstrate that stochastic models using a matched runs approach are likely to reflect the most precautionary approach (Fig. 3). Furthermore, it is important that uncertainty is accounted for and discussed in the consenting process (Sanderson and Petersen, 2002). It is likely to be more straightforward to achieve this where stochastic rather than deterministic models are used.

It is important to note that density dependence in any of the

Table 3
Sensitivity of decision criteria to uncertainty surrounding population model parameters.

	ABC		DPD		CIU	
	ABC $_{\lambda<1}$	ABC$_n$	DPD$_{\Delta25}$	DPD$_{\lambda<1}$	CIU$_n$	CIU$_\lambda$
Insensitive to population trend	✗	✗	✗	✗	✗	✓
Insensitive to mis-specification of adult survival	✗	✗	✗	✗	✓	✓
Insensitive to mis-specification of 1st year survival	✓	✗	✗	✗	✓	✓
Insensitive to mis-specification of immature survival	✓	✗	✗	✗	✓	✓
Insensitive to mis-specification of productivity	✓	✓	✗	✗	✓	✓
Insensitive to mis-specification of error surrounding demographic parameters	✗	✗	✗	✗	✓	✓

demographic parameters may also influence the conclusions drawn from the models (Fig. 3). It is clear that density dependent processes operate on many seabird populations, and that these processes can take a variety of forms (Cury et al., 2011; Horswill et al. in review). Where density dependence is of a compensatory nature (i.e. demographic rates increase in response to decreases in population size), unless there is strong evidence of the form and strength of this relationship, the most precautionary approach would be to assume no density dependent relationships within the population. However, density dependence may also be depensatory (i.e. demographic rates decrease in response to decreases in population size) meaning omitting it from the models may substantially underestimate potential impacts. In these circumstances, where there is no strong evidence of the form and strength of the relationship, the precautionary approach may be to compare outputs from a variety of plausible models.

The selection of a suitable decision criterion may be driven by the situation under consideration. The binary nature of the ABC criteria may be attractive to decision makers looking for guidance on what is, essentially, a binary decision (to grant, or not grant, planning consent). However, our analyses show that the binary nature of these criteria mean that they are sensitive to uncertainty and mis-specification of the input parameters (Table 3). A similar pattern was observed for the DPD criteria which may, in principle, be preferable to ABC (Green et al., 2016). Demographic parameters may show significant variation between populations of the same species (Frederiksen et al., 2005). Consequently, there is a need to treat conclusions based on these criteria with caution due to the need for confidence about pre-construction population trends and site-specific demographic data, often not available for key populations. However, a particular concern is the sensitivity of both ABC and DPD to the mis-specification of error surrounding demographic parameters (Table 3). Our analyses highlighted that for impacts of an equivalent magnitude, both ABC and DPD were likely to result in predictions of a less significant population-level effect where there was greater uncertainty surrounding demographic parameters (Fig. 4e). Whilst the error surrounding estimates of demographic parameters may, in part, reflect natural variation in the population concerned (process error), observation error (e.g. biases in data collection, sampling variance) may also make a significant contribution to these estimates of error (Gould and Nichols, 1998). This leads to concerns that developments may be deemed acceptable as a result of uncertainty surrounding data collection, as opposed the actual population level effect associated with an impact. Whilst we have estimated ABC using a Leslie Matrix Model, elsewhere it has been derived using a Bayesian framework (e.g. Marine Scotland, 2015). However, we feel our conclusions are broadly applicable to either type of model.

Of the decision criteria we consider, those that are based on CIU appear most applicable to situations where there is significant uncertainty about the status of population under consideration (Table 3). This conclusion is consistent with the arguments set out by Green et al. (2016). Ideally these criteria should be sensitive to stress in the system and respond to that stress in a consistent fashion (Dale and Beyeler, 2001). Whilst CIU$_\lambda$ varies over a limited range (Fig. 2), meaning it is likely to be less sensitive to impacts associated with wind farms than is the case for CIU$_n$, it responds to stress in a more consistent fashion (i.e. linearly) than is the case for CIU$_n$ (asymptotic). Furthermore, CIU$_n$ is more sensitive to the previous population trend than is the case for CIU$_\lambda$. These factors must be taken into consideration when selecting an appropriate decision criterion. It is necessary to consider the trade-off between a criterion offering a more unambiguous assessment of the population level effect (CIU$_n$), and one which is less sensitive to uncertainty surrounding population trend and has a more consistent relationship with impacts of increasing magnitude (CIU$_\lambda$), potentially aiding an understanding of the consequences of over or underestimating the magnitude of any impact.

Ultimately, any decision criteria produced using this framework will be used to guide assessments and determine whether predicted impacts are acceptable or not. This will rely on some sort of threshold being applied to the criteria and legislative pressures often create demand for these thresholds to be biologically meaningful. For example, the EU Habitats Directive (92/43/EEC) states that:

> *“The competent national authorities shall agree to the plan or project only after having ascertained that it will not adversely affect the integrity of the site concerned”*

It has been argued that any impact associated with a development will affect site integrity (Green et al., 2016). However, where a species population at a site is in a favourable status there may be some flexibility in determining what may be described as an adverse effect (EC, 2000). From a biological point of view, it is clear that any criterion indicating a difference between an impacted and unimpacted population will reflect a significant population level effect. However, the question then becomes whether, at a societal level, impacts may be deemed acceptable. Setting thresholds in this way requires close collaboration between ecologists and other disciplines in order to ensure that the values of the public and other stakeholders are reflected (Mavrommati et al. in press). Such thresholds are likely to vary on a case by case basis and may need to consider contexts beyond the designated site boundary. For example, impacts of a greater magnitude may be deemed to be relatively more acceptable in northern gannet *Morus bassanus* populations, which are currently increasing, than in kittiwake *Rissa tridactyla* populations, which are decreasing (JNCC, 2015). However, any thresholds derived in this manner must be subject to continuous scrutiny and redefinition (Mavrommati et al. in press), for example, in relation to the changing conservation status, at regional or local level, of the species concerned.

5. Conclusions

Our analyses focus on the example of offshore wind farms. However, our conclusions are more generally applicable to assessing the impact of other renewable energy industries, both in the terrestrial and marine environments, and more generally when considering the likely impact of different management strategies. Conclusions about the likely impact of developments can be strongly influenced by the 'expert' judgements made by those involved in the process (Wood et al., 2006). This may include, for example, the selection of appropriate demographic values for use in the population models in the framework described above. However, there remain concerns about the lack of transparency in the assessment process to date. Given the significance of the economic and policy decisions that may be informed by the conclusions from these models, it is important to ensure transparency throughout the consenting process, including a clear explanation for any judgments made by practitioners. In order to assess the impacts associated with any development, we suggest that impacted and unimpacted populations should be estimated over the lifetime of the project under consideration. These populations should be derived from stochastic population models using a matched runs approach. In most circumstances a counterfactual approach (CIU) is likely to be the most appropriate method for comparing the impacted and unimpacted populations and quantifying the population level effect associated with any development. Independent review of the data and models used in these models is important to ensure that conclusions are based on sound evidence and are free from bias (Reed et al., 2002; Parker et al., 2016), which is likely to boost confidence in the outcomes of the impact assessment process.

Acknowledgments

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Weak relationship between risk assessment studies and recorded mortality in wind farms

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Summary

1. Wind farms generate little or no pollution. However, one of their main adverse impacts is bird mortality through collisions with turbine rotors.

2. Environmental impact assessment (EIA) studies have been based on observations of birds before the construction of wind farms. We analysed data from 53 EIAs in relation to the actual recorded bird mortalities at 20 fully installed wind farms to determine whether this method is accurate in predicting the risk of new wind farm installations.

3. Bird data from EIAs were compared with bird collisions per turbine and year at functional post-constructed wind farms to identify any relationship between pre- and post-construction studies.

4. Significant differences in birds recorded flying among the 53 proposed wind farms were found by the EIAs. Similar results were obtained when only griffon vultures *Gyps fulvus* and other raptors were considered. There were significant differences in indexes, including the relative index of breeding birds close to proposed locations, among the 53 proposed wind farm sites.

5. The collision rate of birds with turbines was one of the highest ever recorded for raptors, and the griffon vulture was the most frequently killed species. Bird mortality varied among the 20 constructed wind farms.

6. No relationship between variables predicting risk from EIAs and actual recorded mortality was found. A weak relationship was found between griffon vulture and kestrel *Falco* sp. mortality and the numbers of these two species crossing the area.

7. *Synthesis and applications*. There was no clear relationship between predicted risk and the actual recorded bird mortality at wind farms. Risk assessment studies incorrectly assumed a linear relationship between frequency of observed birds and fatalities. Nevertheless, it is known that bird mortality in wind farms is related to physical characteristics around individual wind turbines. However, EIAs are usually conducted at the scale of the entire wind farm. The correlation between predicted mortality and actual mortality must be improved in future risk assessment studies by changing the scale of these studies to focus on the locations of proposed individual wind turbine sites and working on a species specific level.

Key-words: bird collision, environmental impact assessment, *Falco* sp., *Gyps fulvus*, mortality data, raptors, Tarifa, wind farm

Introduction

Wind farms have received public and government support as alternative energy sources because they do not contribution to air pollution, which is typically associated with fossil fuel technologies (Huntley *et al.* 2006). At the end of 2008, the global wind energy capacity surged by 28·8% and the total installed capacity reached 120·8 GW. Spain is the world's third largest wind energy market with 16·8 GW of installed electric generation capacity (Pullen, Qiao & Sawyer 2008).

Wind farms can affect birds through collisions with turbines (Orloff & Flannery 1992; Everaert & Stienen 2007; Smallwood 2007; Thelander & Smallwood 2007) or through

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displacement because of disturbance (Hötker, Thomsen & Köster 2004; Drewitt & Langston 2006). Although low collision rates have been recorded at many wind farms (Erickson *et al.* 2001; Percival 2005; de Lucas *et al.* 2008), some poorly sited wind farms have caused high collision mortality rates (de Lucas *et al.* 2008) and the potential for wind farms to cause problems for bird populations should not be underestimated (Hunt 2002; Madders & Whitfield 2006).

The prevention of bird collisions in newly built wind farms is a critical issue. When a wind project is proposed in European countries, an environmental impact assessment (EIA) is required by environmental authorities (either the Ministry or Environmental Department of a region). EIAs must include a section assessing the impact that the development is likely to have on the site's bird populations (Environmental Impact Assessment Directive 97/11/EC). Environmental authorities use the overall assessment to reach a 'declaration on the environmental impact' stating the significance and acceptability of the predicted effects. Mostly, these declarations identify additional measures to mitigate and compensate potential negative environmental consequences and other conditions that should be met by the project developer such as the monitoring of the environmental impacts.

Baseline data collection must be adjusted to different requirements depending on the area, so a fixed baseline survey is not possible. Langston & Pullan (2003) recommended that EIAs should include, at a minimum, a 12-month baseline field survey to determine the bird populations that use the development site annually. In some cases, 24 months of baseline surveys may be required for EIA assessment, where the bird species likely to be affected are subject to protective legislation (e.g. in Scotland for raptors listed on Annex I of the EC Birds Directive, SNH 2005).

This basic procedure is followed in European countries like Spain (RDL 2008), the UK (Percival, Band & Leeming 1999; SNH 2005), Denmark (Bro 2008) and Norway (NORAD 2003), and in other countries like Canada (Kingsley & Whittam 2005) and Mexico (Martínez 2008). Wind power regulations and wildlife guidelines in the United States vary by state (Stemler 2007), sometimes using voluntary guidelines to provide recommendations for minimising the potential impacts of wind development. Other states like Colorado, Maryland, Minnesota, Ohio and Oregon have mandatory and comprehensive guidelines for pre-construction evaluation, design, construction recommendations and monitoring post-construction.

In Andalusia, risk assessment studies are regulated by both regional and national legislation, although additional data can be required by local authorities. In the case of wind farm developments, additional data on bird and bat presence are often requested.

Nevertheless, we failed to find any study comparing previous risk evaluation with actual mortality recorded after a wind farm was operational. This lack of pre- and post-construction mortality comparisons is alarming because these previous risk evaluations are an integral part of the procedure of accepting or rejecting installations of new wind farms in several countries.

The main objective of our study was to analyse the relationship between risk prediction according to the environmental impact assessment studies (i.e. at the scale of entire wind farm) and the actual recorded mortality of birds in wind farms located in southern Spain after they became operational. The aim was to determine whether the assessment methods were accurate and to recommend improvements where necessary.

Materials and methods

STUDY AREA

The 53 potential locations for wind farms studied (20 finally approved and 33 rejected) were all located in Tarifa, Andalusia region, southern Spain, near the Strait of Gibraltar (Fig. 1). The Strait of Gibraltar is one of the most important migrating routes of the Palearctic birds (Bernis 1980; Finlayson 1992; Bildstein & Zalles 2000).

Cliff-breeding species such as griffon vultures *Gyps fulvus*, common kestrels *Falco tinnunculus*, Bonelli's eagles *Hieraaetus fasciatus*,

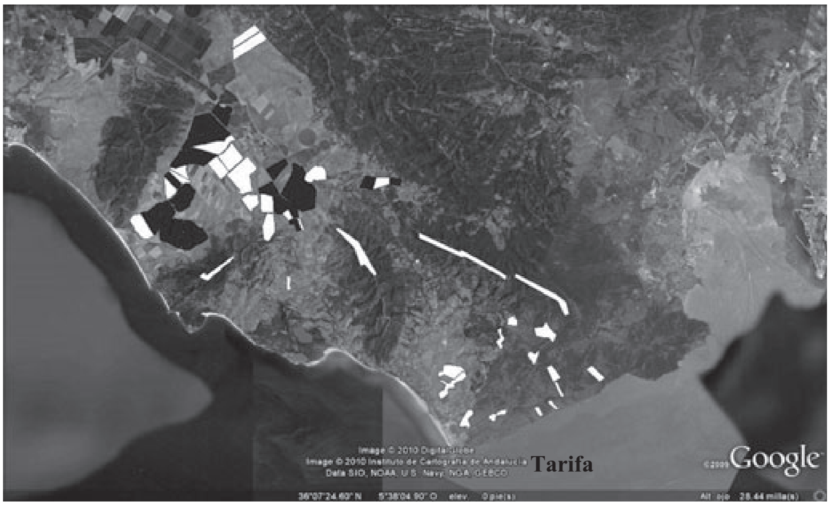

Fig. 1. Google Earth map with two different areas shows permitted wind farms (20 wind farms, in black colour) and unpermitted wind farms (33 wind farms, in white colour). Study area was in the extreme south-west of Spain.

peregrine falcons *Falco peregrinus*, Egyptian vultures *Neophron pernopterus*, short-toed eagles *Circaetus gallicus* and eagle owls *Bubo bubo* are common in this area. Three of these species are endangered in Spain: Bonelli's eagle, Egyptian vulture and peregrine falcon (Madroño, González & Atienza 2004). In addition, during the migration period, thousands of soaring species such as honey buzzards *Pernis apivorus*, black kites *Milvus migrans*, white storks *Ciconia ciconia*, booted eagles *Hieraaetus pennatus* and short-toed eagles pass through this area. The vegetation in the study area is characterised by brushwood and scattered trees (*Quercus suber, Q. rotundifolia*) on the mountain ridges and by pasture land used for cattle grazing in the lower areas.

The 20 approved wind farms consisted of 252 wind turbines and nine different models (see Table 1). Turbines were arranged in rows running north–south, so they optimised the use of prevailing east and west winds. The total height of the turbines (including the blades) ranged from 106 to 170 m, rotor diameters ranged from 56 to 90 m, and distances between turbines ranged from 115 to 180 m with the distance between turbines double the rotor diameter.

FIELD METHODS DURING RISK ASSESSMENT STUDIES

All the methods we used were in accordance with environmental administration requirements and were very similar to those of other autonomous administrations in Spain as well as other parts of the world. The studies of environmental impacts prior to wind farm construction were carried out from 1999–2000. During these studies, the 53 proposed wind farm locations (i.e. areas where turbines were due to be installed) were sampled to estimate bird use of the areas over an entire year. Eventually 20 wind farms received construction licences from the local environmental authorities, and they were built at locations where the least risk was expected based on several criteria, including bird use, proximity to breeding and roosting sites, and the presence of endangered species or potential collision victims (i.e. raptors).

The numbers of birds crossing the 53 potential wind farm locations were recorded from fixed observation points inside each area. At these points, birds were observed using binoculars that allowed detection up to 1 km away. Because of relief in some cases, the whole location could not be covered from one observation point, in which case bird crossings were recorded from different points and average use of the entire site was calculated.

Table 1. Wind farm characteristics

Wind farm Number	Number of turbines	Turbine power (MW)	Height (Without blades)	Rotor diameter
1	11	1·91	67	87
2	11	1·91	80	90
3	17	0·8	57	59
4	30	0·8	57	59
5	9	2·2	67	87
6	20	0·8	50	56
7	11	2·0	80	90
8	28	1·6	80	80
9	15	0·8	57	59
10	8	2·0	67	87
11	6	2·0	67	87
12	6	1·6	74	74
13	9	1·7	80	90
14	9	1·7	80	90
15	9	1·7	80	90
16	16	0·8	50	56
17	9	2·1	80	90
18	10	2·0	57	71
19	8	2·1	80	90
20	9	2·2	57	71

Birds were recorded by two observers during 3621 independent observation sessions of 1–3 hours each. Sessions were evenly distributed over the study area (about 60 sessions per observation point), the year (about 1800 hour per season) and daylight hours. Days with similar meteorological conditions for visibility were selected to avoid biases in our ability to detect birds. Observations from different locations were conducted on the same days but at different times to ensure complete coverage of all daylight hours. Mean observation time per potential wind farm location varied between 107 and 228 hours (total effort: 7267 hours and 42 minutes). During observation sessions, any bird or group of birds detected was recorded, and its flight altitude through the area was recorded. Each record included (i) species, (ii) number of birds and (iii) categorised flight altitude at each location (beneath rotor/rotor height/above rotor).

For each proposed wind farm location, we recorded the total number of birds observed per hour and the total number of birds at rotor height (risk). Additionally, these values were calculated separately for griffon vultures, kestrels and other raptors (not vultures).

Classification of the mortality threat of the 53 potential wind farm locations was made using two indexes, the Relative Risk Collision Index (RRCI) and the Breeding Birds Relative Risk Index (BBRRI):

$$\mathrm{RRCI} = [(\mathrm{birds/hour}) * (\%\mathrm{bird\ at\ risk/hours}) * (\exp(\mathrm{CS})) * (\exp(\mathrm{ES}))]$$

where CS equalled the percentage of bird species sensitive to collision according to the literature and ES the percentage of endangered species according to the Spanish Red Book. The RRCI was standardised to 0–1 range.

$$\mathrm{BBRRI} = [\textstyle\sum \exp(-d) * (N.nests)]$$

where $-d$ was the distance from the potential wind farm area to a breeding site (with negative value) and *N. nest* was the number of nests in the breeding area. The BBRRI was standardised to 0–1 range.

Based on these indexes, the locations were classified into three risk levels (1, 2 or 3), according to a subjective judgment by planning authorities. Most of the 20 projected wind farms that finally obtained positive environmental impact evaluations had a risk level of 1, but some had risk levels of 2 if they showed low values of RRCI and BBRRI.

FIELD METHODS WITH OPERATING WIND FARMS

Wind farms started to operate between 2005 and 2008 in the study area (Fig. 1). Every operational wind farm activated a Surveillance Program with the main goal of registering the actual bird mortality by finding all of the dead birds that had collided with turbines. This mortality monitoring was made on a daily basis from the day the wind farms began to operate. Every wind farm was monitored from dawn to dusk by 13 trained observers who coordinated their observations to ensure that they surveyed the total area of influence of each wind farm.

Searches for birds killed by collisions were made on a daily basis at every turbine. As we were interested in mid-sized and large birds (pigeon-sized or bigger) and the surveys were conducted daily, we did not conduct carcass disappearance experiments. All turbines in our study area were arranged in rows; therefore, the most efficient search method was to walk transects and/or drive unpaved roads along the rows (Smallwood & Thelander 2004). The data included the number of birds killed by each turbine, with information on species, age and sex if possible, injuries, distance to turbine and weather conditions when the mortality occurred.

As not all wind farms became operational at the same time, and to standardise mortality data, the daily mortality rate was calculated for each wind farm by dividing the sum of fatalities recorded in a given wind farm by the number of turbines and by the number of days which it was operating; annual mortality rate was calculated by multiplying by 365.

We calculated two different mortality rates: total bird mortality across all observed species and raptor mortality, with raptors selected because of their importance in conservation. Both variables are used in EIAs. We also selected two additional species for the analyses: griffon vultures and kestrels (*Falco tinnunculus* plus *Falco naumanni*) to determine whether the aggregation of raptor species would introduce errors in estimating mortality risk. Griffon vultures were selected because they are the most common large raptor species in the area. Kestrels were the most frequent raptor after vultures.

STATISTICAL METHODS

Because of the distributional characteristics of our data, we used nonparametric statistics to perform most of the analyses. The data set for the 53 proposed locations was tested for differences in the number of birds crossing each site and any of the indexes used to determine risk level.

Any possible relationships between the total number of birds killed by collisions with each turbine and year, and the number of birds crossing per hour or the number of birds at risk crossing the area per hour in the same location, as determined in the pre-construction studies, were analysed using nonparametric correlations. The number of raptors killed and the number of griffon vultures and kestrels killed were analysed in the same way. To account for inter-seasonal variation, we conducted correlation analyses between the total number of birds across all species, raptors, and vultures and kestrels counted in each season during pre-construction studies and mortality recorded in the same season when wind farms were operating.

To estimate the ability of EAI studies to predict the potential for collision rates, we used the linear relationship between pre-construction flight activity and subsequent frequency of collisions in the 20 authorised wind farms, log or square transforming variables when necessary to meet normality criteria (Shapiro–Wilk test). We used the prediction limits (95%) of this regression to estimate the potential predictive power according to pre-construction information, which is a common approach in EIAs. It was possible that we might not have been able to detect a significant relationship between pre-construction risk and actual recorded mortality at wind farms because only those wind farms with low risk were constructed; therefore, we attempted to predict the likely collision rates of the non-authorised wind farms by assessing the overlap between the two single species we analysed. For mortality variables, we also calculated the coefficient of variation among turbines inside the same wind farm and the coefficient of variation of mean mortality values among wind farms.

We also performed ANOVA analyses, log or square transforming variables when necessary to meet normality criteria (Shapiro–Wilk test), to analyse differences in mean mortality per turbine and year among operating wind farms. All tests were two tailed. Statistica 7.0 was used to perform statistical procedures, and we used an alpha value of 0·05 to assess the significance of results.

Results

RISK ASSESSMENT STUDIES

A total of 291 278 birds were counted in the 53 study locations, averaging 40·08 birds per hour. Of those birds, 111 180 (38·17%) were griffon vultures and 4682 were raptor species.

According RRCI and BBRRI, and after a subjective judgment by planning authorities (i.e. weighting species according to regional status as well as national conservation status), the 53 locations were classified into three levels of mortality danger (Fig. 2). These levels of danger showed significant differences in the rates of bird crossings (birds per hour: Kruskal–Wallis test = 7·75, $P = 0·0207$; birds at risk per hour: Kruskal–Wallis test = 14·77, $P < 0·001$; raptors per hour: Kruskal–Wallis test = 16·91, $P < 0·001$; raptors at risk per hour: Kruskal–Wallis test = 22·21, $P < 0·001$; vultures per hour: Kruskal–Wallis test = 6·36, $P = 0·041$; vultures at risk per hour: Kruskal–Wallis test = 15·52, $P < 0·001$; kestrel per hour: Kruskal–Wallis test = 10·54, $P = 0·005$; kestrel at risk per

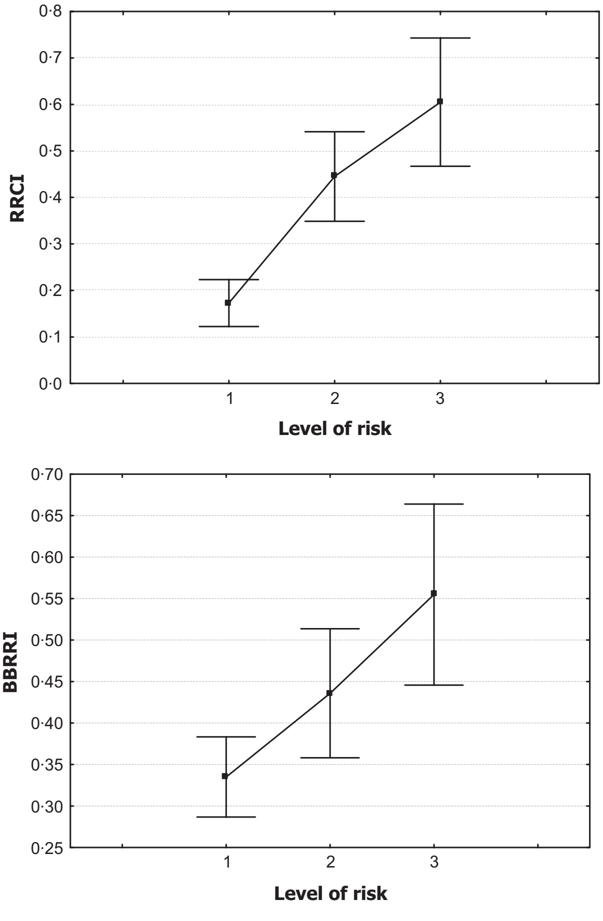

Fig. 2. Classification of the 53 potential wind farm locations according to level of risk, using the Relative Risk Collision Index and the Breeding Birds Relative Risk Index. See text for definitions.

Table 2. Differences between authorised (Yes, n = 20) and non-authorised (No, n = 33) wind farm locations. Kruskal–Wallis test was used. See text for calculation of the Relative Risk Collision Index (RRCI) and the Breeding Birds Relative Risk Index (BBRRI)

	Mean (SD)		
Variables	No	Yes	P
RRCI	0·52 (0·26)	0·24 (0·13)	< 0·001
BBRRI	0·49 (0·21)	0·35 (0·07)	0·009
Birds/hour	45·65 (27·65)	30·89 (8·74)	0·004
Birds at risk/hour	22·53 (20·84)	11·54 (5·35)	0·001
Raptors/hour	7·90 (5·36)	3·81 (2·02)	0·002
Raptors at risk/hour	3·84 (3·24)	1·71 (0·61)	0·005
Vultures/hour	14·93 (9·87)	11·51 (4·50)	0·018
Vultures at risk/hour	7·05 (3·31)	4·24 (3·08)	0·003
Kestrels/hour	0·65 (0·06)	0·36 (0·05)	0·001
Kestrels at risk/hour	0·17 (0·01)	0·08 (0·01)	0·002

hour: Kruskal–Wallis test = 8·92, P = 0·011). For all the variables analysed, the mean values of birds flying were higher at the highest, most dangerous sites.

Based on the findings, 20 wind farm locations were finally authorised. Mean values of the rates of bird crossing and the statistical differences between authorised and unauthorised wind farm locations are shown in Table 2. In all cases, the 20 authorised locations showed significantly lower values than did the unauthorised ones.

MORTALITY IN OPERATING WIND FARMS

We found a total of 596 dead birds at all of the wind farms during the time they were operational (see Table S1 in Supporting Information). The griffon vulture was the most frequently killed species with 138 individuals (23·15%) colliding with turbines. Another 76 raptors other than vultures were found killed (23 common kestrels, 13 lesser kestrels and 16 short-toed eagles, among others). Raptors including vultures represented 36% of the total number of birds found dead, which was the same proportion as passerines (36%). Taking into account the time that the different wind farms have been operating (from 11 to 34 months), 337 birds of all species and 124 raptors died annually because of collisions with turbines, including 87 griffon vultures. The mean number of bird mortalities per turbine per year was 1·33, and there were significant differences in the mortality of all birds (ANOVA F = 5·185, d.f. = 19, 232, P < 0·001) as well as raptors alone (ANOVA F = 2·245, d.f. = 19, 232, P = 0·002) among the 20 wind farms. Differences were also significant when only griffon vulture (ANOVA F = 8·276, d.f. = 19, 232, P < 0·001) and only kestrel (ANOVA F = 9·004, d.f. = 19, 232, P = 0·009) mortalities were considered.

COMPARING ESTIMATED AND RECORDED MORTALITY THREATS

No significant relationship was found between bird mortality and either RRCI (r_s = −0·291, n = 20, P = 0·212) or BBRRI (r_s = −0·403, n = 20, P = 0·177). Recorded bird mortality in operating wind farms showed no differences according to the level of threat of the wind farm (Kruskal–Wallis test, H = 0·353, P = 0·552). Also, no difference between wind farm risk level and vulture mortality (Kruskal–Wallis test, H = 2·391, P = 0·122) was found. Bird flying rates in pre-construction locations and actual subsequent mortality in operating wind farms are given in Table S2, Supporting Information. No relationship was found between birds per hour and bird collisions per turbine and year (Spearman correlation; r_s = −0·118, n = 20, P = 0·617), nor was there a difference between birds at risk and bird mortality (r_s = 0·163, n = 20, P = 0·491). Raptor mortality was not related to raptors per hour (r_s = −0·376, n = 20, P = 0·101) nor to raptors at risk per hour (r_s = 0·024, n = 20, P = 0·917).

Considering single-species analyses, a marginal relationship was found between vultures per hour and mortality of vultures (r_s = 0·443, n = 20, P = 0·0503) but not between vultures at risk and mortality (r_s = 0·304, n = 20, P = 0·191). Kestrel mortality again was marginally related to kestrels per hour, although not significantly (r_s = 0·395, n = 20, P = 0·084), but not to kestrels at risk (r_s = 0·376, n = 20, P = 0·102).

To control for seasonal variation, we conducted comparisons between different bird flying rates during pre-construction studies and actual subsequent bird mortality, separately by season. Again, no relationship was found for any of the variables analysed (Table S3 Supporting Information).

EIA PREDICTION POWER

A non-significant regression between birds counted per hour and birds killed per turbine and year (both variables transformed to meet normality) was found, with birds per hour explaining only four per 10 000 of the variance (r = −0·0209, n = 20, P = 0·930). Given the lack of positive slope (see Fig. 3), this regression did not produce useful predictions. When we considered only raptors, again a non-significant

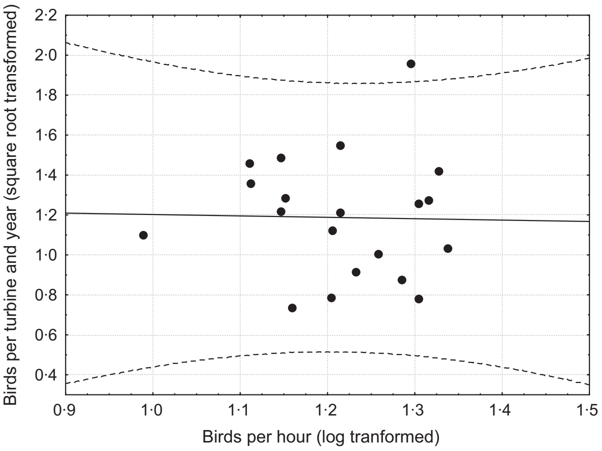

Fig. 3. Non-significant correlation between bird mortality recorded in operating wind farms and observations of birds flying over the area [both variables transformed to meet normality, (r = −0.0209, n = 20, P = 0.930)]. Dotted lines represent 95% of prediction.

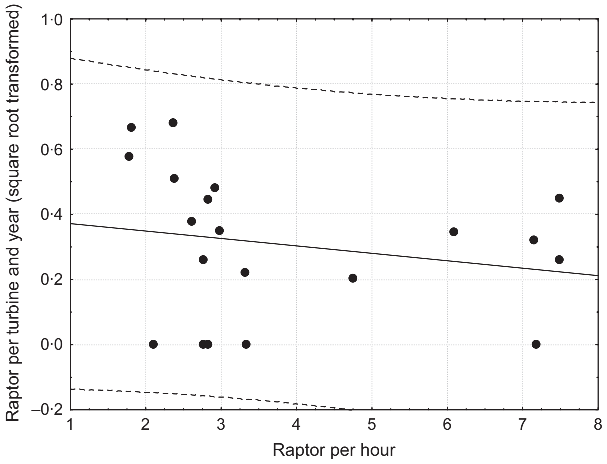

Fig. 4. Non-significant correlation between raptor mortality recorded in operating wind farms and observations of raptors flying over the area [both variables transformed to meet normality, ($r = -0.204$, $n = 20$, $P = 0.382$)]. Dotted lines represent 95% of prediction.

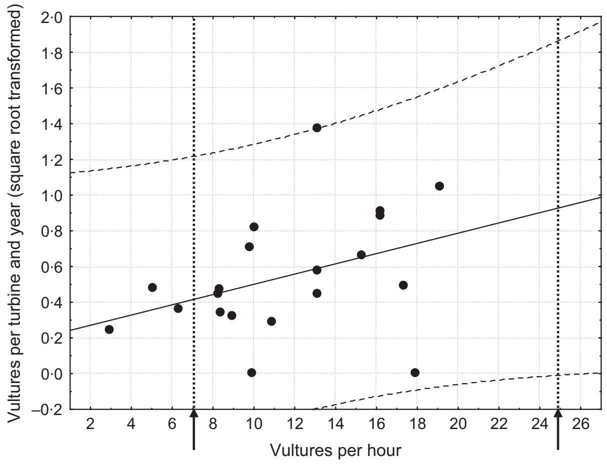

Fig. 5. Non-significant correlation between vulture mortality recorded in operating wind farms (square root transformed) and observations of vultures flying over the area ($r = 0.379$, $n = 20$, $P = 0.099$). Dotted curves represent 95% of prediction. Arrows and dotted lines represent the range limits of vultures per hour in the refused wind farms.

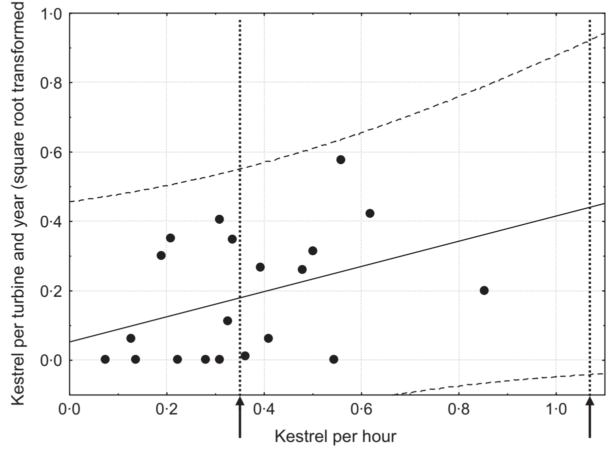

Fig. 6. Non-significant correlation between kestrel mortality recorded in operating wind farms [square root transformed) and observations of kestrels flying over the area ($r = 0.380$, $n = 20$, $P = 0.0979$]. Dotted curves represent 95% of prediction. Arrows and dotted lines represent the range limits of kestrels per hour in the unpermitted wind farms.

regression was found (raptor per hour and square root of raptor per turbine and year, $r = -0{\cdot}204$, $n = 20$, $P = 0{\cdot}382$, 4% of variance explained). Using the 95% prediction limits for this regression (see Fig. 4), we again obtained the limits of predictions for raptor mortality values for raptors recorded per hour. Prediction limits were the same for any value of raptors counted per hour, including those values recorded in the proposed wind farm sites that did not receive permits. No relationships between RRCI and raptor mortality ($r = 0{\cdot}1663$, $n = 20$, $P = 0{\cdot}4835$, 3% of variance explained) or BBRRI and raptor mortality ($r = 0{\cdot}1714$, $n = 20$, $P = 0{\cdot}469$, 3% of variance explained) were found.

A marginal although non-significant regression between vultures per hour and mortality of vultures per turbine and year (this last variable squared root transformed to meet normality) was found, with vultures per hour explaining 14% of the variance ($r = 0{\cdot}379$, $n = 20$, $P = 0{\cdot}099$). Using the 95% prediction limits for this regression (see Fig. 5), we obtained the limits of predictions expected according to increasing values for the independent variable, assessing the overlap of those wind farms receiving and not receiving permits (see Fig. 5). Considering the highest values of vultures per hour recorded during EIAs in the unauthorised wind farms (24·6 vultures per hour), expected vulture mortality would be between 0 and 3·4 vultures per turbine and year (back transformed). With the lowest values for vultures per hour recorded, the expected mortality would be between 0 and 1·46.

Mortality of kestrels showed a marginal, although non-significant, relationship with kestrels per hour (kestrels per hour and square root of kestrels per turbine and year, $r = 0{\cdot}380$, $n = 20$, $P = 0{\cdot}0979$, 14% of variance explained, Fig. 6). As before, considering the highest values of kestrels per hour recorded during EIAs in the unauthorised wind farms (1·04 kestrels per hour), expected mortality would be between 0 and 0·828 kestrels per turbine and year (back transformed). With the lowest recorded values for kestrels per hour, the expected mortality would be between 0 and 0·336. To summarise, Figs 5 and 6 indicate that the previous weak correlations between bird activity and subsequent mortality after wind farm construction do not appear to be a consequence of a large difference in the initial activity levels, and hence likely collision levels, between operating and unauthorised wind farms.

Variance, measured as the coefficient of variation, of the mortality values among wind farms was 113·8% in total bird mortality, 101·5% in only raptors and 51·9% in only vultures. Coefficients of variation among turbines inside the same wind farm were calculated, and mean values for all the wind farms were 178·6% in bird mortality, 212·8% in raptor mortality and 150·5% in vulture mortality. Consequently, coefficients of variation among turbines were more than twice than those among wind farms.

Discussion

Avian fatalities in wind farms have been reported around the world (Orloff & Flannery 1992; Erickson *et al.* 2001; Fiedler *et al.* 2007; de Lucas *et al.* 2008). Herein, we present one of the highest mean collision rates ever reported for all bird species (1·33 birds per turbine per year) and for raptor species alone (Orloff & Flannery 1992; Drewitt & Langston 2006; Lekuona & Ursúa 2007; de Lucas *et al.* 2008). These mortality rates are similar to those reported in coastal areas of Belgium, the Netherlands and Great Britain where principal victims are sea birds (Still *et al.* 1994; Musters, Noordervliet & Terkeus 1996; Everaert & Stienen 2007). All of these high-mortality wind farms have been, according to the law, licensed after risk assessment studies were conducted prior to construction according to accepted methodology.

Our results suggest that there is no clear relationship between predicted risk identified during EIAs and actual mortality of birds (particularly raptors) after wind farms have been constructed. Only weak relationships were found for single species. Although there were significant differences among wind farms in the frequency of birds observed flying, and there were significant differences in mortality rates when the wind farms were operating, no relationship between both groups of variables was evident. Consequently, some of the more *a priori* safe sites showed some of the higher collision rates when operating and vice versa.

This finding is relevant because the location of a wind farm is one of the few certainties known to affect the impact of a wind energy facility on birds (Madders & Whitfield 2006; de Lucas *et al.* 2008). Therefore, if criteria used in the prediction of the greatest risk areas are not valid tools for planning wind farm developments, as in our case, there are at least two problems. First, during at least the last decade, environmental administrations might have been giving licences to construct wind farms based on the wrong criteria, enabling them to be constructed in unsuitable places, as well as in safe ones. Second, there is an urgent need for a new or modified tool to adequately select locations for new wind farms to be constructed in future years.

Some factors could explain the lack of correlation between estimated risk *a priori* and actual mortality recorded after wind farms are constructed. It is possible that only those locations showing low to medium potential risk were authorised. As no 'unsuitable' places were authorised, we do not have any data for wind farms operating in these potentially high-risk areas. The lack of extreme data may affect the probability of detecting significant correlations, but regression analysis with prediction limits of 95% clearly showed that the power of prediction for mortality based on bird counts, raptor counts or indexes (including distances to nest sites) at the scale of the complete wind farm was very low because of the lack of positive slopes. As a result, the amplitude for predictions was the same for the lowest and the highest values of the independent variable (including unauthorised wind farms). In the single-species analyses (i.e. vultures and kestrels), the overlap of independent variable values (i.e. observation of vultures or kestrel per hour) between wind farms receiving permits and those not receiving permits was so large that we can conclude that we have enough data for the analyses. Consequently, we can suggest that this approach (estimation of potential mortality according to bird observation records during EIA) is inaccurate, demonstrating low predictive power.

Obviously, we cannot discount the fact that with more mortality data coming from unpermitted wind farm sites, some of the regression would be significant or marginally significant, but it is difficult to believe because the mortality recorded at the approved wind farms was among the highest ever recorded. In considering only the approved wind farms, mortality records in some locations were so high that clearly the risk assessment method was inadequate if the aim was to grant permission for low-impact installations only.

Another potential source of error is selection of groups of birds or the definition of indexes that are in fact aggregates of different species. Our results showed that assessment based on a single-species approach would be much more accurate than those used in the EIAs. Interestingly, regressions were improved using total observations of vultures or kestrels per hour compared to observations of birds flying at risk (i.e. at rotor height). This suggests that estimating risk introduces more variability and error into the assessment. In conclusion, we recommend the use of species-specific approaches (as in other countries like UK; SNH 2005) and total observations in the area.

It is possible that data collection during previous studies did not take into account the real distribution of wind direction during the year, a variable that affects bird behaviour and use of space mainly in soaring birds' species (Barrios & Rodríguez 2004; de Lucas *et al.* 2008). Indeed, a comparison of the study hours by wind direction showed that east and west wind directions were surveyed more than their occurrence. These potential differences between wind directions recorded during EIAs and the annual pattern of wind direction would explain discrepancies between *a priori* studies and actual recorded mortality after wind farms became operational.

Additionally, the methods used in our risk assessment studies might have been inappropriate. Records taken at fixed observation points are potentially skewed towards the locations of observers, and the actual use of some areas might be underestimated because of the large distance between the area and the observation points, especially for medium- to small-sized birds. Estimates based on transect counts or supported by radar data might have provided more accurate data. However, this should have not been an issue for vultures, and our results would have been similar to those for kestrels, clearly a smaller bird than vultures.

We contend that there are some weaknesses in the common methodology used in risk assessment studies because they wrongly assume a linear relationship between the frequency of observed birds and fatalities of birds (Langston & Pullan 2003;

Smallwood & Thelander 2004; Tellería 2009). There is clear evidence that the probability of bird collisions with turbines depends critically on species behaviour and topographical factors, and not only on local abundance (Barrios & Rodríguez 2004; de Lucas *et al.* 2008). This challenges the main assumption of wind farm assessment studies: birds do not move over the area at random, but follow main wind currents, which are affected by topography. Consequently, certain locations of wind turbines could be very dangerous for birds even where there is a relatively low density of birds crossing the area, whereas other locations would be relatively risk free even with higher densities of birds. If relevant factors affecting the frequency of collisions with turbine rotor blades are operating at the individual turbine scale, and not at the entire wind farm scale, EIAs must be conducted at the level of individual proposed turbines. Our results demonstrate that mortality variation among wind turbines inside the same wind farm was more than double the variation among wind farms. As turbine locations were not defined in the EIA in our study, it was impossible to focus observations on birds crossing close to future individual turbine locations. Differences in working scales would explain the low correlation between predicted risks and observed actual mortality. In the future, it would be useful to map bird flight paths at the scale of proposed turbines as well as recording the number of birds observed crossing proposed development sites. We realise that this will probably need a higher intensity of pre-construction fieldwork than is usually conducted, in order to be accurate.

An interesting development is proposed by de Lucas (2007) to test a model of the proposed development area in a wind tunnel to determine the location of the main passages for birds prior to construction. At a finer scale, these models could be used to evaluate the relative effects of individual turbines within particular locations, using data from a meteorological mast recording wind speed in the area. The use of this kind of aerodynamic model, as well as any statistical model using existing wind and topographical data, at an early planning stage could help to streamline the process of selecting potential locations and reduce the uncertainty associated with wind farm development.

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Supporting Information

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

Table S1. Mortality records in the 20 approved wind farms.

Table S2. Number of birds observed flying and number of dead birds per wind farm location.

Table S3. Correlations by season between bird flying rates and bird mortality in the 20 approved wind farms.

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Scaling possible adverse effects of marine wind farms on seabirds: developing and applying a vulnerability index

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Summary

1. Marine wind farms have attracted substantial public interest. The construction of wind facilities offshore may become Europe's most extensive technical development in marine habitats. Due to political pressure to complete construction soon, assessments of possible wind farm locations, for example in the German sectors of the North Sea and Baltic Sea, have to be based on existing knowledge.

2. In this study, we developed a wind farm sensitivity index (WSI) for seabirds. We applied this index to the Exclusive Economic Zone and the national waters of Germany in the North Sea. We chose nine factors, derived from species' attributes, to be included in the WSI: flight manoeuvrability; flight altitude; percentage of time flying; nocturnal flight activity; sensitivity towards disturbance by ship and helicopter traffic; flexibility in habitat use; biogeographical population size; adult survival rate; and European threat and conservation status. Each factor was scored on a 5-point scale from 1 (low vulnerability of seabirds) to 5 (high vulnerability of seabirds). Five of these factors could be dealt with by real data but four could only be assessed by subjective considerations based on at-sea experience; in the latter cases, suggestions of the first author were independently modulated by experts.

3. Species differed greatly in their sensitivity index (SSI). Black-throated diver *Gavia arctica* and red-throated diver *Gavia stellata* ranked highest (= most sensitive), followed by velvet scoter *Melanitta fusca*, sandwich tern *Sterna sandvicensis* and great cormorant *Phalacrocorax carbo*. The lowest values were recorded for black-legged kittiwake *Rissa tridactyla*, black-headed gull *Larus ridibundus* and northern fulmar *Fulmarus glacialis*.

4. A WSI score for areas of the North Sea and Baltic Sea was calculated from the species-specific sensitivity index values. Coastal waters in the south-eastern North Sea had values indicating greater vulnerability than waters further offshore throughout the whole year.

5. Derived from the frequency distribution of the WSI, we suggest a 'level of concern' and a 'level of major concern' that are visualized spatially and could act as a basis for the selection of marine wind farm locations.

6. *Synthesis and applications.* The wind farm sensitivity index might be useful in strategic environmental impact assessments (EIA). Results of small-scale EIA from wind installations should be considered within a more global perspective, provided, for example, by large mapping projects and detailed behavioural studies. This is difficult in normal EIA, particularly in highly dynamic coastal/marine habitats, and the results of this study fill an important gap by providing information on the potential sensitivity of seabirds and the importance of locations of wind installations.

Key-words: birds, conservation, management, sensitivity analysis, wind energy

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Introduction

The first wind farms at sea were established in the early 1990s, off Denmark and Sweden (Larsson 1994). There are now at least nine operational marine wind farms in Europe, as well as proposals to build many more around the UK and off Germany and Denmark (Anonymous 2000a,b, 2001, 2002; ICES 2003). Plans for marine wind farms indicate that each may consist of up to 1000 turbines, extending as far offshore as 100 km, and in waters up to 40 m deep (Anonymous 2002). Within northwest Europe, there are in total about 260–270 turbines in existing marine wind farms (as of December 2003, http://www.offshorewindenergy.org/) but many thousands are planned for construction in the next few years (Hüppop, Exo & Garthe 2002; ICES 2002). According to the current development plan for the German parts of the North and Baltic Seas, marine wind farms will require an area of 13 000 km^2 between 2002 and 2030 (BMU 2001). The UK has recently issued licenses for the development of about 40 marine wind farms in its waters. Thus, erection of wind facilities offshore may become Europe's most extensive technical development in marine habitats (Merck & von Nordheim 2000).

With so few wind farms established in the sea to date, there is very limited information on their effects on the marine environment (Merck & von Nordheim 2000; ICES 2002) and none on marine sites located more than 10–20 km from the coast. As applications for offshore wind farm construction will be decided in many locations before comprehensive, medium- to large-scale, ecological studies on the status of marine wildlife are completed, predicted effects have to be based on limited current knowledge. This should include conclusions from studies of wind farms on land, from the few inshore wind farms and from knowledge of the spatiotemporal patterns of abundance of organisms at sea that might be at risk.

Birds are assumed to be among the taxa affected most heavily by wind farms. Studies on land, and the first results from marine sites, suggest that both birds on migration and those resting or foraging locally may be affected (Barrios & Rodríguez 2004). At sea, this therefore includes both migrating birds, from the smallest songbirds to large birds such as cranes and birds of prey, and seabirds during their local movements (Anonymous 2000b; Garthe 2000; Exo, Hüppop & Garthe 2003; Hüppop, Exo & Garthe 2002). From extensive studies of seabirds at sea over the past 20 years, the distribution and abundance of seabirds in the North Sea is well-known over large and medium scales. Available data can therefore advise the site selection of wind farm locations. However, different habits and activities of birds at sea have to be taken into account. For example, species flying frequently at altitudes of 50–200 m a.s.l. are much more vulnerable to wind turbines than species that swim most of the time. However, there are many effects to be considered in addition to direct collisions.

In order to assess the possible impacts on seabirds of a range of factors, several indices have been applied in recent years, for example with regard to oil pollution (King & Sanger 1979; Williams *et al*. 1994) and the sandeel fishery (Furness & Tasker 2000). One of the indices of vulnerability to oil pollution has been applied to the whole North Sea, separating areas of high and low vulnerability for seabirds over the year (Carter *et al*. 1993). While the lack of long-term data, the inability to infer causality from monitoring studies and the limited spatial and temporal scales of experimental studies make such indices difficult to derive with confidence, more are required (Forde 2002). Debate on the effects of human activities on wildlife necessitates risk and impact assessments (Stillman *et al*. 2001) even where the database might be poor (Tuck *et al*. 2001).

In this study, we developed a wind farm sensitivity index (WSI) for seabirds. In terms of rationale and methodology, this index follows those of Williams *et al*. (1994) and Furness & Tasker (2000). We applied this index to the Exclusive Economic Zone and national waters of Germany in the North Sea. This area was selected for three reasons. First, there has not been any governmental advice prior to the beginning of the planning process (as was the case in Denmark, for example; Anonymous 1996) so site selection by the applicants might not have taken into account all of the important environmental issues. Secondly, there is an urgent need to evaluate risks in this area, in which 24 project applications comprising more than 11 000 single turbines were in place as of January 2003 (in the Exclusive Economic Zone). Thirdly, due to the complex geomorphology and hydrography under the existing intensive human usage (e.g. shipping lanes), a small- and medium-scale environmental assessment is urgently needed as a solid basis for future licensing.

Methods

VULNERABILITY FACTORS

We chose nine different factors, derived from species' attributes, to be included in the WSI, all taking into account the risks of seabirds colliding with wind turbines and/or being disturbed by wind farms. Each factor was scored on a 5-point scale from 1 (low vulnerability to seabirds) to 5 (high vulnerability to seabirds). Five of these factors could be based on real data, four could only be assessed by subjective considerations based on at-sea experience. In the latter cases, our suggestions were sent for independent evaluation by 10 experts per factor. The experts were chosen according to their experience (e.g. in ship-based and aerial seabird surveys) from a total of eight national and five international experts (listed in the Acknowledgements). Following Furness & Tasker (2000), we made changes if two or more experts suggested alterations to the original categorization in the same direction. The nine factors included are outlined below.

(a) Flight manoeuvrability

This factor took into account flight properties with regard to the potential to avoid collision with wind farms at sea. It was assessed subjectively, based on extensive field experience, and was modulated by experts as described above. Species were ranked from a very high flight manoeuvrability (score 1) to low flight manoeuvrability (score 5). A fast-flying, relatively heavy species such as the common guillemot *Uria aalge* (Pontoppidan) is thus considered much more vulnerable compared with a very agile species such as the Arctic tern *Sterna paradisaea* Pontoppidan, which is assumed to be able to escape wind turbines much better.

(b) Flight altitude

This factor was based on flight altitude assessments made during regular seabirds at sea surveys (see Distributional data). Flight altitudes were estimated using binoculars, distance meters and comparative height measures on the ships according to the following height classes: 1, 0–5 m; 2, 5–10 m; 3, 10–20 m; 4, 20–50 m; 5, 50–100 m; 6, > 100 m. A further separation of high flight altitudes was not useful. Flight altitude data were converted to a 5-point scale by using two different percentiles of the flight altitude data distributions, the median (= 50 percentile) and 90 percentile. The 90 percentile was chosen in addition to the median to take into account the few birds that flew high (i.e. 90% of the birds flew in the same or lower height classes, 10% of the birds flew in the same or upper height classes). The scores were classified as follows: 1, height class 1 for the median; 2, height class 2 for the median; 3–5, height class 3 for the median but the 90 percentile differed, score 3 had height classes < 5 for the 90 percentile, score 4 height class 5 for the 90 percentile and score 5 height class 6 for the 90 percentile.

(c) Percentage of time flying

The percentage of time flying was obtained from seabirds at sea counts, with numbers of swimming birds corrected for individuals overlooked at larger distances (see below). Species were scored 1 if 0–20% of the individuals in the transect were flying, 2 if 21–40% in the transect were flying, 3 if 41–60%, 4 if 61–80% and 5 if 81–100% of the individuals in the transect were flying.

(d) Nocturnal flight activity

Nocturnal flight activity could not be quantified by real data and was thus classified subjectively from 1 (hardly any flight activity at night) to 5 (much flight activity at night). Information for this classification was taken from comprehensive handbooks such as Glutz von Blotzheim & Bauer (1982) and Cramp & Simmons (1983). Field experience as well as personal observations were also used (Garthe & Hüppop 1996). Our classification was subsequently modulated by experts as indicated above.

(e) Disturbance by ship and helicopter traffic

Species react differently to the ship and helicopter traffic that occurs during the construction and maintenance of wind farms. Such behaviour might also give an indication of the general behaviour of birds towards disturbances. Due to the paucity of data, this factor was scored subjectively from 1 (hardly any escape/avoidance behaviour and/or none/very low fleeing distance) to 5 (strong escape/avoidance behaviour and/or large fleeing distance) and classifications were modulated by experts. Our scores resulted from extensive surveys at sea from boats, where the reactions of birds to the approaching platform were experienced constantly. Also, species' reactions to counts from aerial surveys (low-flying aeroplanes) as well as to over-flying aeroplanes and helicopters in coastal areas were used.

(f) Flexibility in habitat use

Habitats at sea are often defined by hydrographic characteristics. Because these hydrographic characteristics, e.g. water masses and fronts, depend on wind direction and speed as well as tidal stage, they often vary in location and may shift over many tens of kilometres. This factor scored the flexibility in habitat use of the different species. It could only partially be based on real data, such as published in Garthe (1997) and Skov & Prins (2001). Thus, in addition, unpublished data on seabird–habitat relationships were analysed. We scored this factor from 1 (very flexible in habitat use) to 5 (reliant on specific habitat characteristics) based on the information sources listed above. Species scored low were those occupying large sea areas with no specific habitat preference (e.g. lesser black-backed gull *Larus fuscus* Linnaeus), while species that scored high were those relying on specific habitat features (e.g. sea ducks occurring over bivalve banks on shallow grounds). Our classifications were again modulated by experts.

(g) Biogeographical population size

This factor was scored according to the respective biogeographical population size of each species. Population sizes were taken, if available, from either Rose & Scott (1997) or by collating area-specific data species by species from Lloyd, Tasker & Partridge (1991). Score 1 was given for population sizes exceeding 3 million individuals; score 2 for > 1 million up to 3 million individuals; score 3 for > 500 000 up to 1 million individuals; score 4 for > 100 000 up to 500 000 individuals; and score 5 for less than 100 000 individuals.

(h) Adult survival rate

As additional mortality due to collisions affects species with high annual survival rates more than species with low survival rates, we included this factor. A score of 1 was given if the annual survival rate ≤ 0·75; 2, > 0·75–0·80;

Table 1. Annual adult survival rates of the bird species involved in the index

Species name (English)	Species name (scientific)	Annual adult survival	Source
Red-throated diver	*Gavia stellata*	0·84	Hemmingsson & Eriksson (2002)
Black-throated diver	*Gavia arctica*	0·85	Nilsson (1977), Hemmingss on & Eriksson (2002)
Great crested grebe	*Podiceps cristatus*	0·7	Fuchs (1982)
Red-necked grebe	*Podiceps grisegena*	0·7	Estimate
Northern fulmar	*Fulmarus glacialis*	0·986	del Hoyo, Elliott & Sargatal (1992)
Northern gannet	*Morus bassanus*	0·94	del Hoyo, Elliott & Sargatal (1992)
Great cormorant	*Phalacrocorax carbo*	0·84	Krementz, Sauer & Nichols (1989)
Common eider	*Somateria mollissima*	0·895	Krementz, Barker & Nichols (1997)
Black scoter	*Melanitta nigra*	0·773	Krementz , Barker & Nichols (1997)
Velvet scoter	*Melanitta fusca*	0·77	Estimate
Arctic skua	*Stercorarius parasiticus*	0·84	del Hoyo, Elliott & Sargatal (1996)
Great skua	*Catharacta skua*	0·90	del Hoyo, Elliott & Sargatal (1996)
Little gull	*Larus minutus*	0·80	Estimate
Black-headed gull	*Larus ridibundus*	0·825	Glutz von Blotzheim & Bauer (1982)
Mew gull	*Larus canus*	0·80	Glutz von Blotzheim & Bauer (1982)
Lesser black-backed gull	*Larus fuscus*	0·93	Wanless *et al.* (1996)
Herring gull	*Larus argentatus*	0·93	Glutz von Blotzheim & Bauer (1982)
Great black-backed gull	*Larus marinus*	0·93	Glutz von Blotzheim & Bauer (1982)
Black-legged kittiwake	*Rissa tridactyla*	0·81	del Hoyo, Elliott & Sargatal (1996)
Sandwich tern	*Sterna sandvicensis*	0·88	Estimate
Common tern	*Sterna hirundo*	0·88	del Hoyo, Elliott & Sargatal (1996)
Arctic tern	*Sterna paradiasea*	0·875	del Hoyo, Elliott & Sargatal (1996)
Black tern	*Chlidonias niger*	0·88	Estimate
Common guillemot	*Uria aalge*	0·885	del Hoyo, Elliott & Sargatal (1996)
Razorbill	*Alca torda*	0·905	del Hoyo, Elliott & Sargatal (1996)
Atlantic puffin	*Fratercula arctica*	0·95	del Hoyo, Elliott & Sargatal (1996)

3, > 0·80–0·85; 4, > 0·85–0·90; 5, > 0·90. For survival rates see Table 1. Due to a lack of data, for red-necked grebe *Podiceps grisegena* (Boddaert), velvet scoter *Melanitta fusca* (Linnaeus), little gull *Larus minutus* Pallas, sandwich tern *Sterna sandvicensis* Latham and black tern *Chlidonias niger* (Linnaeus), values from closely related species had to be taken.

(i) European threat and conservation status

This factor reflected both threat and conservation status of the species in Europe as given by Tucker & Heath (1994). Species were scored 1 if the threat status was 'secure' and no species of European concern (SPEC) status given. A score of 2 was given for species with a threat status of 'secure' but a SPEC status of 4 (species whose global populations are concentrated in Europe). Species judged 'localized' for threat status were scored 3, those listed as 'declining' 4 and those judged 'vulnerable' 5.

SENSITIVITY INDEX CALCULATION

We organized the nine vulnerability factors into three groups, comprising (A) flight behaviour (factors a–d), (B) general behaviour (factors e–f) and (C) status (factors g–i). For each group, an average score of the respective factors was calculated. These average scores were subsequently multiplied by each other to give the species-specific sensitivity index (SSI) for each species:

$$\mathrm{SSI} = \frac{(a+b+c+d)}{4} \times \frac{(e+f)}{2} \times \frac{(g+h+i)}{3}$$

DISTRIBUTIONAL DATA

Distribution at sea was assessed by counts from boats following the methods of Tasker *et al.* (1984), Webb & Durinck (1992) and Garthe, Hüppop & Weichler (2002). Transects were always 300 m wide and were set to one or both sides of the vessels. Because some birds were overlooked in the outer areas of the transect, the density of swimming birds was corrected using the values provided by Stone *et al.* (1995). For grebes, being quite rare in North Sea waters and not dealt with by Stone *et al.* (1995), we used a correction factor of 1·3 based on our own, more extensive, data sets from the western Baltic Sea. The density of flying birds was not corrected, assuming that flying birds were recorded more or less completely within the transects (Stone *et al.* 1995; Garthe 1997). Data originated from the European Seabirds at Sea Database version 3.0 (July 2002) and the German Seabirds at Sea Database version 3.06 (April 2003). Databases are described in Stone *et al.* (1995) and Garthe, Hüppop & Weichler (2002).

VULNERABILITY MAPS

Seabird vulnerability to offshore wind farms is presented in maps with grids of 6′ latitude × 10′ longitude each, amounting to a total grid size of *c.* 120 km^2. Only data collected under good detectability conditions (in sea states 0–4; Garthe, Hüppop & Weichler 2002) from January 1993 to May 2003 were used for analyses. Data were summarized per season: summer = June–August, autumn = September–November, winter = December–February, spring = March–May. Coverage was not equal across the study area. Thus, data were corrected for different survey effort: for each species, the density per grid cell was obtained by dividing the sum of individuals recorded in the transect by the total transect area covered by cruises. To reduce bias due to only short visits to some grid cells, all grid cells with less than 1 km^2 covered were excluded. For each grid cell with sufficient data, the vulnerability was determined as:

$$\mathrm{WSI} = \Sigma_{\mathrm{species}} (\ln (\mathrm{density}_{\mathrm{species}} + 1) \times \mathrm{SSI}_{\mathrm{species}})$$

Thus, for each species, the respective SSI value was multiplied with the natural logarithm of its density (+1, to avoid undefined values) and subsequently summed over all species.

We also defined three levels for a final evaluation of the area under investigation. The levels were established with the following considerations. The median divides the whole sample into two equally large parts, i.e. half of the grid cells have a WSI larger than the 'average'. By definition these are the areas with a wind farm vulnerability higher than the average. To end up with a more conservative estimate for areas of 'concern' we decided to use the 60 percentile rather than the 50 percentile (= median). Accordingly we assumed a 'level of major concern' for the fifth of all grid cells with the highest WSI indices.

SENSITIVITY ANALYSIS FOR THE WSI

In order to verify how the WSI might be affected by inaccurate scores for any of the nine factors listed above, a sensitivity analysis was carried out. We chose three species, one with a high SSI, one with a medium SSI and one with a low SSI. Randomly, each score for any of the eight factors was altered. In a first run, the scores were altered either by upgrading or downgrading them by 1 (determined by random and only if applicable, e.g. score 5 could not be increased and thus remained). In a second run, the scores were altered either by upgrading or downgrading them by 2 (again determined randomly and only to the extent possible).

Results

SENSITIVITY INDEX

The species had strongly differing sensitivity index values (Table 2). Black-throated diver *Gavia arctica* (Linnaeus) and red-throated diver *Gavia stellata* (Pontoppidan) ranked highest (i.e. were most sensitive), followed by velvet scoter, sandwich tern and great cormorant *Phalacrocorax carbo* (Linnaeus). The lowest SSI values were

Table 2. Score of the nine vulnerability factors and the resulting species sensitivity index (SSI) values for each of the 26 seabird species. For details see text

Bird species	Flight manoeuvrability	Flight altitude	% flying	Nocturnal flight activity	Disturbance by ship and helicopter traffic	Habitat use flexibility	Biogeographical population size	Adult survival rate	European threat and conservation status	SSI
Black-throated diver	5	2	3	1	4	4	4	3	5	44·0
Red-throated diver	5	2	2	1	4	4	5	3	5	43·3
Velvet scoter	3	1	2	3	5	4	3	2	3	27·0
Sandwich tern	1	3	5	1	2	3	4	4	4	25·0
Great cormorant	4	1	4	1	4	3	4	3	1	23·3
Common eider	4	1	2	3	3	4	2	4	1	20·4
Great crested grebe	4	2	3	2	3	4	4	1	1	19·3
Red-necked grebe	4	2	1	1	3	5	5	1	1	18·7
Great black-backed gull	2	3	2	3	2	2	4	5	2	18·3
Black tern	1	1	4	1	2	3	4	4	4	17·5
Common scoter	3	1	2	3	5	4	2	2	1	16·9
Northern gannet	3	3	3	2	2	1	4	5	3	16·5
Razorbill	4	1	1	1	3	3	2	5	2	15·8
Atlantic puffin	3	1	1	1	2	3	2	5	5	15·0
Common tern	1	2	5	1	2	3	3	4	1	15·0
Lesser black-backed gull	1	4	2	3	2	1	4	5	2	13·8
Arctic tern	1	1	5	1	2	3	3	4	1	13·3
Little gull	1	1	3	2	1	3	5	2	4	12·8
Great skua	1	3	4	1	1	2	5	4	2	12·4
Common guillemot	4	1	1	2	3	3	1	4	1	12·0
Mew gull	1	3	2	3	2	2	2	2	4	12·0
Herring gull	2	4	2	3	2	1	2	5	1	11·0
Arctic skua	1	3	5	1	1	2	4	3	1	10·0
Black-headed gull	1	5	1	2	2	2	1	3	1	7·5
Black-legged kittiwake	1	2	3	3	2	2	1	3	1	7·5
Northern fulmar	3	1	2	4	1	1	1	5	1	5·8

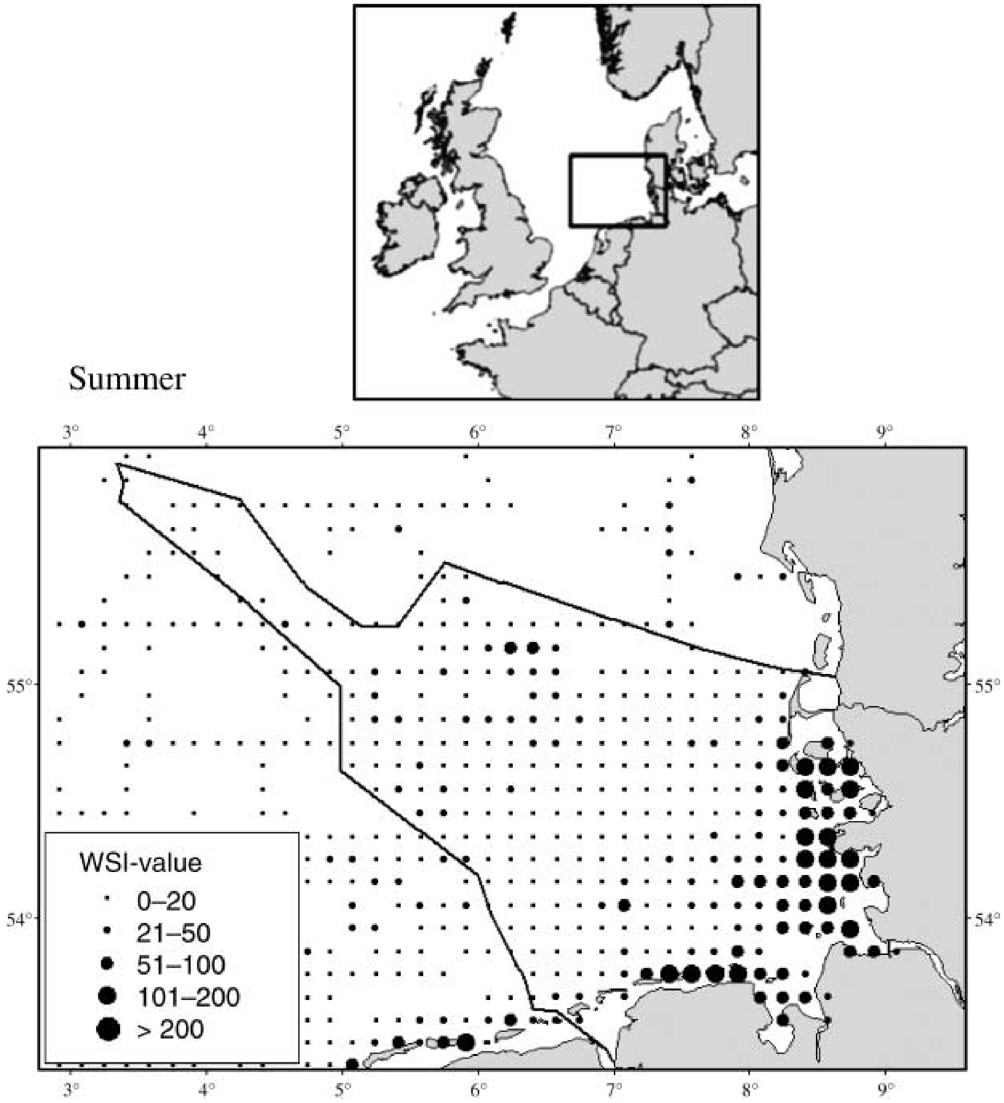

Fig. 1. Spatial distribution of the wind farm sensitivity index (WSI) values (all seabird species combined) in the south-eastern North Sea in summer (June–August) 1993–2002. For assumptions and calculations see text.

calculated for black-legged kittiwake *Rissa tridactyla* (Linnaeus), black-headed gull *Larus ridibundus* Linnaeus and northern fulmar *Fulmarus glacialis* (Linnaeus).

Sensitivity analyses for the three species selected resulted in moderate deviation when the scores where randomly altered by 1 (mean of 10 runs per species). The SSI for red-throated diver changed from 43·3 to 44·8, the SSI for common eider *Somateria mollissima* (Linnaeus) from 20·4 to 23·0 and that for black-headed gull from 7·5 to 10·0. When the scores were randomly altered by 2, the changes were more pronounced (mean of 10 runs per species). The SSI for red-throated diver was reduced from 43·3 to 24·4, the SSI for common eider increased from 20·4 to 30·3 and that for black-headed gull from 7·5 to 16·4.

AREAS OF VULNERABILITY

Throughout the whole year, WSI values in coastal waters of the south-eastern North Sea were considerably higher than those of waters further offshore. Focusing on the German sector, the coastal zone had consistently moderate to high WSI values in summer (Fig. 1). The area around Helgoland showed some moderate values whereas vulnerabilities further offshore were low. In autumn, WSI values were generally lower than in summer, but a number of coastal sites reached moderate to high vulnerabilities (Fig. 2). Some offshore areas gained importance compared with summer because species' distributions were less confined to breeding sites so that they were more widely distributed. Also, autumn migration certainly created higher densities in areas far away from the coast. In winter, the south-easternmost part of the German Bight was less vulnerable than in summer and autumn (Fig. 3). Nevertheless, it was obvious that WSI values in the coastal zone in winter were usually moderate to high whereas the values far away from the coast were low or very low. In spring, vulnerabilities were again quite high in most areas of the coastal zone but were also moderate to high in areas up to 70–80 km off the northern part of the German coast (Fig. 4).

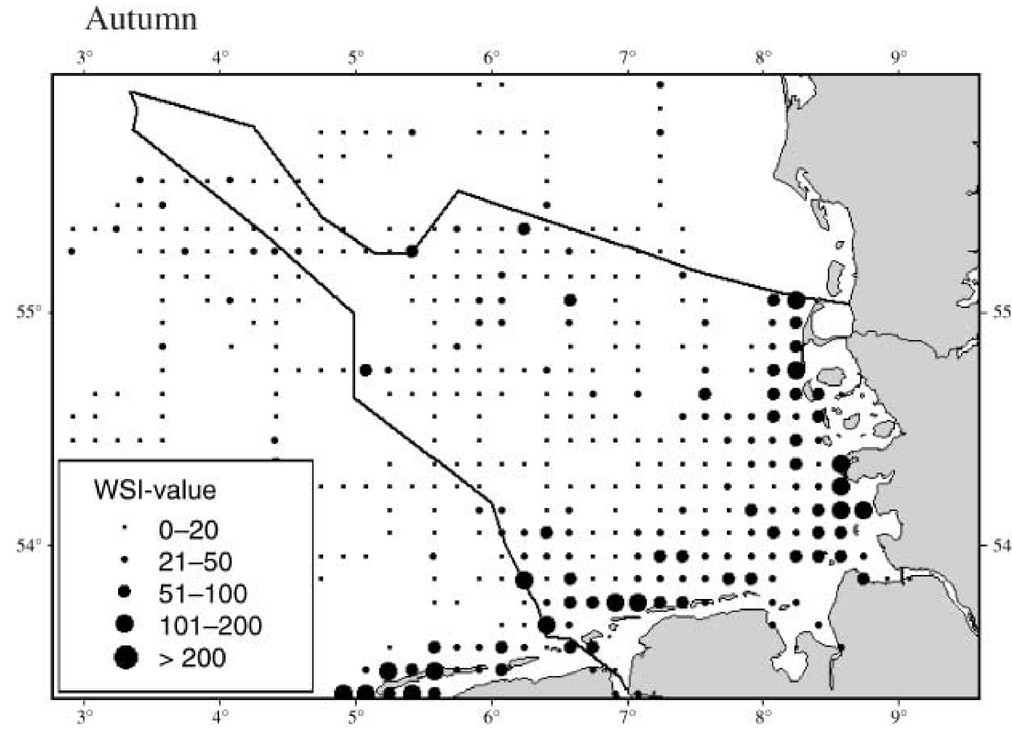

Fig. 2. Spatial distribution of the wind farm sensitivity index (WSI) values (all seabird species combined) in the south-eastern North Sea in autumn (September–November) 1993–2002. For assumptions and calculations see text.

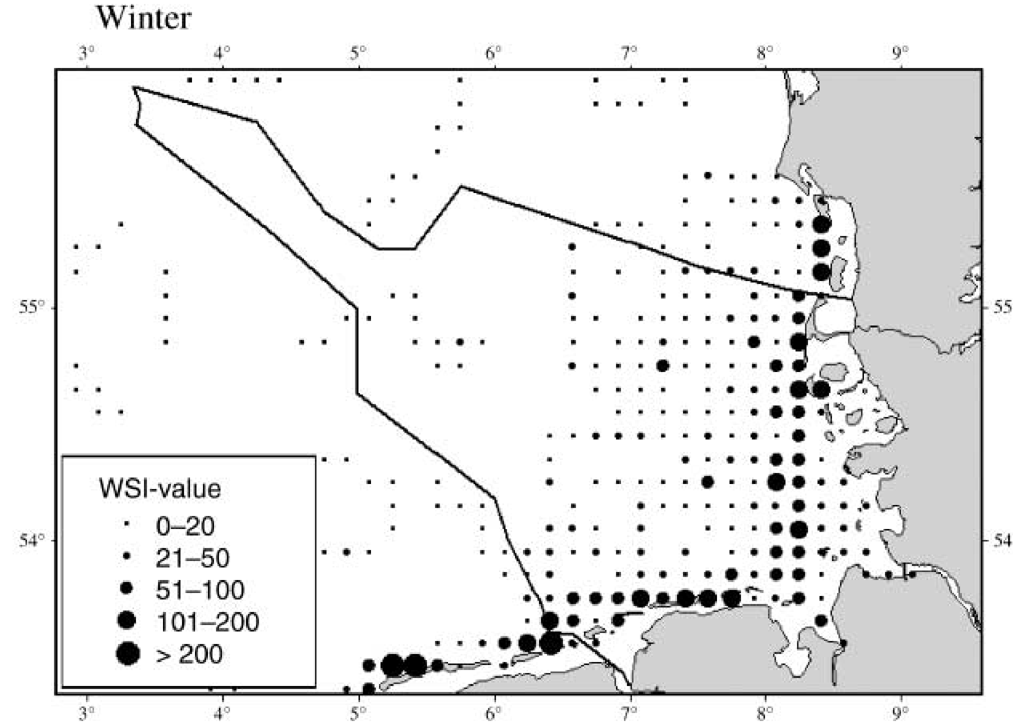

Fig. 3. Spatial distribution of the wind farm sensitivity index (WSI) values (all seabird species combined) in the south-eastern North Sea in winter (December–February) 1993–2003. For assumptions and calculations see text.

Discussion

SUITABILITY OF THE WSI

Five factors incorporated into the WSI were based on real data but this was not possible for the remaining four factors. We therefore decided to assess these factors subjectively based on at-sea experience. Experts then evaluated our initial scores by reading and considering our values presented on a list, based on their own experiences related to that topic. Such a procedure, called the Delphi technique, has been applied broadly

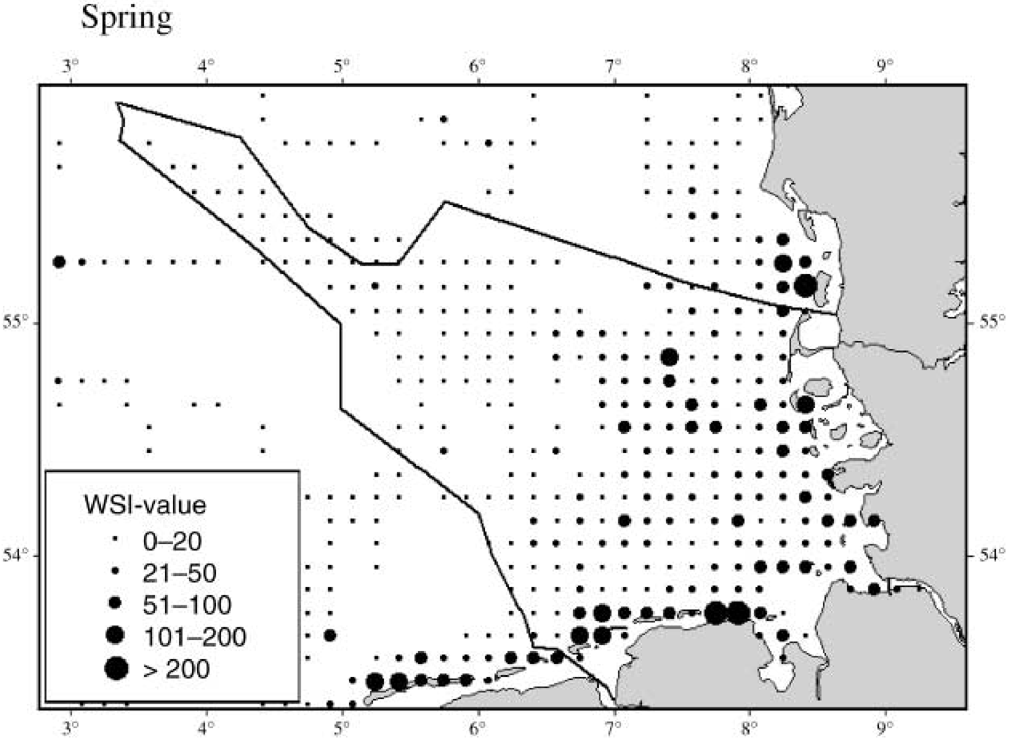

Fig. 4. Spatial distribution of the wind farm sensitivity index (WSI) values (all seabird species combined) in the south-eastern North Sea in spring (March–May) 1993–2003. For assumptions and calculations see text.

before, for example in habitat suitability indices of the US Fish and Wildlife Service (Crance 1987). Recently Cowling *et al*. (2003) compared an expert-based and a systematic algorithm-based approach to identifying priority areas for conservation in the Cape Floristic Region. They concluded that 'rather than emphasize the dichotomy between expert and systematic approaches, conservation planners should devise ways of integrating them'. Although not without difficulties, these data based on expert judgement are currently the best available.

In general, indices depend strongly on the factors selected and the way they are weighed against each other. The WSI proposed in this paper is no exception in this respect. Nevertheless, there are at least four reasons why this index seems to be well-suited to fulfil the urgent need to assess the vulnerability of all seabirds that occur in a large area. First, the final SSI values show substantial differences between the species. Hence, the WSI combines numerical abundance data with evaluations of the sensitivity and importance of the different species. Secondly, not too many alterations were required on the basis of the evaluations made by national and international experts (four changes for flight manoeuvrability, nine changes for nocturnal flight activity, 18 changes for disturbance by ship and helicopter traffic and 15 changes for habitat use flexibility), suggesting that the species-specific scores for each vulnerability factor were well chosen. Interestingly, it has been shown by morphometric measurements and behavioural observations in the literature (Verbeek 1977; Camphuysen 1995) that lesser black-backed gulls have a higher flight manoeuvrability than herring gulls *Larus argentatus* Pontoppidan, providing an example of how accurate our expert judgement system was. Thirdly, sensitivity analyses showed that minor changes in the scores did not affect the SSI much, although major changes may do so. Fourthly, the spatial representation of the WSI values fits well with previous evaluations of the location of important bird areas (Skov *et al*. 1995).

SUGGESTIONS FOR CONSERVATION AND MANAGEMENT

In the German Bight, seabird vulnerability towards wind farms seems to be a function of distance to the coast. The highest values were found relatively close to the coast and lowest values (very) far from the shore. However, there are differences between the two coasts. North of the East Frisian Islands (the southern part of the German sector), the values decrease at much shorter distances from the coast than they do west of the North Frisian Islands (in the eastern part of the German sector). These patterns originate from the density distribution of all species, with a bias towards those species that are ranked high in the SSI and with less emphasis towards species having a low SSI. However, in no case did high densities of a single species create very high WSI values. Thus, the vulnerability maps are more a summary of all species contributing to the final WSI value to a different degree. Species' attributes do not exhibit much variation over the study area so that they contribute primarily via the input factors for the SSI. Thus, spatial variation of the WSI values is basically a reflection of summarized species' density distributions.

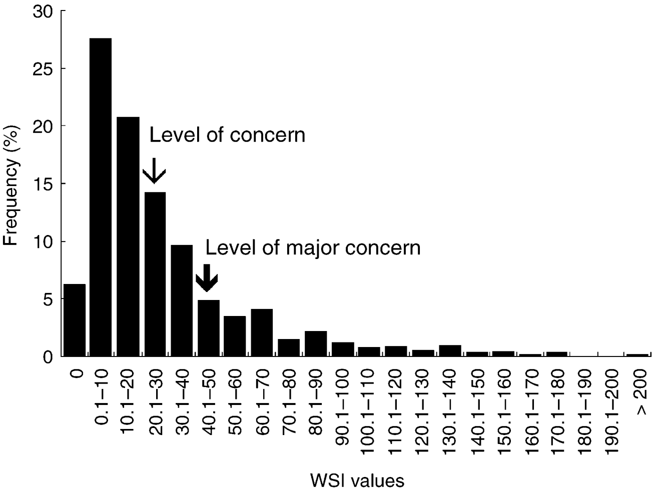

Fig. 5. Average frequency distribution (in percentage) of the WSI values in the German sector of the North Sea. The values were obtained by calculating means per size class over the fours seasons presented in Figs 1–4.

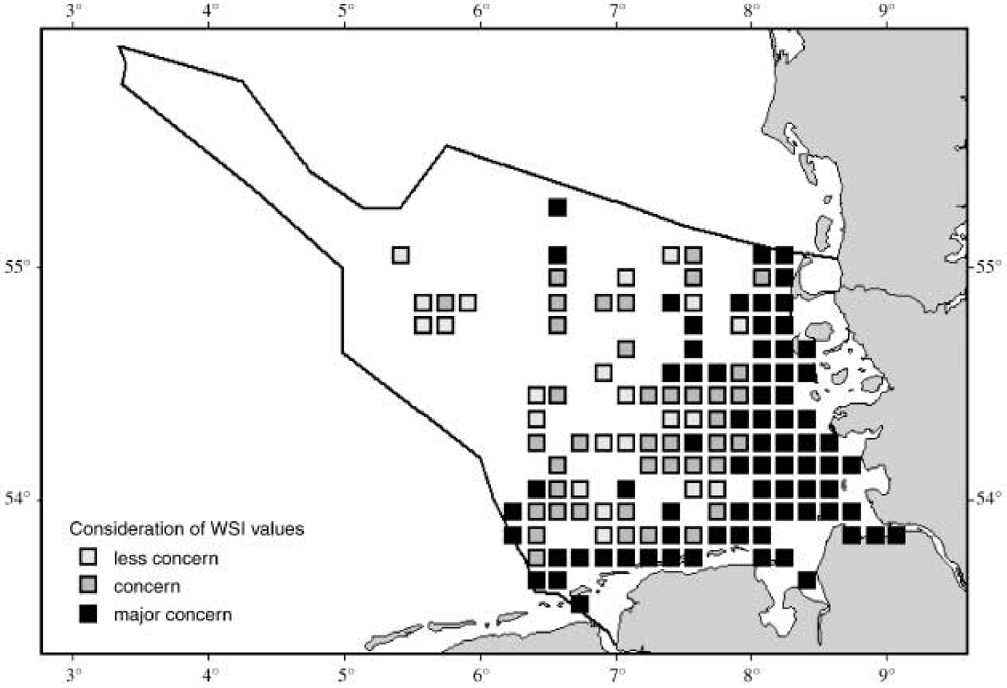

Fig. 6. Areas in the German sector of the North Sea where wind energy utilization is considered to be of 'no (less) concern', 'concern' or 'major concern'. Areas not studied in at least one of the seasons are left blank.

From the average frequency distribution of the WSI values over the four seasons (Fig. 5) it is apparent that most areas do not hold important concentrations of seabirds and thus do not appear particularly vulnerable to marine wind farm construction. However, there is no doubt that some areas have high to very high WSI values and hence are unsuited for such constructions. We suggest a level of concern set at the 60 percentile of the average frequency distribution (= WSI of 24) and a level of major concern set at the 80 percentile (= WSI of 43).

Wind farms operate the whole year round. Threats might therefore also be important even if they only affect species in a single season. Thus, spatial information from Figs 1–4 has been compiled for the whole year. The values of the most important season per grid cell are visualized spatially in Fig. 6 in relation to the levels of concern discussed above.

SYNTHESIS AND APPLICATIONS

This index has been developed primarily for marine wind farm site selection purposes and comparative area assessments. It might be a useful tool for strategic environmental impact assessments (EIA). However, it cannot substitute for proper, detailed EIA, which usually only cover small areas over a limited time period

(BSH 2003). Results from such small-scale EIA should therefore be set into a more global perspective, provided, for example, by large mapping projects and detailed behavioural studies. Because this is without doubt difficult in normal EIA, particularly in highly dynamic coastal/marine habitats, this study aims to fill this gap by providing comprehensive and up-to-date background information.

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PRACTITIONER'S PERSPECTIVE

Lack of sound science in assessing wind farm impacts on seabirds

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Introduction

Electrical power generation from wind farms has grown rapidly in the UK and European Union (EU) in the last decade and is set to grow further. By 2020, the EU proposes to source 20% of energy from renewable sources (Directive 2009/28/EC). Wind energy is expected to provide 9–14% of global electricity generation by 2050 (IPCC 2011). This may eventually reduce climatic change and its negative impacts on biodiversity, but there are also several poorly quantified negative effects on wild species of renewable energy generation, including wind turbines. For example, birds and bats are killed by colliding with turbine blades or towers and there may be effects of wind farms on mortality and reproductive rates of a wide range of species from avoidance and displacement. Birds may incur additional costs or forego benefits because of reduced transit or foraging within or near to wind farms (Drewitt & Langston 2006; Searle *et al*. 2014). Depending upon the strength of density-dependent compensatory processes, these effects could reduce the population to a lower stable level or cause its extinction (Wade 1998; Niel & Lebreton 2005). Except in the rare circumstances where density dependence is exactly compensating, such effects would always diminish population size. Positive effects of renewable energy infrastructure on populations of wild species have also been proposed and, in a few cases, quantified. These include possible enhancement of food resources of seabirds by protection from fishing from the presence of offshore installations and the provision of artificial substrates as habitat for fish and invertebrates (Inger *et al*. 2009; Langhamer, Wilhelmsson & Engström 2009).

The UK has the best wind resources in Europe (DECC 2011). Although the cost per megawatt-hour of electricity generation from offshore wind turbines averages about twice that for onshore installations (Bilgili, Yasar & Simsek 2011; Chu & Majumdar 2012), offshore wind power is currently favoured over onshore by the present UK government because of public perceptions of nuisance and landscape consequences of onshore turbines. The UK also has internationally important breeding populations of seabirds. It holds more than 10% of the world's breeding population of eight species, of which three have more than half of their global breeding population in the UK (Brown *et al.* 2015). Because seabirds range over long distances, there may be cumulative impacts on a breeding colony from several wind farms (Masden *et al.* 2010). Seabirds are long-lived and late-maturing, which renders their population growth rate particularly sensitive to additional mortality from collisions or displacement (Niel & Lebreton 2005). The importance of these seabird populations and their sensitivity places a heavy responsibility on those conducting and acting upon scientific assessments of the impacts of offshore wind farms on seabirds to comply with the protection measures and the precautionary principle enshrined in the EU Birds and Habitats Directives (Directive 2009/147/EC and Council Directive 92/43/EEC).

For the UK, and other countries within the European Union, the regulation of wind farm construction requires the assessment of possible damage to the integrity of sites and populations under the EU Habitats and Birds Directives. Consideration must be given to impacts on bird populations of a project on its own and in combination with others already in existence, given consent or planned. Governments give or refuse consent for the construction of wind farms after taking into account the scale and level of certainty of the impacts indicated by these assessments.

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However, there are no definitive quantitative thresholds or criteria defining how large or likely expected impacts must be for damage to the integrity of sites and populations to be anticipated and for consent for wind farm construction to be denied or limited. Consent can be granted only if it is ascertained that there will not be an adverse effect on the integrity of a Natura site, excepting in cases where there are imperative reasons of overriding public interest for consent and no alternative solutions (Article 6 (4) of Directive 92/43/EEC). In recent years, several plans for large offshore wind farms have been approved and some built in UK and EU waters close to large seabird populations because the competent authority judged there was no expected adverse effect on the integrity of the Natura sites involved. For example, in 2014 approval was granted for several extensive wind farms at Hornsea (England, UK Government) and the Firth of Forth (Scotland, Scottish Government), close to internationally important breeding populations of seabirds. This approach contrasts with that in some other EU states. In Germany and Denmark, for example, offshore wind farms have been subject to rigorous marine spatial planning with the aim of avoiding potential conflict with nature conservation as part of the required Strategic Environmental Assessment (SEA) process recommended in EU Commission guidance (European Commission 2011). The German Cabinet approved Europe's first maritime spatial plan in September 2009, after a considerable effort in terms of surveys and research to identify marine sites of high nature value and potential conflict areas with wind farms and to establish zones for various activities and infrastructure. The offshore SEA covering UK waters is not of comparable quality.

In this perspective, we argue that the methods and data used in these cases for estimating effects upon seabird demographic rates and translating them into potential impacts on seabird populations do not allow adequate assessment of effects on site integrity. As a result, sound science and its logical interpretation are lacking in Environmental Impact Assessments of this large and expanding industry.

Estimates of the effects of wind farms on seabird demographic rates are neither robust nor validated

Collision risk models (CRMs) are used to predict the number of fatal collisions of flying birds with wind turbines and per capita additional mortality rates. In the UK, the most widely used CRM is that of Band (2012) (see review by Masden & Cook (2016)). The model requires estimates or assumptions about bird numbers and ages at the wind farm, attribution of birds at the wind farm to source populations, sizes and age structure of source populations, flight behaviour and avoidance rates. Data specific to the project and species being assessed are usually collected on seabird numbers and flight heights, judged by eye, but these estimates are subject to substantial uncertainties, variability and potential biases (Johnston *et al.* 2014), including:

1. accuracy of input variables is rarely quantified, is often poor, and the CRM outputs are highly sensitive to the values used, including flight speed (Masden 2015), and avoidance rate estimates;

2. in many cases, birds at risk are not attributed to source populations because recently developed tracking technologies are either not deployed at all or not on a sufficient scale for robust estimation;

3. count and flight height data are usually insufficient in quantity and quality for precise estimation of seasonal variation, age structure and age differences (Band 2012).

Total avoidance rates used for CRM calculations for seabirds, including within-wind farm avoidance of individual turbines and macro-avoidance by movement of birds around the turbine array, are most often based upon judgement or extrapolation from other contexts rather than pertinent data. Empirical values are only available from a few species (mostly gulls and terns) and usually extrapolated from studies of onshore wind farms, where different circumstances prevail (Cook *et al.* 2014). Robust direct estimates of within-wind farm avoidance rates are lacking for seabird species frequently present in and near planned and consented offshore wind farms in the UK, such as northern gannet *Morus bassanus* and black-legged kittiwake *Rissa tridactyla* (Cook *et al.* 2014). Macro-avoidance and displacement rates have been estimated using radar, visual surveys and imaging, but robust quantitative estimates with confidence intervals are generally not used in impact assessments. Estimates of macro-avoidance for the same species can be highly variable (e.g. Petersen *et al.* 2006; Krijgsveld *et al.* 2011; Vanermen *et al.* 2012, 2013 for northern gannet). This may well be because macro-avoidance varies with the relative positions of nesting and foraging sites, foraging site quality and seasonal timing of studies.

At onshore wind farms, carcasses of some of the birds killed by collisions with turbines can be collected during systematic searches and probabilities of their detection can be estimated. This allows estimation of numbers of deaths per unit time and confidence intervals, even if with low precision (e.g. Bellebaum *et al.* 2013). These methods help to quantify uncertainty and remove bias, but are currently impractical for offshore wind farms. Alternatives that use video or thermal camera systems have not yet been deployed sufficiently to substitute for them. Where direct measurements of avoidance rates are lacking, Band (2012) recommends use of a range of plausible values. However, this can result in a 20-fold variation in assumed per capita mortality rates (APEM 2015).

Overall, CRM outputs are sensitive to the combined effects of multiple assumptions of unknown accuracy, sampling errors and unquantified biases. Only for species that almost completely avoid entering wind farms can the annual per capita mortality rate from collisions be estimated reliably and with robust confidence limits (Desholm & Kahlert 2005). Validation tests of offshore seabird CRM outputs, in which expectations from pre-consent data and modelling are compared with independent robust post-construction measurements of numbers of collision deaths, have not been conducted.

Estimation of effects on seabird demographic rates of the displacement and barrier effects of wind farms is even less well developed. Avoidance of wind farms by foraging and migrating birds can be substantial and operate over long distances from the turbines (Desholm & Kahlert 2005; Petersen *et al.* 2006; Percival 2010), but the degree to which this affects travel times and costs, access to food and mortality and reproductive rates of breeding seabirds has not been measured reliably. In the case of migrating birds, the displacement and increased travel costs caused by avoidance of a single wind farm may be trivial relative to the total length and cost of the journey (Masden *et al.* 2009), but effects on demographic rates have not been robustly quantified by empirical studies for central-place foraging breeding seabirds repeatedly subjected to barrier or displacement effects. Simulation modelling has been performed of potential effects of displacement by as yet unconstructed wind farms on seabird time and energy budgets and demographic rates (Searle *et al.* 2014). Modelled potential effects of displacement included considerable declines in adult survival of up to 2·1% for black-legged kittiwake and up to 4·9% for Atlantic puffin *Fratercula arctica* (both for the Forth Islands cumulative effects: table 3·3 of Searle *et al.* 2014), though simulated effects on survival for other species and sites and for breeding productivity generally were small. The species for which collision mortality can be reliably estimated as low, because of strong avoidance, are those for which displacement and barrier effects upon demographic rates are potentially the largest, but currently unquantified.

In summary, the procedures currently used to calculate expected effects of proposed wind farms on seabird per capita mortality rates and breeding success largely involve modelling with little firm empirical data. Moreover, actual outcomes at wind farms that have been constructed have not been measured, so model predictions are not tested and there is no adaptive improvement of the decision-making process (Nichols *et al.* 2015). As a result, scientifically robust and defensible calculations of effect sizes for changes in seabird demographic rates caused by collision, displacement and barrier effects of offshore wind farms, with confidence intervals, are currently lacking.

Procedures for translating effects on demographic rates into projected impacts on seabird population size and trends are inappropriate and untested

Assessments of the impacts of offshore wind farms in the UK on seabirds require that the highly uncertain estimates of effects on demographic rates are translated into projections of impacts on population size or trend. Decisions about UK offshore wind farms have been based upon, or influenced by, the following *effect–impact translation procedures*.

POTENTIAL BIOLOGICAL REMOVAL (PBR)

The recommended and robust application of this method is to identify a level of additional mortality *above* which a decline of the affected population to eventual extinction would be likely (Niel & Lebreton 2005). In recent cases, such as Hornsea, the UK statutory conservation agencies advised using this method in wind farm assessments to identify demographic rate thresholds *below* which additional mortality estimated from CRMs and related methods is unlikely to adversely impact the population (Natural England 2014). This reverse application involves faulty logic because PBR's value of maximum potential excess growth may not be realizable in the ecological circumstances of a particular population of interest. In addition, PBR does not estimate the effect of additional mortality on population size.

Potential biological removal provides thresholds of additional mortality that are sensitive to assumptions made about the form of density dependence. The studies of Wade (1998) and Bellebaum *et al.* (2013) show that the shape parameter of the generalized logistic equation has a strong effect on PBR results. Details of the form of density-dependent relationships are rarely known for animal populations and are unknown for any of the UK seabird populations to which PBR has been applied. These uncertainties have prompted the use of 'recovery factors', which are constants by which the maximum possible value of the PBR threshold is multiplied to give a safety margin (Dillingham & Fletcher 2008). The values used for these recovery factors are based upon judgement. There has been no empirical validation of their safety by observation of the effects on population size of known additional mortality rates from any source in any bird species.

ACCEPTABLE BIOLOGICAL CHANGE (ABC)

This method, which has not yet been published in the peer-reviewed scientific literature, was developed by Marine Scotland, a Scottish government agency, and used in a recent assessment of the impact of wind farms on internationally important seabird populations in the Firth of Forth (Marine Scotland, 2015). It uses probabilistic forecasts from stochastic seabird population models to assess the probability of a particular level of population size occurring at some future time, such as the end of the period of operation of a wind farm, in the absence of the wind farm. In practice, this probability is obtained from a simulation model of the population in which variation in expected future population size arises from supposed future demographic and environmental stochasticity in demographic rates, when applied to the population of a specified initial size over a period of 25 years, which is the usual licence period for an offshore wind farm. If the best estimate of future population size, after the expected effects of the wind farm on demographic rates are taken into account, equals or exceeds the population size that is 66·7% likely to be equalled or exceeded in the absence of the wind farm, then ABC deems that the impact of the wind farm is acceptable.

The weaknesses of this approach are severe. First, the accuracy of projections of the demographic rates used in the model of the unimpacted seabird population long into the future is highly uncertain and untested. Perversely, the greater the estimated uncertainty, the larger the acceptable population decline. Secondly, it does not address the uncertainties in size of the effects of the wind farm on demographic rates, which are mostly unquantified. Hence, ABC does not assess the risk or probability that the wind farm itself will cause a particular specified outcome or change at all. It simply proposes that an event half as likely to occur as not *if there is no wind farm* should be the threshold for acceptability. Thirdly, the threshold probability for acceptance is arbitrary and is plucked from an unrelated context: IPCC guidelines about the appropriate language to describe the likelihood of an event or outcome of at least given size happening, based upon available evidence (Mastrandrea *et al.* 2010). The threshold chosen for ABC is described as 'unlikely' in the IPCC lexicon. However, this lexicon was not developed for the purpose of determining acceptable levels of risk, which also requires that the societal costs and benefits of possible outcomes are evaluated. It is not only the chance of being wrong that is important, but also the scale of the damage caused by being wrong. No justification is given by the proponents of ABC for using as a tolerable risk threshold for damage to important nature conservation sites and their species a term selected arbitrarily from a lexicon developed by IPCC for a different purpose.

DECLINE PROBABILITY DIFFERENCE (DPD) METHOD

Large uncertainties in predicting future seabird population changes might not matter if *differences* in the probability of a specific population outcome between scenarios with and without wind farms could be predicted reliably and used as criteria for acceptability. This focus on differences in risk has been proposed by the Joint Nature Conservation Committee & Natural England (2012). It was suggested that assessments of acceptable impact should be based upon an arbitrary threshold level of absolute difference between the impacted and unimpacted scenarios in the probability that a population decline by an arbitrary proportion of the initial level would occur. In principle, this approach is preferable to ABC because it takes the uncertainty in the predicted magnitude of the effect of the wind farm into account. However, the results of this procedure are sensitive to the selection of unpredictable baseline (unimpacted) demographic rates. For example, in a model in which the selected values of baseline demographic rates imply a rapid increase in projected population size, it is unlikely that even large additional mortality would give rise to an appreciable absolute difference in the probability of population decline between impacted and unimpacted scenarios. Both probabilities would be very small. If the selected rates were inaccurate and the true values instead led to the unimpacted population being approximately stable, the same level of additional mortality could result in a large difference in the probability of population decline between impacted and unimpacted scenarios.

In practice, uncertainties in future projections of both unimpacted and impacted populations are mostly unquantified, so the probability distribution of an outcome for population size cannot be calculated. This problem makes approaches, such as ABC and DPD, which are based upon assessments of probability or difference in probability unworkable, given present knowledge.

The danger of acceptability thresholds without a logical or empirical basis

All the effect–impact translation procedures described above have a built-in threshold for an acceptable impact. Such thresholds are naturally attractive to decision-makers because they appear to offer a clear-cut, evidence-based way to establish whether damage to the integrity of a designated site will or will not occur. However, in the case of ABC and DPD, the thresholds offer only false security because they are arbitrary, have no foundation in population biology and embed the acceptance of some adverse impact on population size. Whilst PBR does identify a threshold based upon population biology, it is one that is misapplied to the problem at hand. PBR could be used to identify a threshold level of effect of wind farms on demographic rates above which a decline of an affected closed population to eventual extinction would be almost certain. However, population declines of a wide range of magnitudes, short of extirpation, could be caused by effects of wind farms on demographic rates well below this. How large these declines would be depends upon the form and strength of density dependence, which are unlikely to be measured with sufficient precision, and the magnitude of such declines has not been quantified using PBR in any UK wind farm assessment. We argue that such declines would constitute adverse effects on site integrity. Hence, PBR is not an appropriate method for assessing population impacts of a development in a manner that is relevant to the concerns of the public and decision-makers.

A robust effect–impact translation procedure without a built-in threshold

A more robust procedure for evaluating population-level impacts of wind farms on seabirds is to calculate, using a density-independent Leslie matrix model (LMM), expected population sizes, with and without the expected effects on demographic rates of the wind farm, at the end of its lifetime. The ratio of the expected population size with the wind farm to that without it (the counterfactual of population size) is a robust metric for likely population-level impact of a specified set of effects of the wind farm on seabird demographic rates. This LMM-ratio approach is relatively insensitive to the assumptions made about the magnitude, variability and trends of demographic rates in the model from which it is calculated, because the same uncertainties apply to both the impacted and unimpacted scenarios. Hence, this effect–impact translation procedure contributes little to the uncertainty in the difference in population size caused by the wind farm.

Density dependence tends to reduce the impact on population size of a given effect of the wind farm on demographic rates, so the LMM ratio calculated from the density-independent model is a precautionary worst-case outcome. We think it probable that density-dependent compensation occurs in UK seabird populations and that including it in LMMs (e.g. Miller, Jensen & Hammill 2002) could lead to more accurate estimates of population impact than those based upon density-independent LMMs. However, accuracy would only be increased if robust estimates of the form and strength of density dependence were available or population outcomes could be shown to be insensitive to assumptions made about density dependence in the absence of reliable quantification. In practice, no assessments of population impacts of additional mortality from wind farms on UK seabirds have included empirical estimates of the form and strength of density dependence because applicable estimates seem not to be available. Until adequate quantification of density dependence is available, we recommend the use of density-independent LMM ratios.

Whether density dependence is included or not, there is no threshold value of acceptability built into the LMM-ratio metric. Population estimates from Leslie matrix models, for example Trinder (2014), and population models fitted using a Bayesian approach (Marine Scotland, 2015) have been calculated as part of offshore wind farm impact assessments, but their results have not been used explicitly as counterfactuals in decision-making about the acceptability or otherwise of UK offshore wind farm projects. Based upon the documentation of UK wind farm assessments, we believe that methods such as PBR, ABC and DPD have been used in preference to LMMs because they provide thresholds which can be used to argue that site integrity will not be affected by the project, whilst LMMs deliberately do not provide a threshold. We argue that, because the thresholds offered by the other methods are arbitrary and invalid, LMMs should be used as the standard, best-practice method, and we note that any of the potential positive effects of offshore wind farms on seabird demographic rates, if quantified, could be included in an integrated assessment using an LMM-ratio metric.

Conclusions

Current procedures for collecting empirical data, modelling effects on demographic rates and translating those effects into projected impacts of offshore wind farms on seabird populations are inadequate. Empirical measurements of effects of offshore wind farms on seabird demographic rates from fieldwork are not sufficiently precise and unbiased. In the case of some important parameters such as turbine avoidance rates and the strength of density-dependent compensation, estimation is rarely even attempted. As a result of these holes in the evidence base, the magnitude of effects of wind farms on seabird demographic rates cannot be estimated accurately and the level of bias and precision in the estimates used cannot be calculated.

To overcome these problems, responsible governments should require the renewable energy industry to co-fund an adequate level of field-based research to estimate effects of wind farms on seabird demographic rates more reliably. The Offshore Renewables Joint Industry Partnership (ORJIP) intends to address this need (Carbon Trust 2015), but the objectives of its project need to be greatly expanded with regard to the number of species covered, proximity to their breeding colonies and robustness of estimation. Further development and deployment of radar, imaging and tracking techniques are likely to be required, including remote download 3D tracking (Cleasby *et al.* 2015). A defensible approach is then needed to translate these effect measurements, and their uncertainties, into expected impacts on populations. We propose that the counterfactual population ratio from a density-independent Leslie matrix model would be an appropriate method for this translation.

Quite separate from these problems of measurement, estimation and modelling, there is a fundamental logical flaw in the link between scientific assessment and decision-making about the acceptability of wind farm impacts. Modelling approaches have been contrived that seek to define an acceptable threshold for a projected negative impact of a wind farm on seabird populations, below which this negative impact is regarded as causing no adverse effect on site integrity. However, the emperor has no clothes: the thresholds used to define the acceptability of projected offshore wind farm impacts are arbitrary, poorly reasoned, not designed for the purpose and have no valid biological basis. Hence, it is necessary to revise decision-making procedures, regardless of what effect-to-impacts translation procedure is used. At present, inadequate data are being combined with arbitrary and scientifically unsupportable thresholds to argue that wind farms will cause no damage to the integrity of sites designated to restore and maintain Europe's biodiversity.

Population viability analysis indicates that the probability of long-term persistence of an animal population and its mean time to extinction generally increases with its average size (Akçakaya, Burgman & Ginzburg 1999). According to European Commission guidance on managing the network of protected sites established by the EU Birds and Habitats Directives (Natura 2000 sites), Article 6 of the Habitats Directive provides that 'The integrity of the site involves its ecological functions. The decision as to whether it is adversely affected should focus on and be limited to the site's conservation objectives' (European Commission 2000). In addition, for the integrity of a site not to be adversely affected, a Court of Justice of the European Union decision (Court of Justice of the European Union 2013; Para. 39) found that the 'site needs to be preserved at a favourable conservation status', which entails 'the lasting preservation of the constitutive characteristics of the site concerned that are connected to the presence of a natural habitat type whose preservation was the objective justifying the designation of that site'. Based upon this reasoning, we argue that some damage to the integrity of a designated site will have been sustained if populations of the seabirds for which it was designated are diminished, even to a small degree, by the effects of a wind farm, compared with what they would otherwise have been. If that is expected to be the case, it does not mean that the competent authority cannot give consent for a wind farm. Article 6(4) of the Habitats Directive sets out tests that determine whether the expected damage can be accepted and compensated for. However, poor science should not be used to avoid those tests by claiming that no damage will occur.

Data accessibility

Data have not been archived because this article does not contain data.

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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 ($\geq$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).

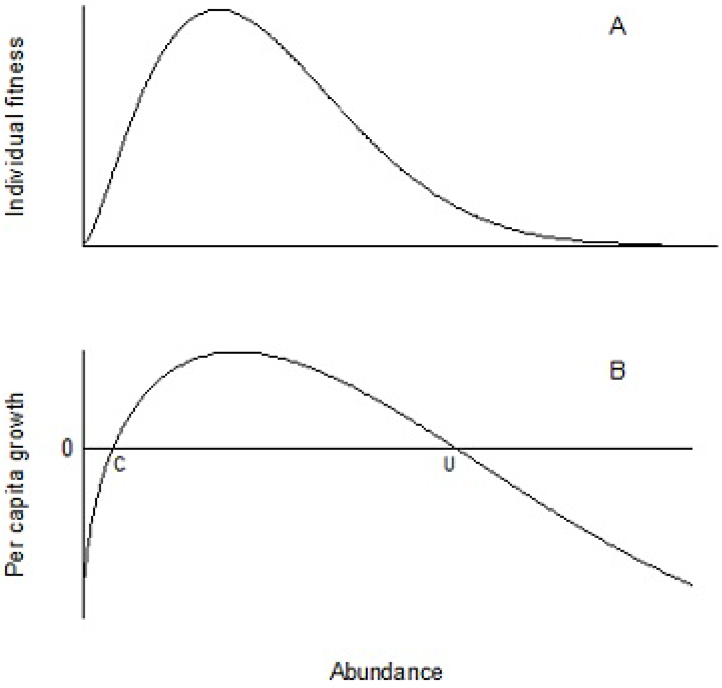

Figure 1. The Allee effect (replotted from Stephens *et al* 1999). A. At low densities, the benefits from the addition of each successive individual outweigh the costs, such that there is a net gain in individual fitness, and fitness is highest at intermediate numbers or densities. B. In this case, population 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.

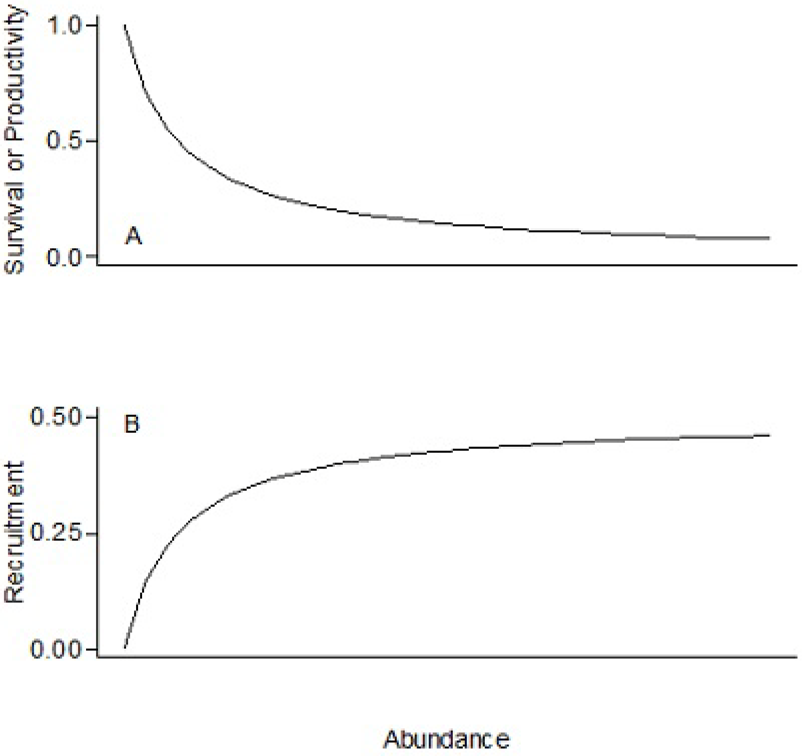

Figure 2. The compensatory (Beverton-Holt function) effect. A. At high population densities, increasing competition and depletion of resources will often result in decreased survival or productivity rates. B. Recruitment increases as the survival or productivity rates decrease resulting in a population 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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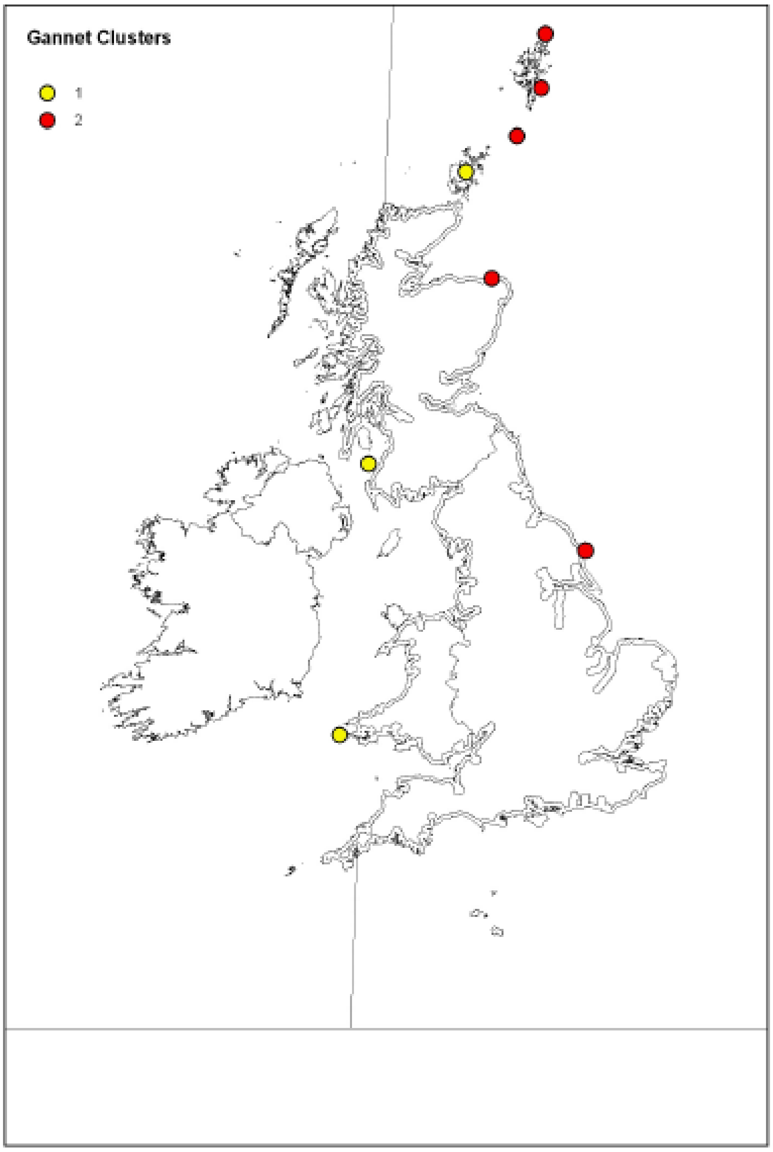
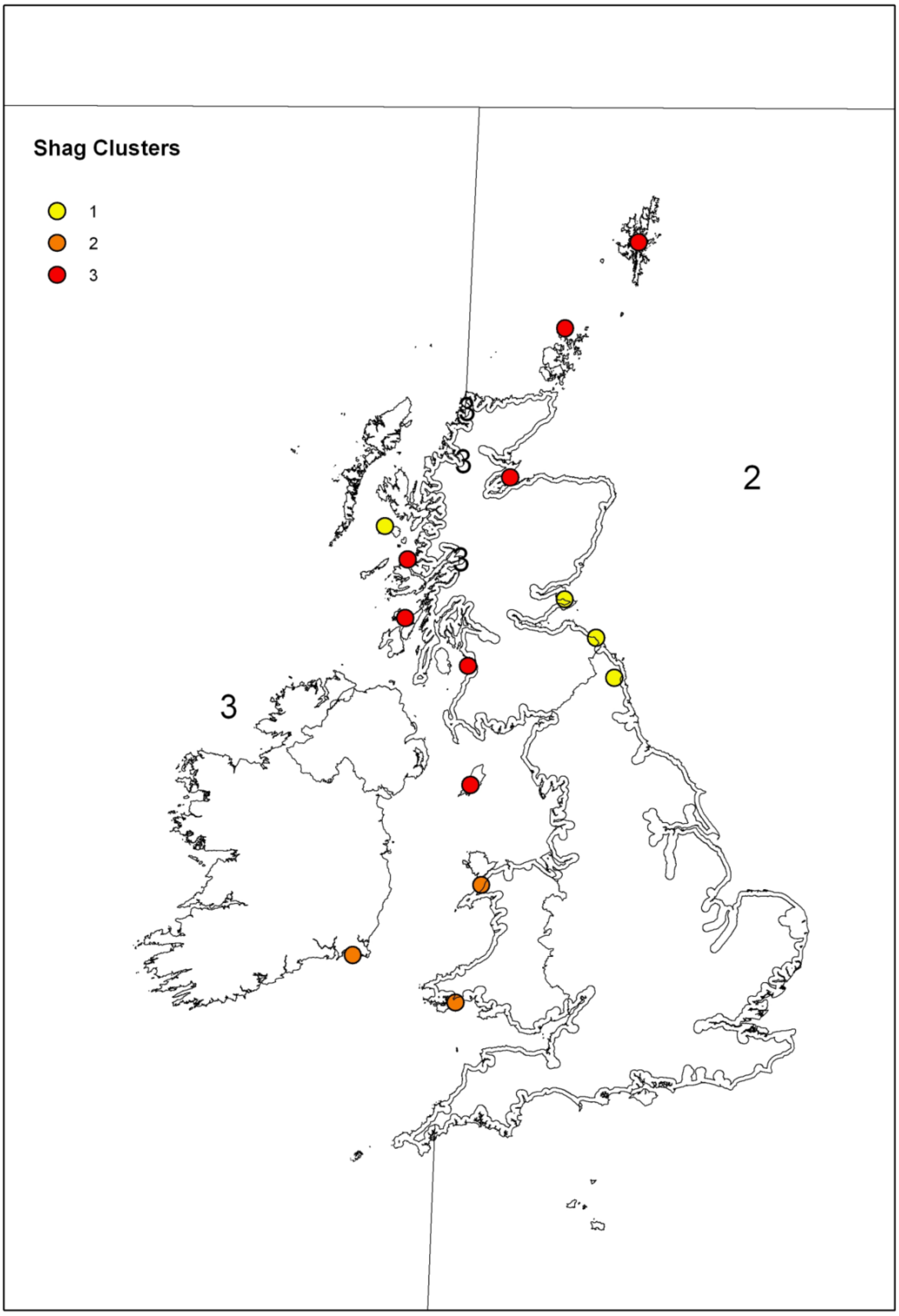
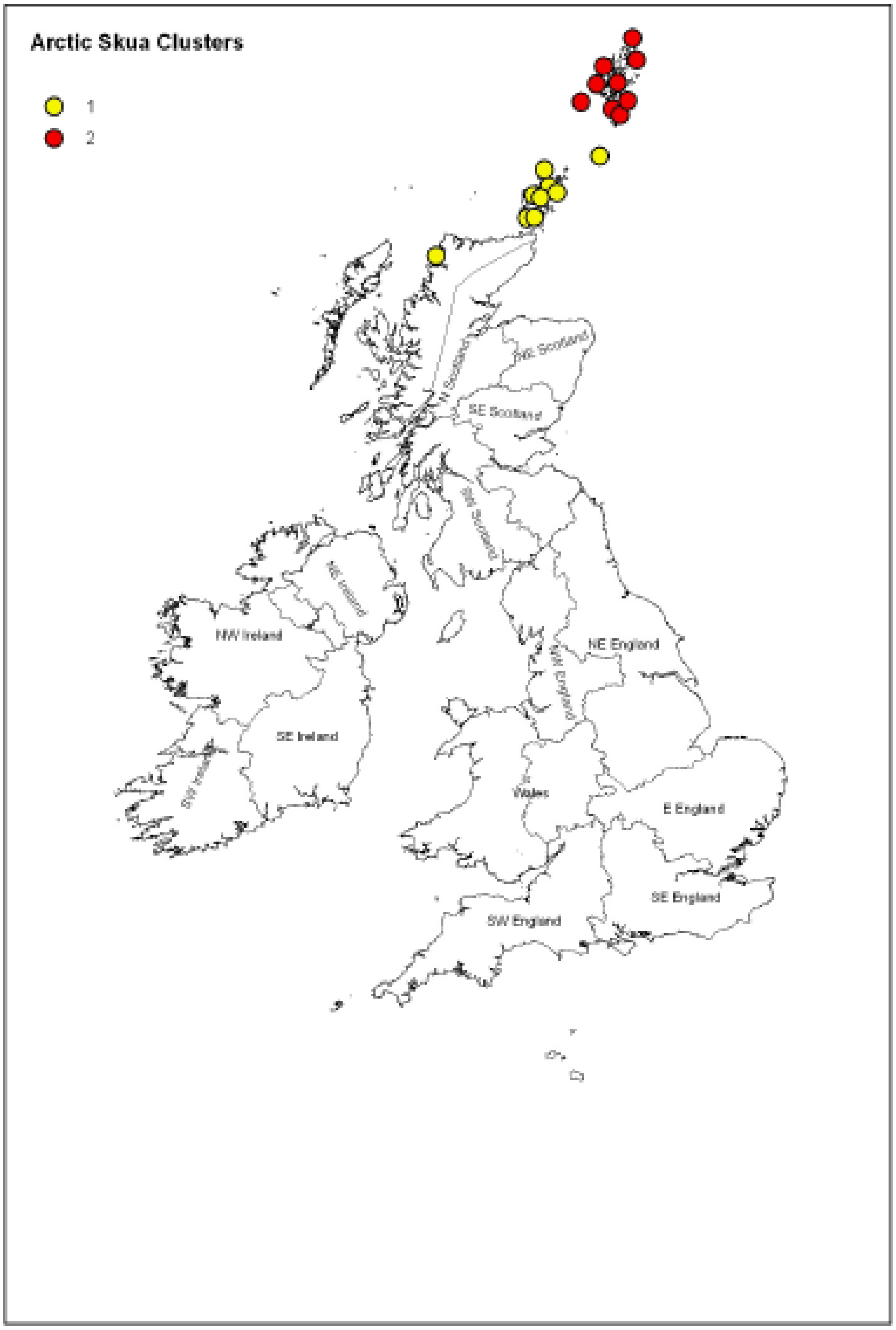
Appendix



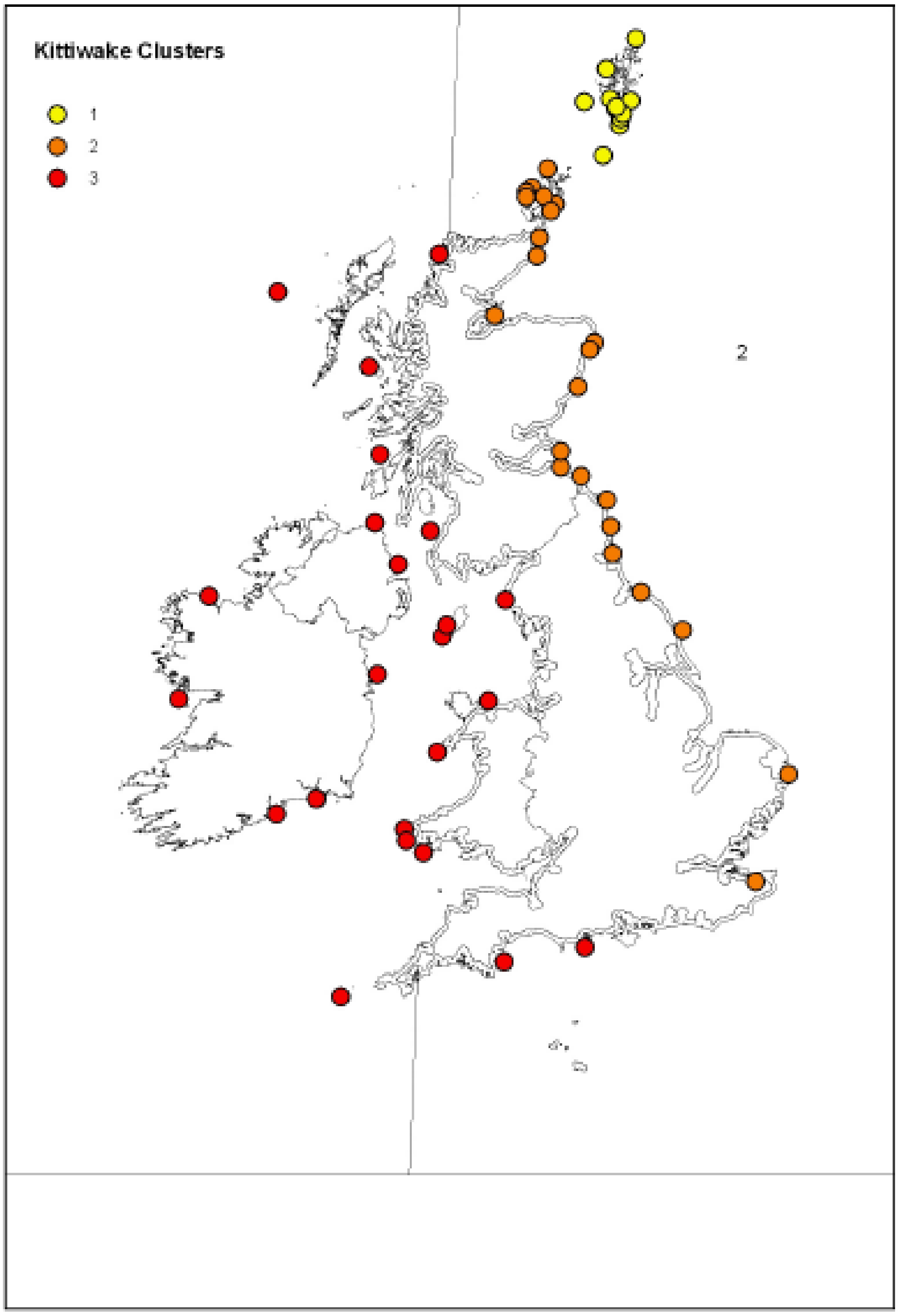
S1. Colony membership of clusters based on analysis of northern gannet breeding success data, overlaid with existing OSPAR monitoring regions (taken from Cook & Robinson 2010). The West Coast cluster is broadly contiguous with the Celtic Sea OSPAR region, with the addition of Orkney, and the East Coast 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.



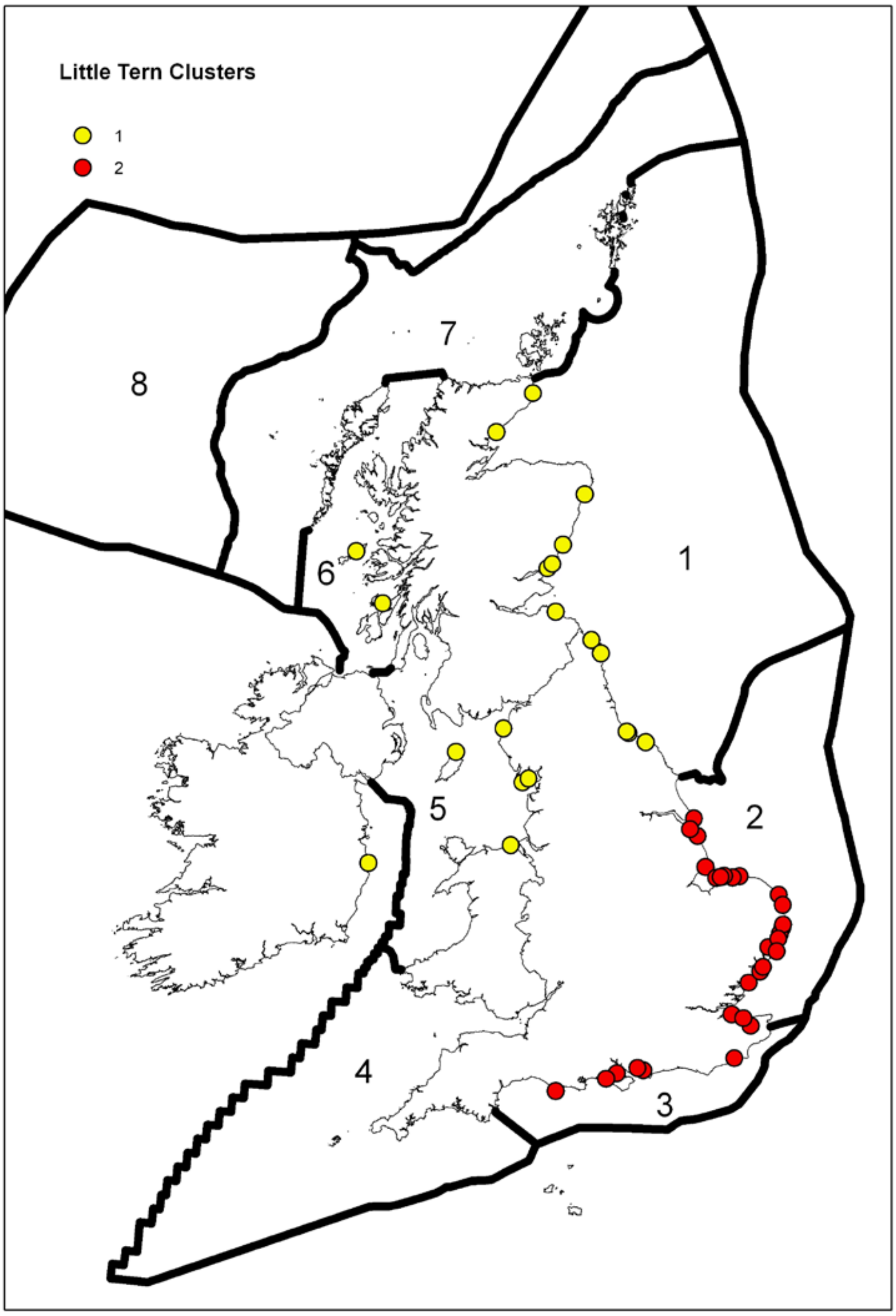
S2. Colony membership of clusters based on analysis of European shag breeding success data, overlaid with existing OSPAR monitoring regions (taken from Cook & Robinson 2010). The East Coast of Scotland and Orkney and the Shetland clusters are within the Greater North Sea OSPAR Region and the West Coast of England and Wales cluster is within the Celtic Sea (Cook & Robinson 2010).



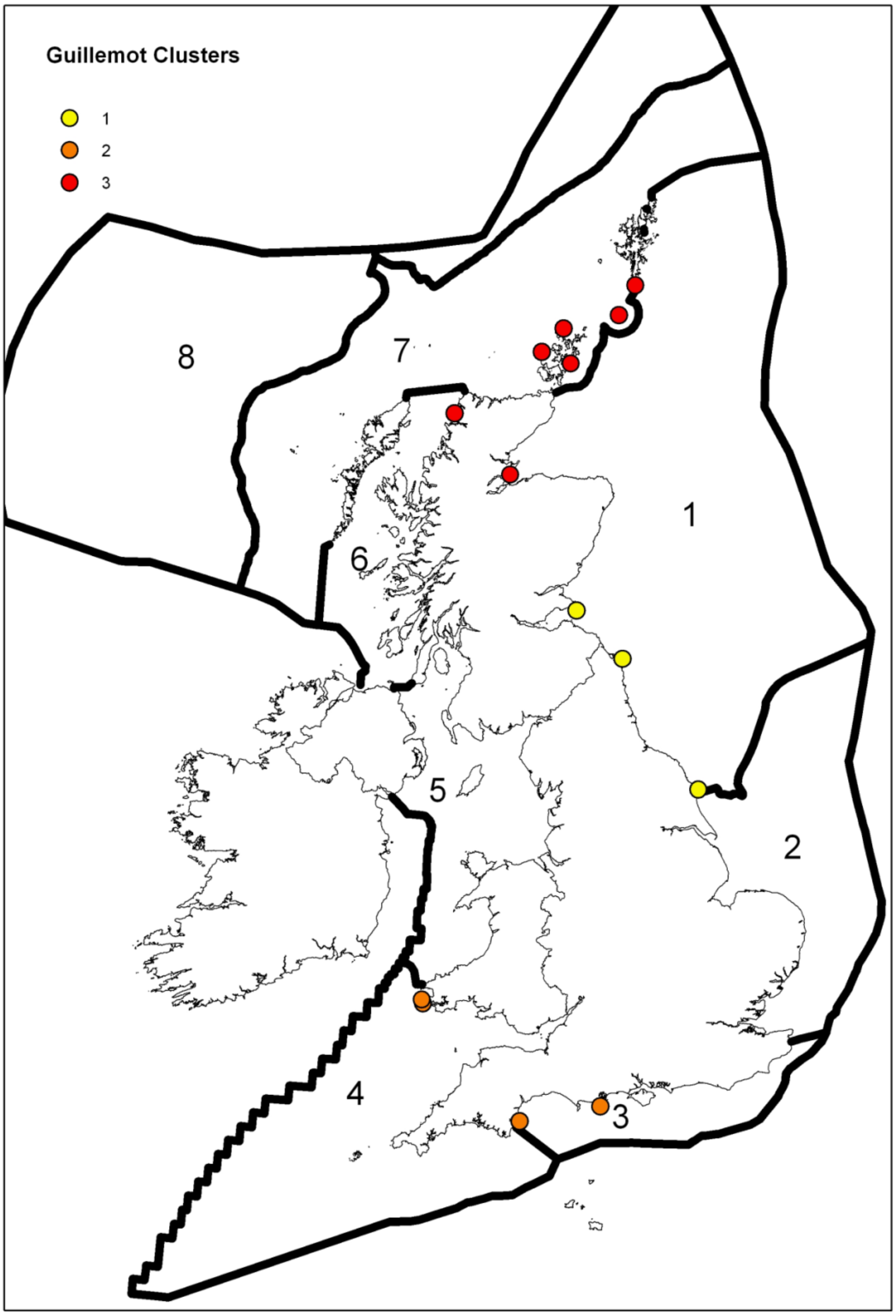
S3. Colony membership of clusters based on analysis Arctic skua breeding success data, overlaid with existing Regional Seas monitoring regions (taken from Cook & Robinson 2010). The Shetland Cluster is within the Greater North Sea OSPAR region and Regional Seas 7. It is contiguous with the Shetland SMP 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.



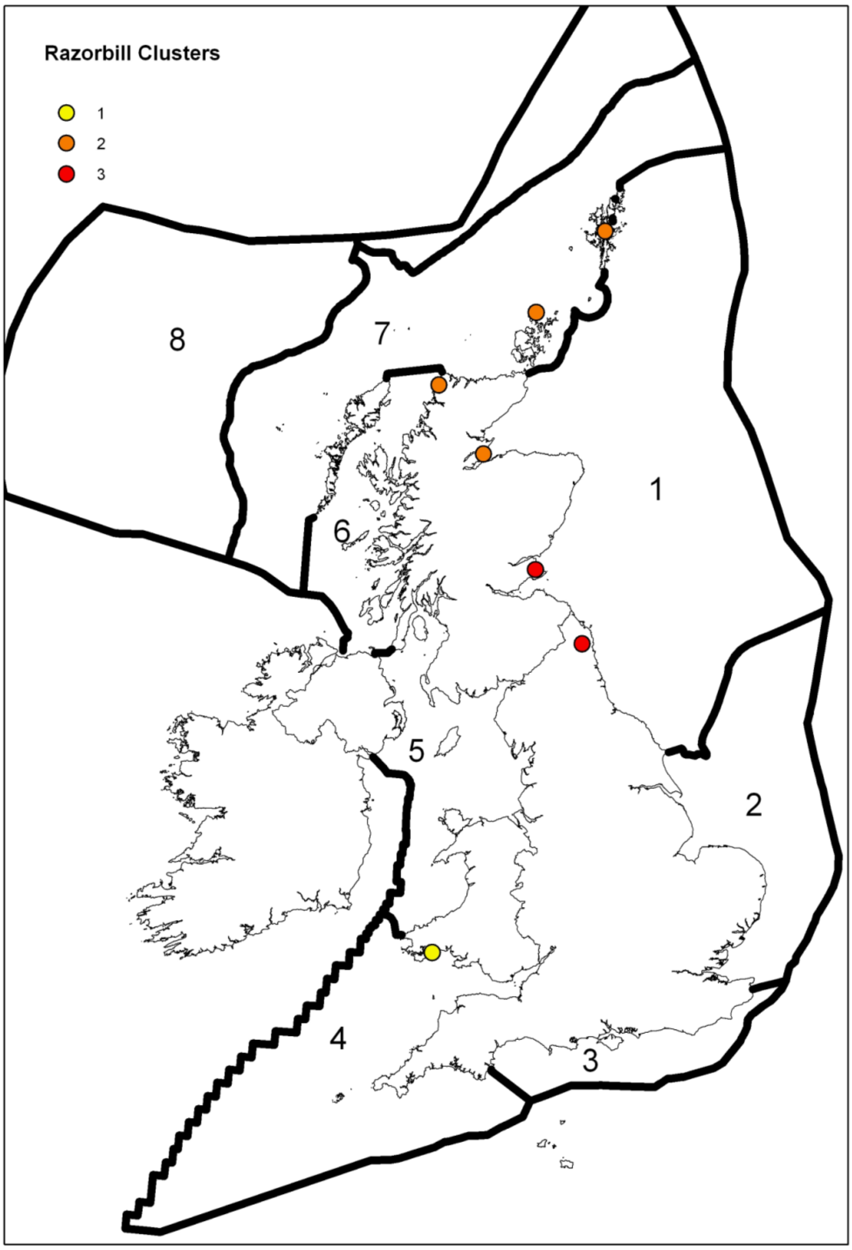
S4. Colony membership of clusters based on analysis of black-legged kittiwake breeding success data, overlaid with existing OSPAR monitoring regions (taken from Cook & Robinson 2010). The Shetland population falls within regional sea 7 and the Shetland SMP region. The Eastern population encompasses regional 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.

Testing and Validating Metrics of Change Produced by Population Viability Analysis (PVA)

Scottish Marine and Freshwater Science Vol 8 No 23

M Jitlal, S Burthe, S Freeman and F Daunt

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Marine Scotland is the directorate of the Scottish Government responsible for the integrated management of Scotland's seas. Marine Scotland Science (formerly Fisheries Research Services) provides expert scientific and technical advice on marine and fisheries issues. Scottish Marine and Freshwater Science is a series of reports that publishes results of research and monitoring carried out by Marine Scotland Science. It also publishes the results of marine and freshwater scientific work that has been carried out for Marine Scotland under external commission. These reports are not subjected to formal external peer-review.

This report presents the results of marine and freshwater scientific work carried out for Marine Scotland under external commission.

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)
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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 $\text{CPD}_{25,10}$ 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.

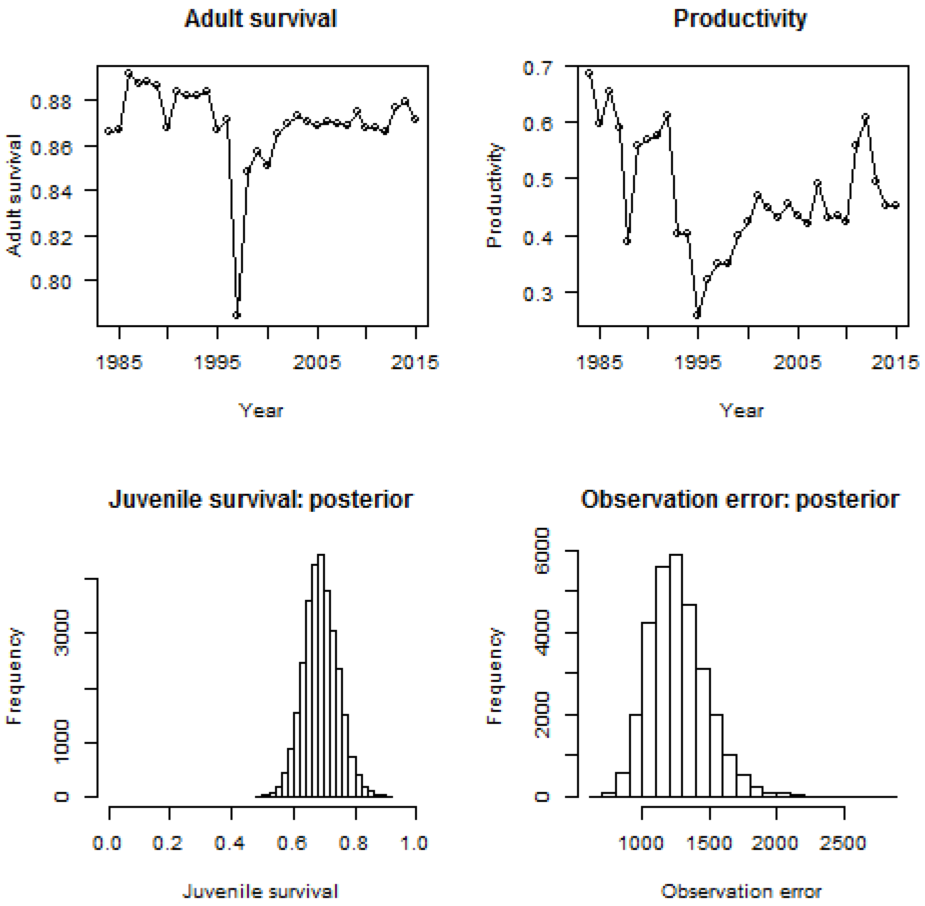


Figure 2a: Estimated total abundance from 1984 to 2016, with an additional 25 years of projections with various declines in productivity and a final five years of projections with no decline in productivity.

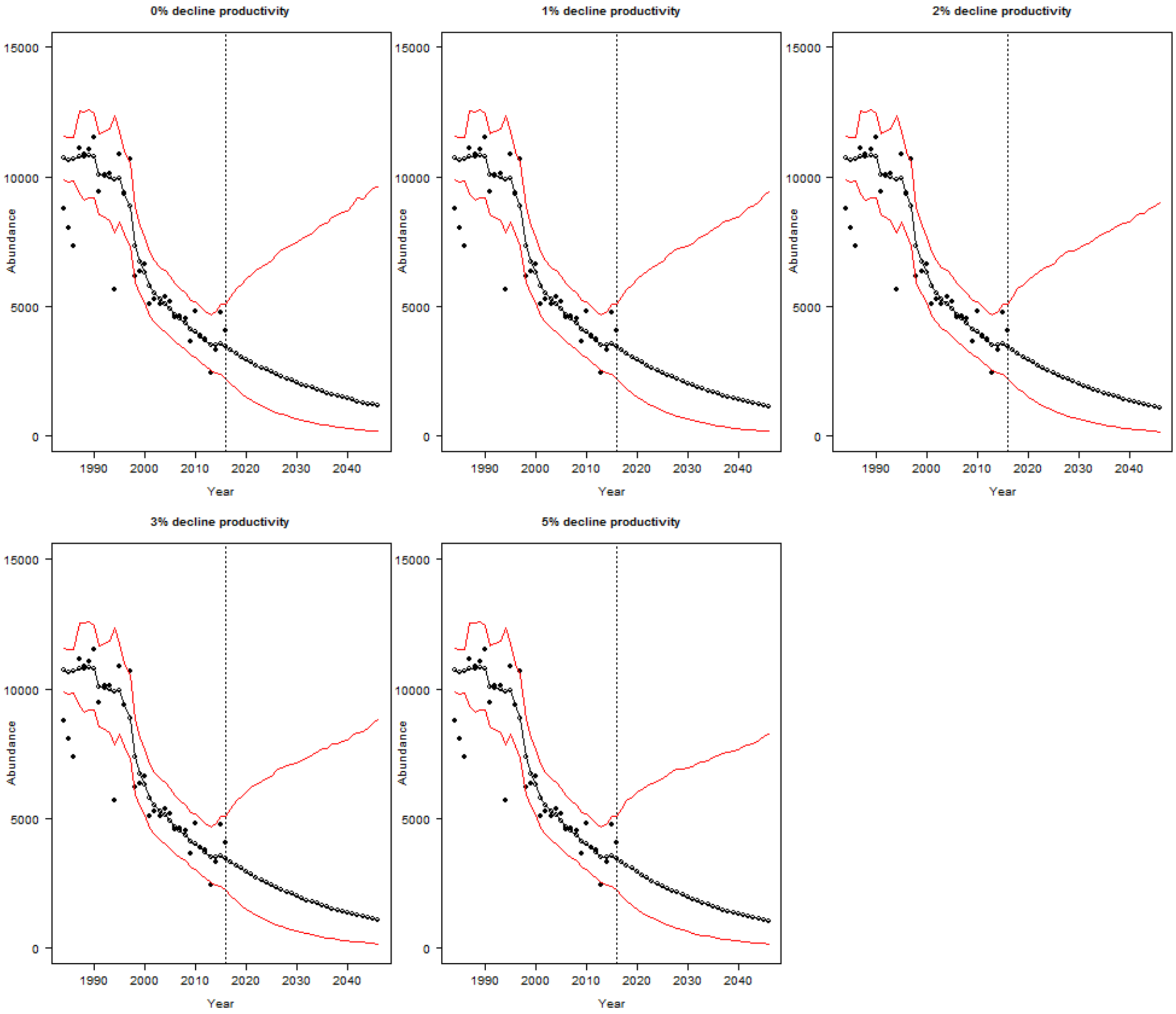


Figure 2b: Estimated total abundance from 1984 to 2016, with an additional 25 years of projections with various declines in adult survival and a final five years of projections with no decline in adult survival.

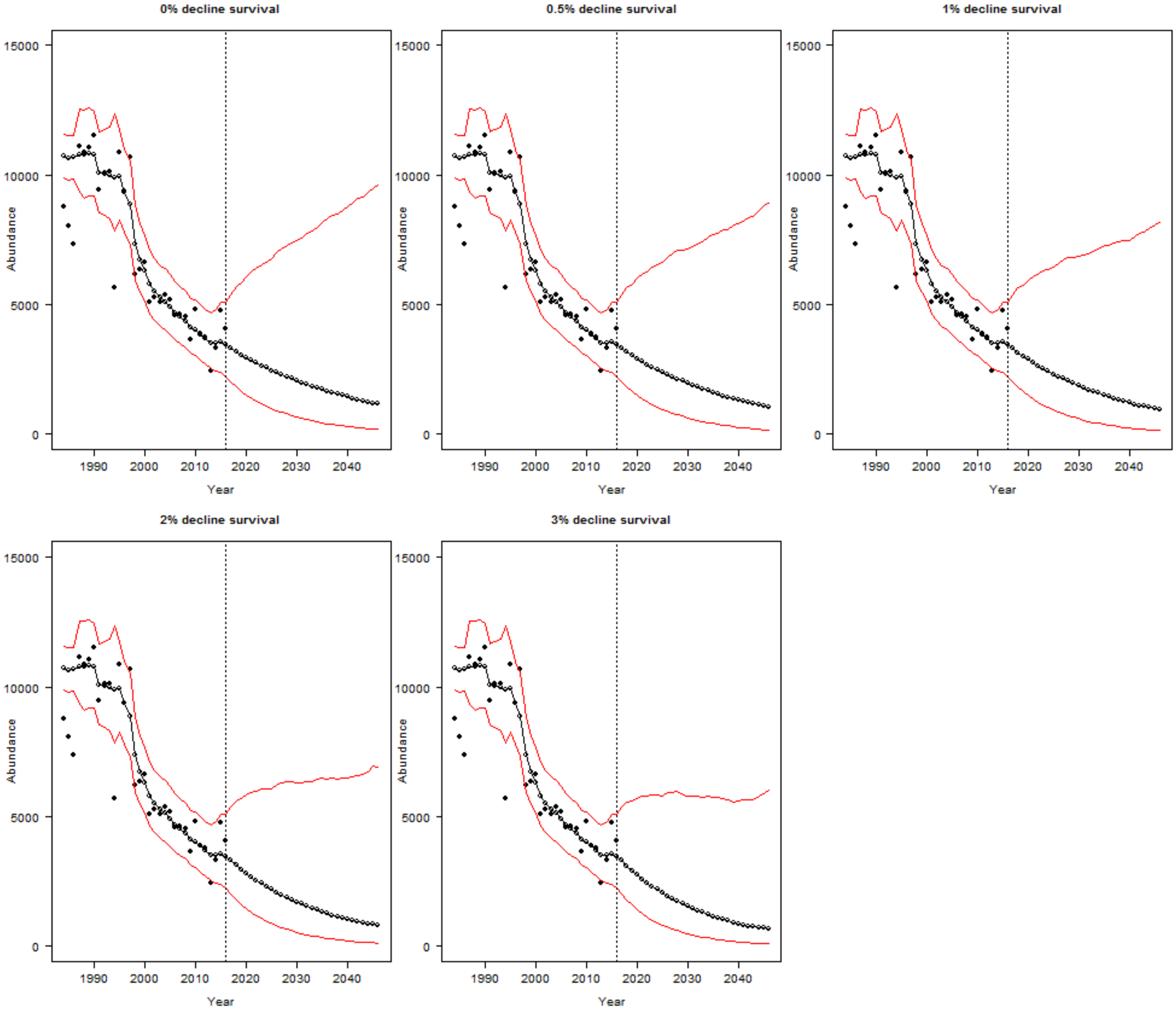


Figure 2c: Estimated total abundance from 1984 to 2016, with an additional 25 years of projections with various declines in both productivity and adult survival and a final five years of projections with no decline in either productivity or adult survival.

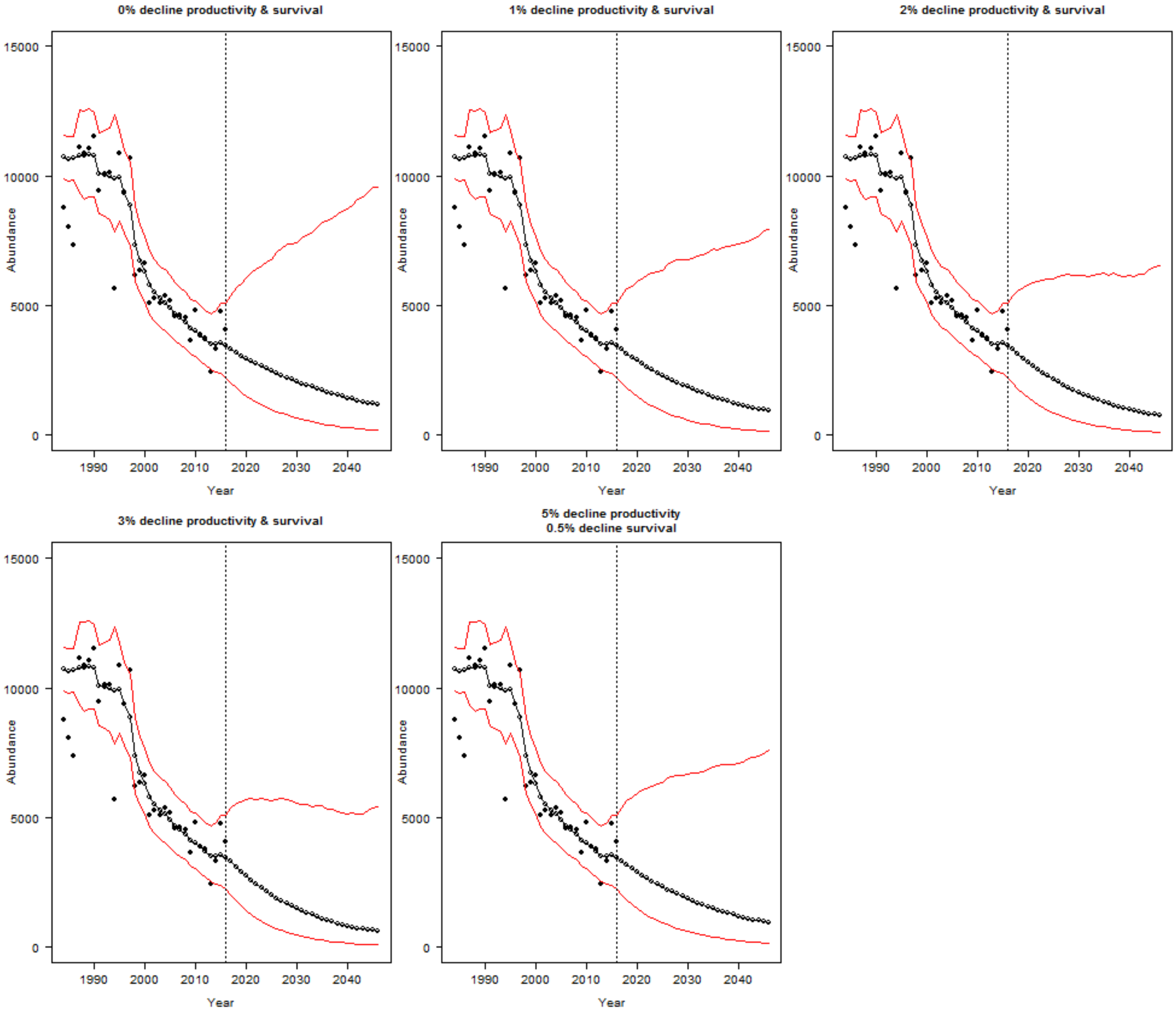


Figure 3: Median impacted population size after 25 years of projections under various scenarios of mis-specification in productivity and adult mortality. Adult mortality mis-specification is illustrated in the upper panels and productivity mis-specification in the lower panels. Mis-specification was 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).

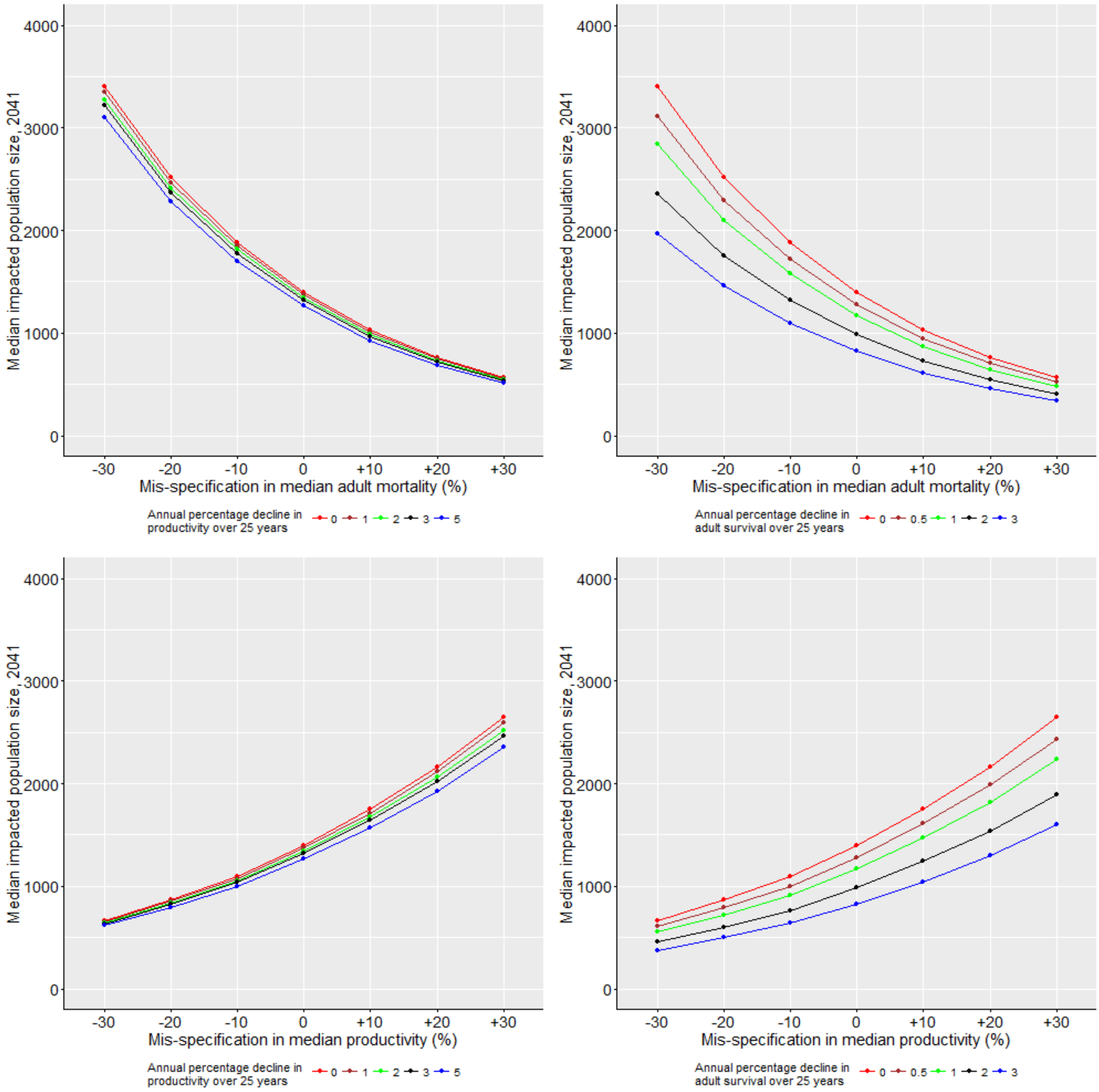


Figure 4a: PVA Metric A – ratio of population growth rate from 2016-2041, comparing impacted population vs. un-impacted population. Adult mortality mis-specification is illustrated in the upper panels and productivity mis-specification in the lower panels. Mis-specification was varied from -30% to +30% (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).

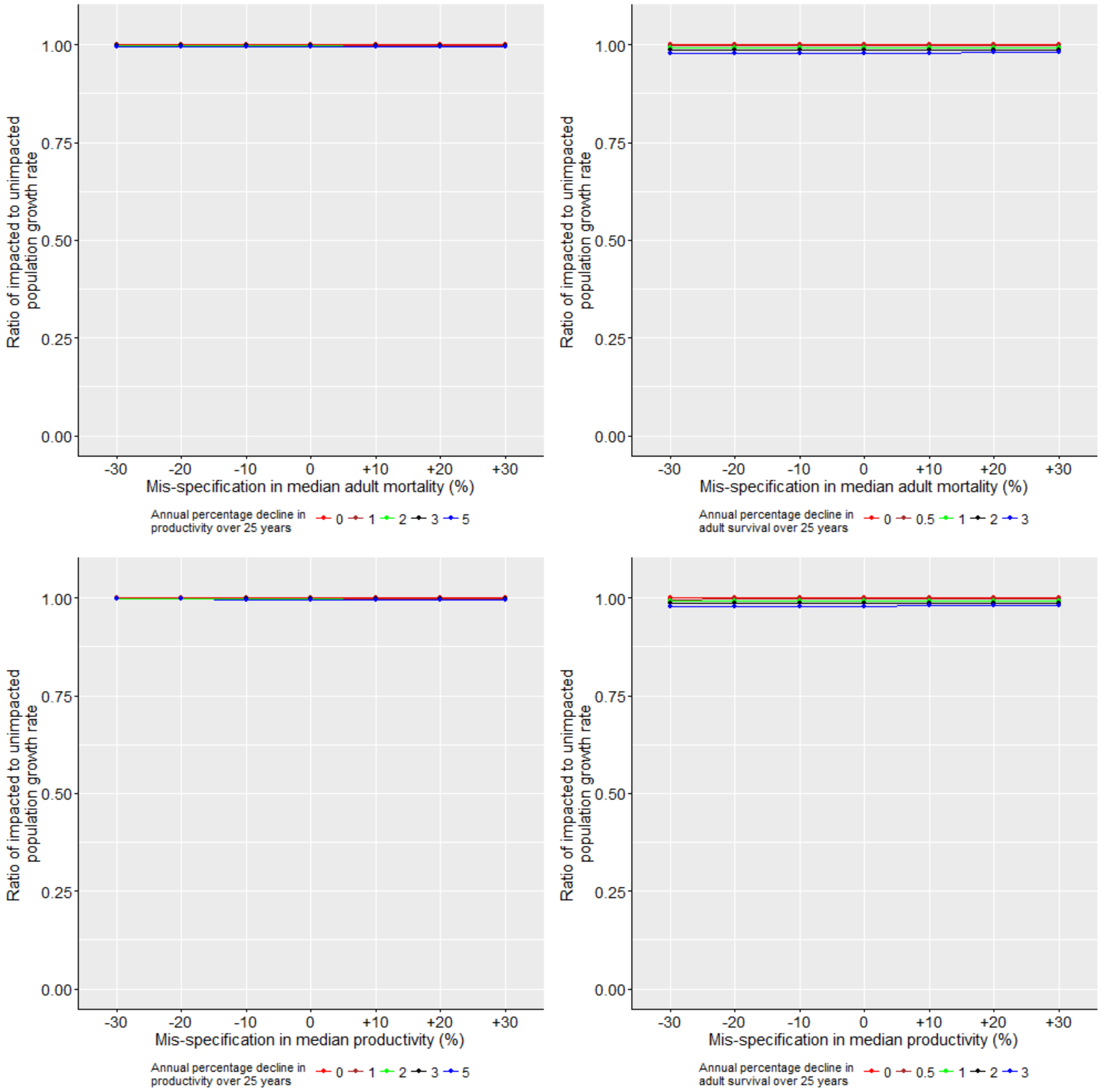


Figure 4b: PVA Metric B – ratio of population size at 2041, comparing impacted population vs. un-impacted population. Adult mortality mis-specification is illustrated in the upper panels and productivity mis-specification in the lower panels. Mis-specification was varied from -30% to +30% (with 0% representing 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).

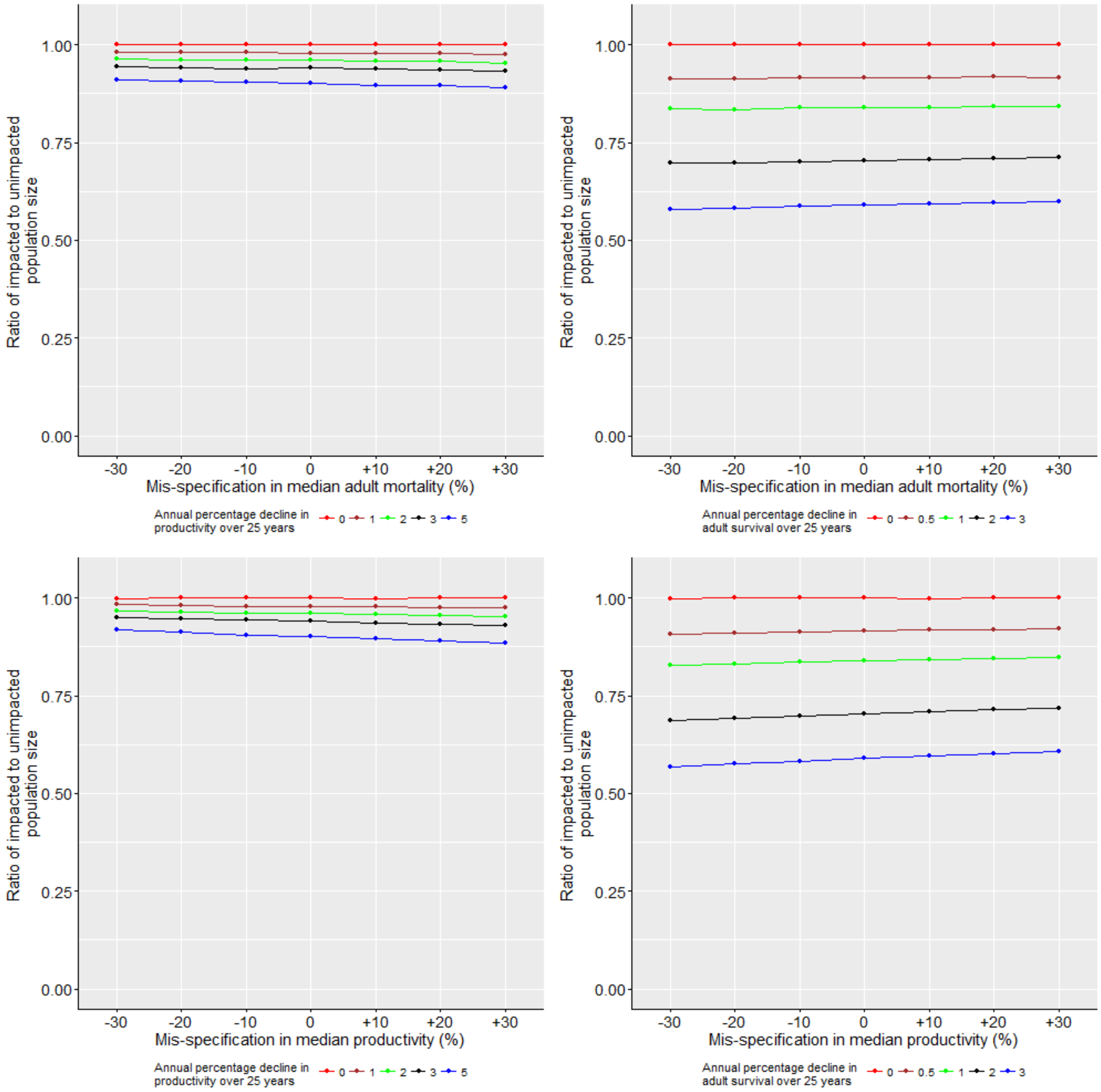


Figure 4c: PVA Metric C – difference in population growth rate from 2016-2041, comparing impacted population vs. un-impacted population. Adult mortality mis-specification is illustrated in the upper panels and productivity mis-specification in the lower panels. Mis-specification was varied from -30% 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).

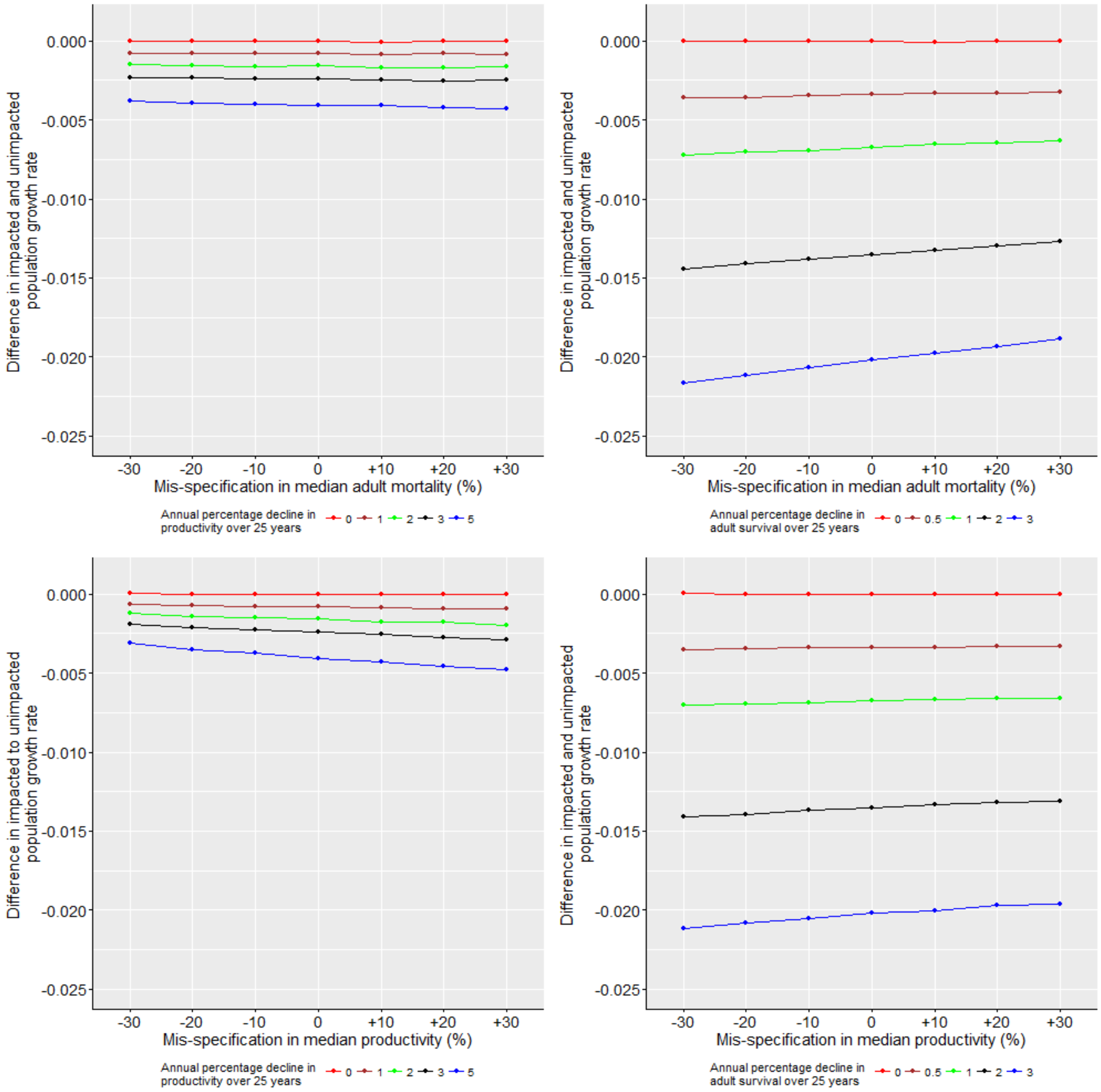


Figure 4d: PVA Metric D – difference in population size at 2041, comparing impacted population vs. un-impacted population. Adult mortality mis-specification is illustrated in the upper panels and productivity mis-specification in the lower panels. Mis-specification was varied from -30% to +30% (with 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).

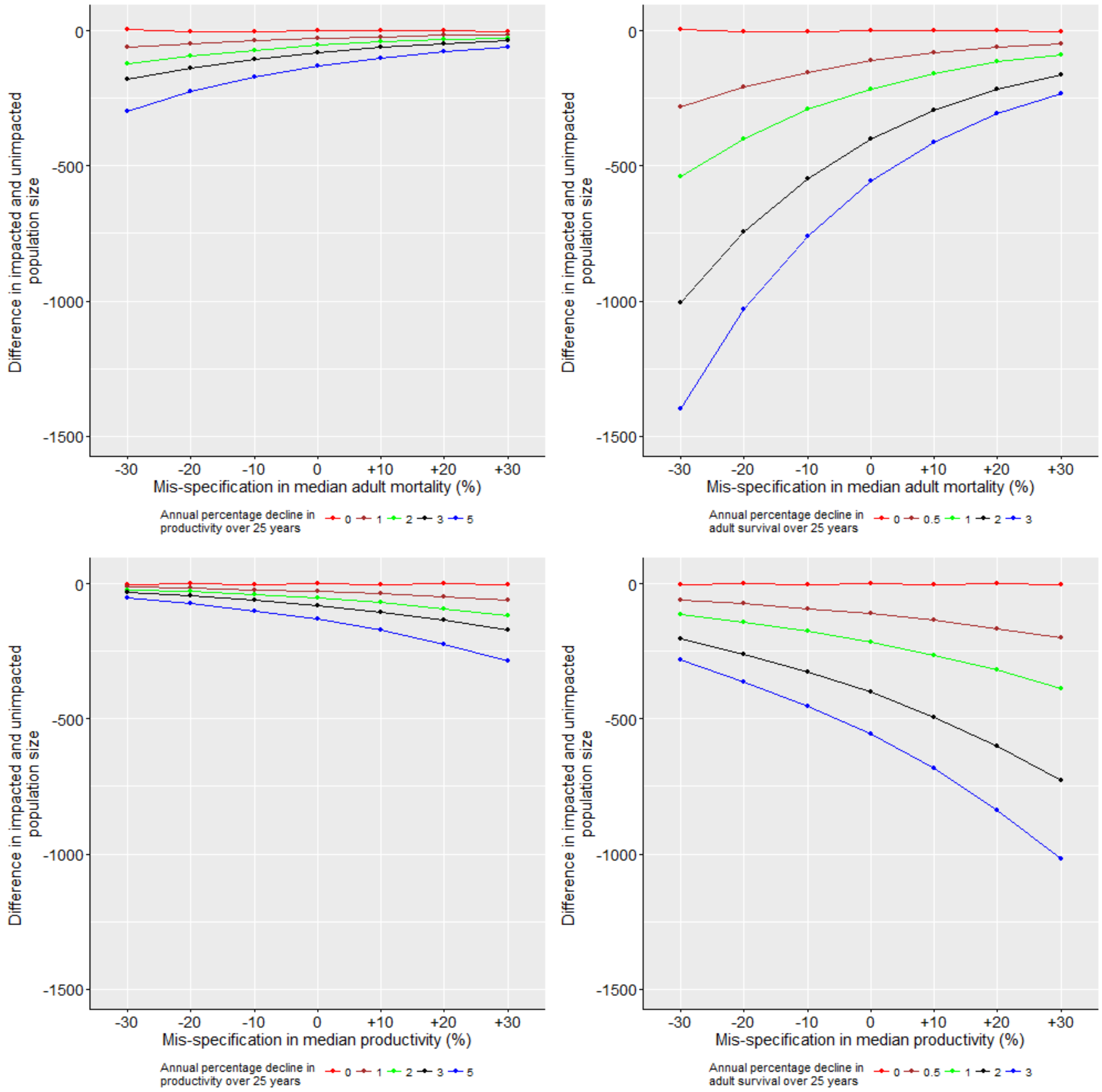


Figure 4e: PVA Metric E1 – probability of population decline greater than 10% from 2016-2041. Adult mortality mis-specification is illustrated in the upper panels and productivity mis-specification in the lower panels. Mis-specification was varied from -30% to +30% (with 0% representing no mis-specification). 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).

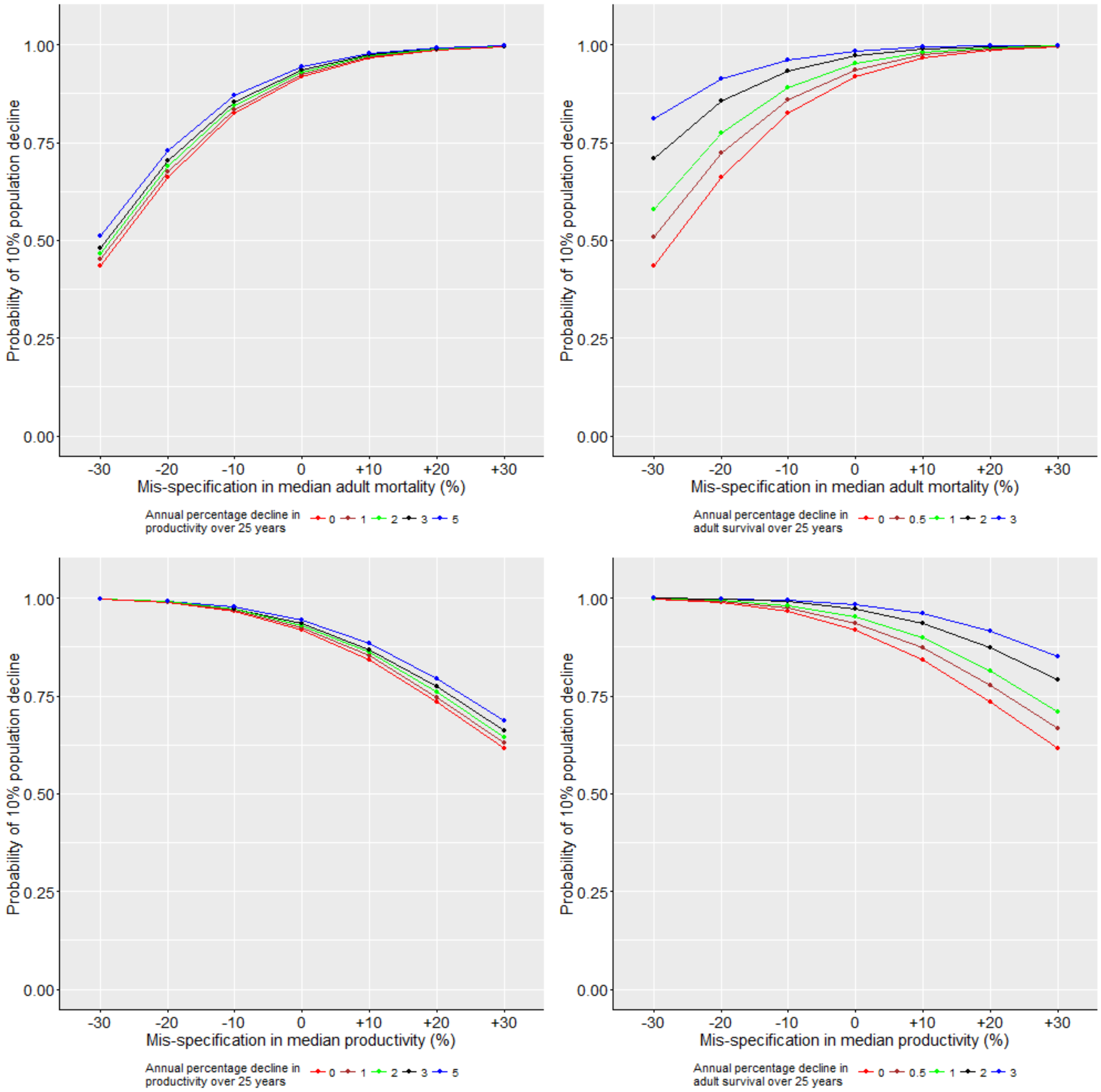


Figure 4f: PVA Metric E2 – probability of population decline greater than 25% from 2016-2041. Adult mortality mis-specification is illustrated in the upper panels and productivity mis-specification in the lower panels. Mis-specification was varied from -30% to +30% (with 0% representing no mis-specification). 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).

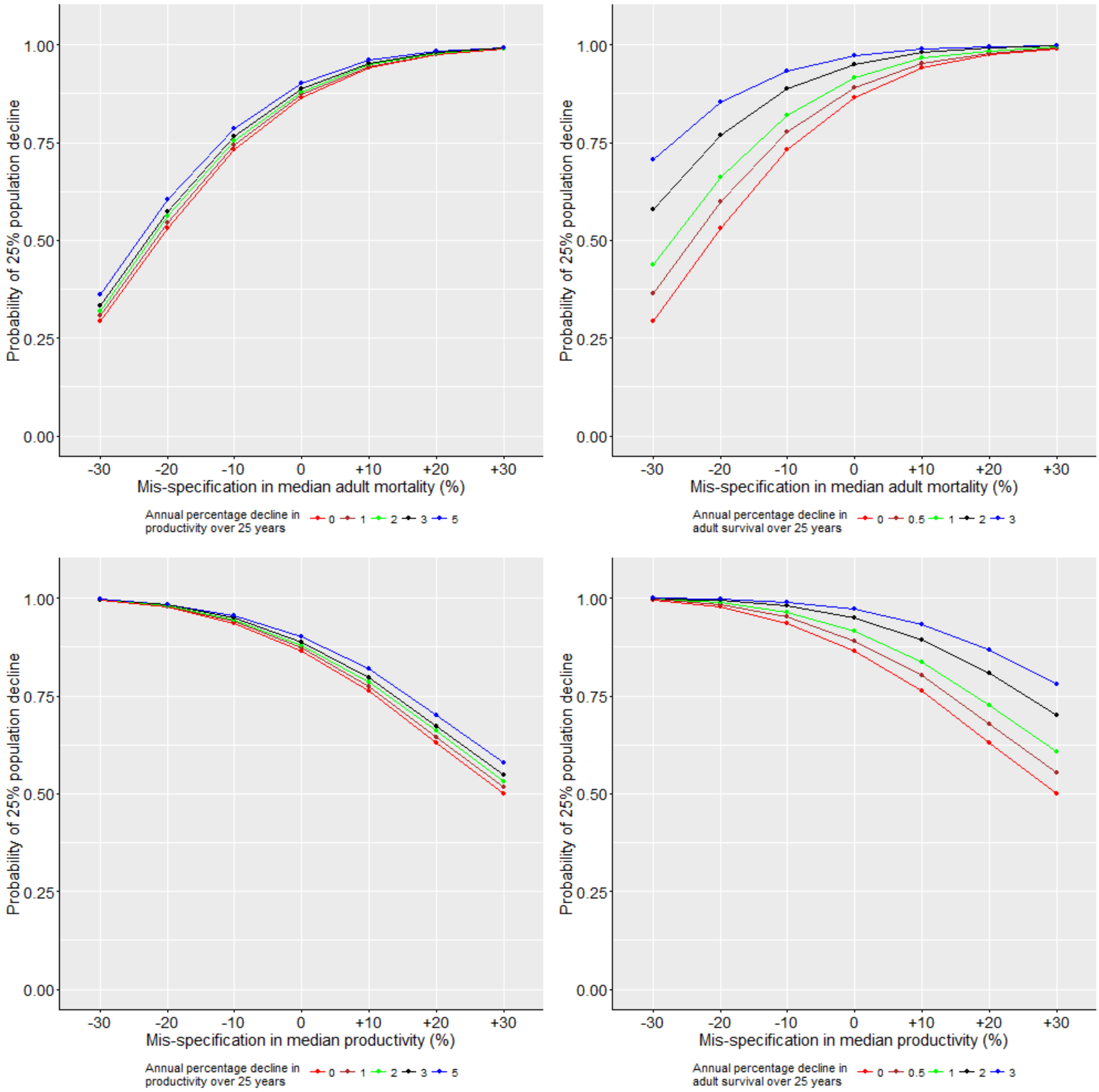


Figure 4g: PVA Metric E3 – probability of population decline greater than 50% from 2016-2041. Adult mortality mis-specification is illustrated in the upper panels and productivity mis-specification in the lower panels. Mis-specification was varied from -30% to +30% (with 0% representing no mis-specification). 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).

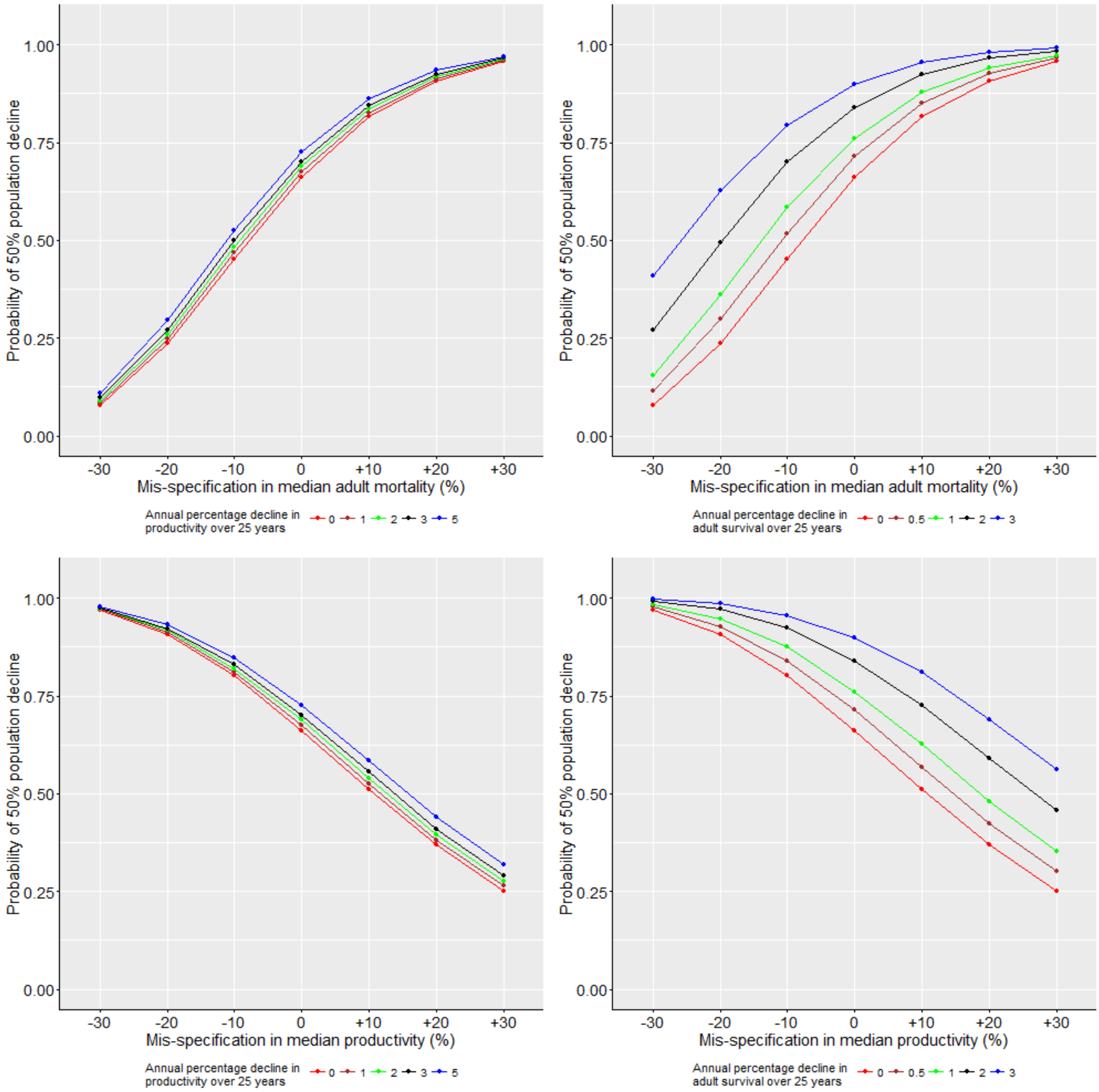
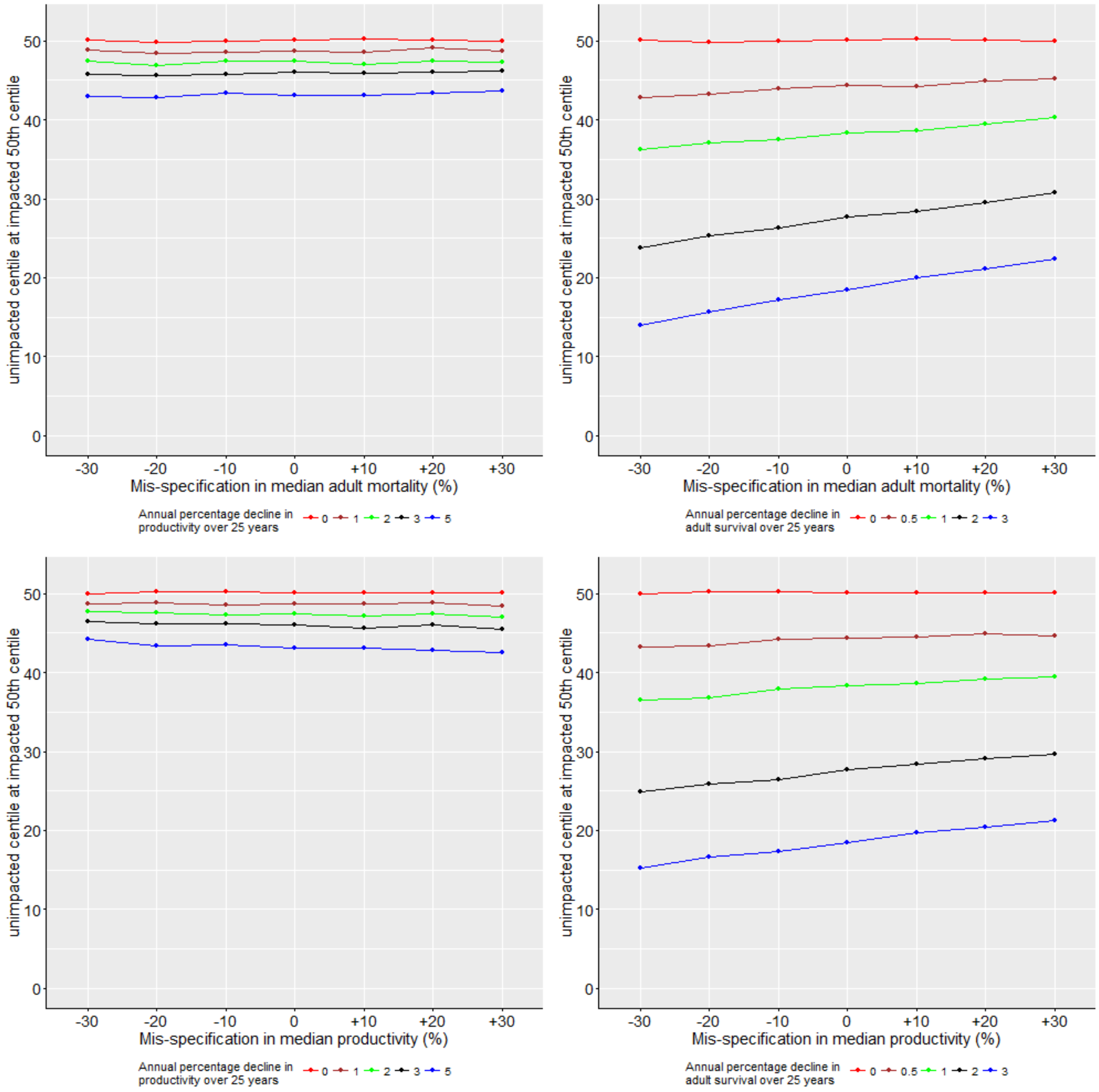


Figure 4h: PVA Metric F – centile from un-impacted population size equal to the 50th centile of the impacted population size, at 2041. Adult mortality mis-specification is illustrated in the upper panels and productivity mis-specification in the lower panels. Mis-specification was varied from -30% to +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.

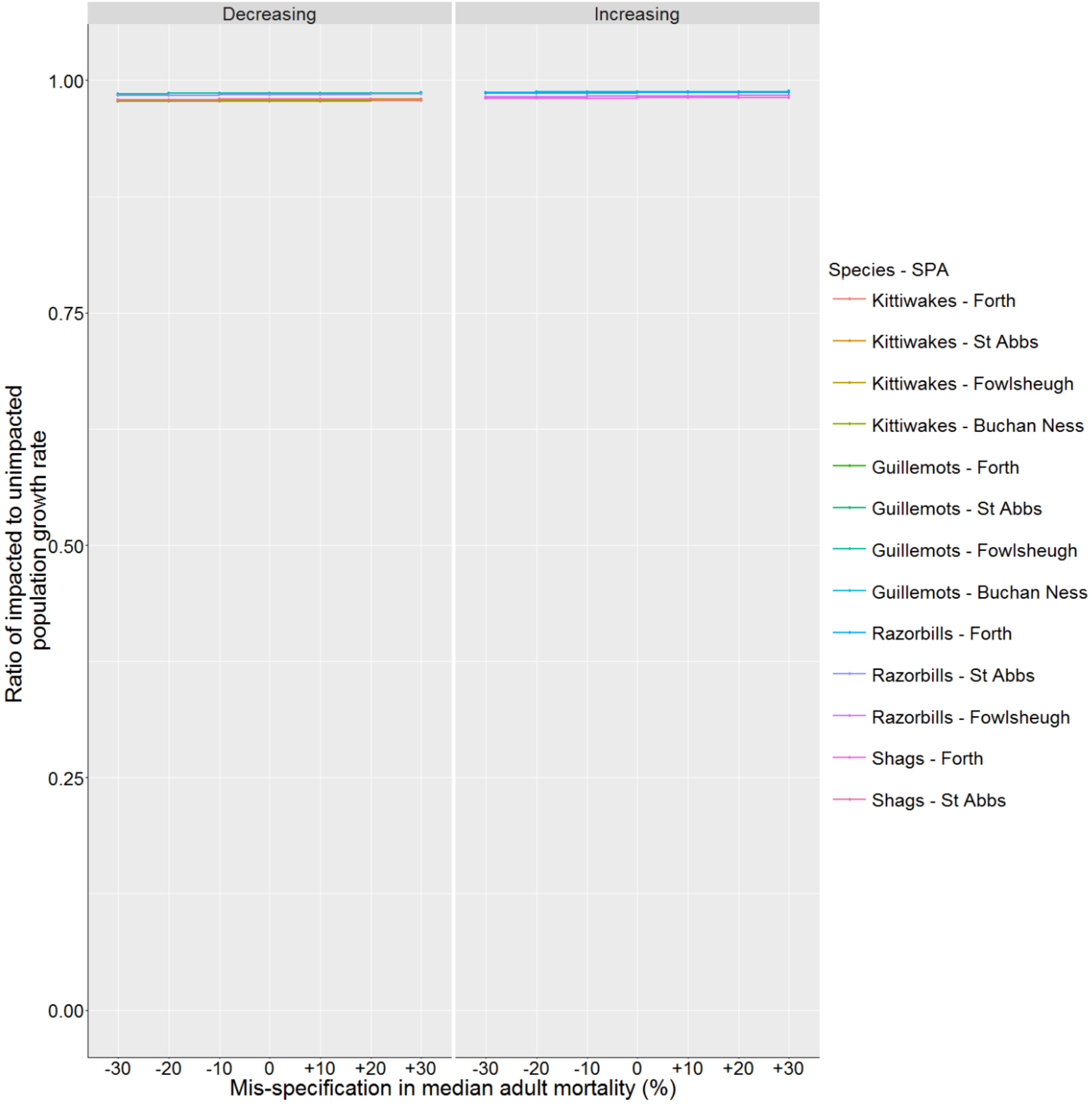


Figure 5b: PVA Metric B – ratio of population size at 2041, comparing impacted population vs. un-impacted population, for changing adult mortality and a 3% decrease in adult survival, across decreasing populations and increasing populations.

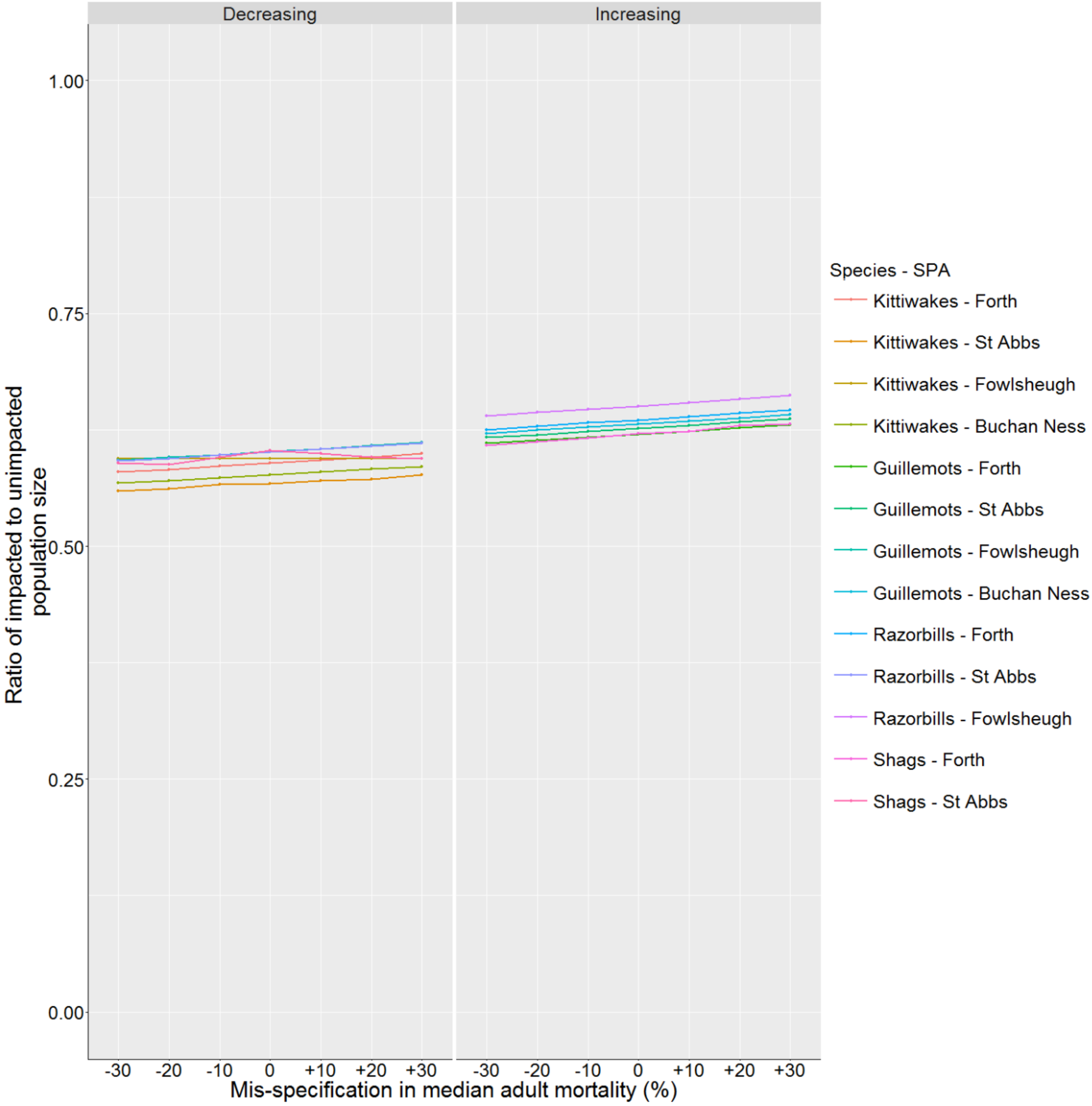


Figure 5c: PVA Metric C – difference in population growth rate from 2016-2041, comparing impacted population vs. un-impacted population, for changing adult mortality and a 3% decrease in adult survival, across decreasing populations and increasing populations.

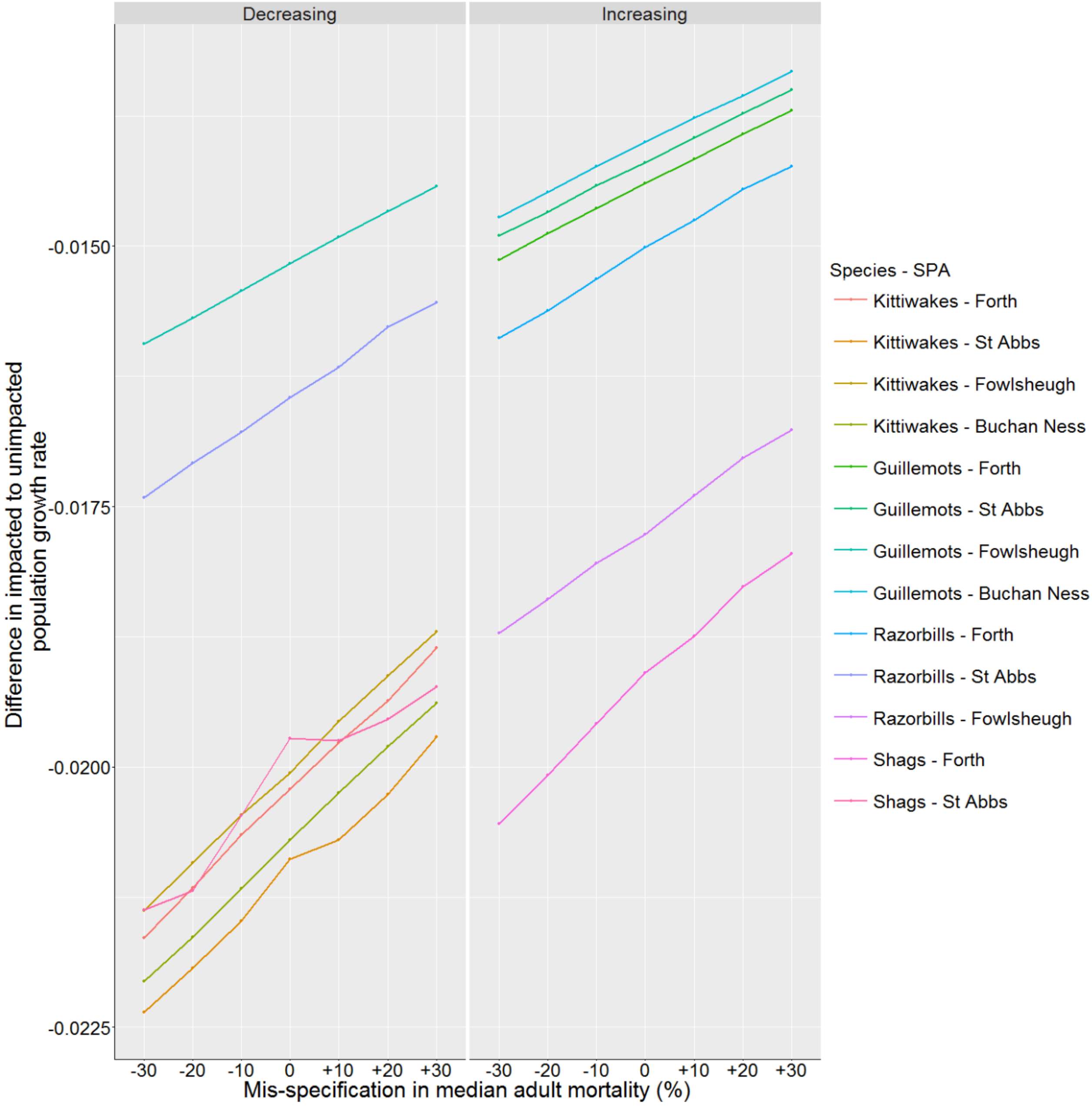


Figure 5d: PVA Metric D – difference in population size at 2041, comparing impacted population vs. un-impacted population, for changing adult mortality and a 3% decrease in adult survival, across decreasing populations and increasing populations.

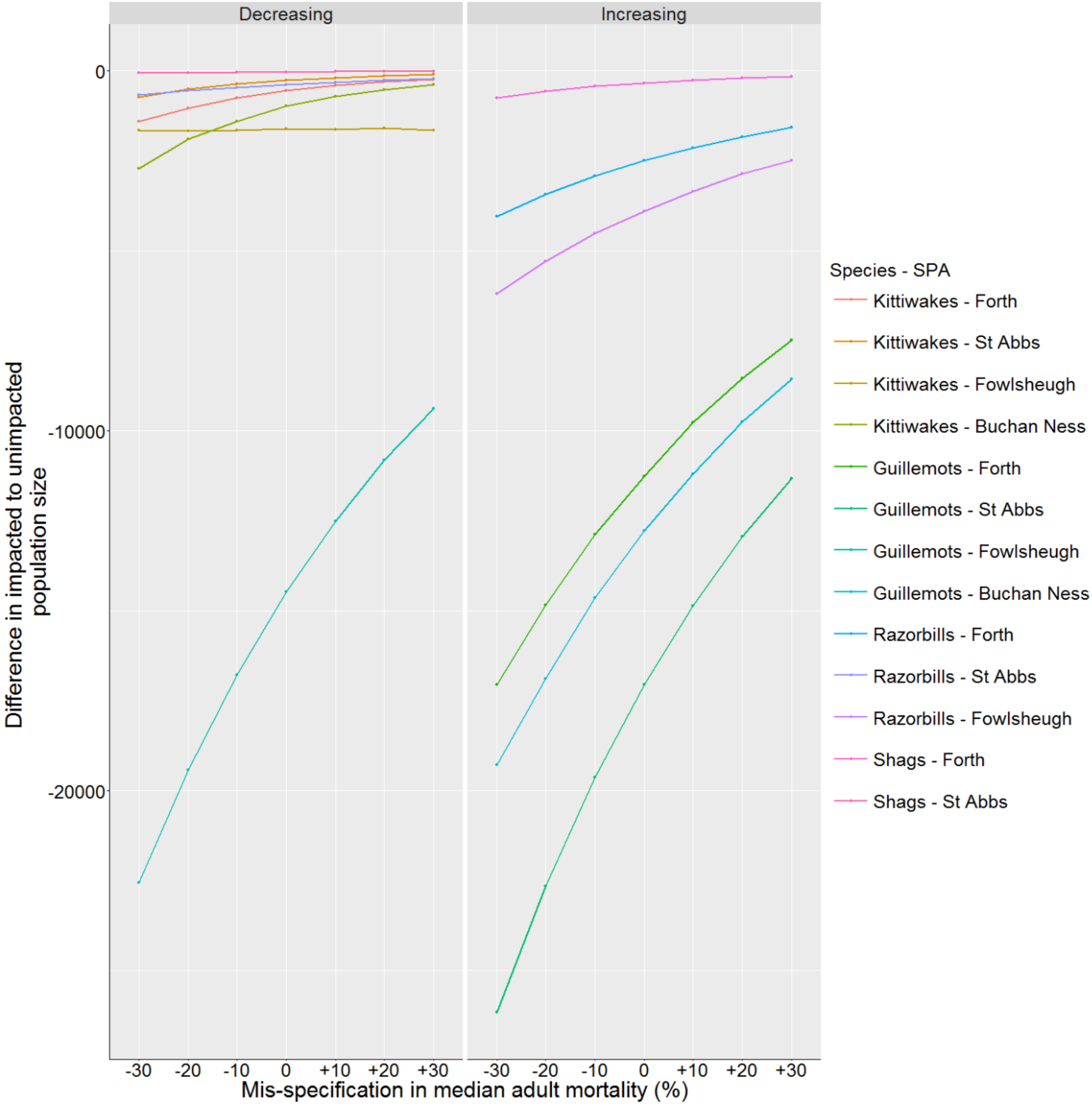


Figure 5e: PVA Metric E1 – probability of population decline greater than 10% from 2016-2041, for changing adult mortality and a 3% decrease in adult survival, across decreasing populations and increasing populations.

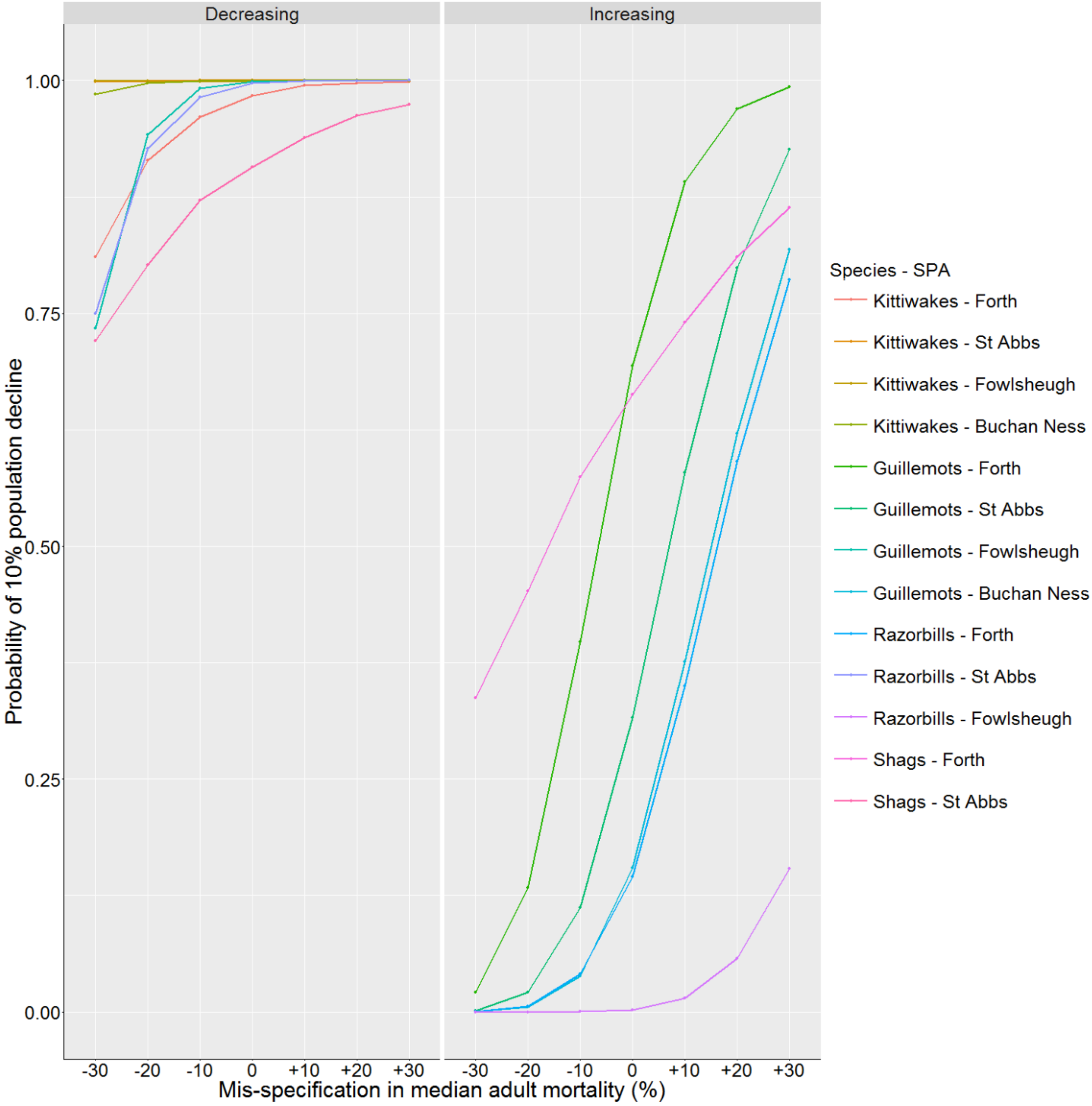


Figure 5f: PVA Metric E2 – probability of population decline greater than 25% from 2016-2041, for changing adult mortality and a 3% decrease in adult survival, across decreasing populations and increasing populations.

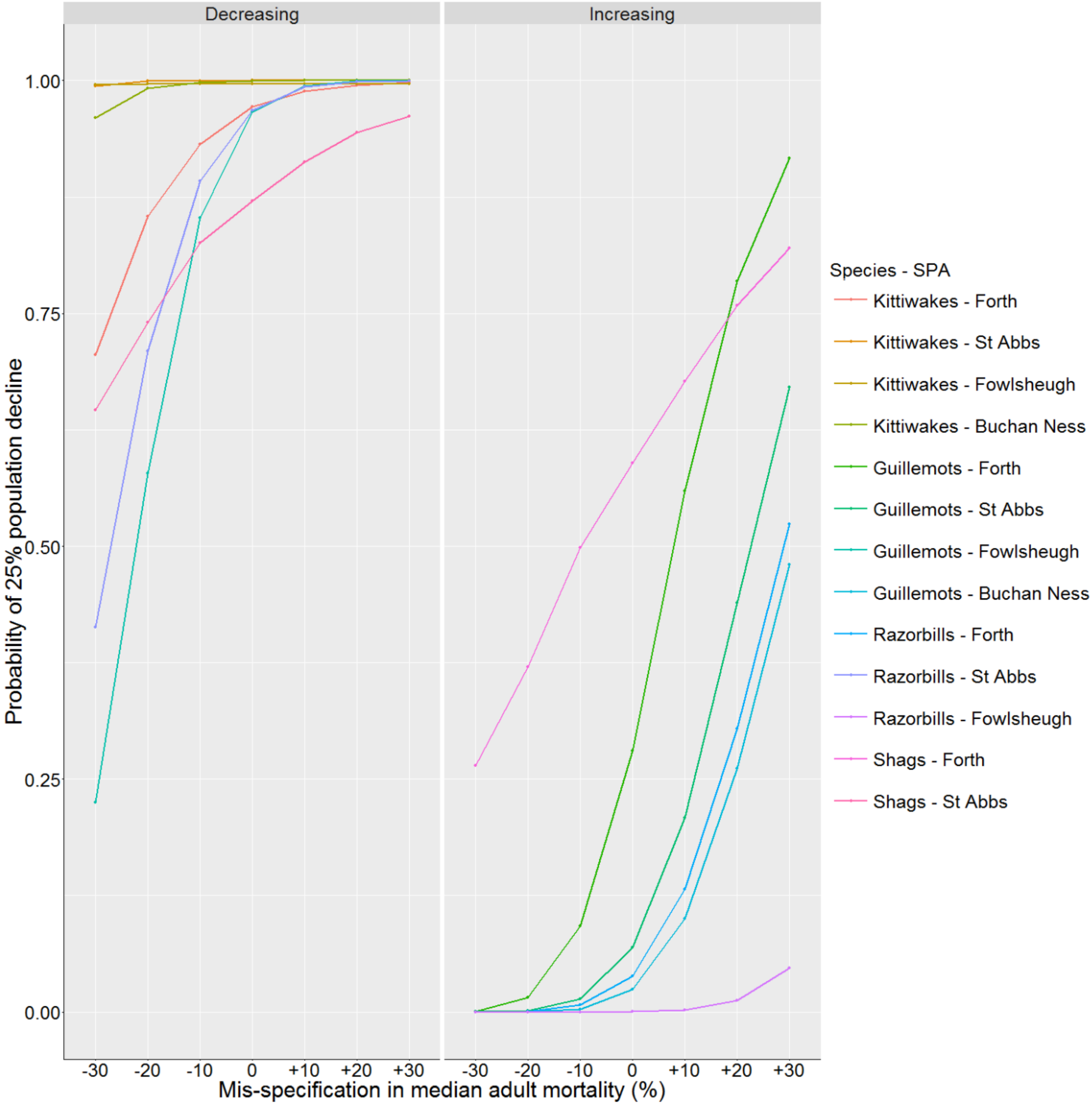


Figure 5g: PVA Metric E3 – probability of population decline greater than 50% from 2016-2041, for changing adult mortality and a 3% decrease in adult survival, across decreasing populations and increasing populations.

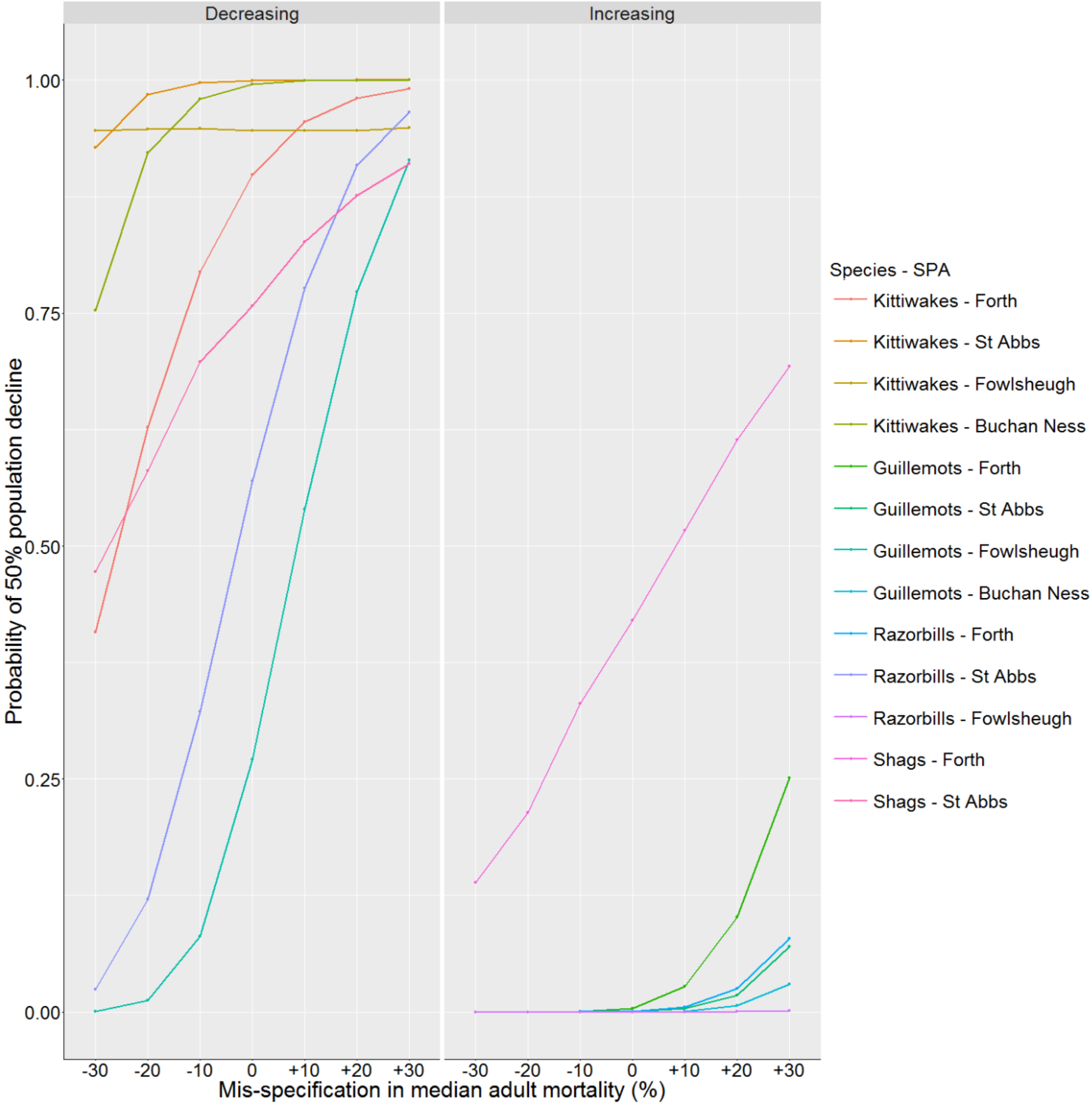


Figure 5h: PVA Metric F – centile from un-impacted population size equal to the 50th centile of the impacted population size, at 2041, for changing adult mortality and a 3% decrease in adult survival, across decreasing populations and increasing populations.

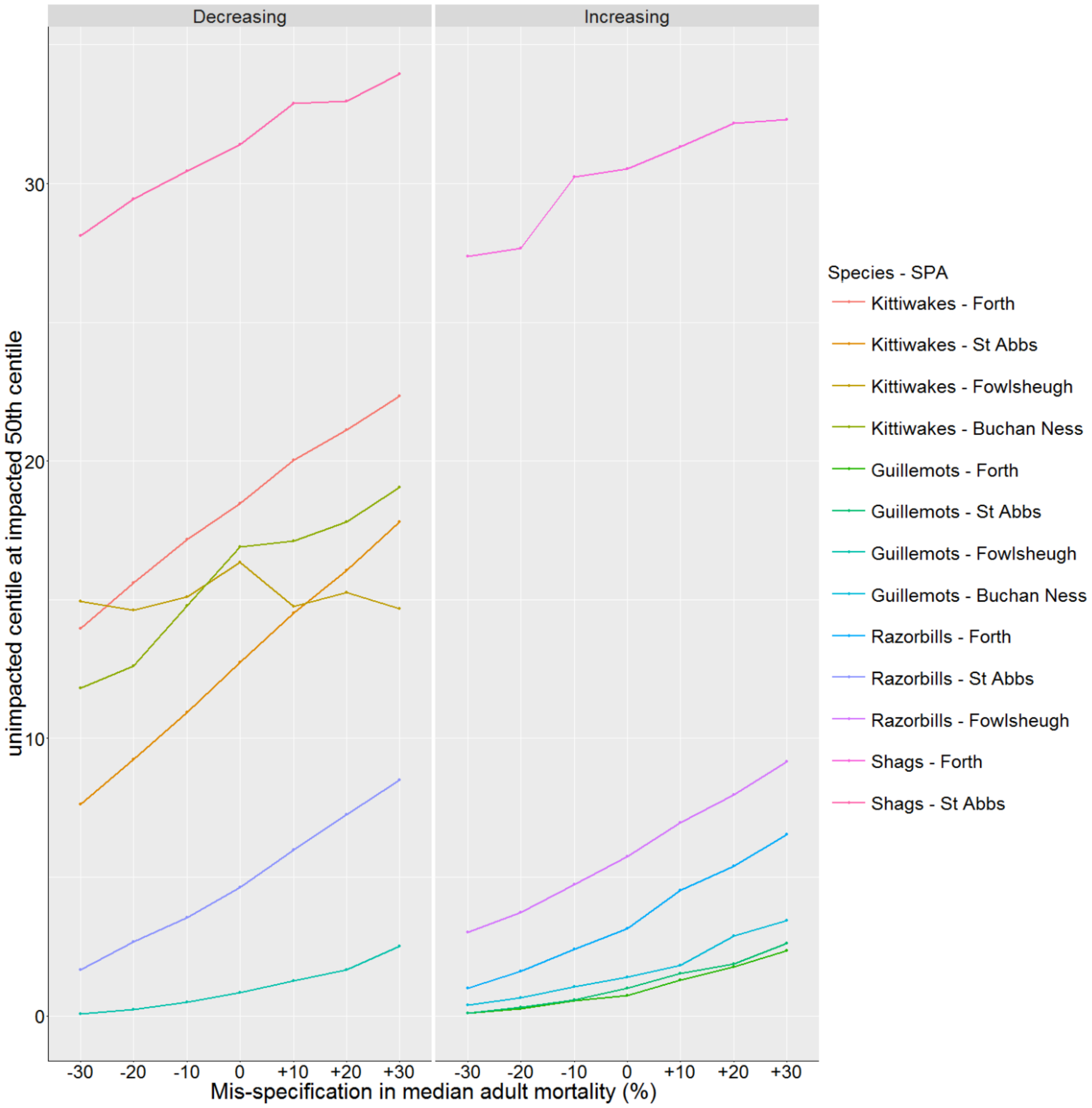


Figure 6a: PVA Metric A – ratio of population growth rate from 2016-2041, comparing impacted population vs. un-impacted population, for changing adult mortality and various decreases in adult survival, across all populations.

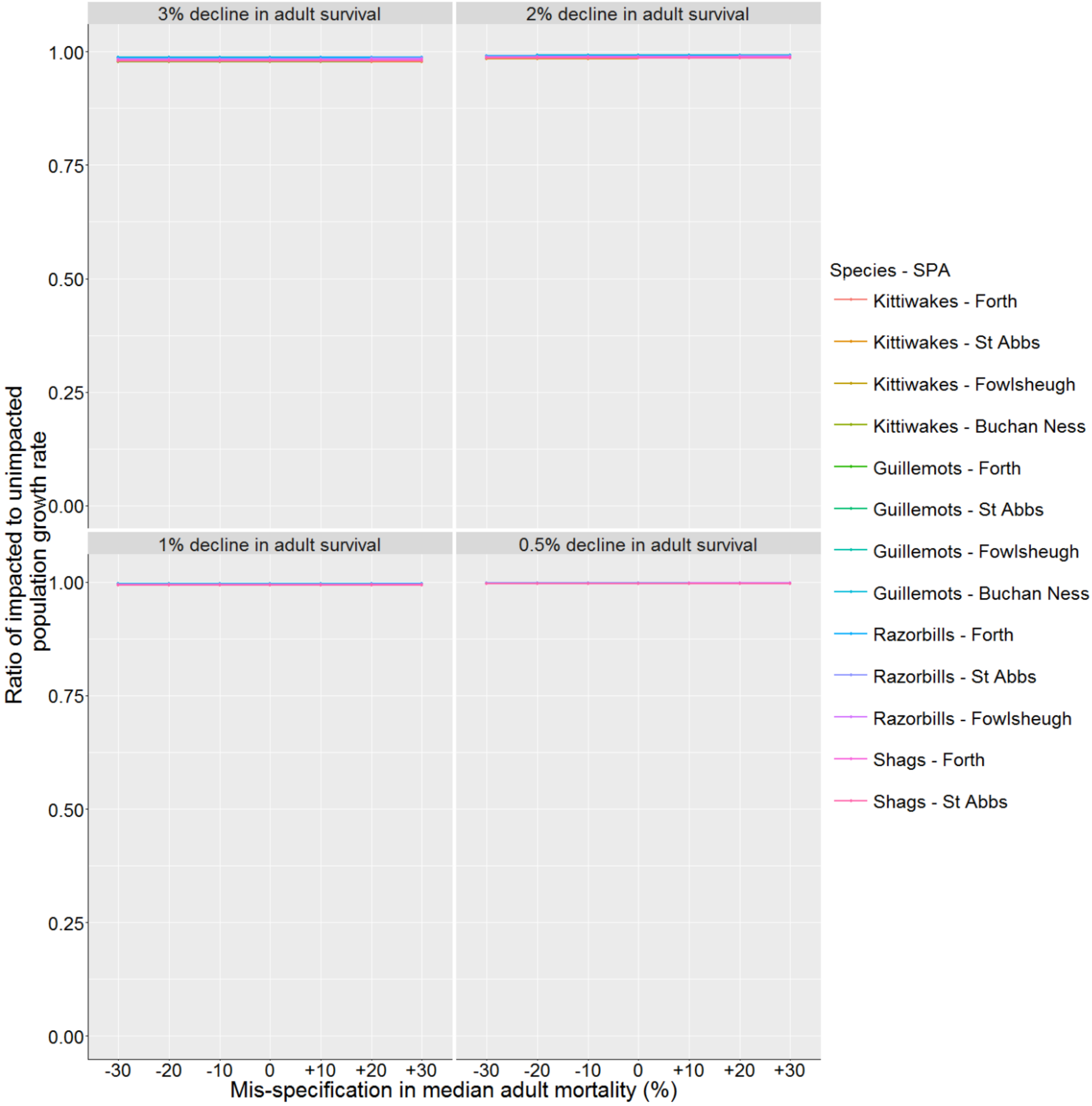


Figure 6b: PVA Metric B – ratio of population size at 2041, comparing impacted population vs. un-impacted population, for changing adult mortality and various decreases in adult survival, across all populations.

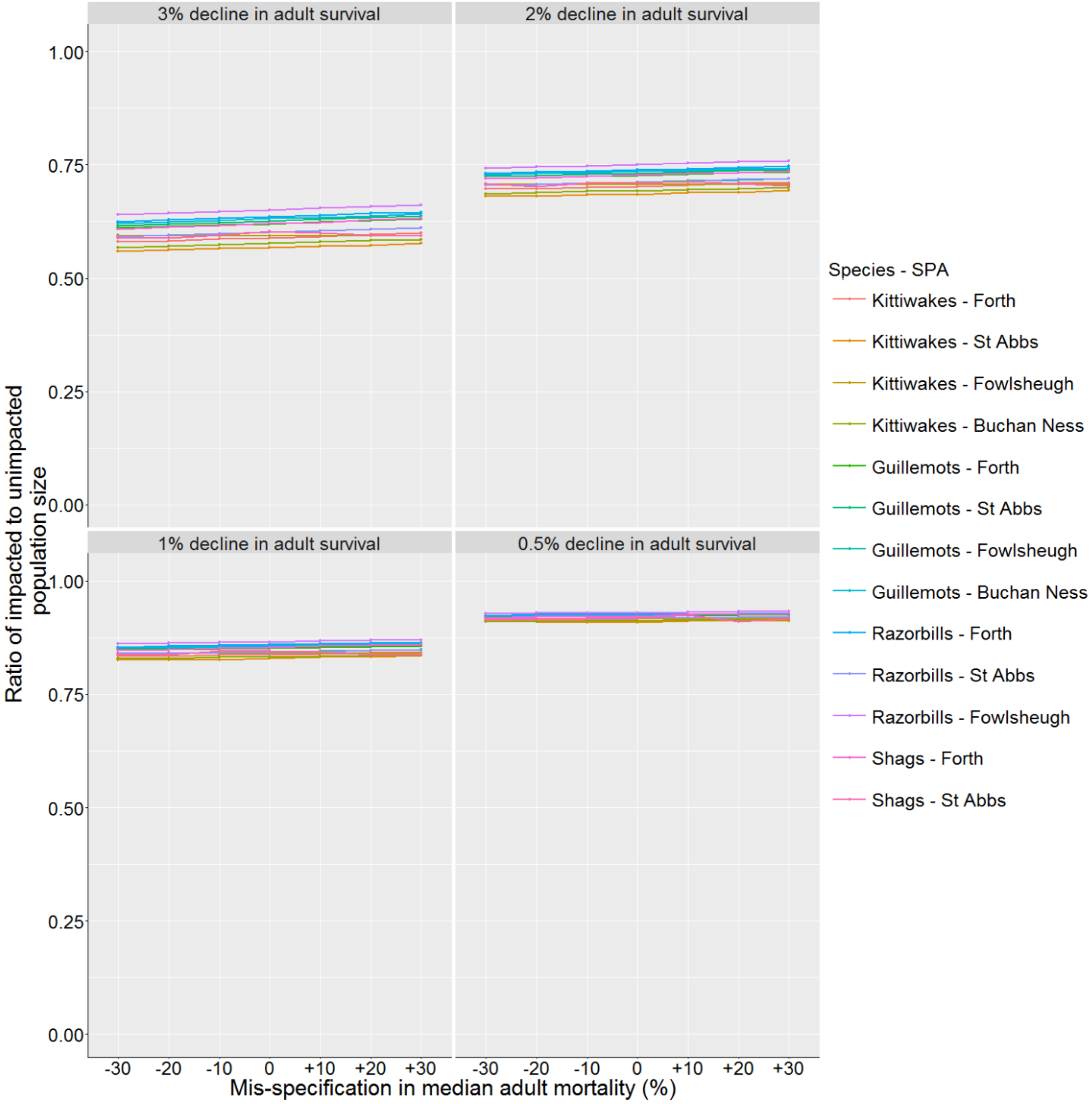


Figure 6c: PVA Metric C – difference in population growth rate from 2016-2041, comparing impacted population vs. un-impacted population, for changing adult mortality and various decreases in adult survival, across all populations.

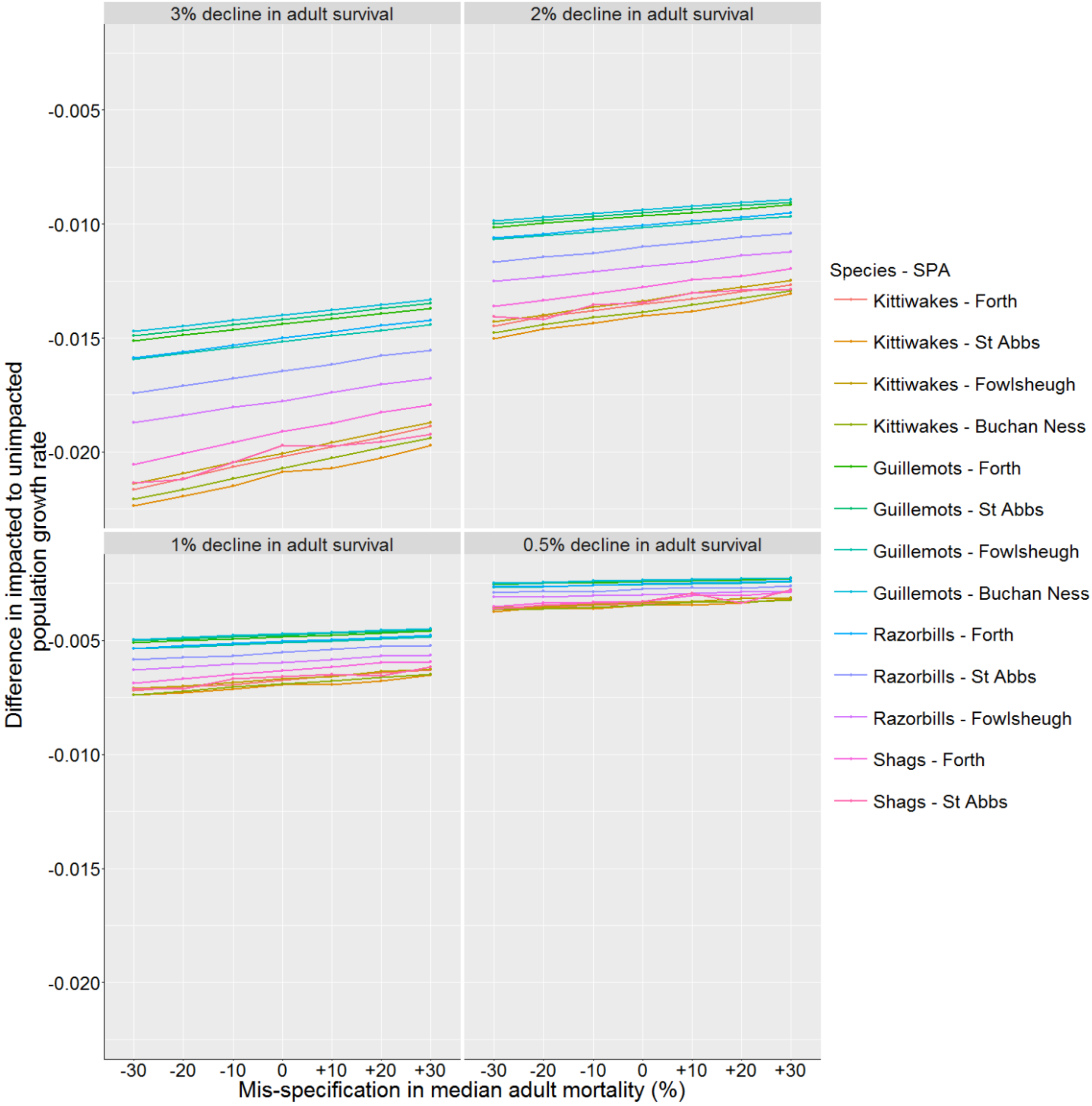


Figure 6d: PVA Metric D – difference in population size at 2041, comparing impacted population vs. un-impacted population, for changing adult mortality and various decreases in adult survival, across all populations.

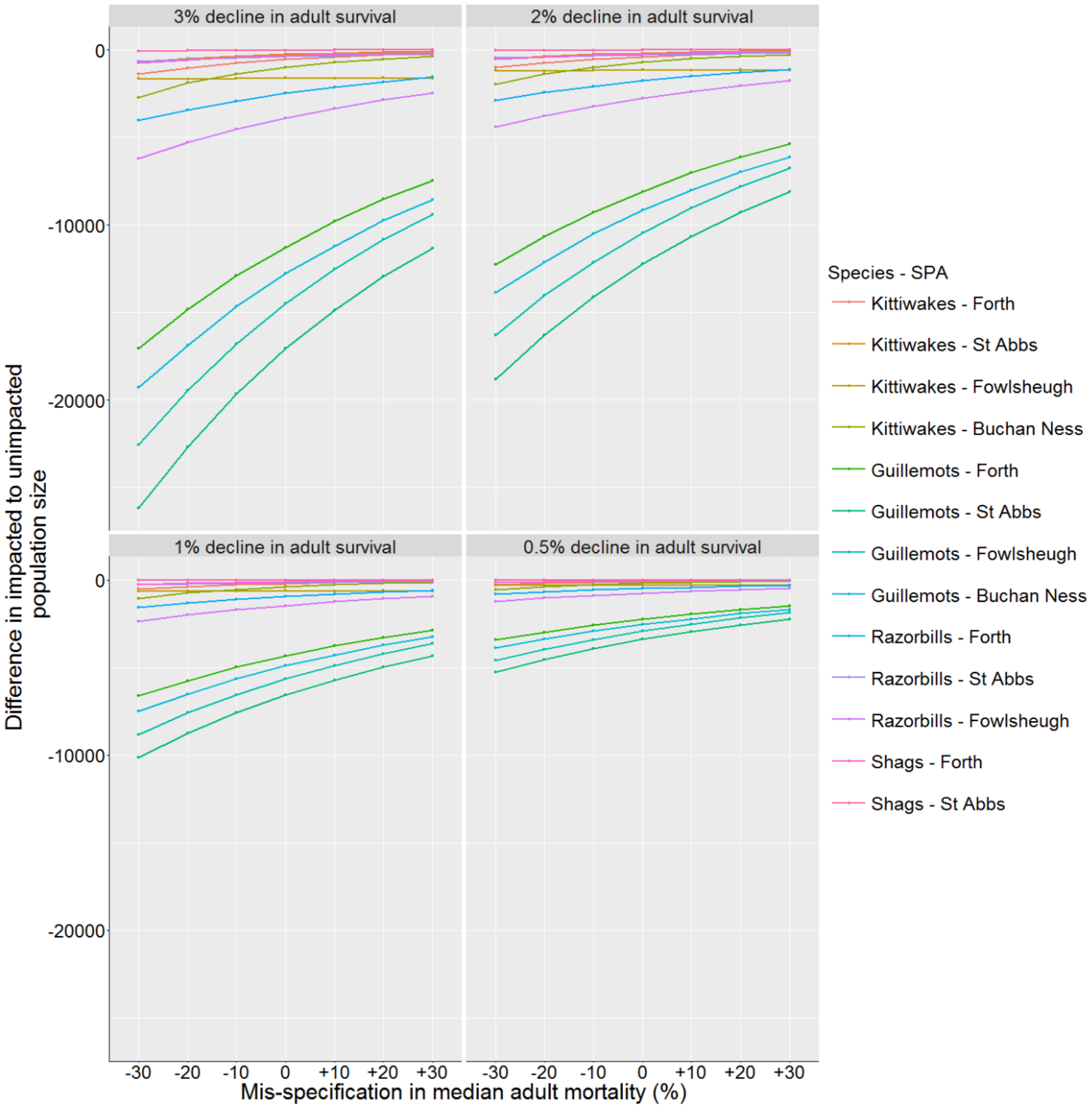


Figure 6e: PVA Metric E1 – probability of population decline greater than 10% from 2016-2041, for changing adult mortality and various decreases in adult survival, across all populations.

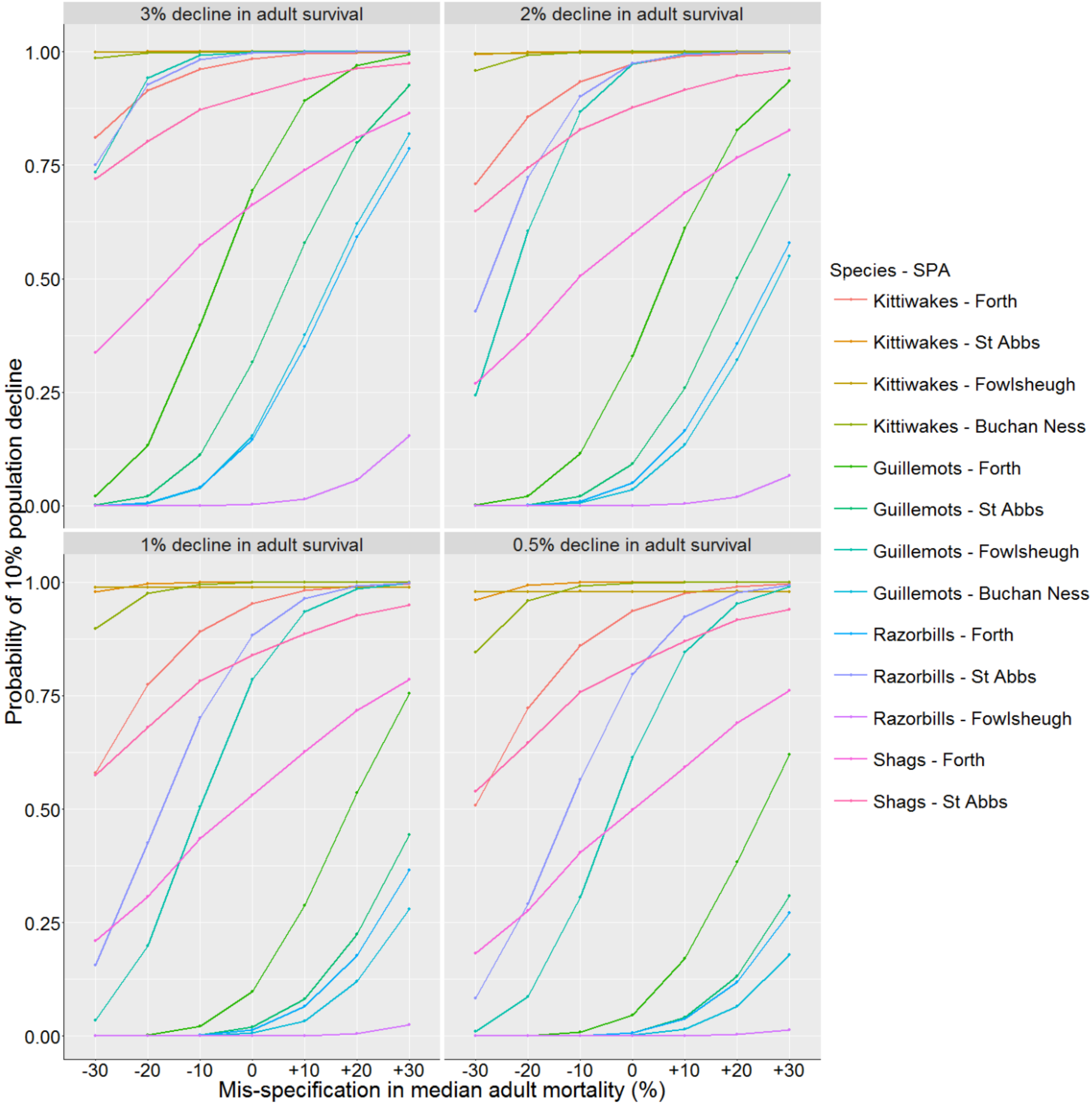


Figure 6f: PVA Metric E2 – probability of population decline greater than 25% from 2016-2041, for changing adult mortality and various decreases in adult survival, across all populations.

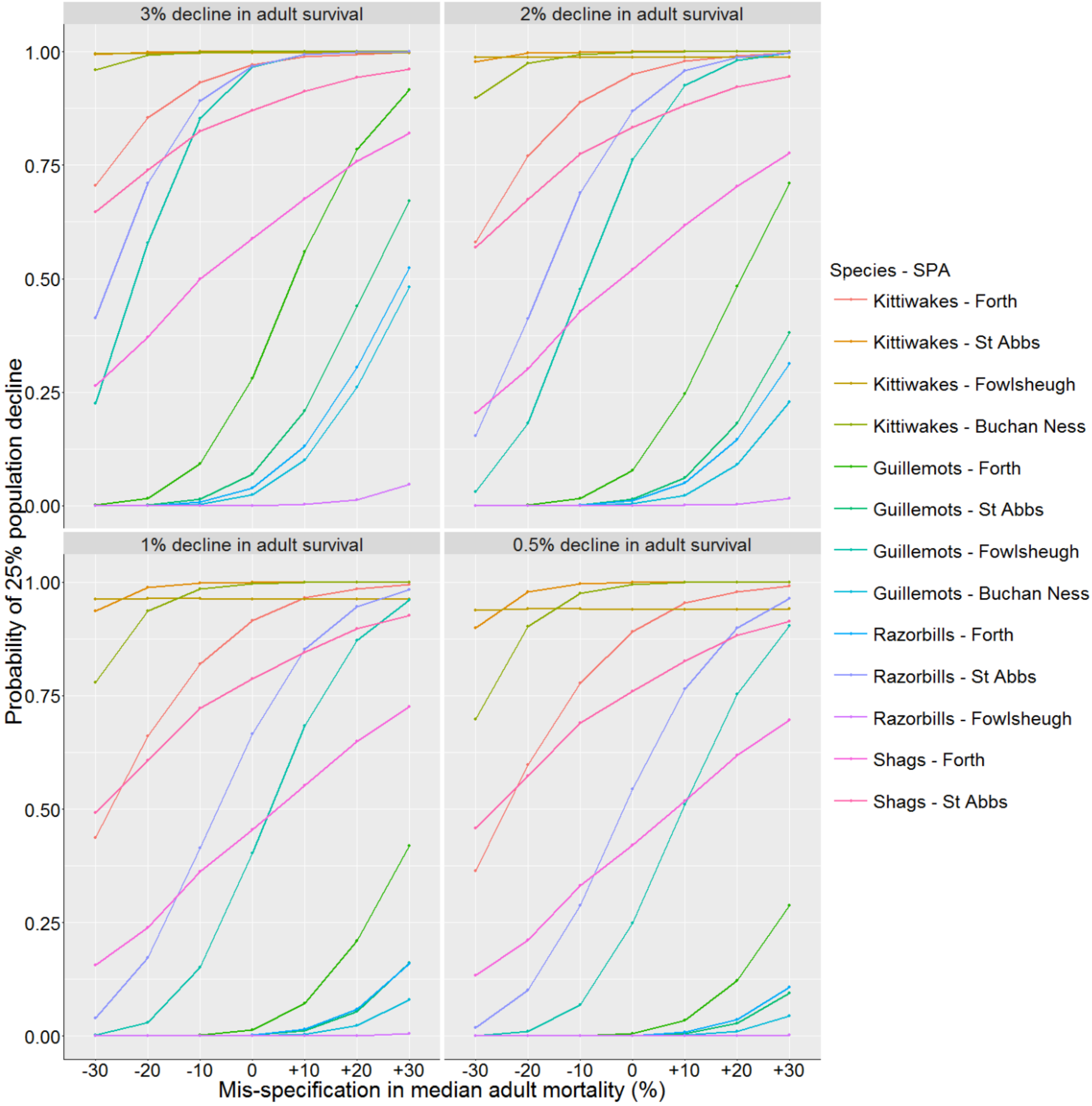


Figure 6g: PVA Metric E3 – probability of population decline greater than 50% from 2016-2041, for changing adult mortality and various decreases in adult survival, across all populations.

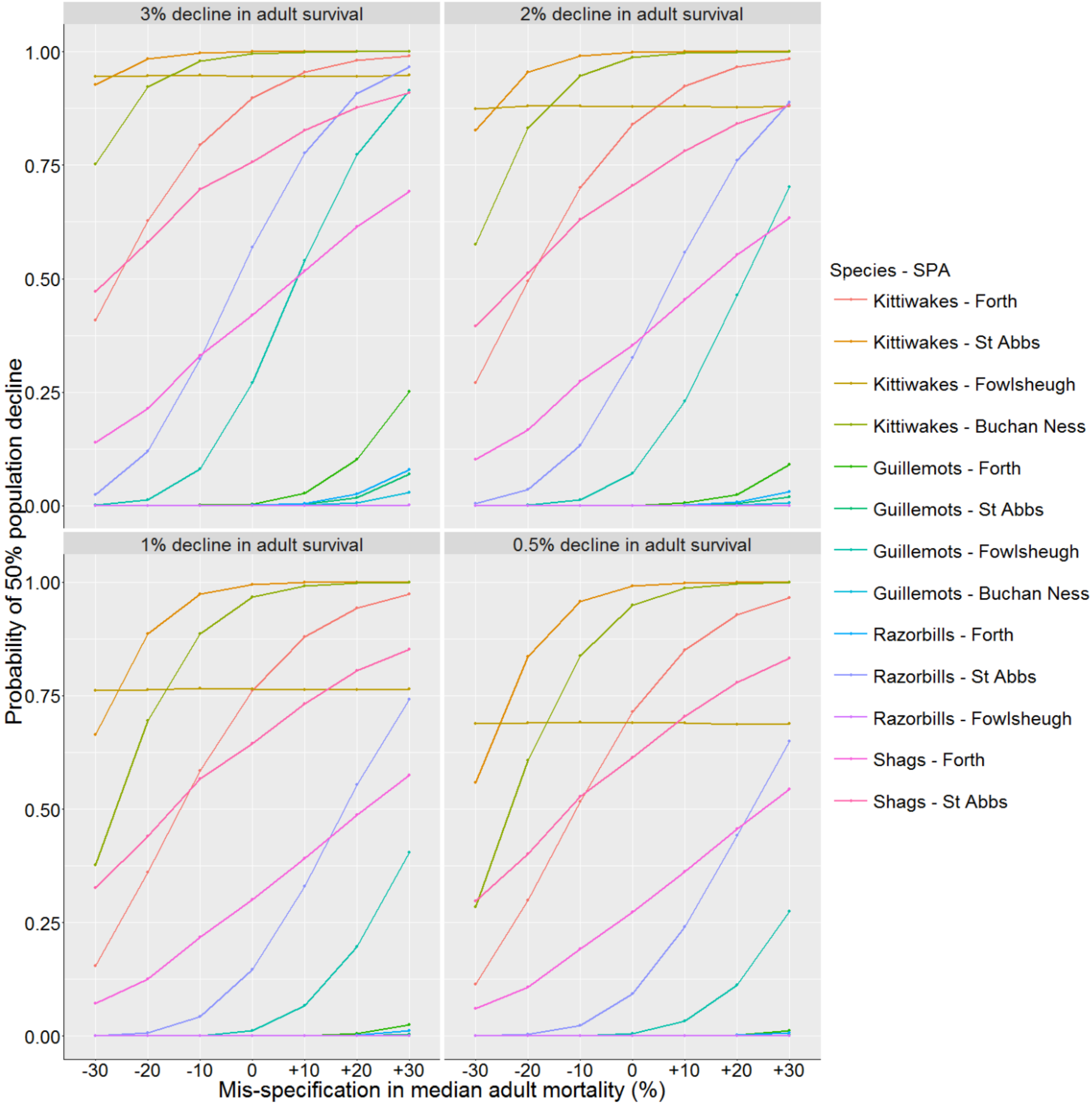
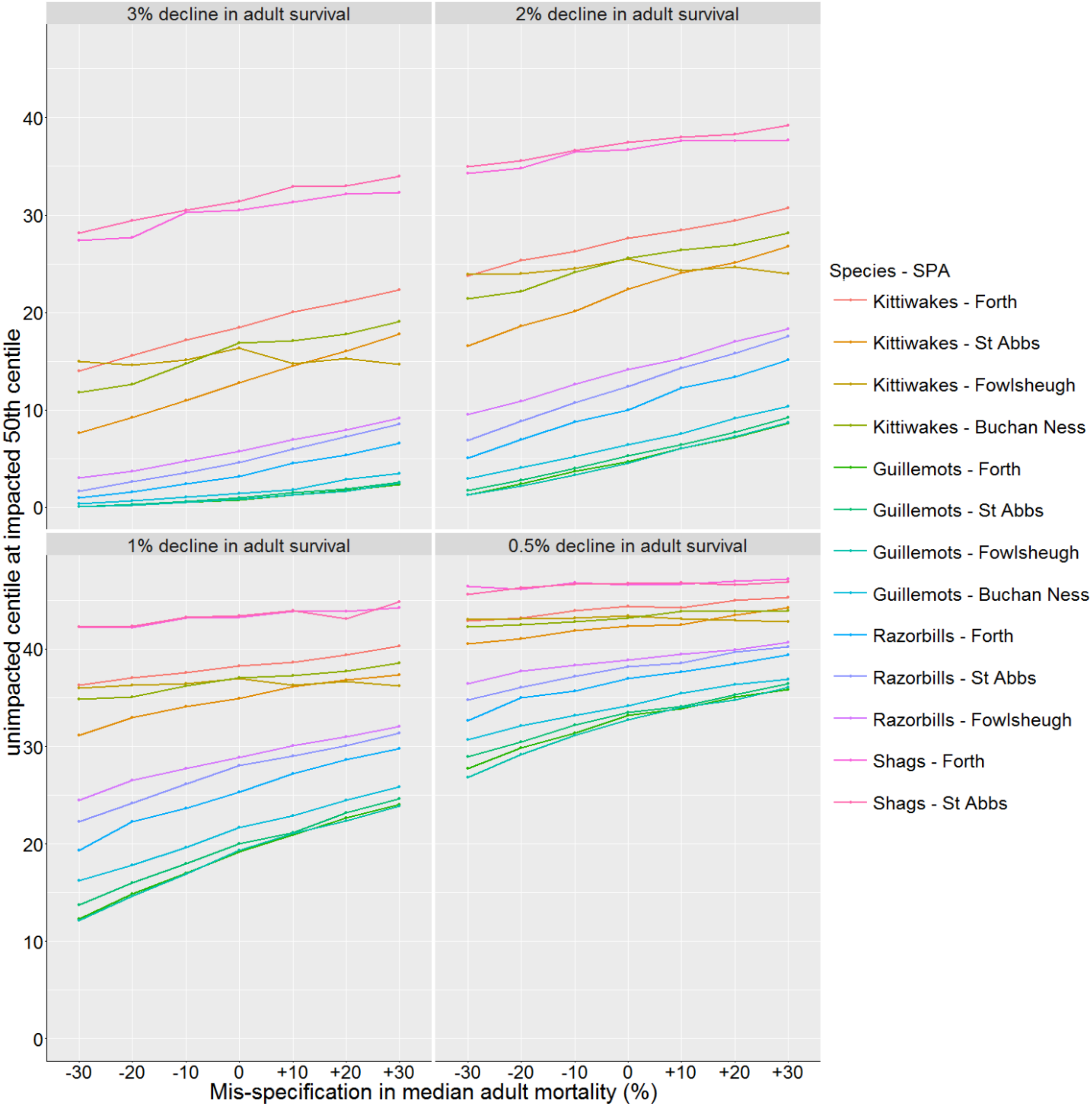


Figure 6h: PVA Metric F – centile from un-impacted population size equal to the 50th centile of the impacted population size, at 2041, for changing adult mortality and various decreases in adult survival, across all populations.



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).

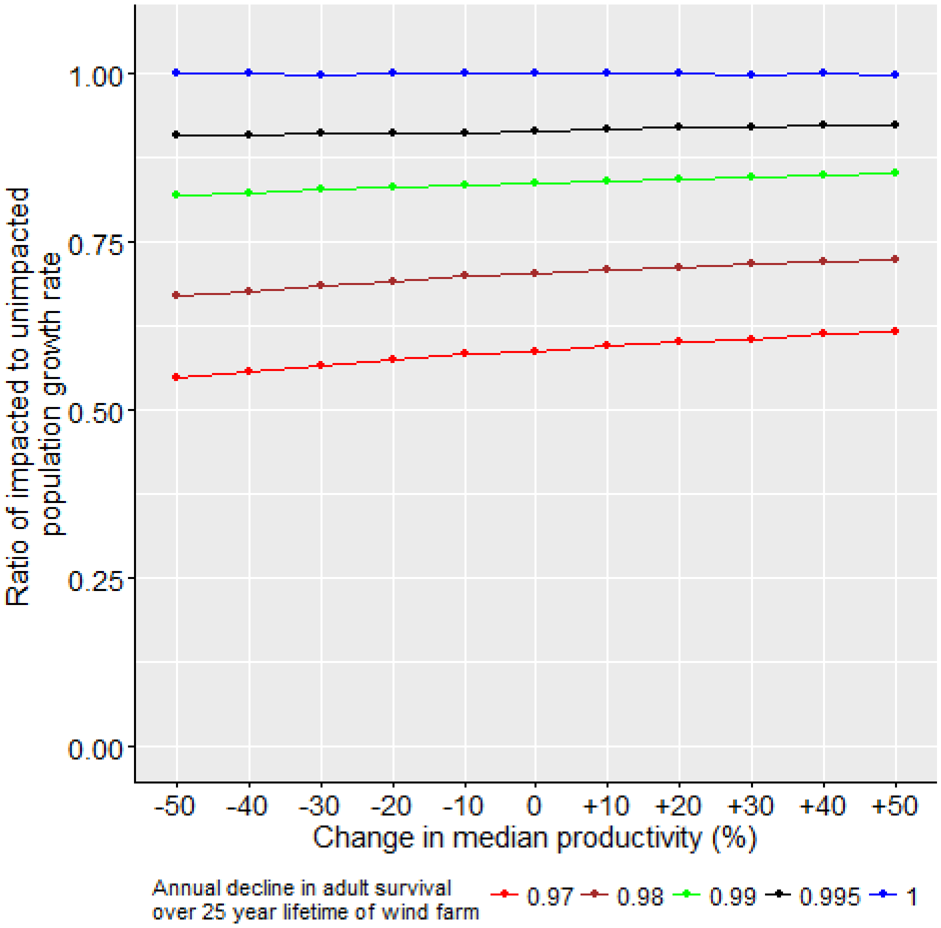


Figure A2.1: PVA Metric A – ratio of 25 year population growth rate, comparing impacted population vs. un-impacted population, showing productivity mis-specification varied from -50% to +50% (with 0% representing no mis-specification) in Forth Islands kittiwakes. The five coloured lines represent the 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.

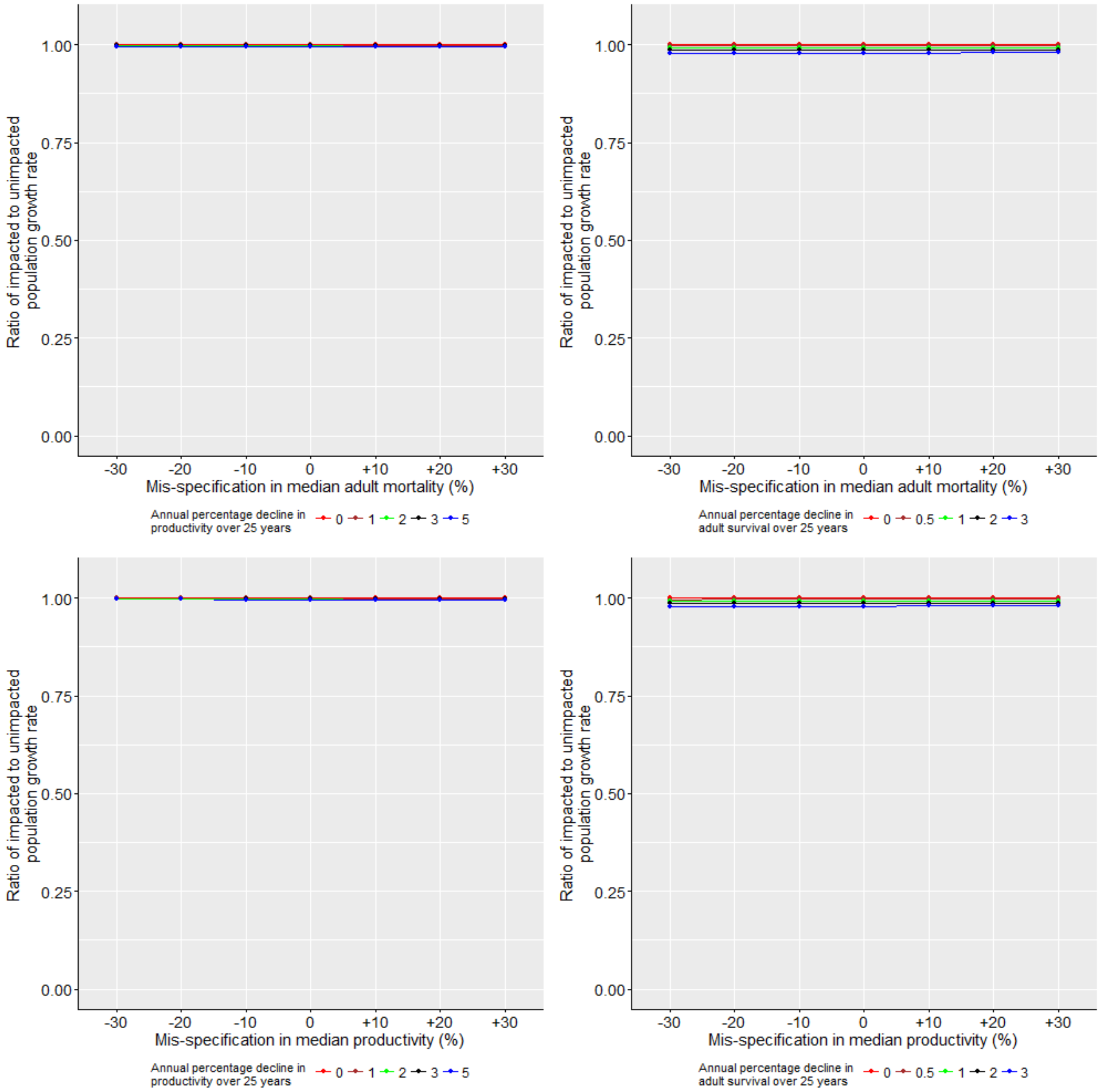


Figure A2.1b: PVA Metric B for Forth Kittiwakes – ratio of population size at 2041, comparing impacted population vs. un-impacted population.

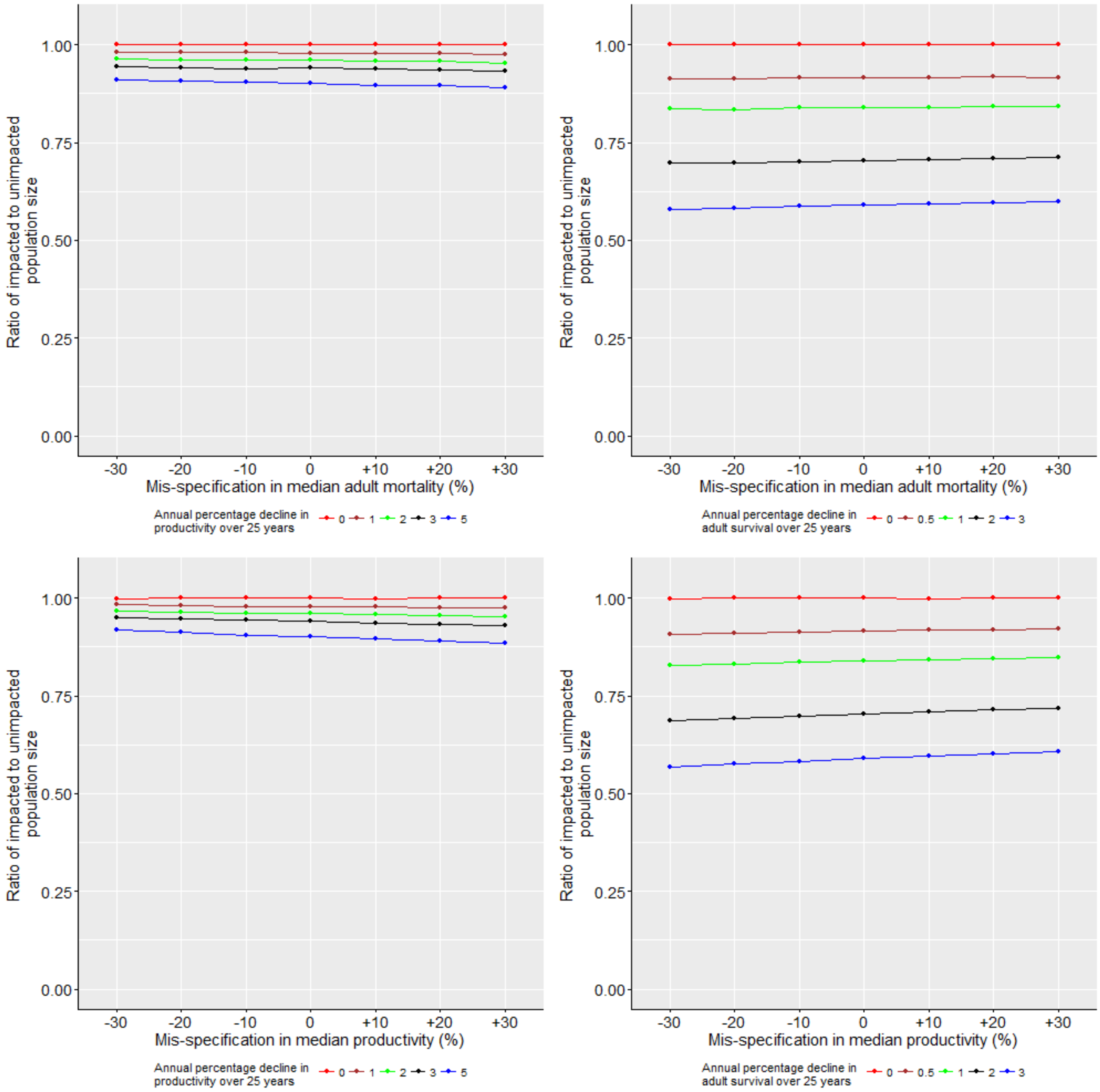


Figure A2.1c: PVA Metric C for Forth Kittiwakes – difference in population growth rate from 2016-2041, comparing impacted population vs. un-impacted population.

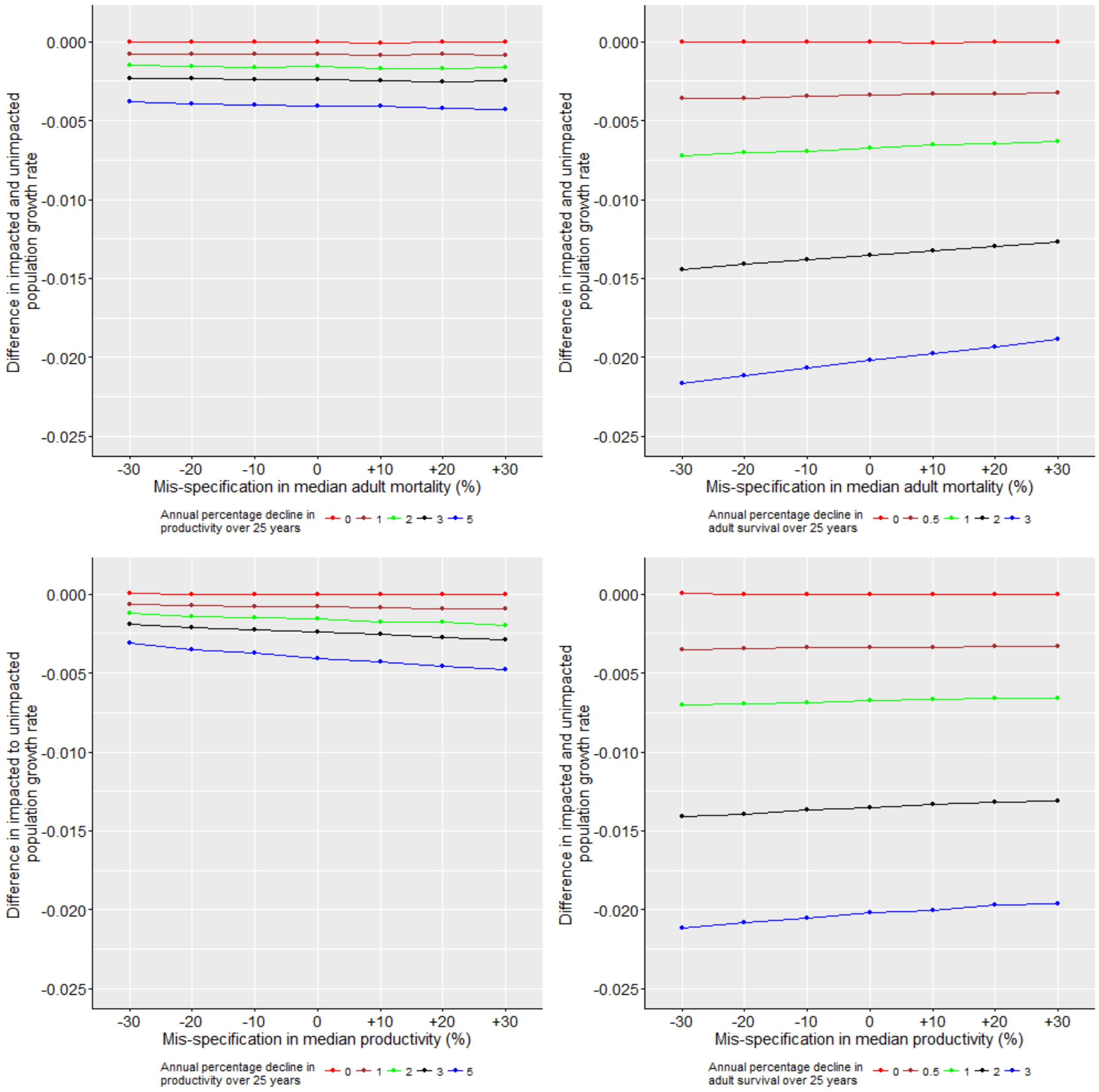


Figure A2.1d: PVA Metric D for Forth Kittiwakes – difference in population size at 2041, comparing impacted population vs. un-impacted population.

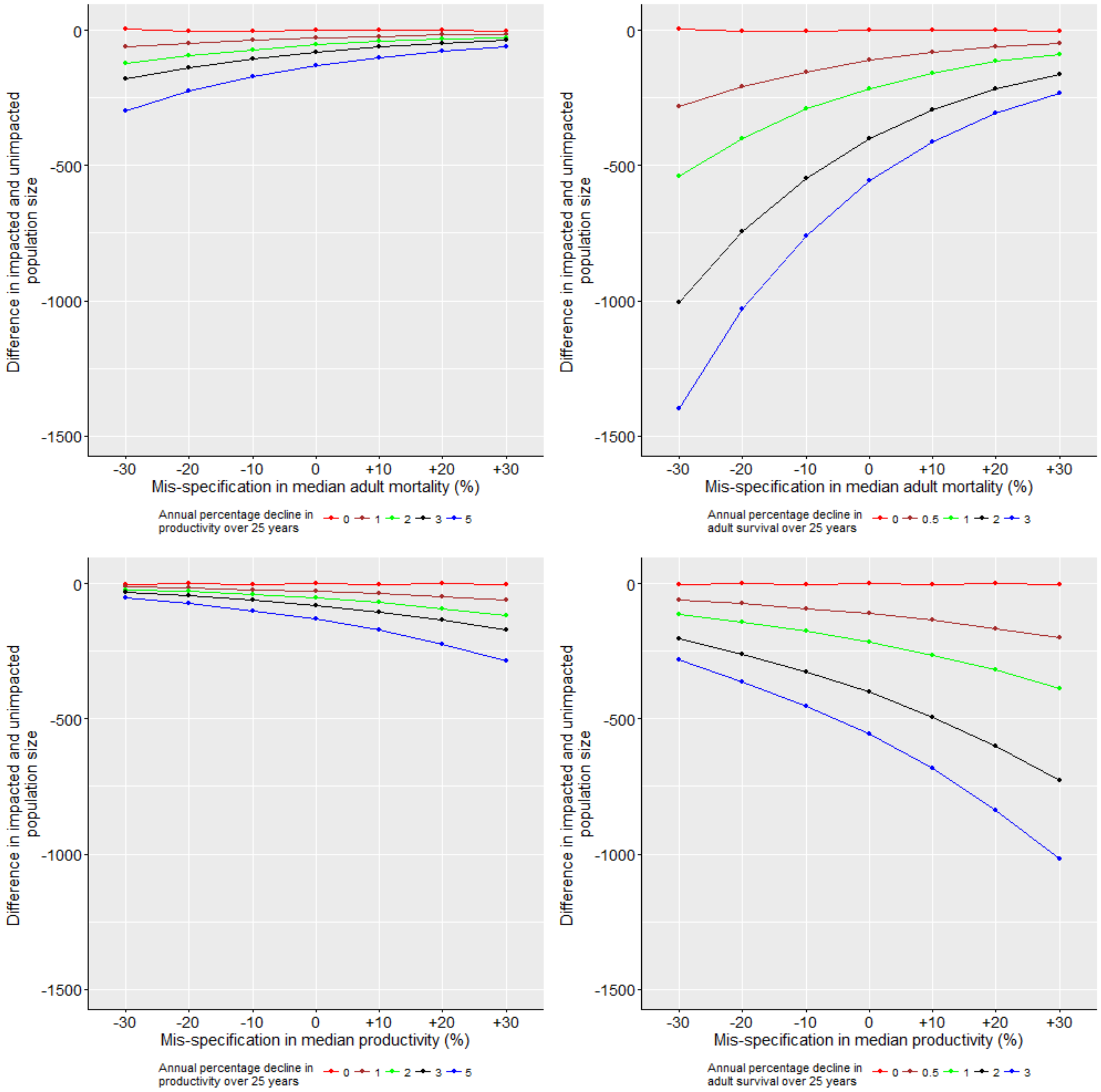


Figure A2.1e: PVA Metric E1 for Forth Kittiwakes – probability of population decline greater than 10% from 2016-2041.

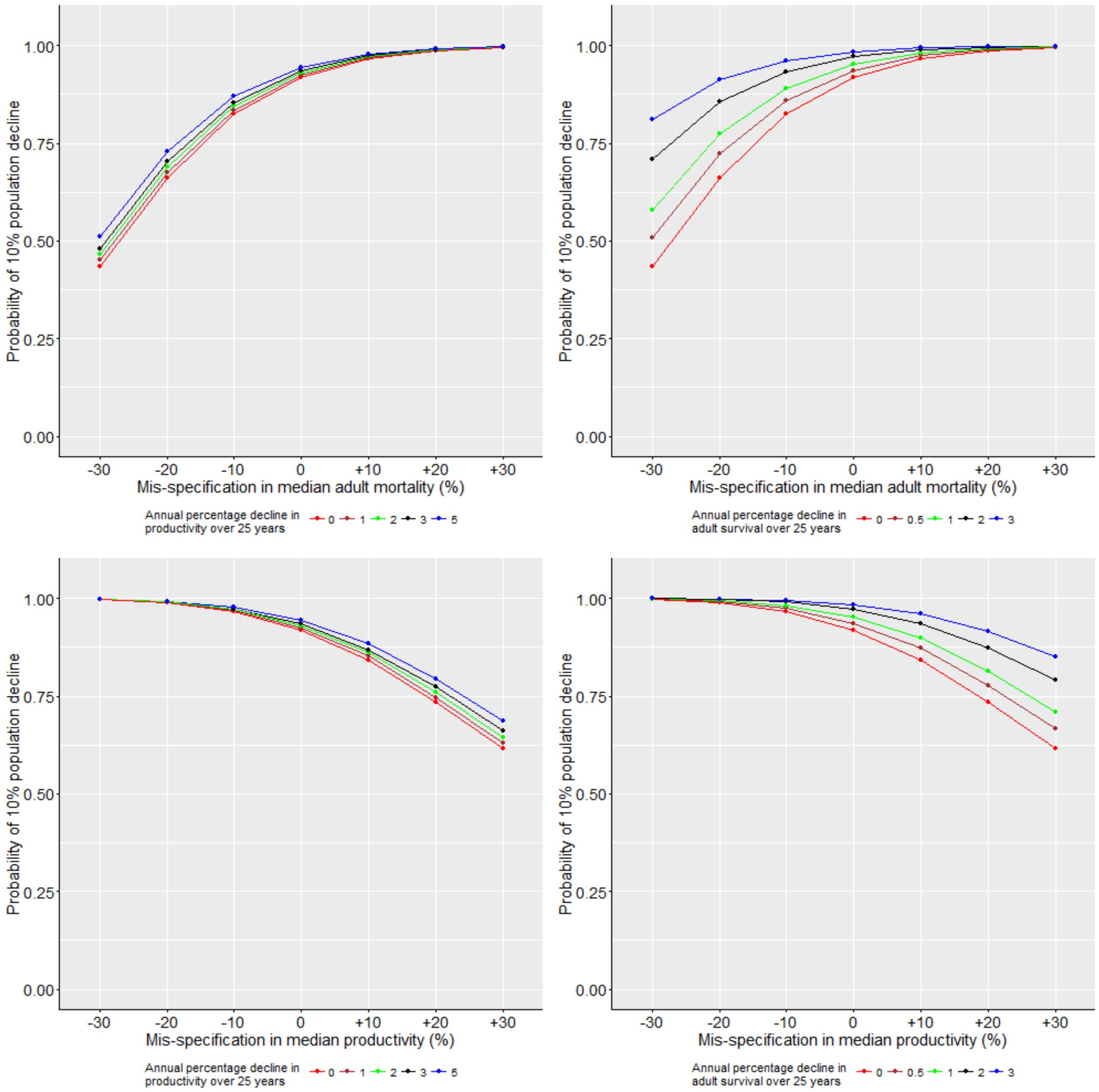


Figure A2.1f: PVA Metric E2 for Forth Kittiwakes – probability of population decline greater than 25% from 2016-2041.

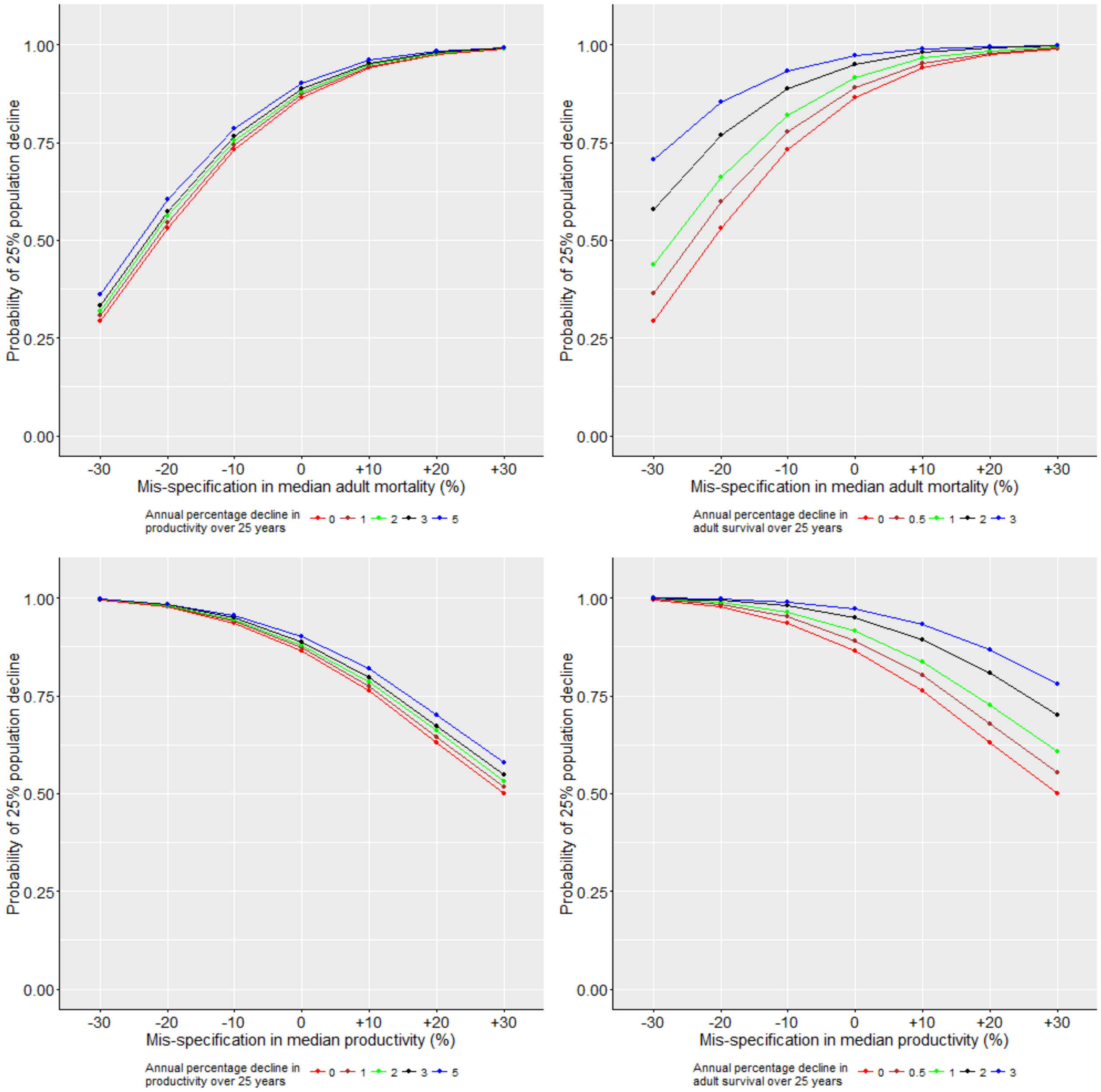


Figure A2.1g: PVA Metric E3 for Forth Kittiwakes – probability of population decline greater than 50% from 2016-2041.

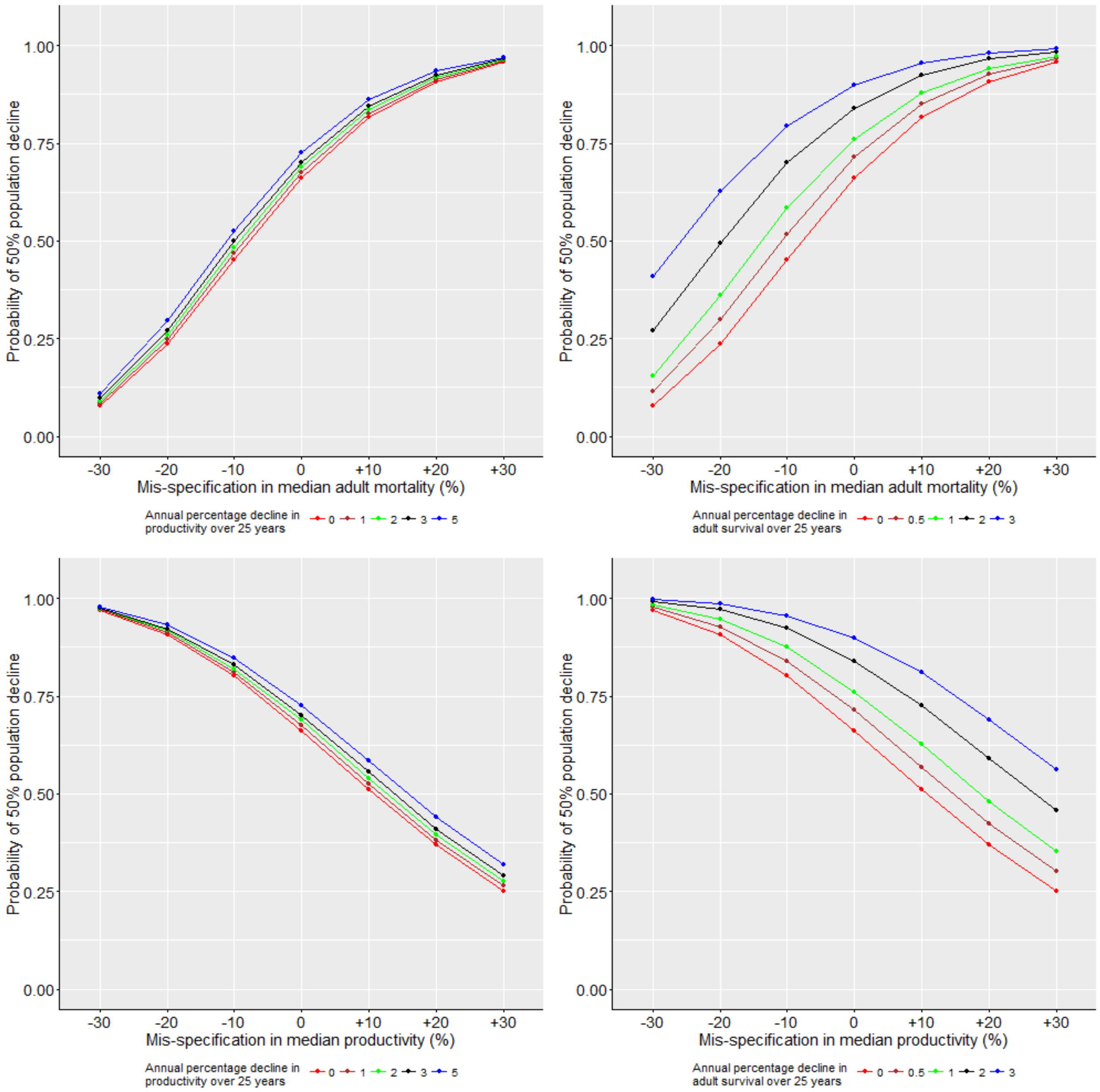
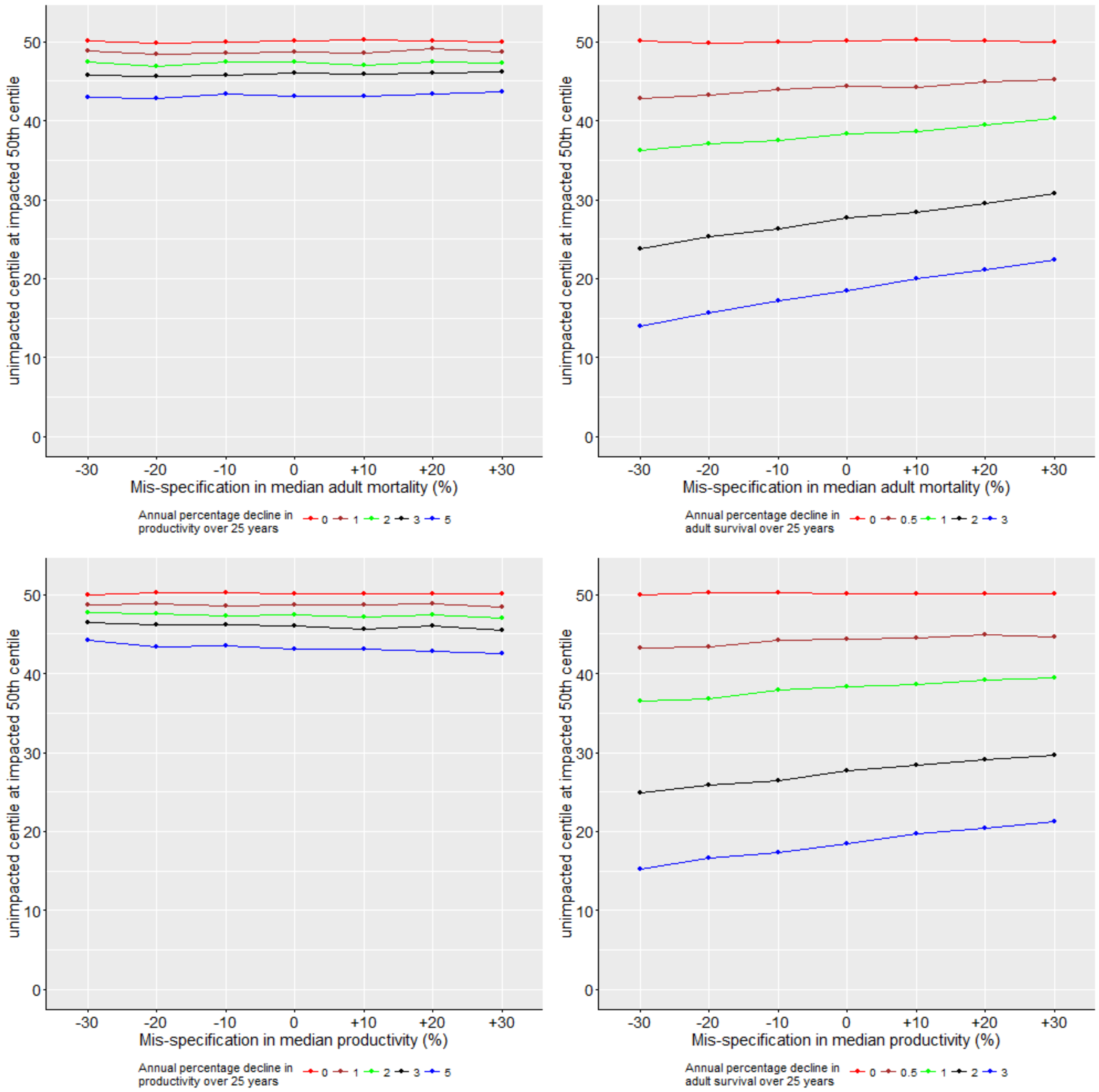


Figure A2.1h: PVA Metric F for Forth Kittiwakes – centile from un-impacted population size equal to the 50th centile of the impacted population size, at 2041.



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.

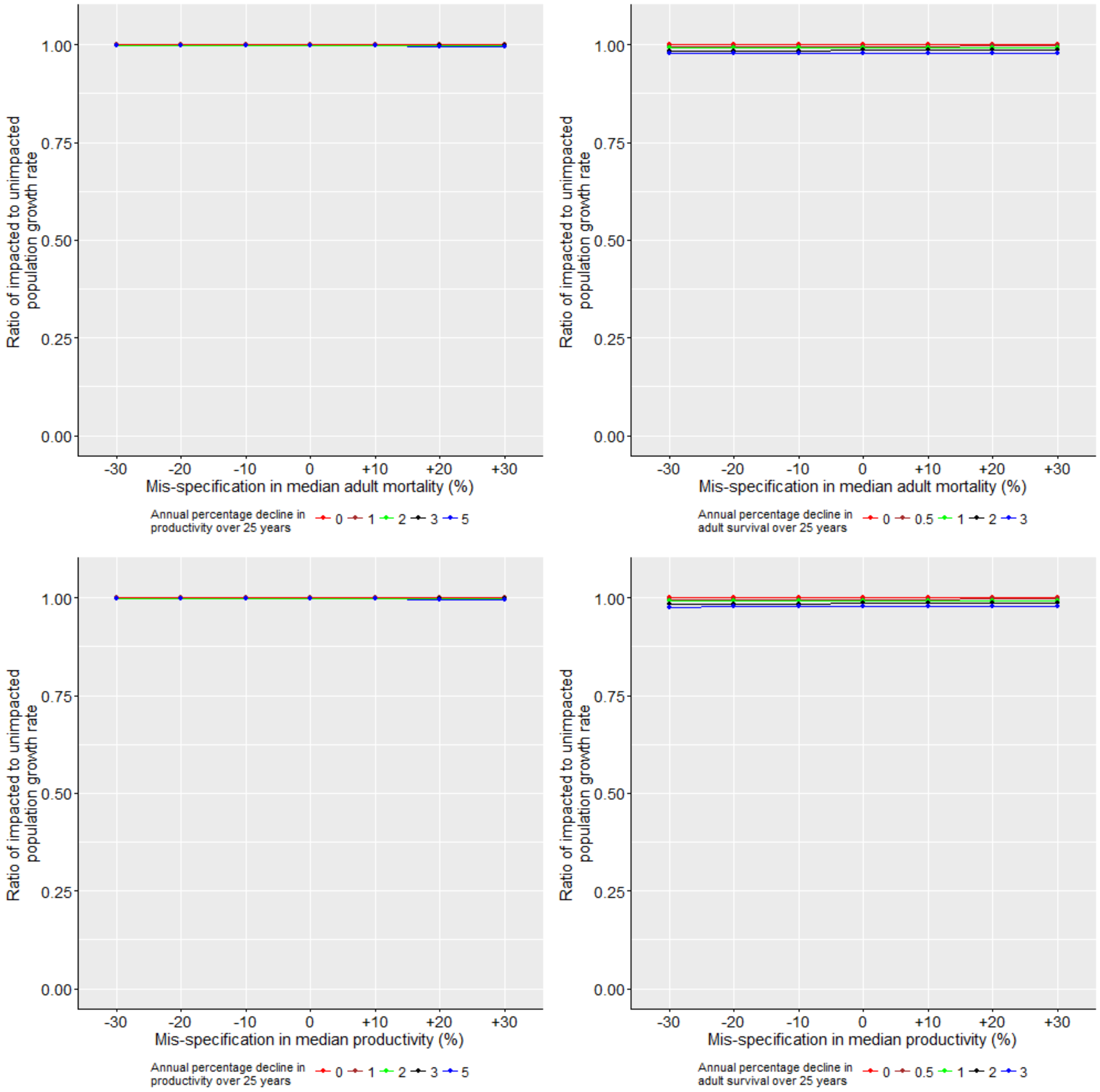


Figure A2.2b. PVA Metric B for St Abb’s Kittiwakes – ratio of population size at 2041, comparing impacted population vs. un-impacted population.

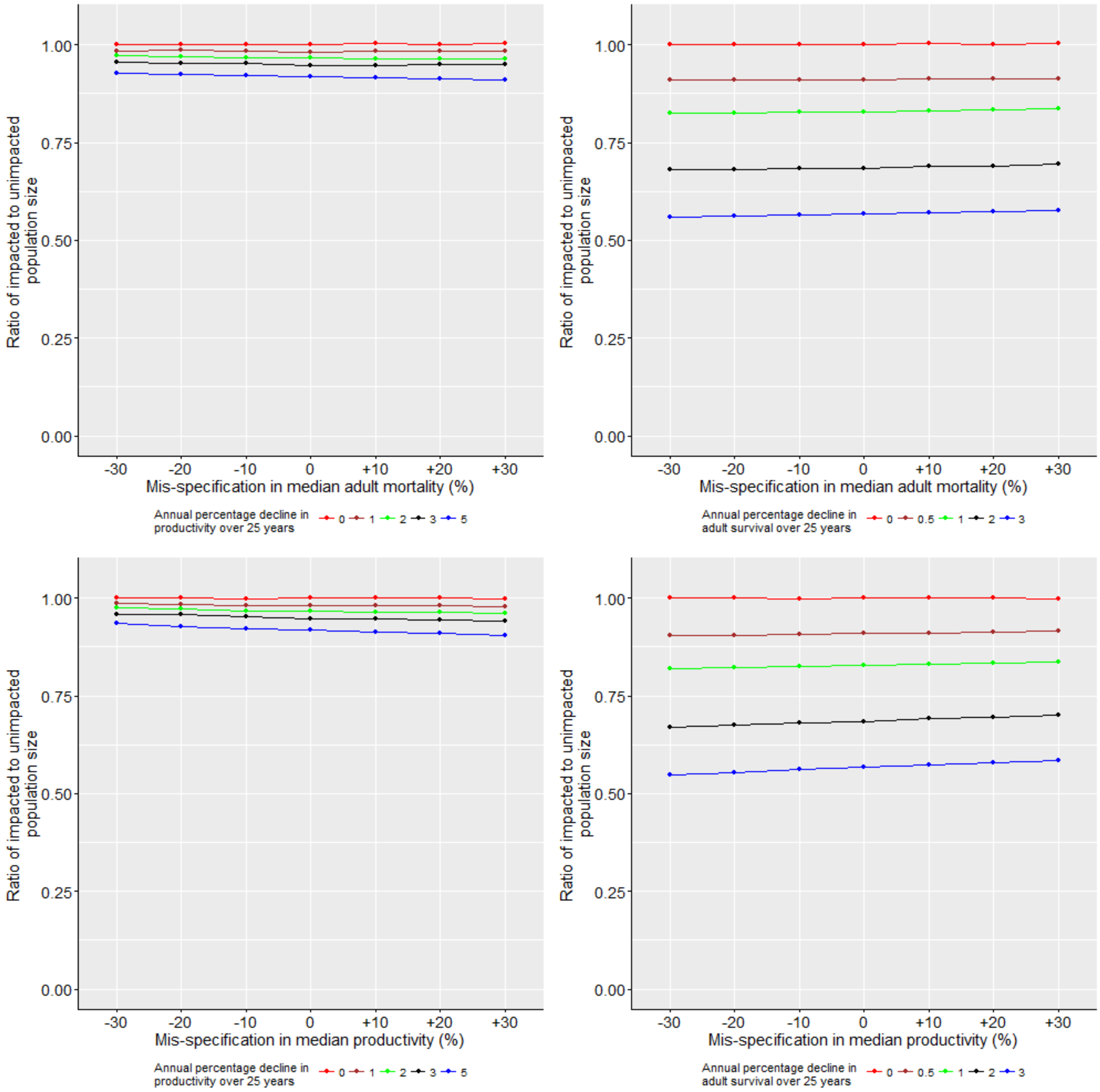


Figure A2.2c. PVA Metric C for St Abb’s Kittiwakes – difference in population growth rate from 2016-2041, comparing impacted population vs. un-impacted population.

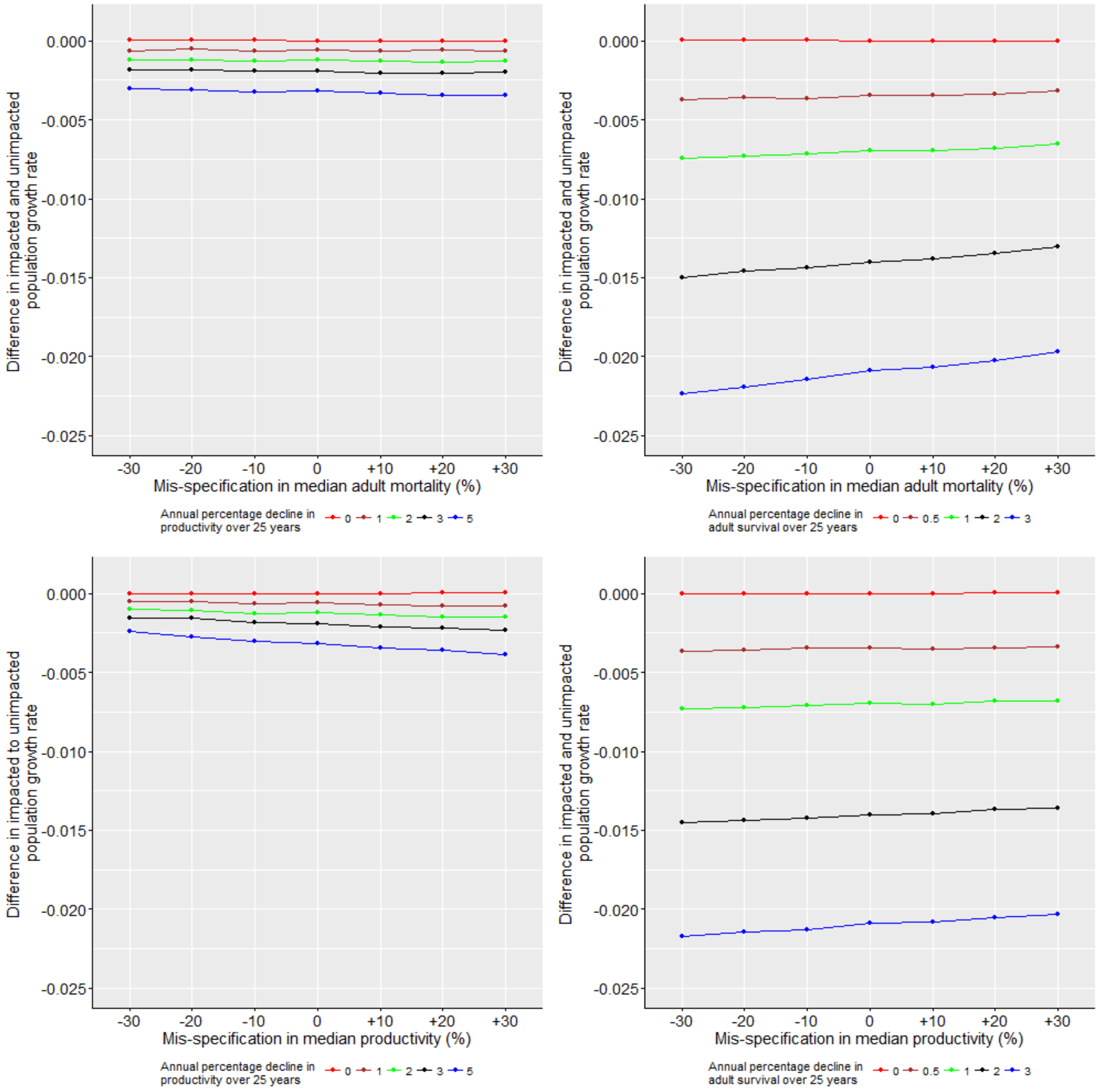


Figure A2.2d. PVA Metric D for St Abb’s Kittiwakes – difference in population size at 2041, comparing impacted population vs. un-impacted population.

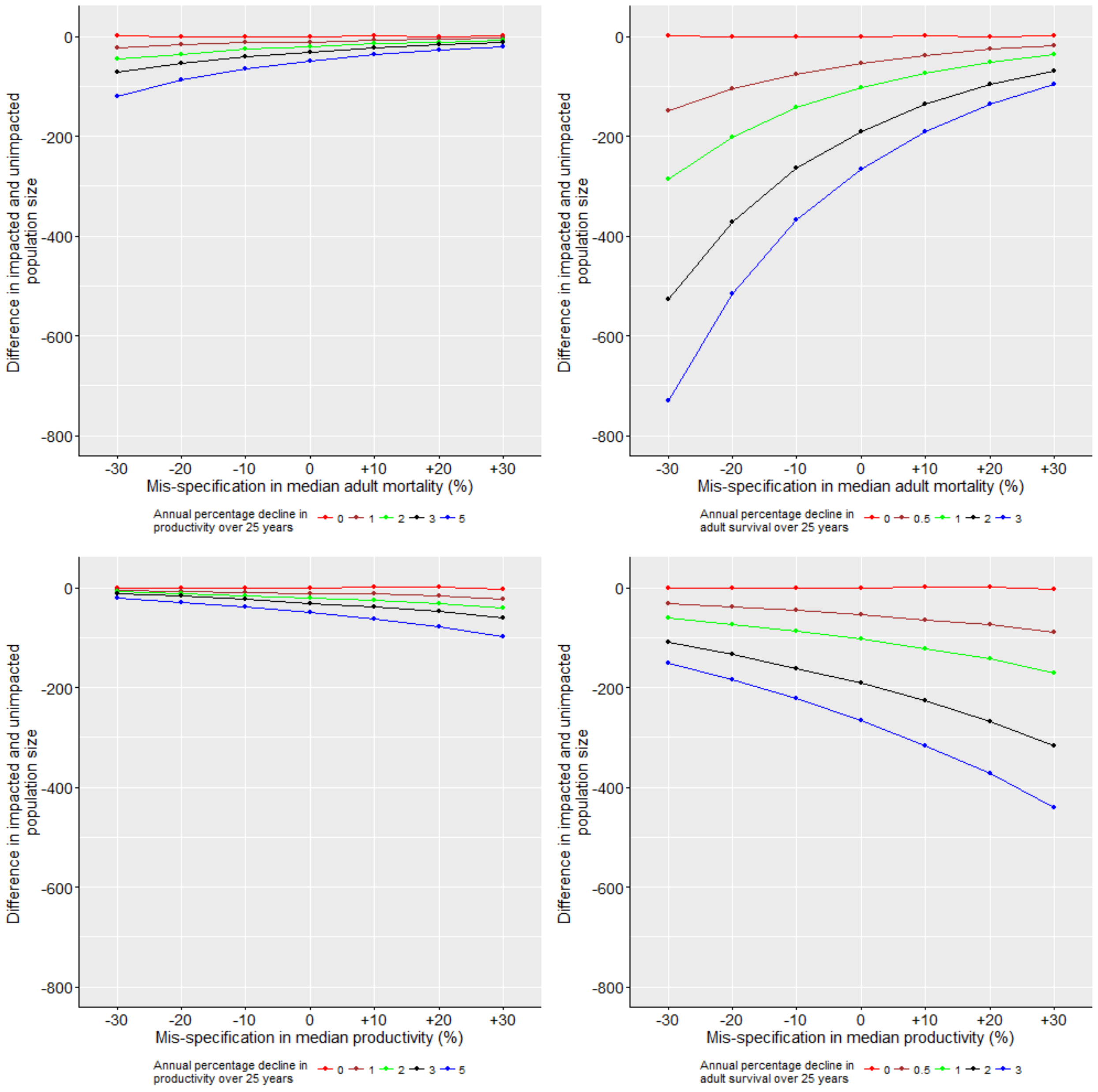


Figure A2.2e. PVA Metric E1 for St Abb’s Kittiwakes – probability of population decline greater than 10% from 2016-2041.

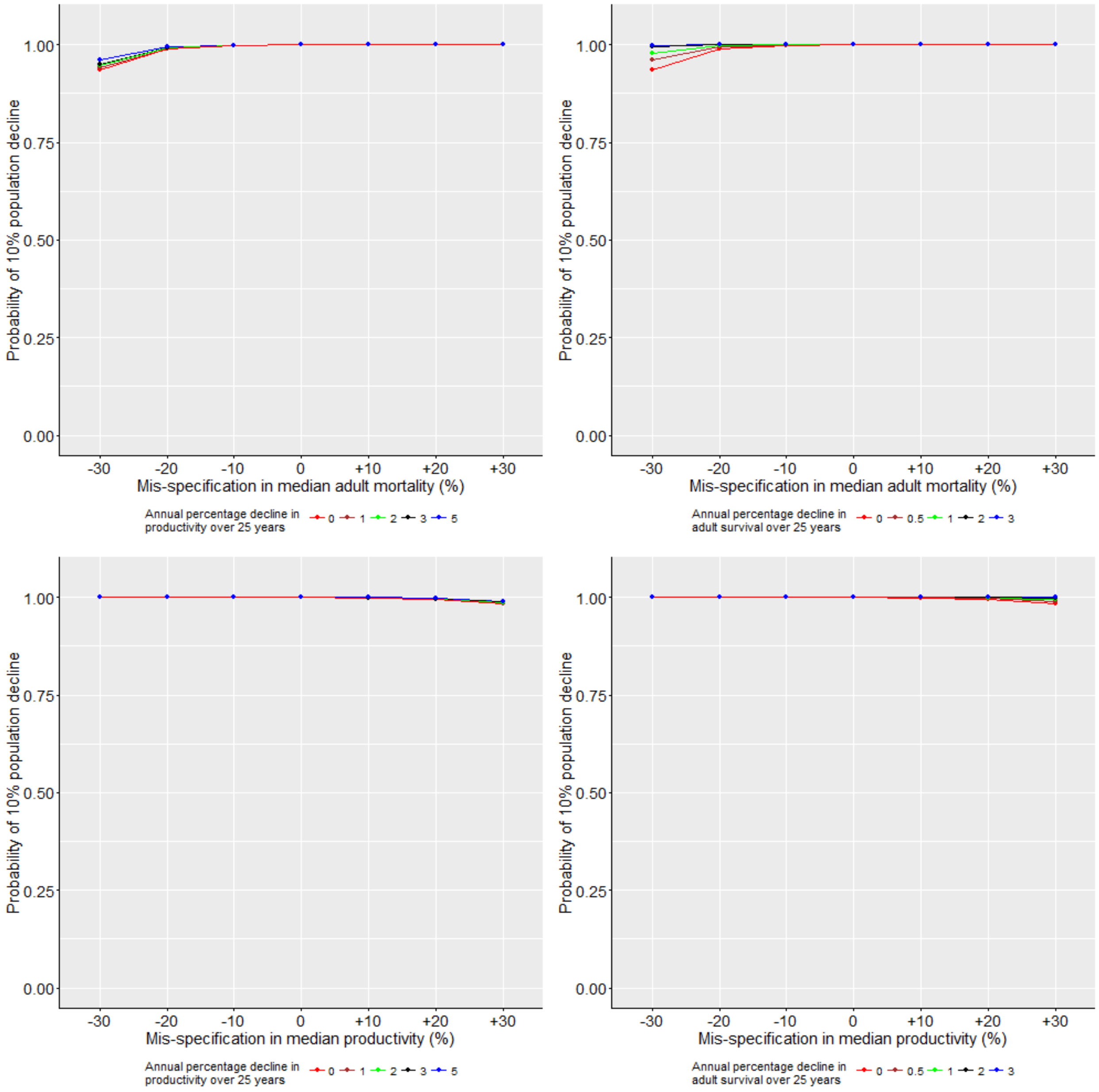


Figure A2.2f. PVA Metric E2 for St Abb’s Kittiwakes – probability of population decline greater than 25% from 2016-2041.

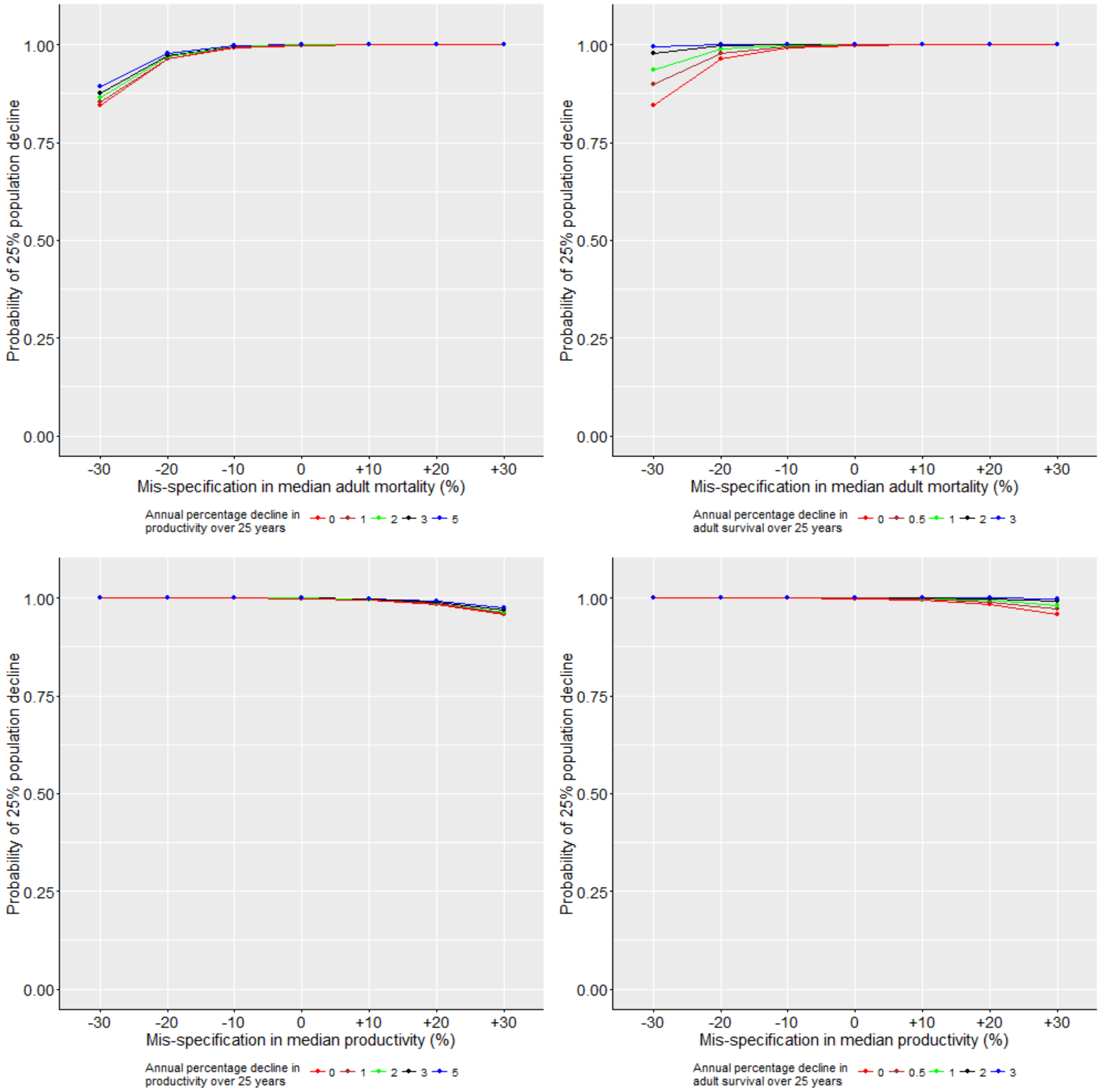


Figure A2.2g. PVA Metric E3 for St Abb’s Kittiwakes – probability of population decline greater than 50% from 2016-2041.

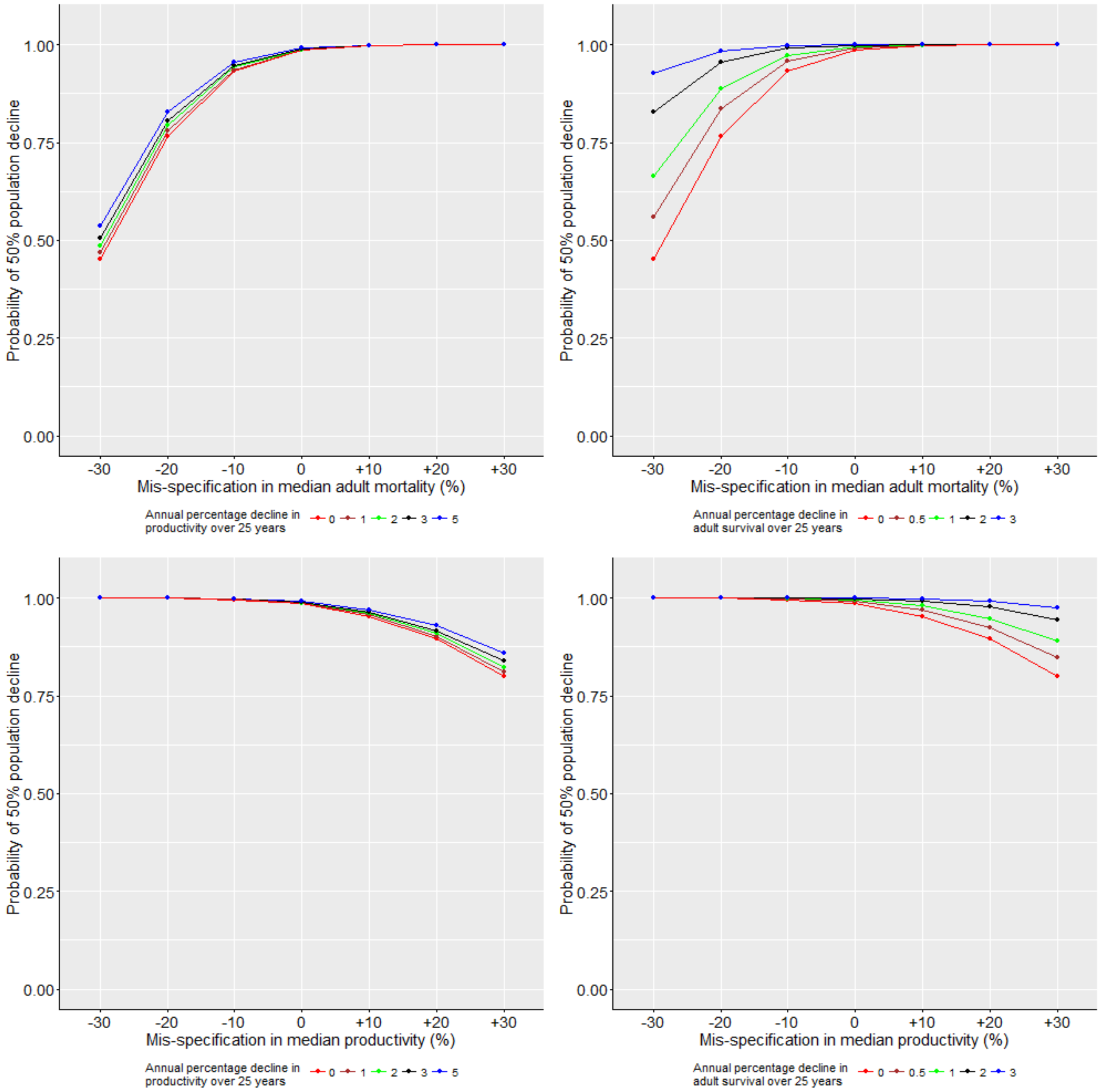
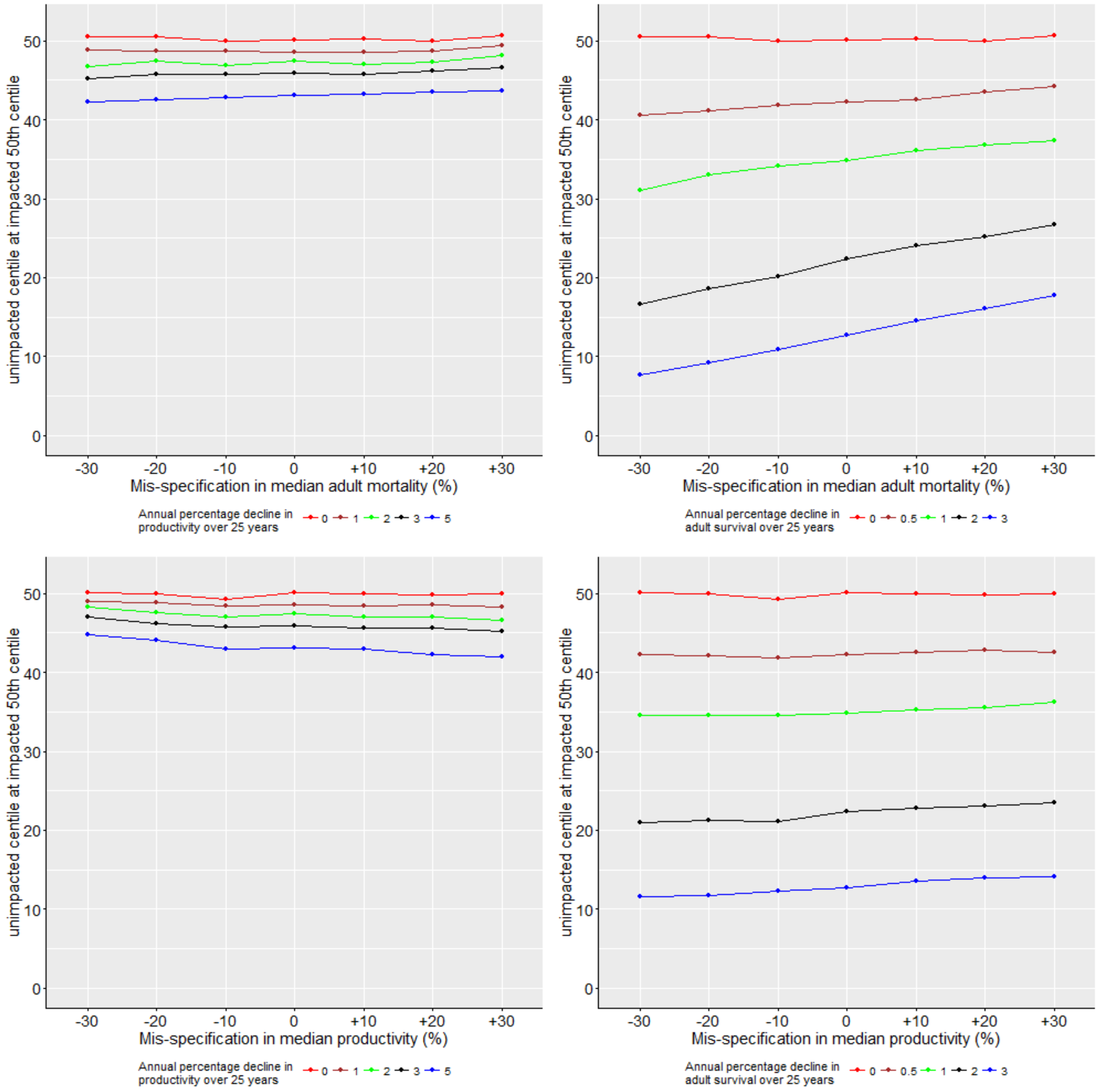


Figure A2.2h. PVA Metric F for St Abb's Kittiwakes – centile from un-impacted population size equal to the 50th centile of the impacted population size, at 2041.



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.

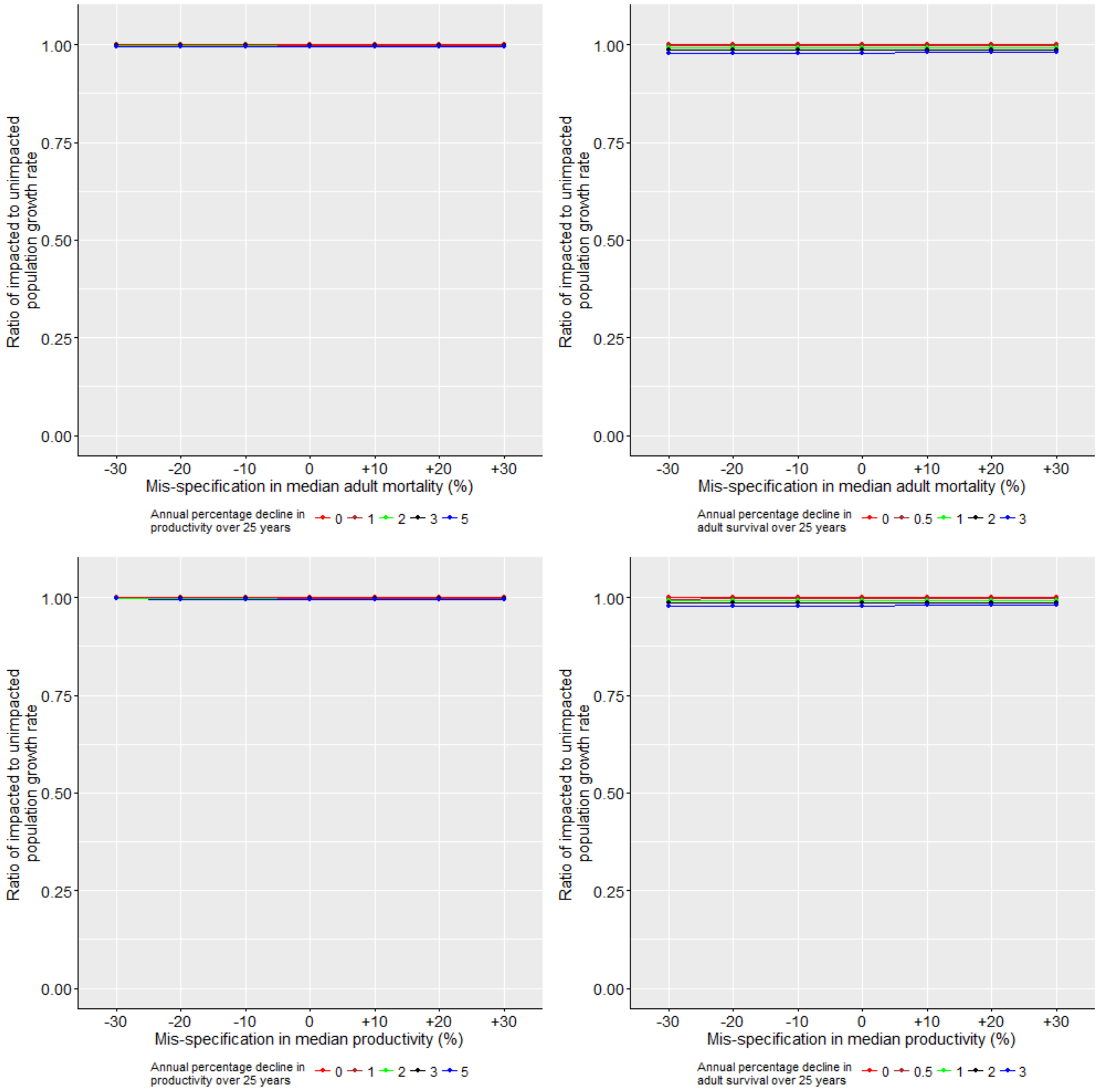


Figure A2.3b. PVA Metric B for Fowlsheugh Kittiwakes – ratio of population size at 2041, comparing impacted population vs. un-impacted population.

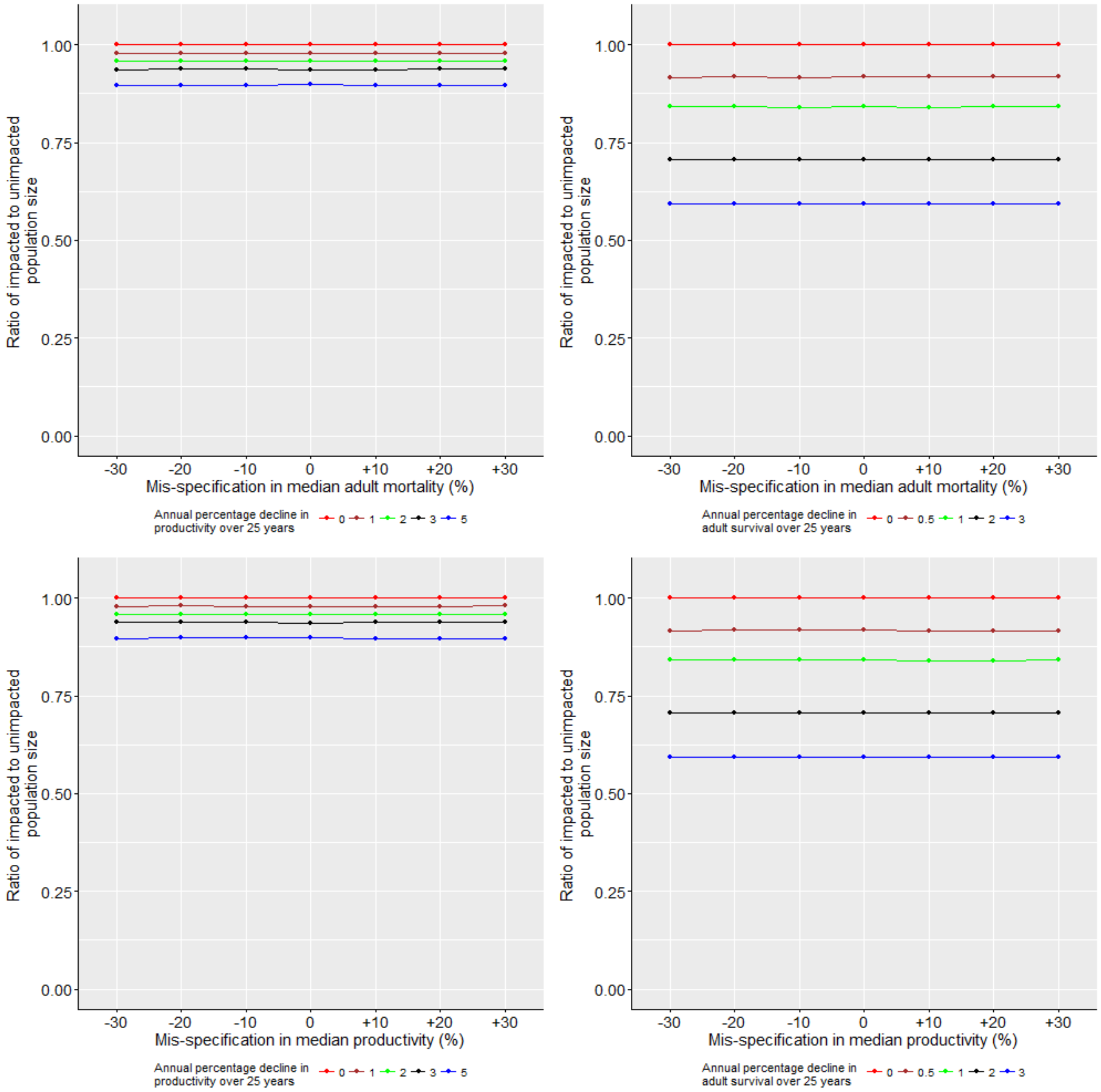


Figure A2.3c. PVA Metric C for Fowlsheugh Kittiwakes – difference in population growth rate from 2016-2041, comparing impacted population vs. un-impacted population.

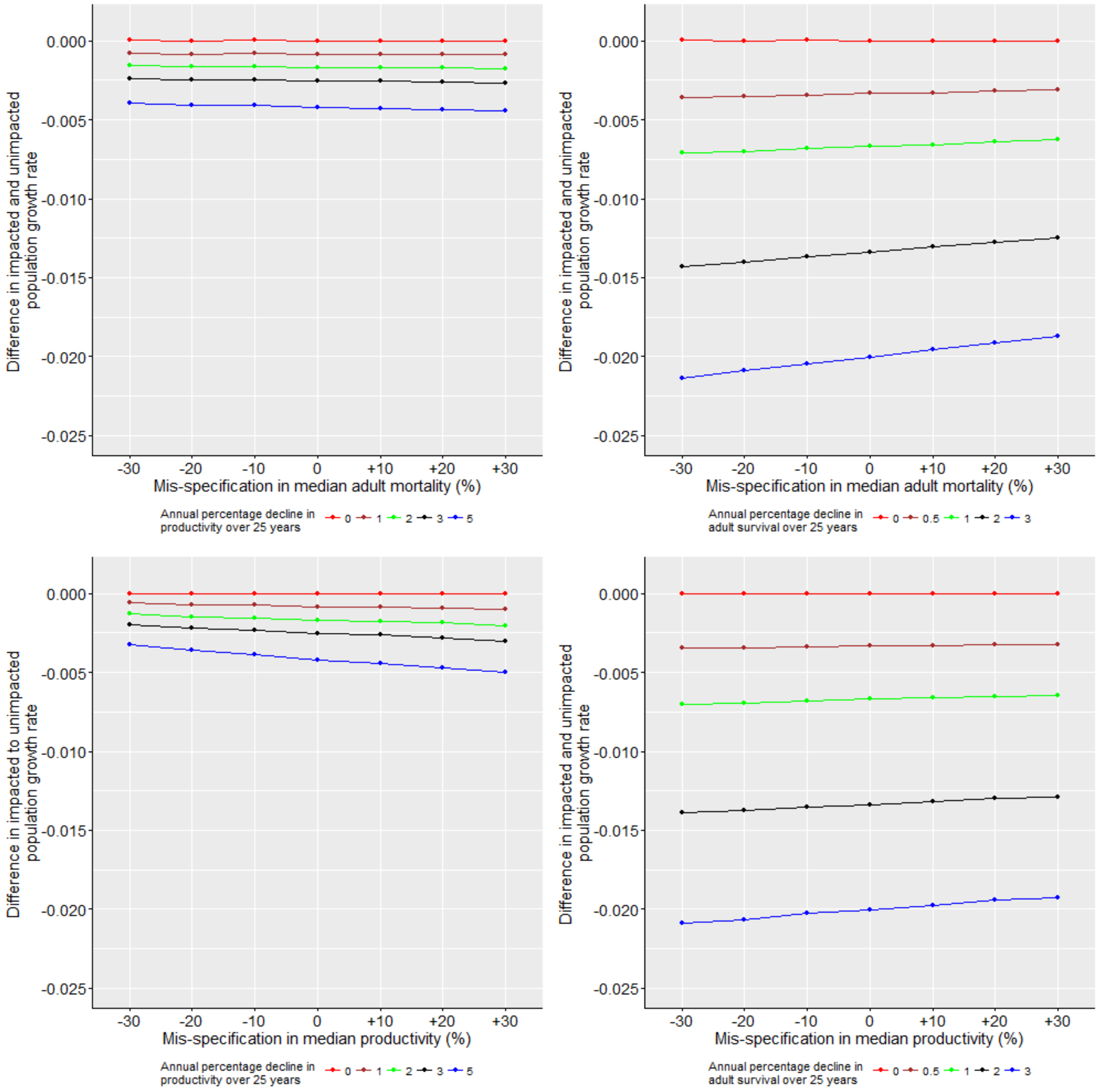


Figure A2.3d. PVA Metric D for Fowlsheugh Kittiwakes – difference in population size at 2041, comparing impacted population vs. un-impacted population.

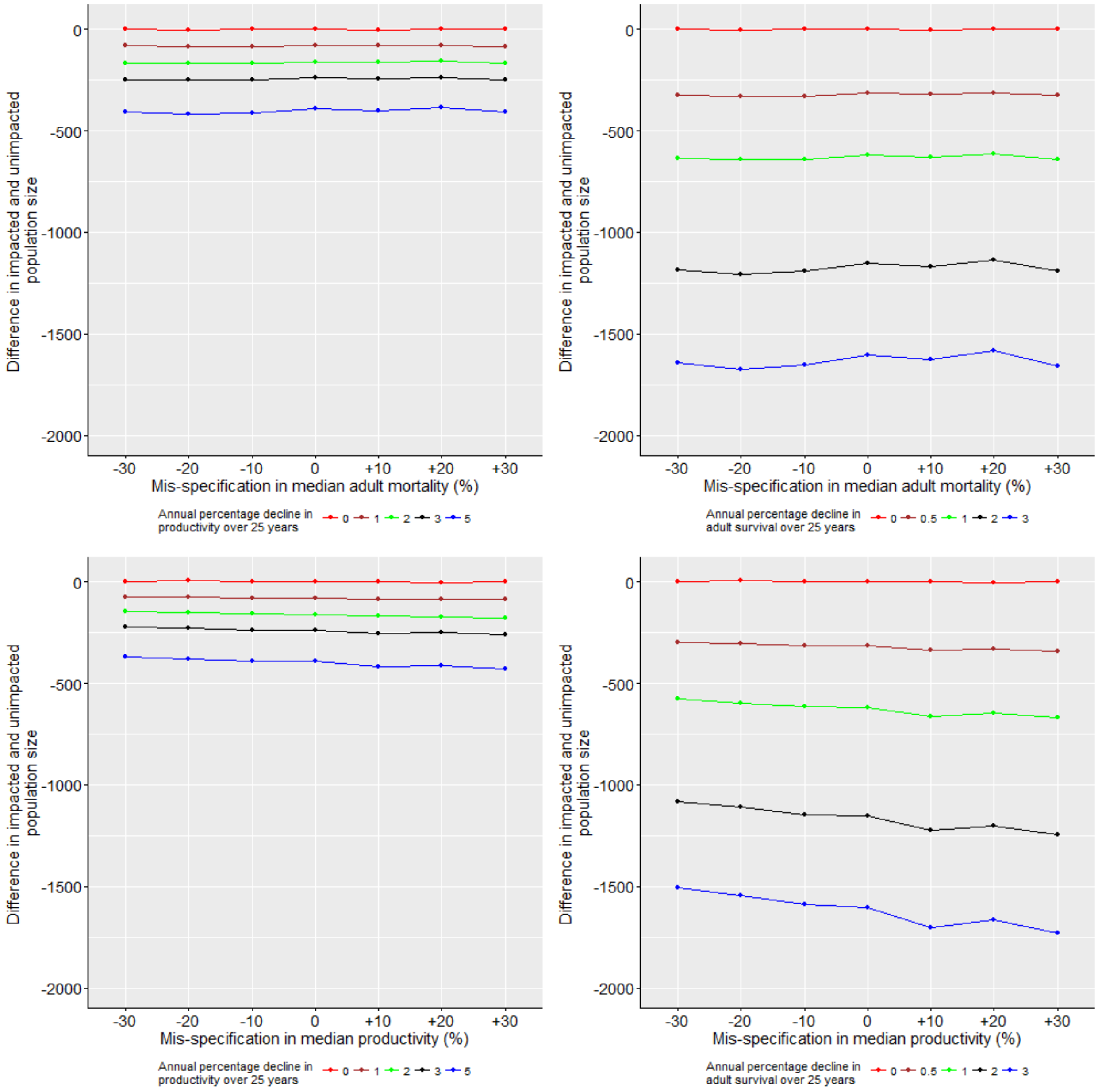


Figure A2.3e. PVA Metric E1 for Fowlsheugh Kittiwakes – probability of population decline greater than 10% from 2016-2041.

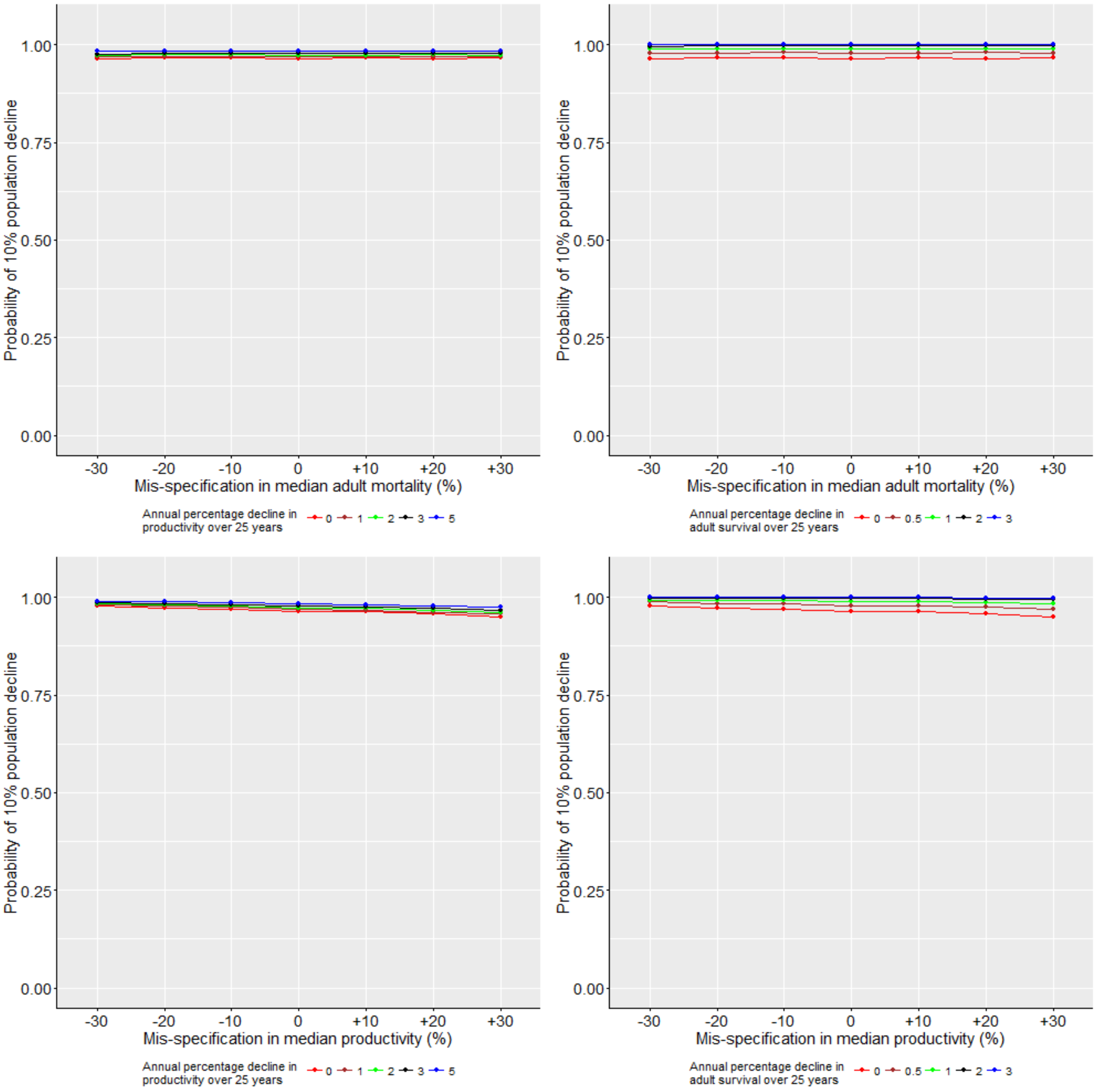


Figure A2.3f. PVA Metric E2 for Fowlsheugh Kittiwakes – probability of population decline greater than 25% from 2016-2041.

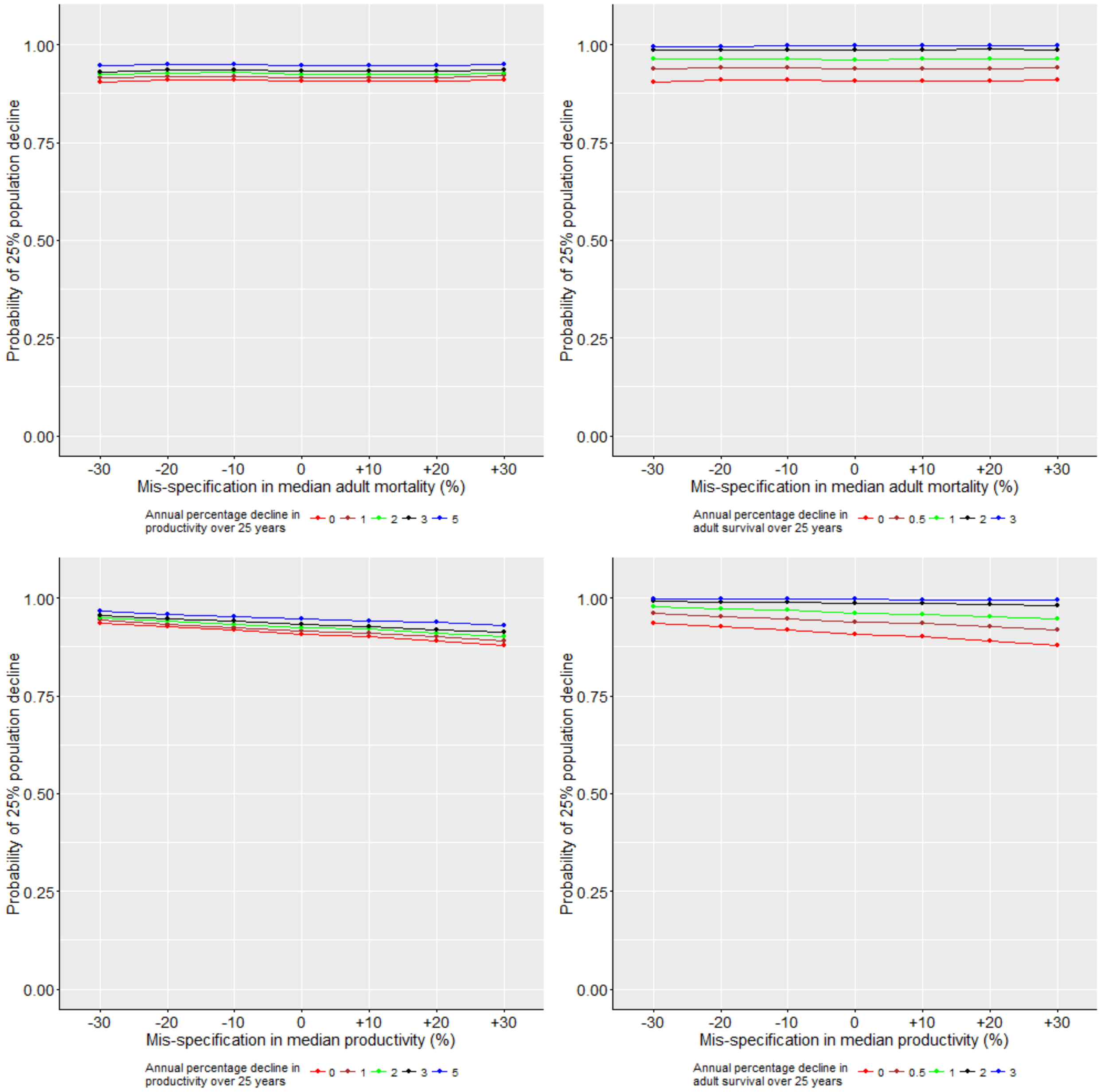


Figure A2.3g. PVA Metric E3 for Fowlsheugh Kittiwakes – probability of population decline greater than 50% from 2016-2041.

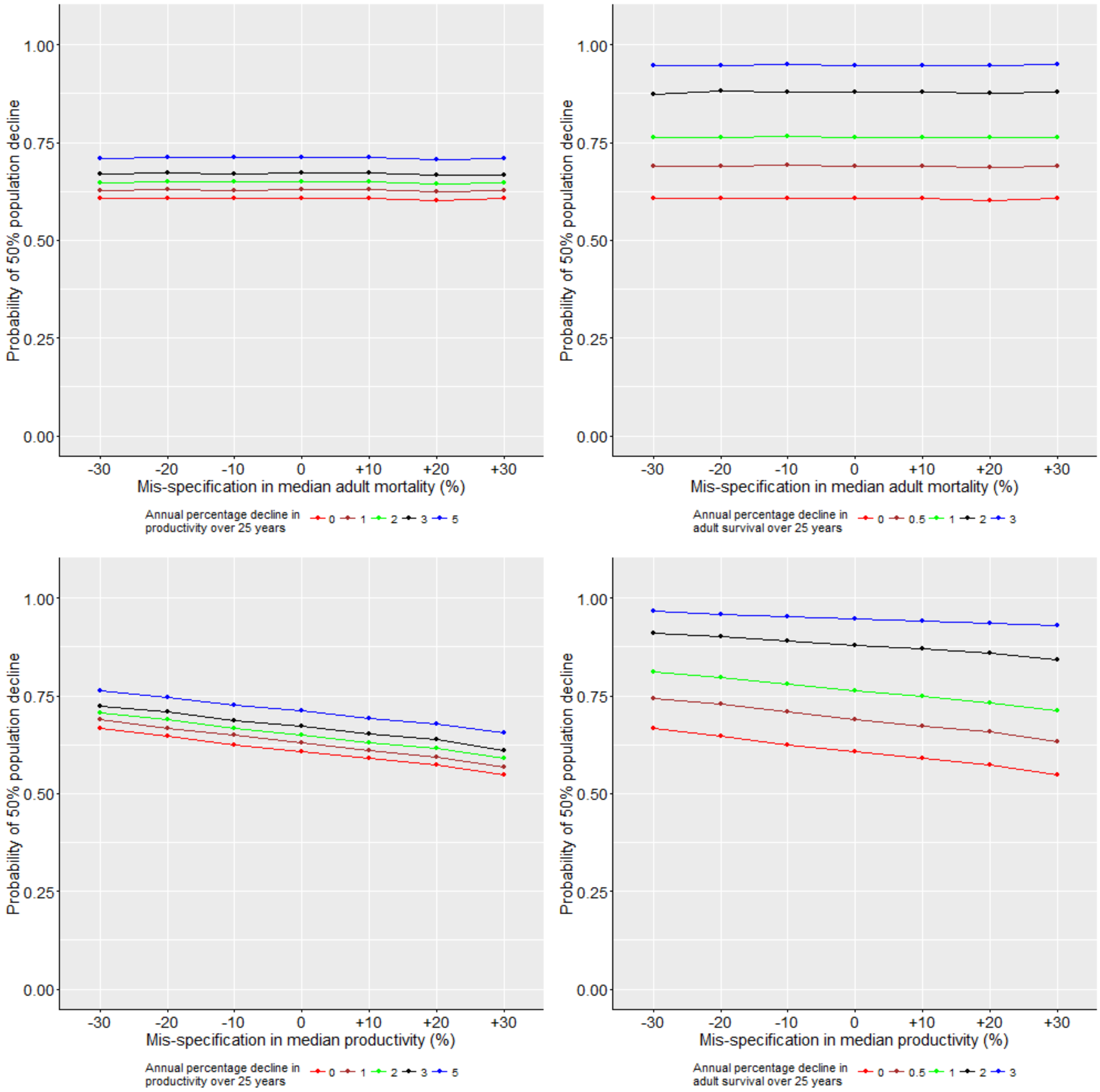
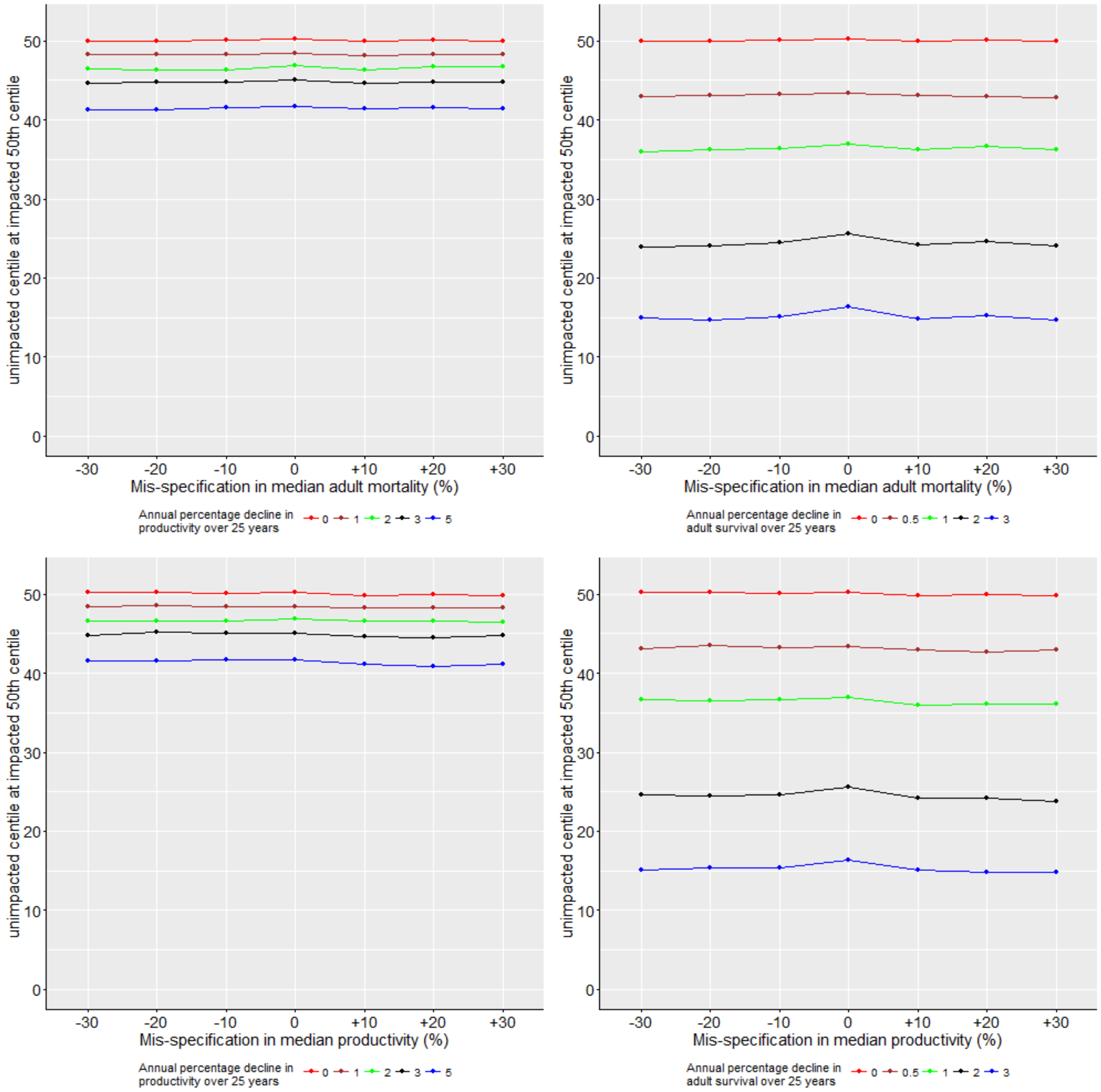


Figure A2.3h. PVA Metric F for Fowlsheugh Kittiwakes – centile from un-impacted population size equal to the 50th centile of the impacted population size, at 2041.



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.

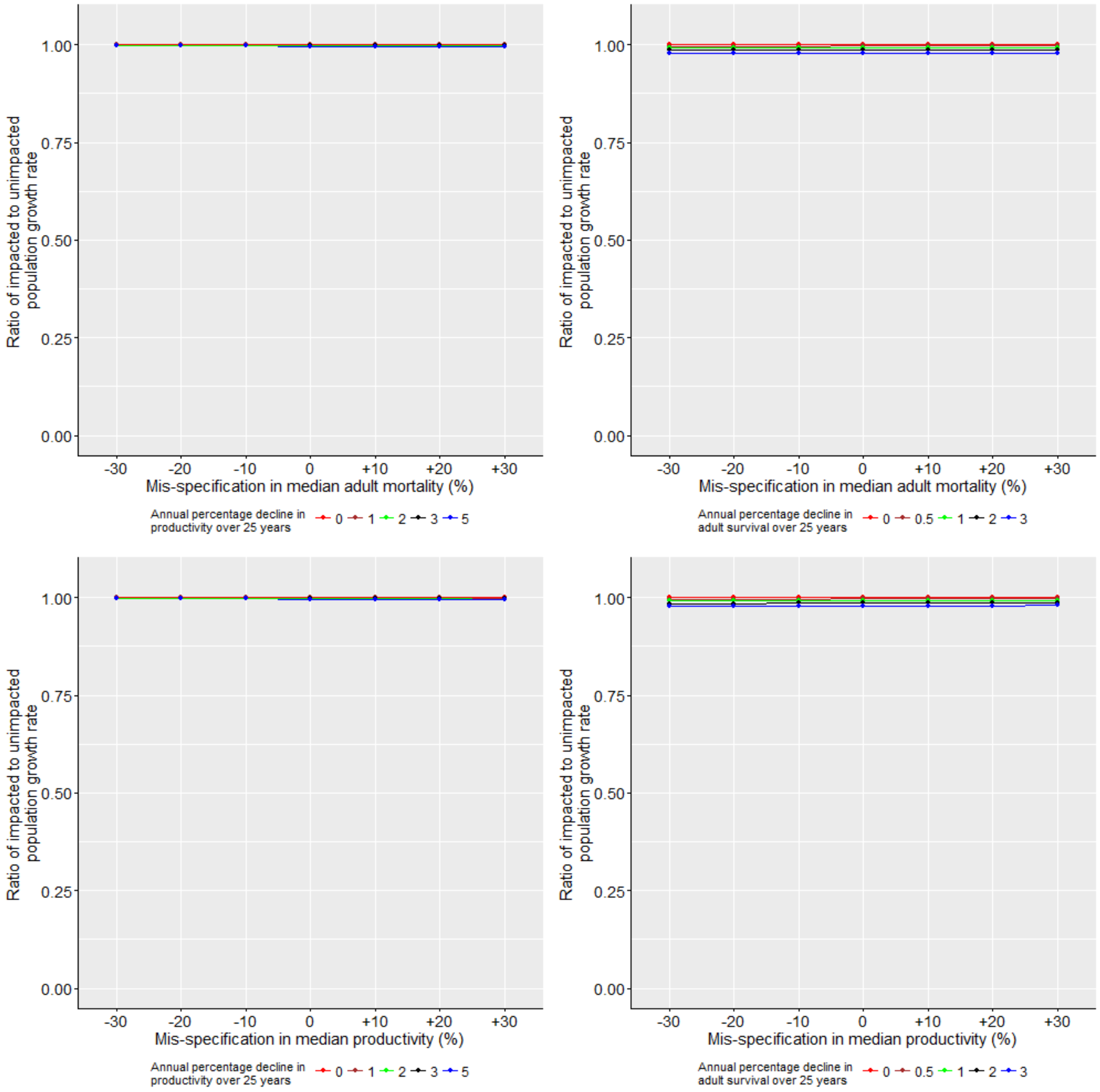


Figure A2.4b. PVA Metric B for Buchan Ness Kittiwakes – ratio of population size at 2041, comparing impacted population vs. un-impacted population.

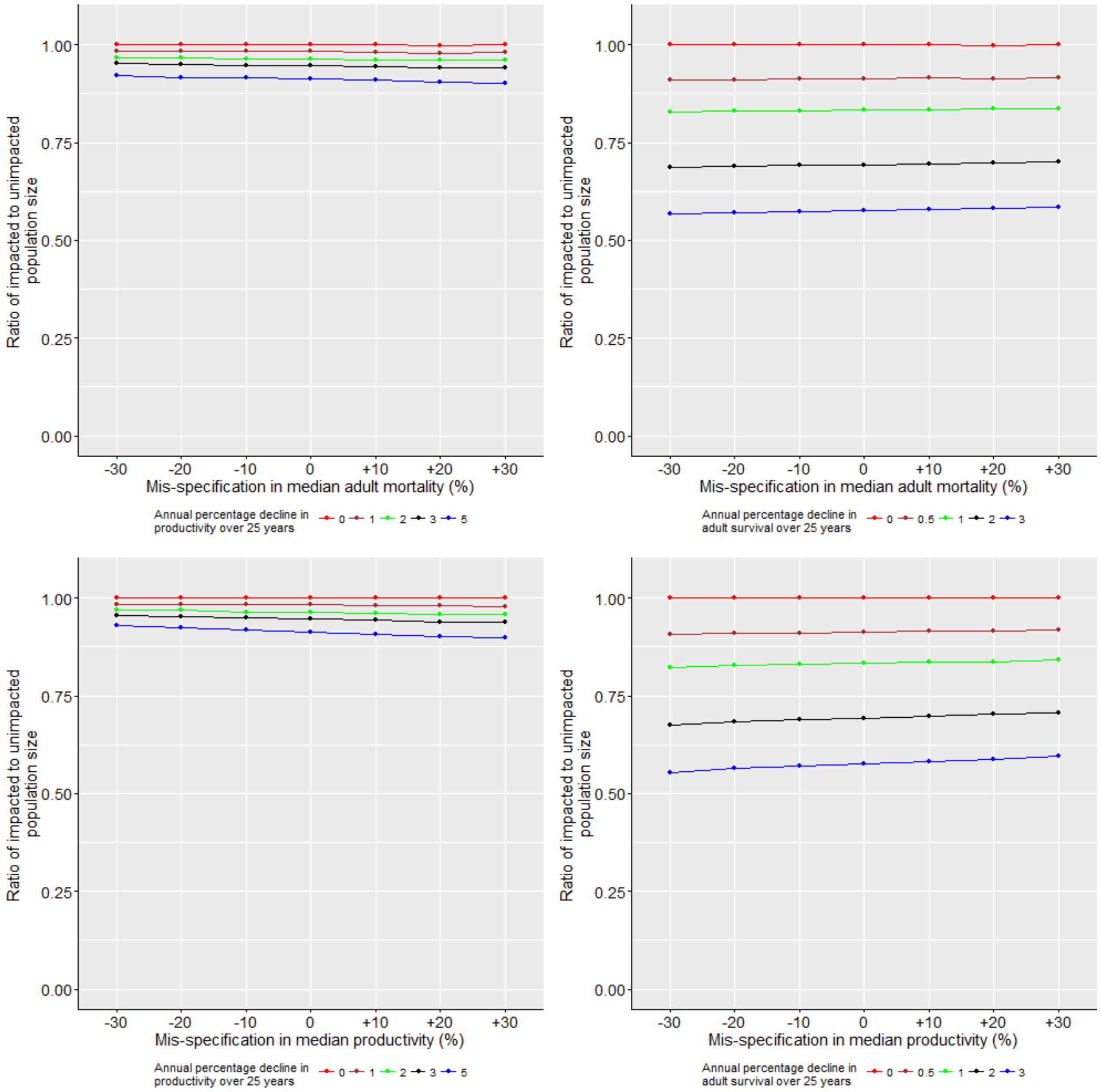


Figure A2.4c. PVA Metric C for Buchan Ness Kittiwakes – difference in population growth rate from 2016-2041, comparing impacted population vs. un-impacted population.

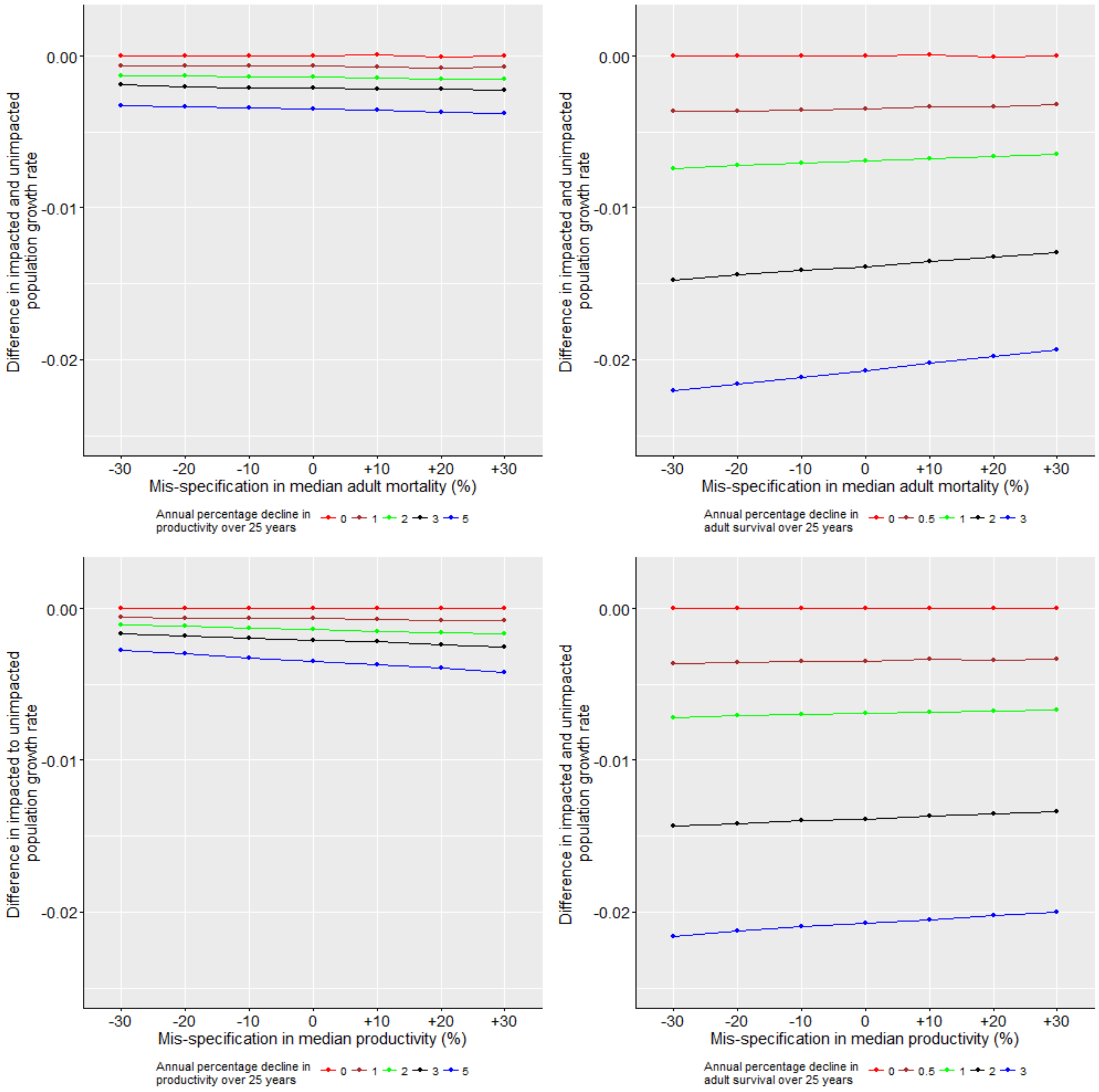


Figure A2.4d. PVA Metric D for Buchan Ness Kittiwakes – difference in population size at 2041, comparing impacted population vs. un-impacted population.

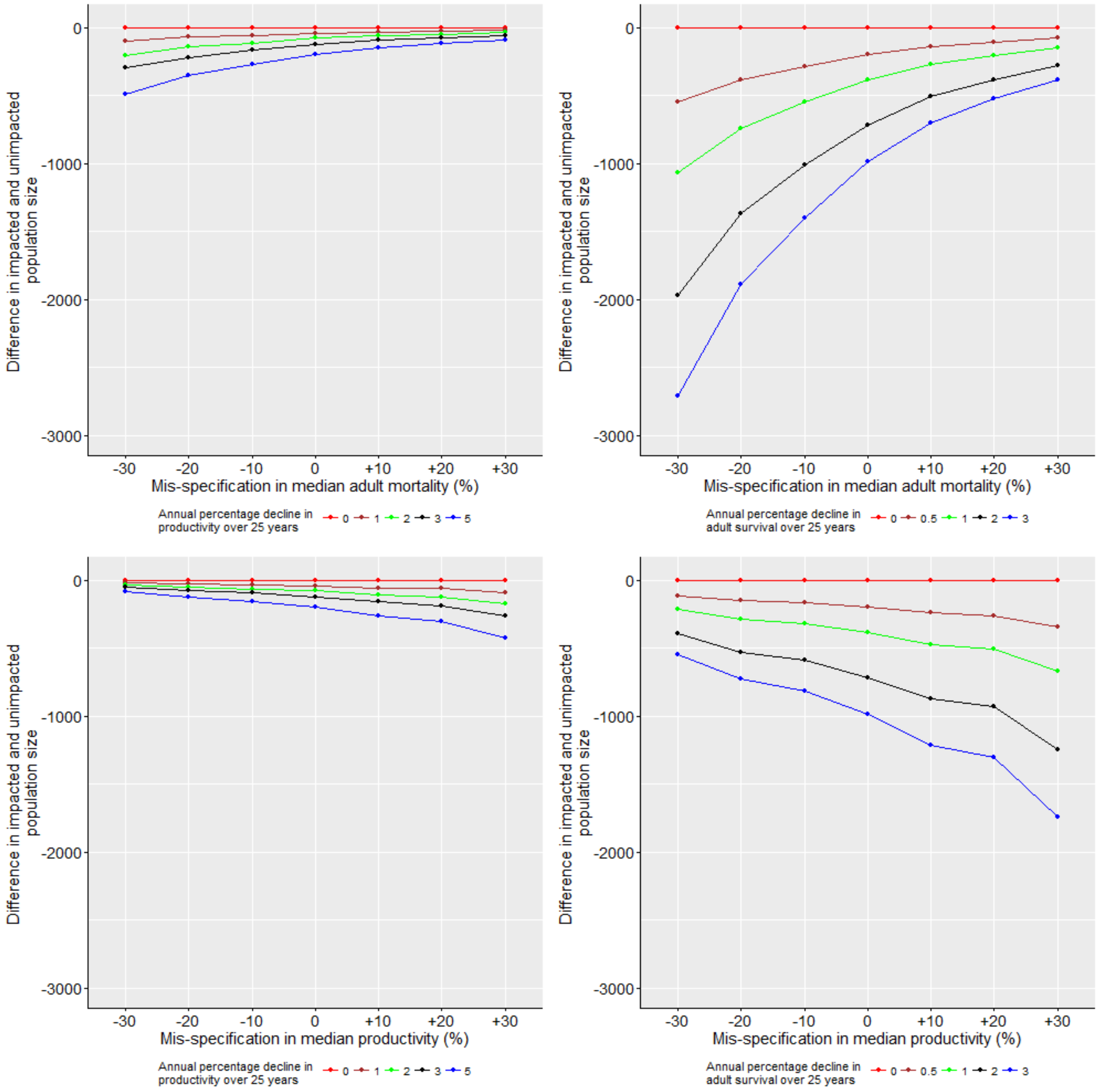


Figure A2.4e. PVA Metric E1 for Buchan Ness Kittiwakes – probability of population decline greater than 10% from 2016-2041.

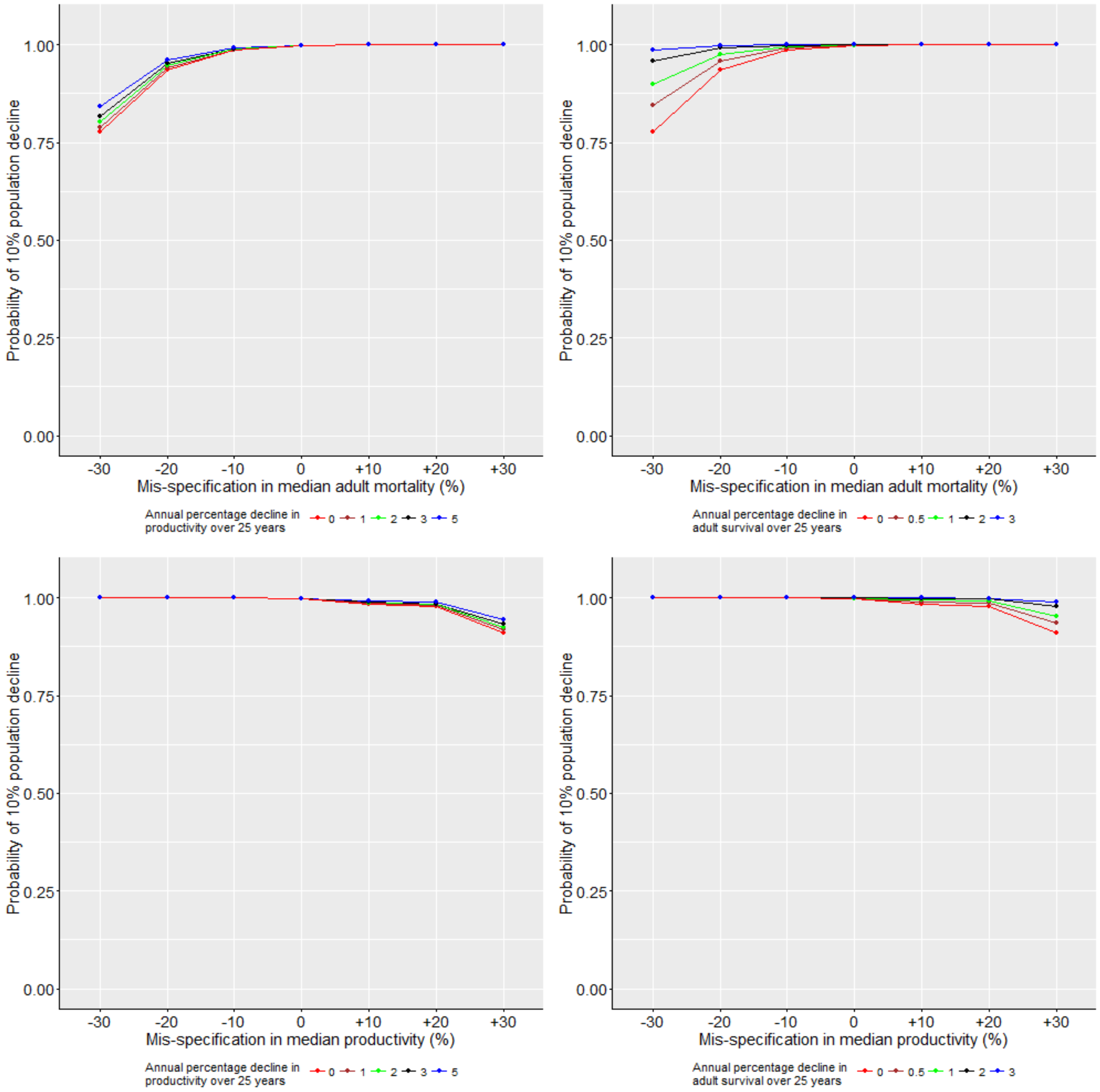


Figure A2.4f. PVA Metric E2 for Buchan Ness Kittiwakes – probability of population decline greater than 25% from 2016-2041.

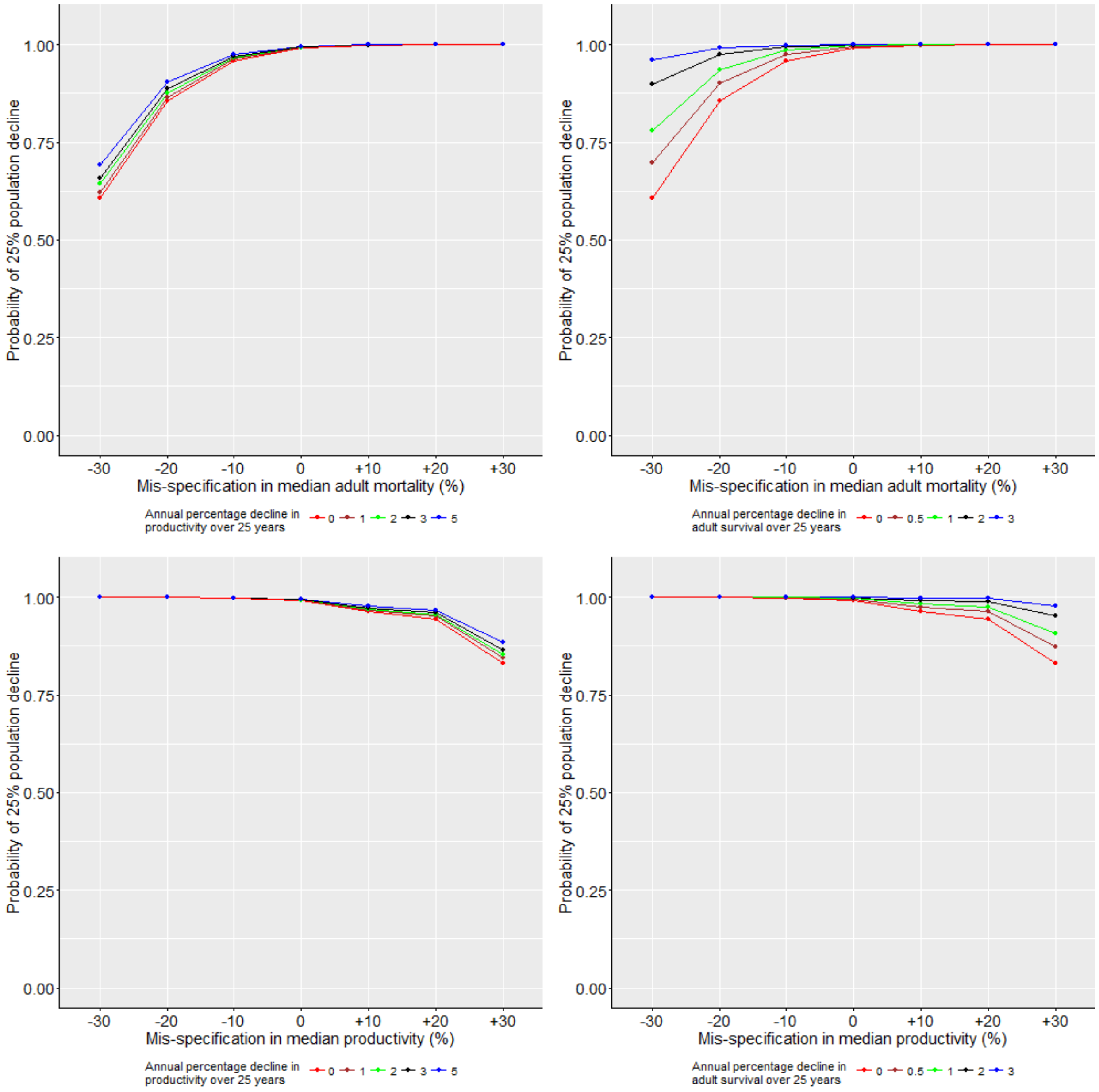


Figure A2.4g. PVA Metric E3 for Buchan Ness Kittiwakes – probability of population decline greater than 50% from 2016-2041.

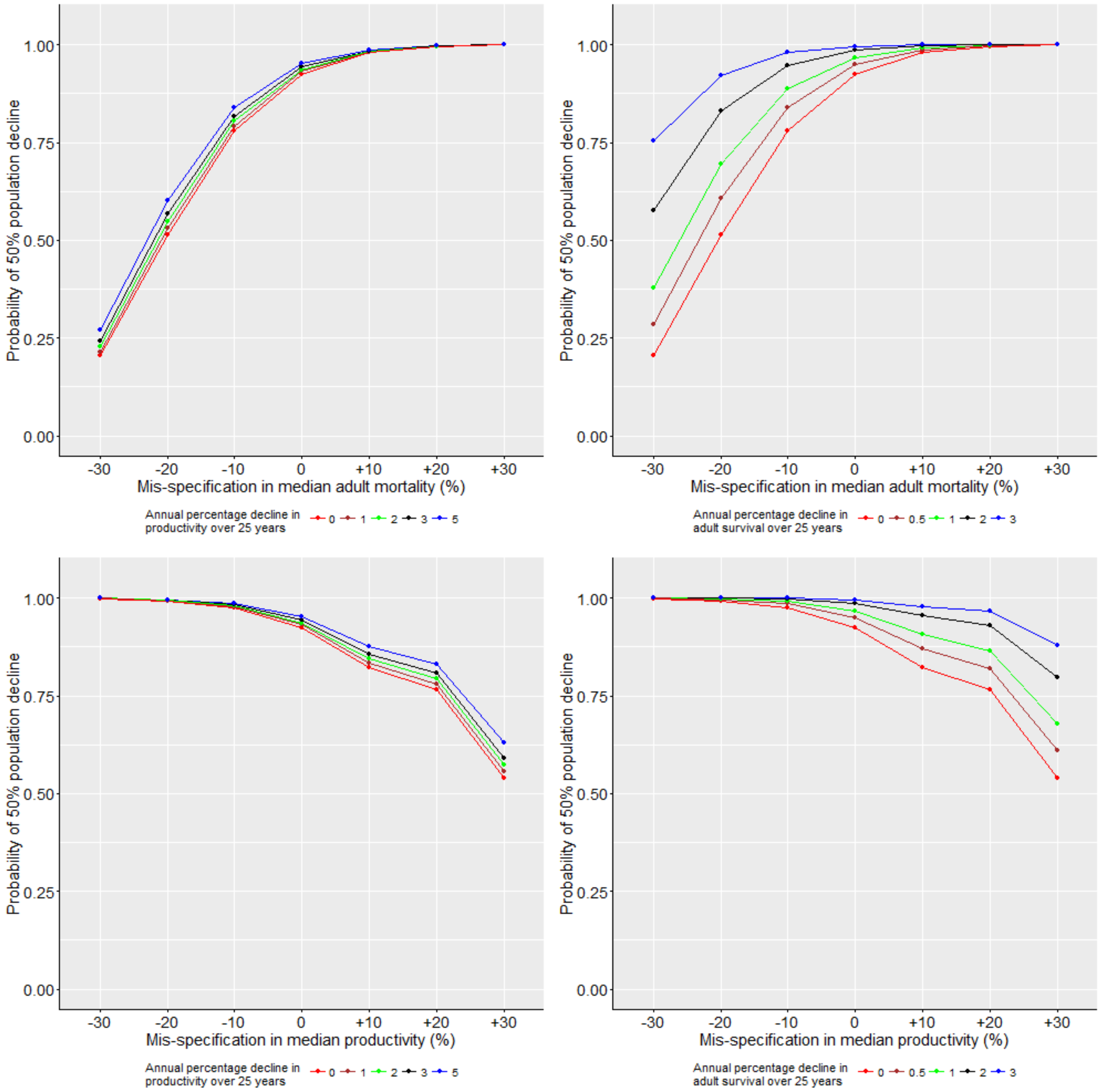
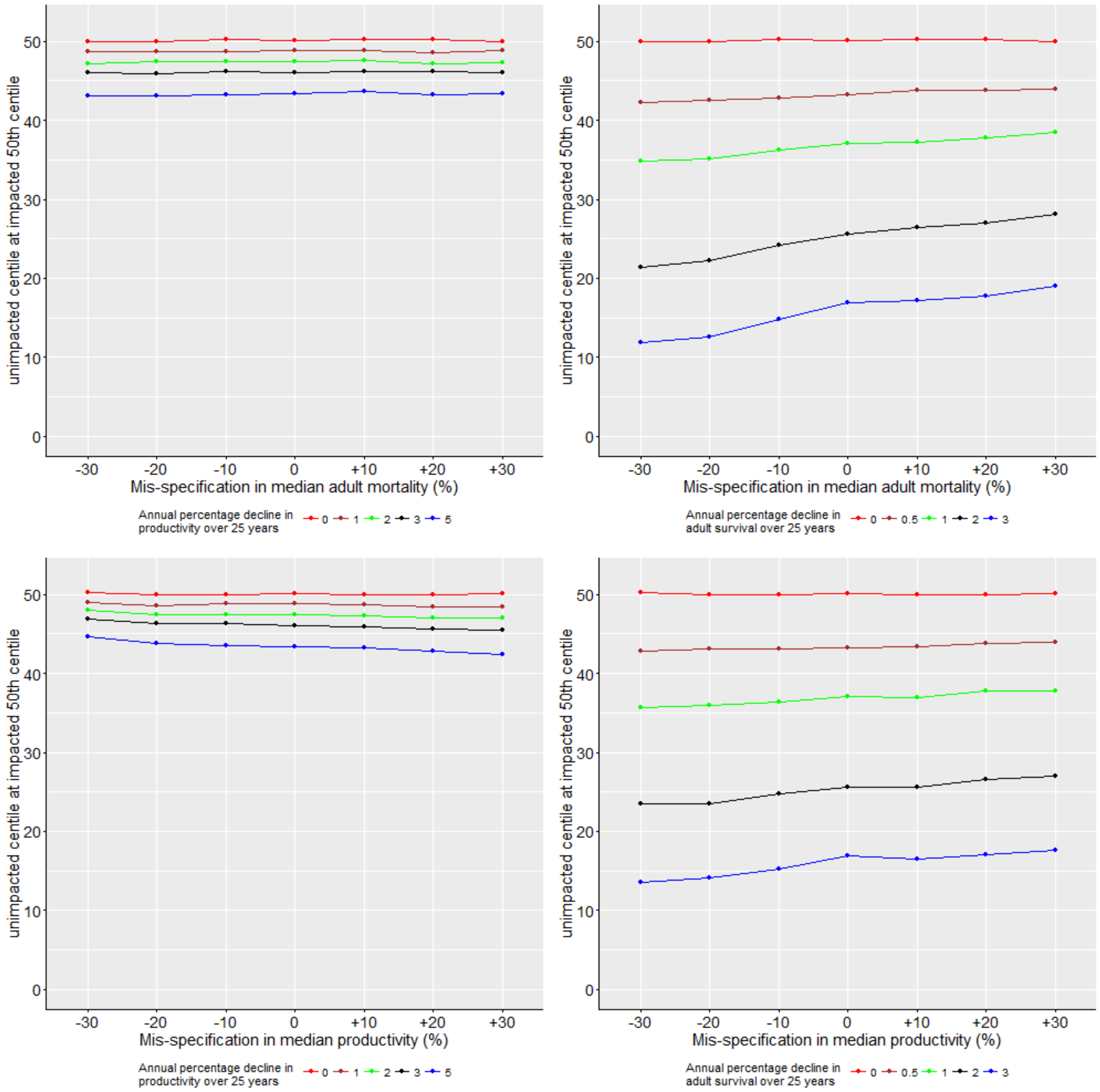


Figure A2.4h. PVA Metric F for Buchan Ness Kittiwakes – centile from un-impacted population size equal to the 50th centile of the impacted population size, at 2041.



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.

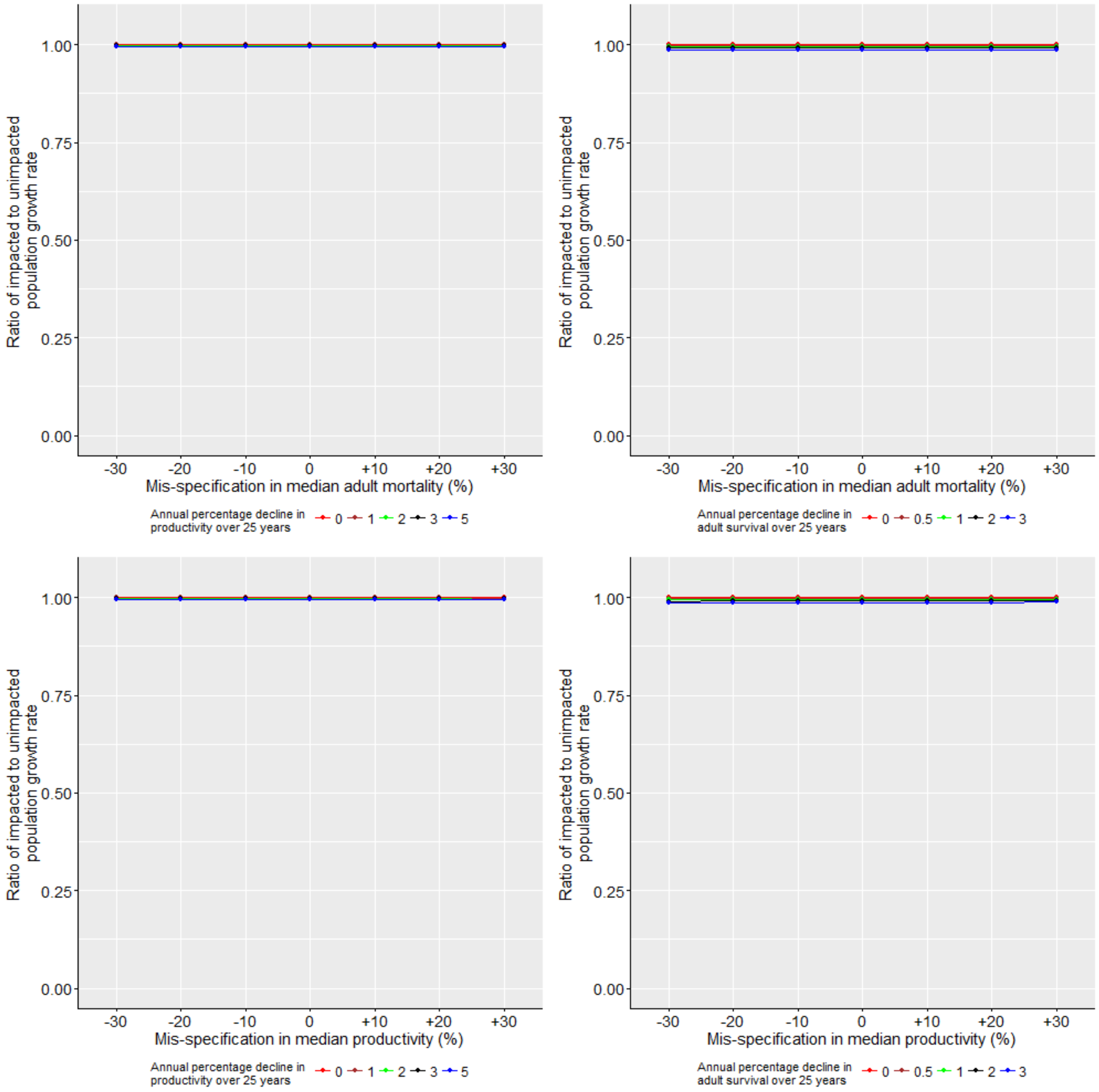


Figure A2.5b. PVA Metric B for Forth Guillemots – ratio of population size at 2041, comparing impacted population vs. un-impacted population.

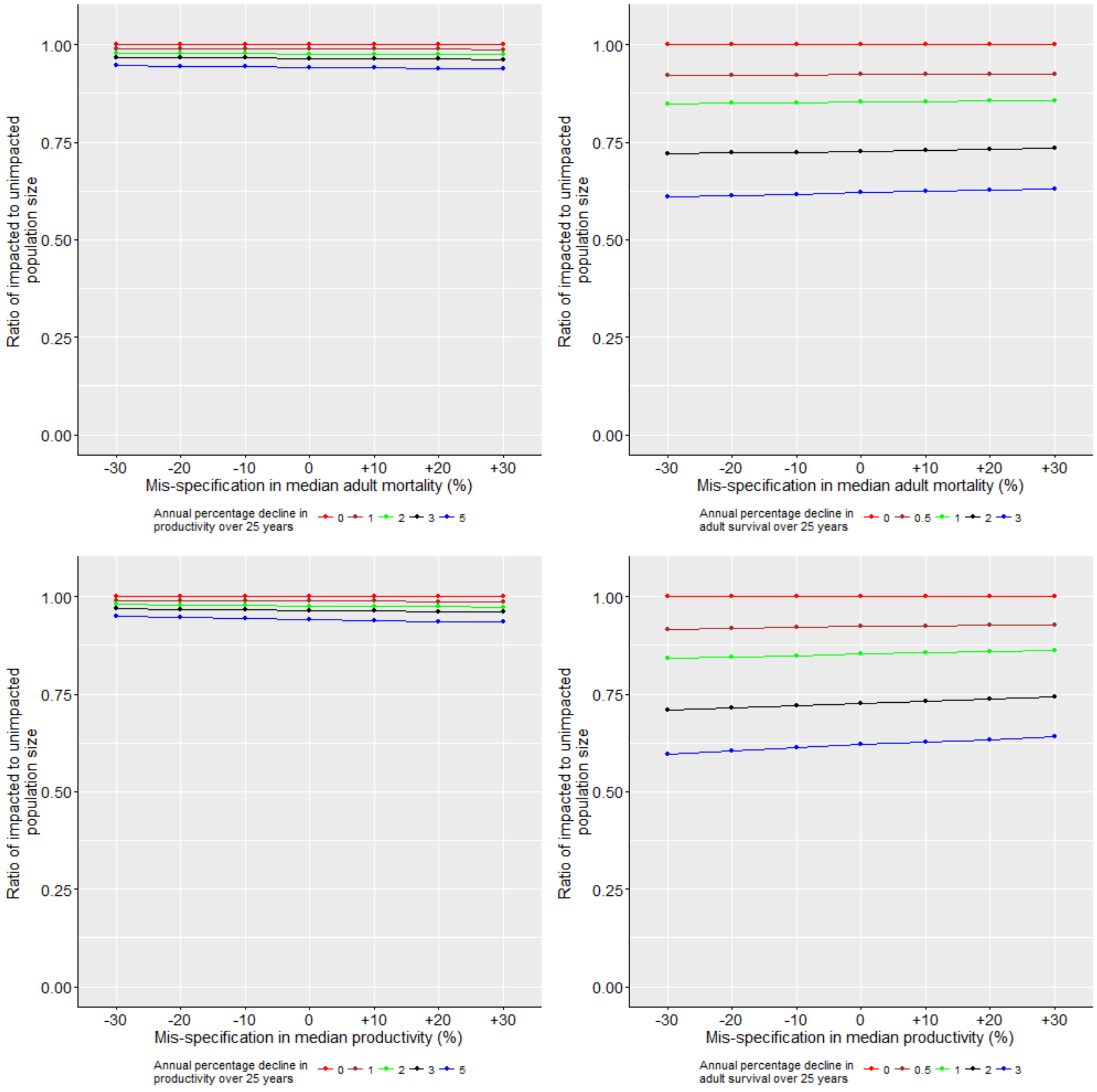


Figure A2.5c. PVA Metric C for Forth Guillemots – difference in population growth rate from 2016-2041, comparing impacted population vs. un-impacted population.

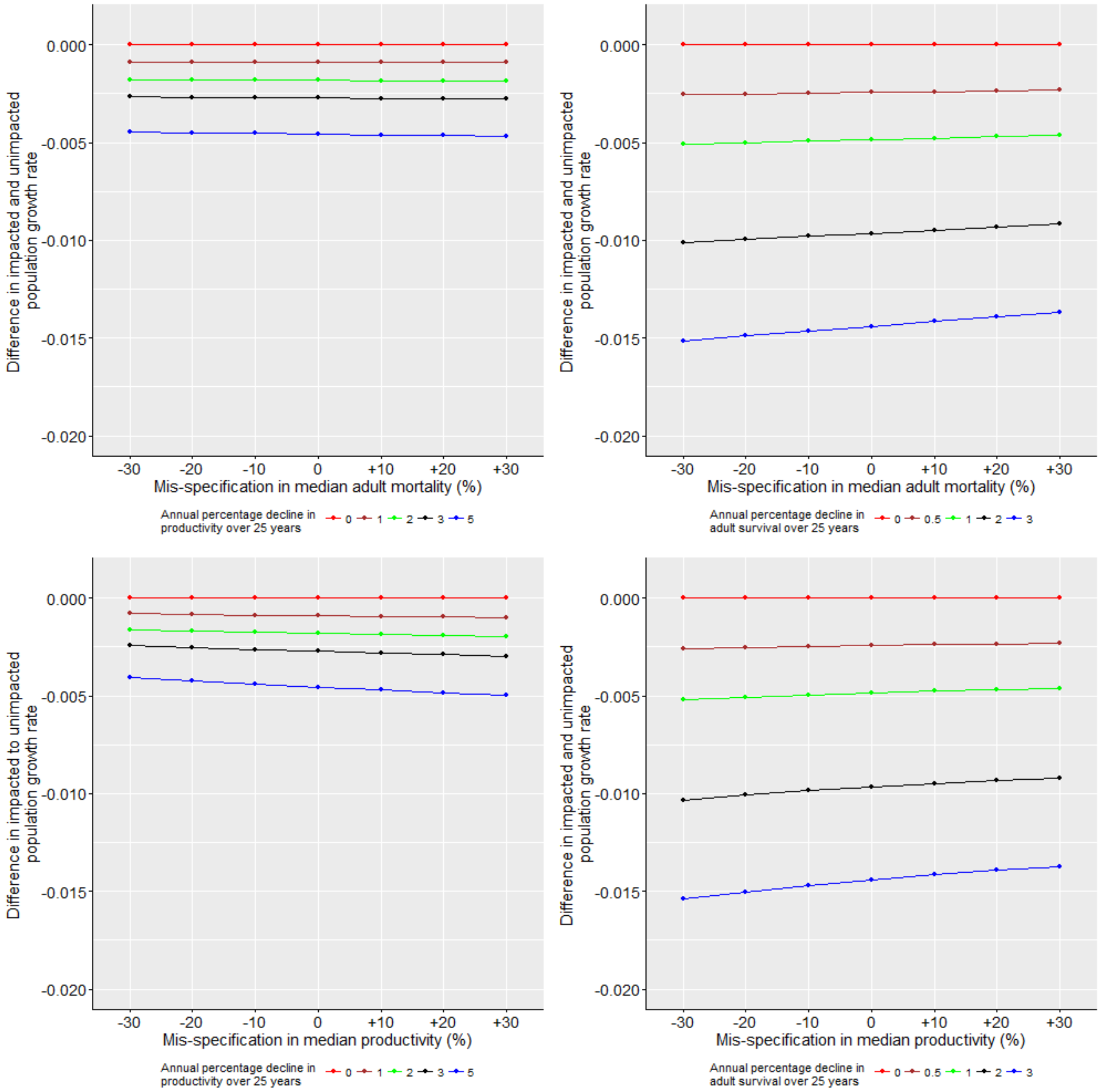


Figure A2.5d. PVA Metric D for Forth Guillemots – difference in population size at 2041, comparing impacted population vs. un-impacted population.

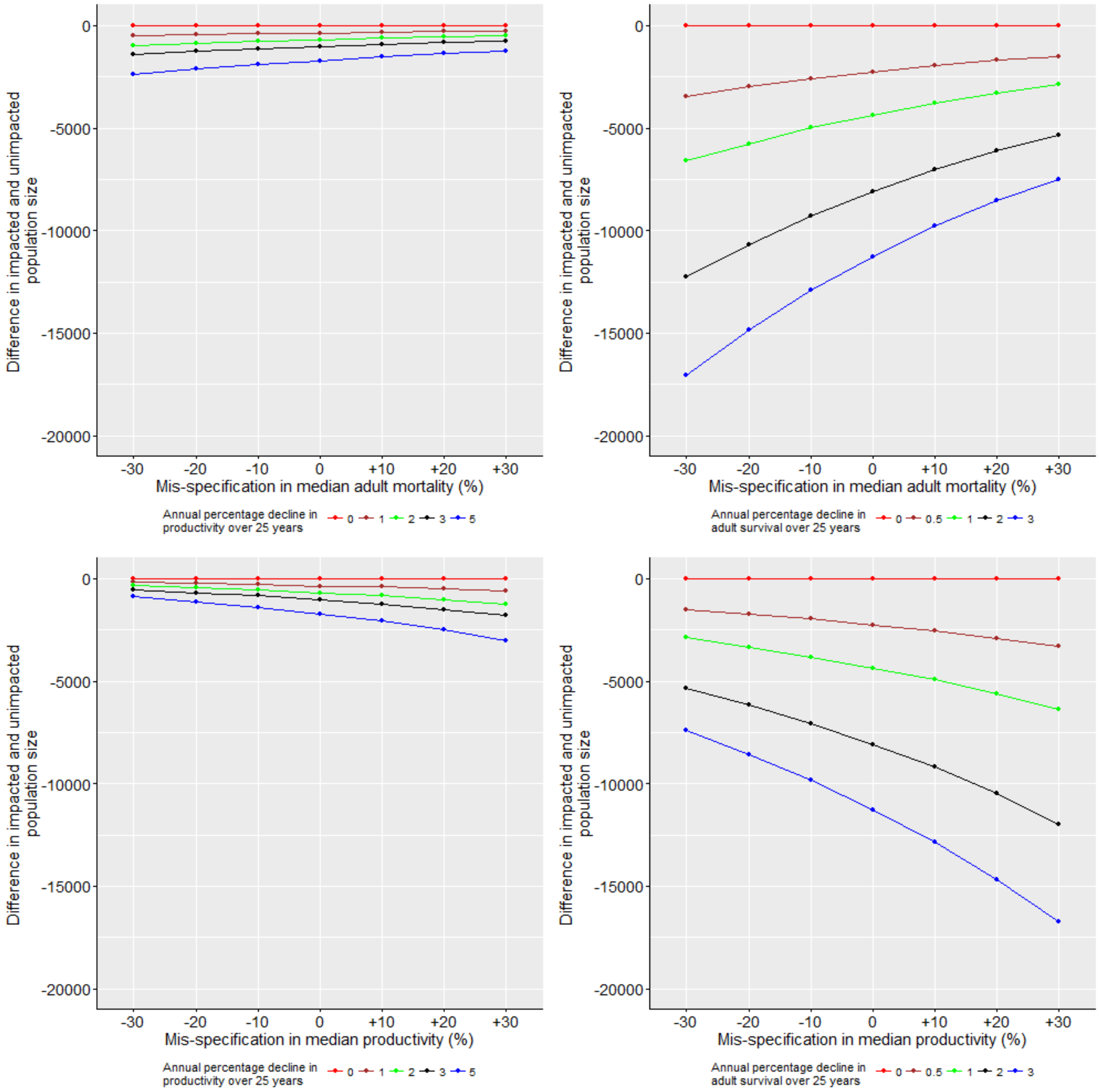


Figure A2.5e. PVA Metric E1 for Forth Guillemots – probability of population decline greater than 10% from 2016-2041.

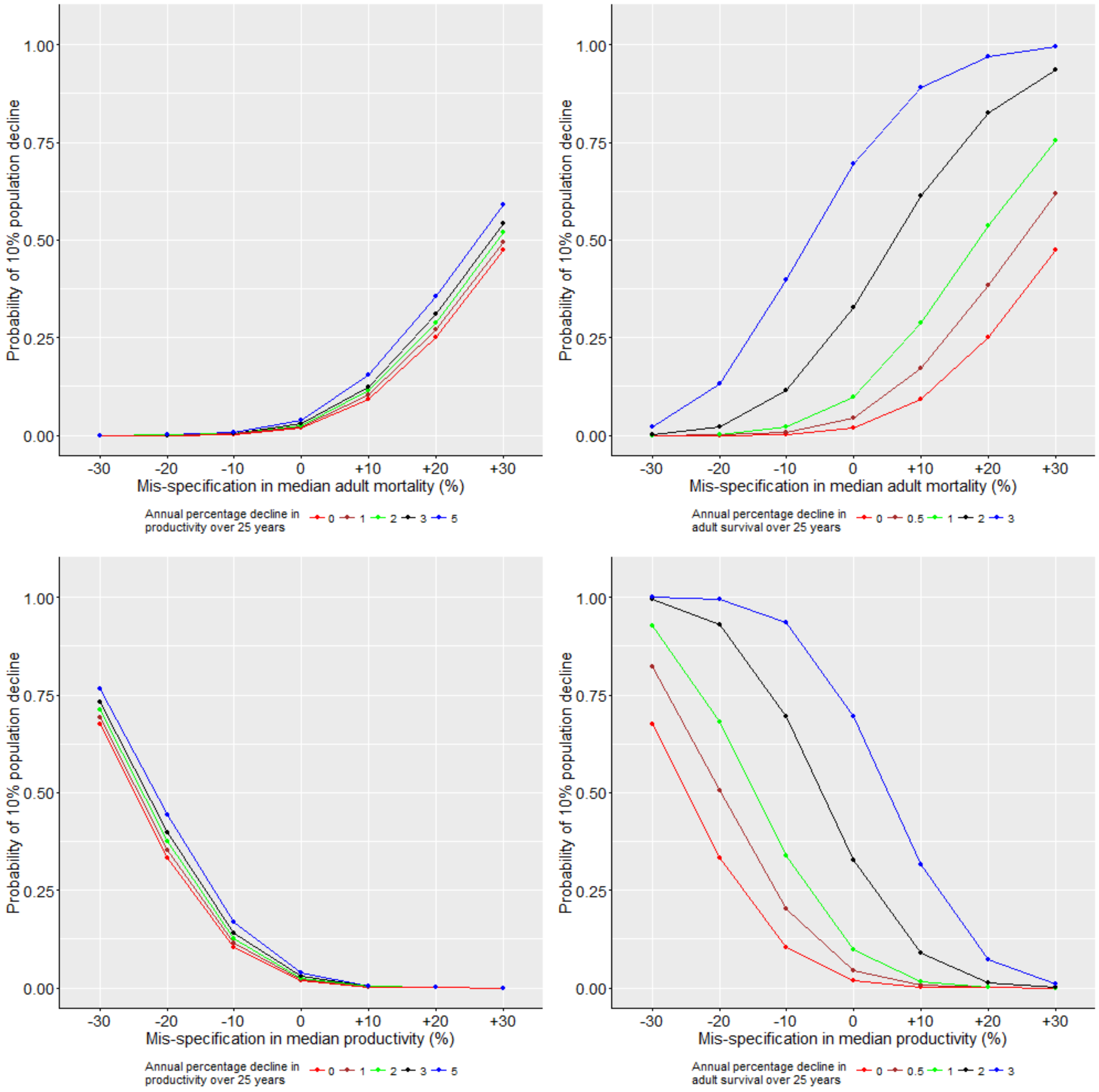


Figure A2.5f. PVA Metric E2 for Forth Guillemots – probability of population decline greater than 25% from 2016-2041.

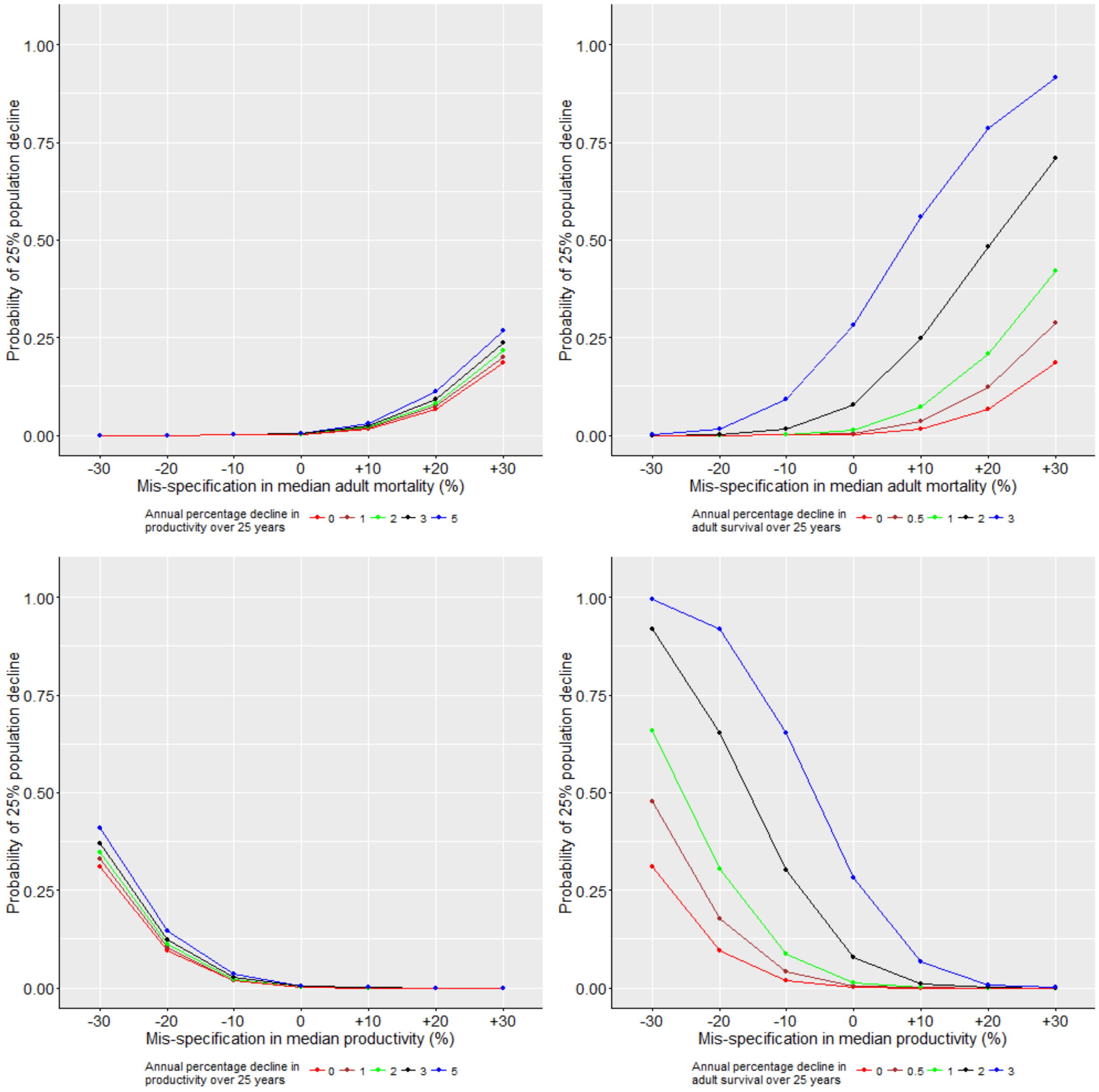


Figure A2.5g. PVA Metric E3 for Forth Guillemots – probability of population decline greater than 50% from 2016-2041.

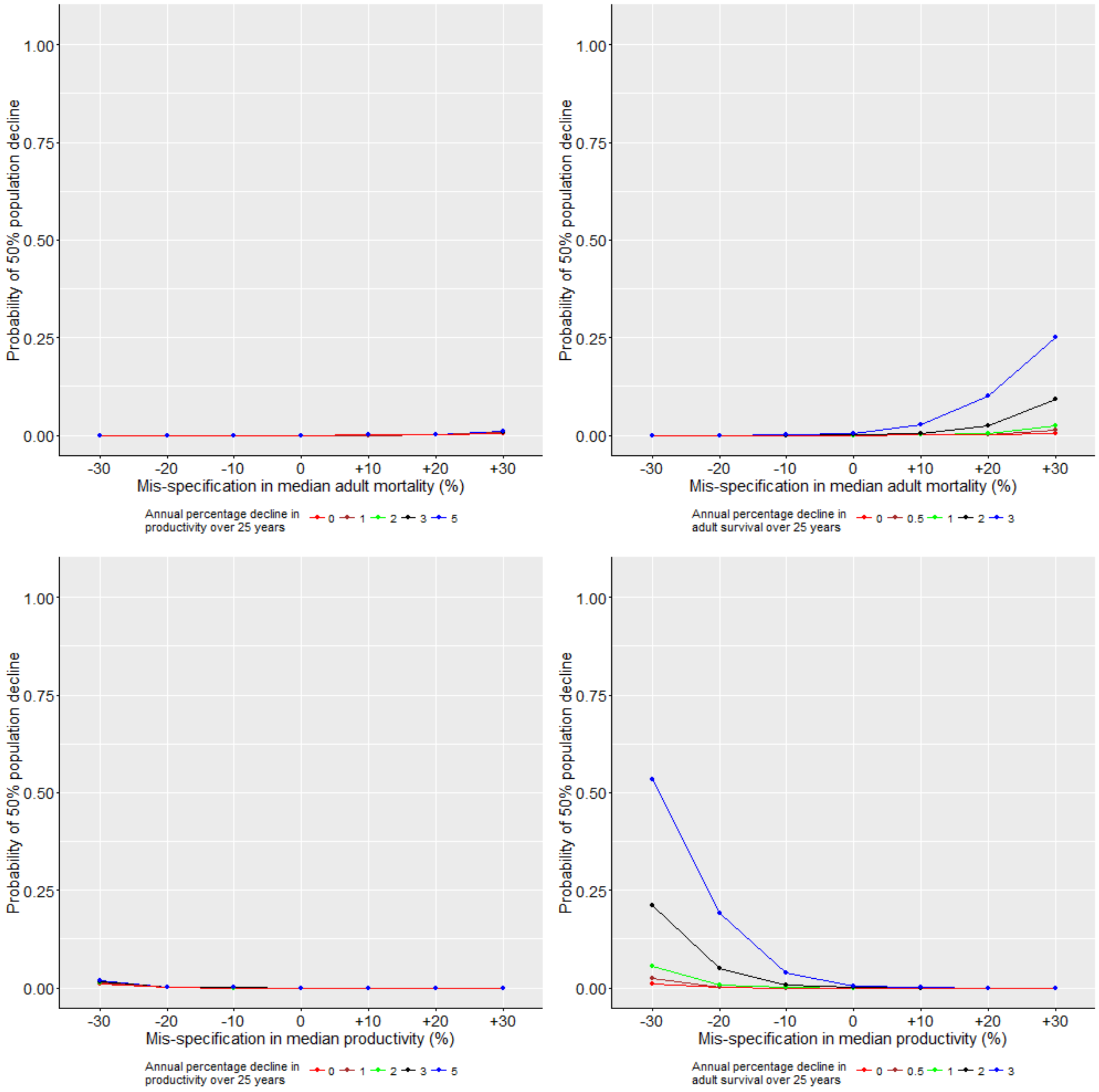
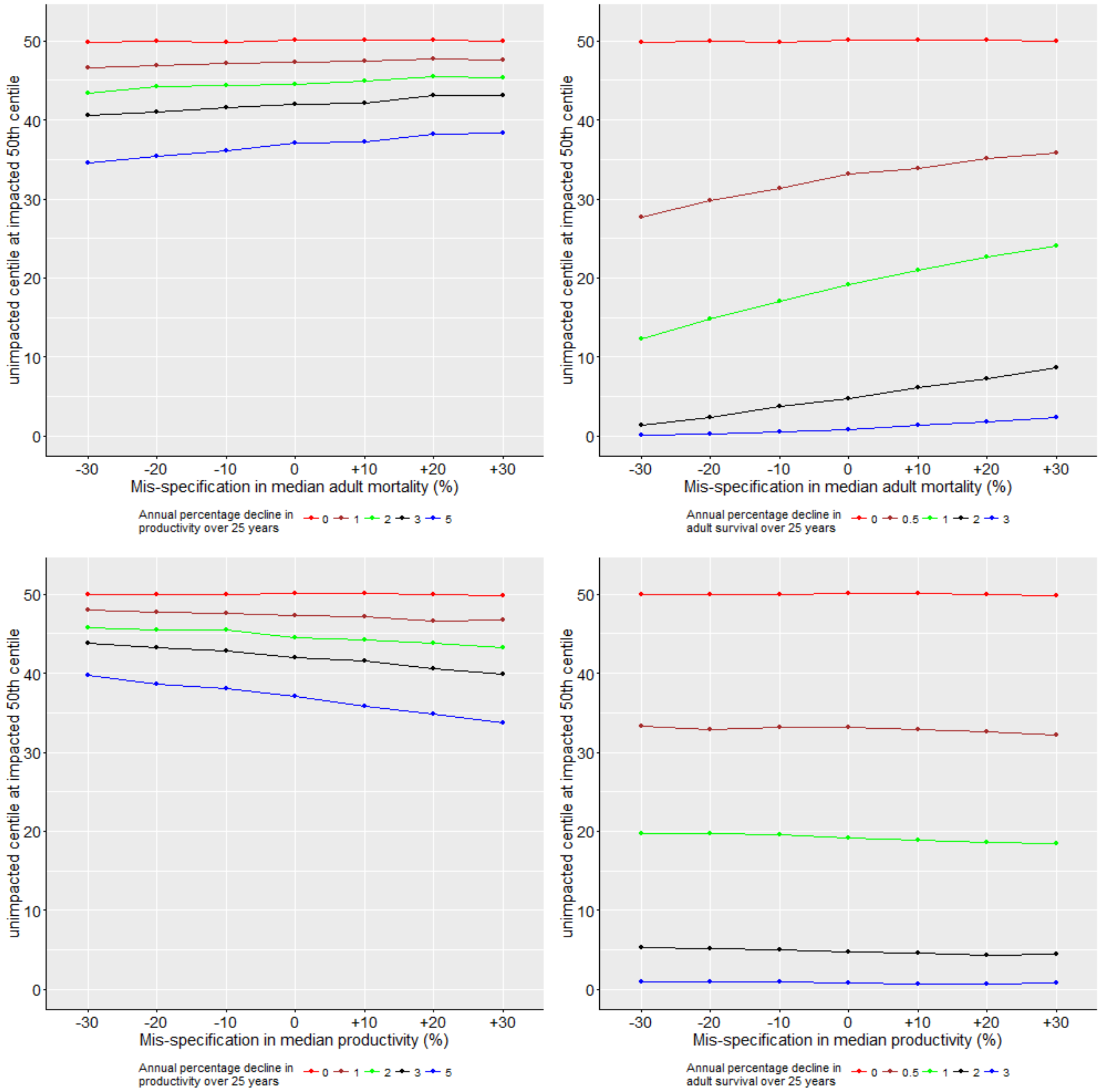


Figure A2.5h. PVA Metric F for Forth Guillemots – centile from un-impacted population size equal to the 50th centile of the impacted population size, at 2041.



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.

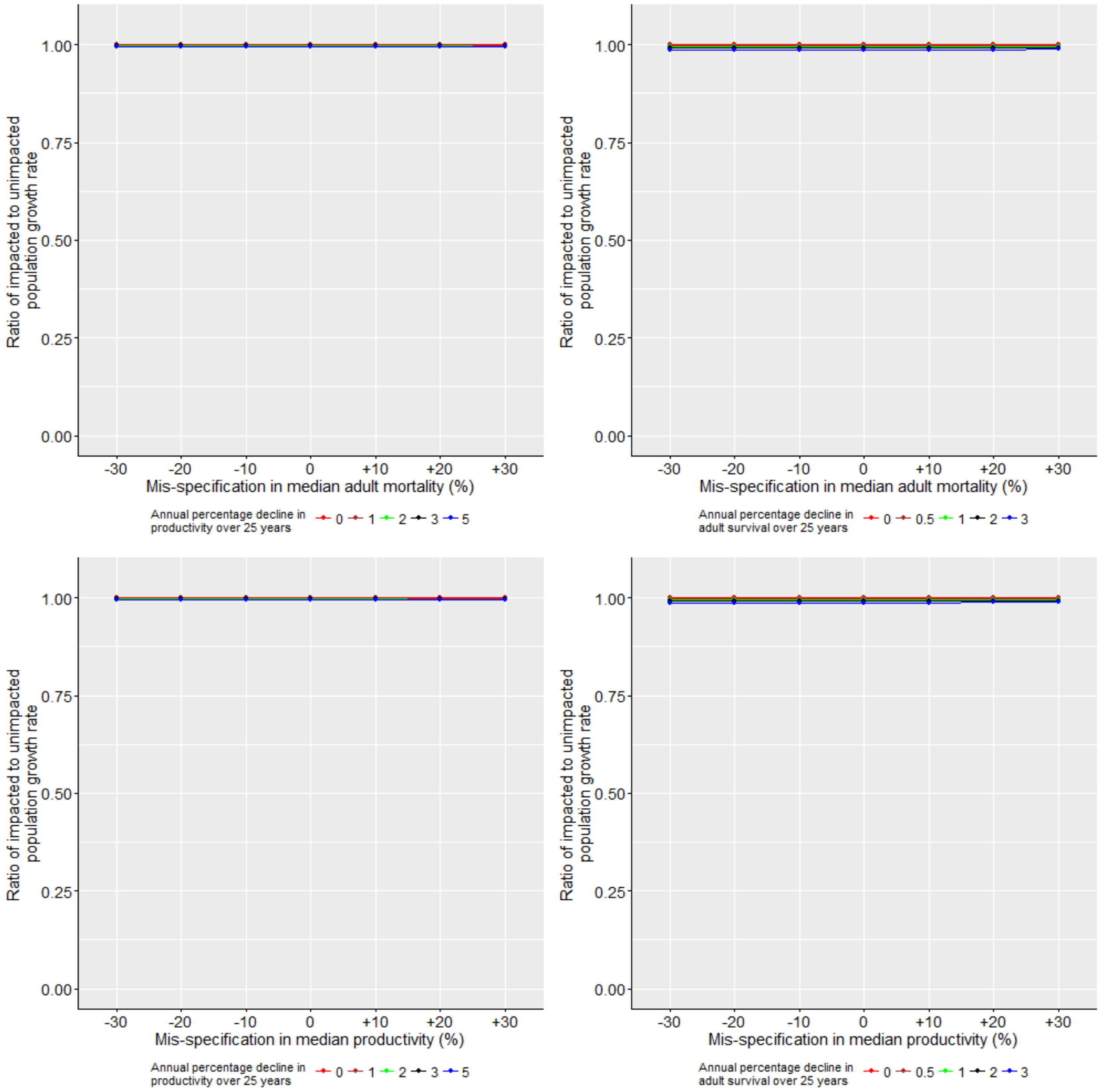


Figure A2.6b. PVA Metric B for St Abb’s Guillemots – ratio of population size at 2041, comparing impacted population vs. un-impacted population.

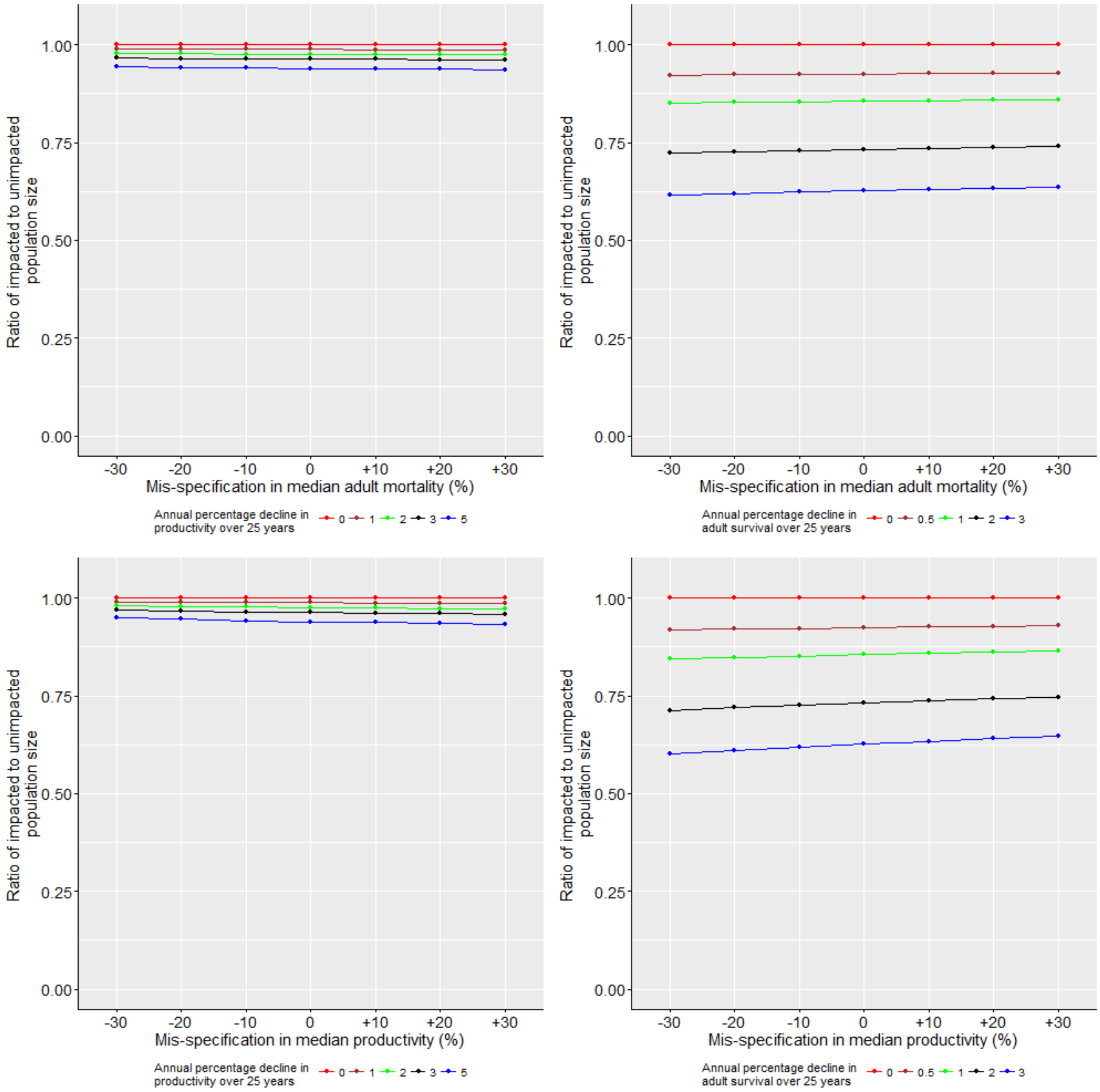


Figure A2.6c. PVA Metric C for St Abb's Guillemots – difference in population growth rate from 2016-2041, comparing impacted population vs. un-impacted population.

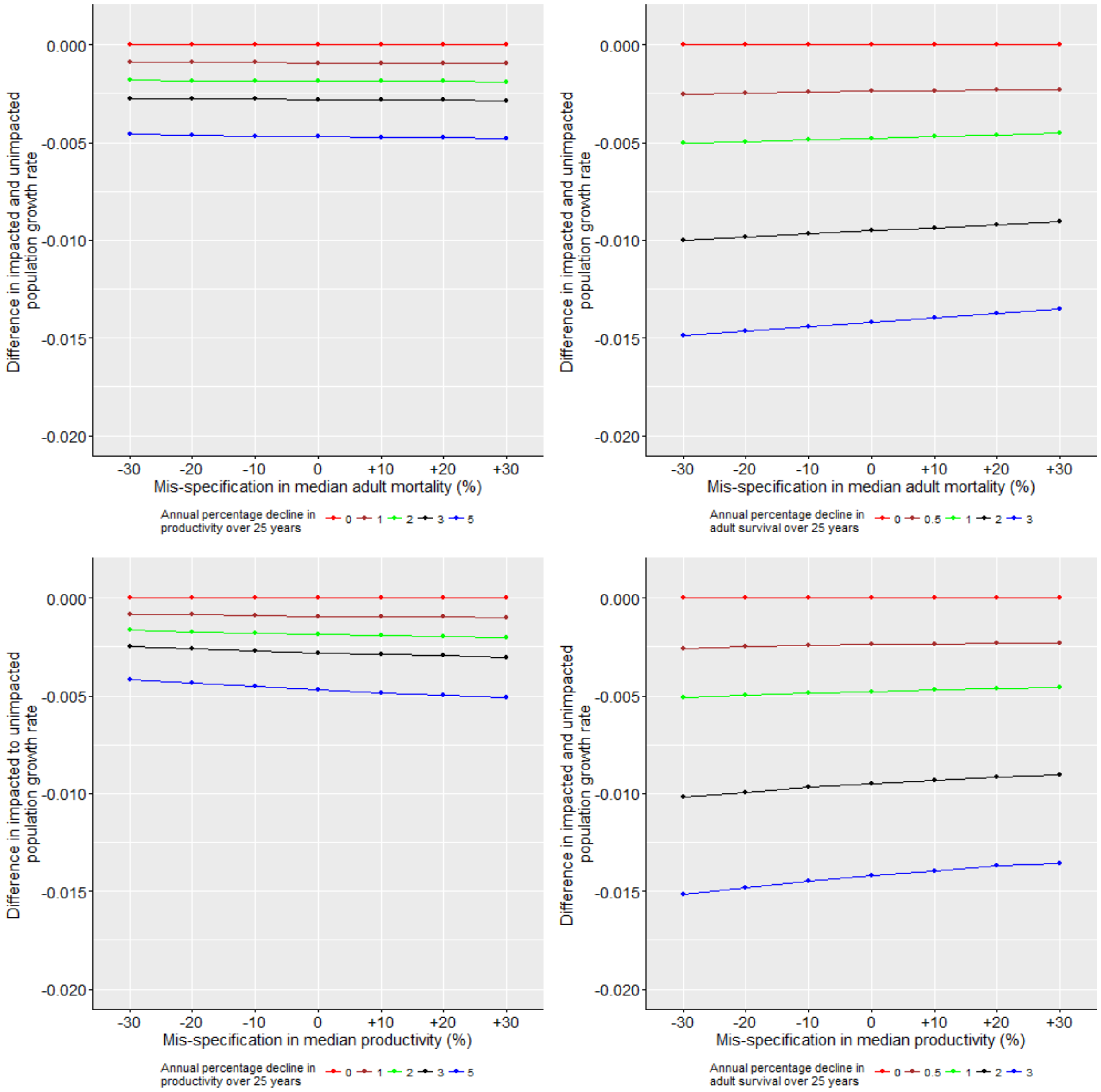


Figure A2.6d. PVA Metric D for St Abb's Guillemots – difference in population size at 2041, comparing impacted population vs. un-impacted population.

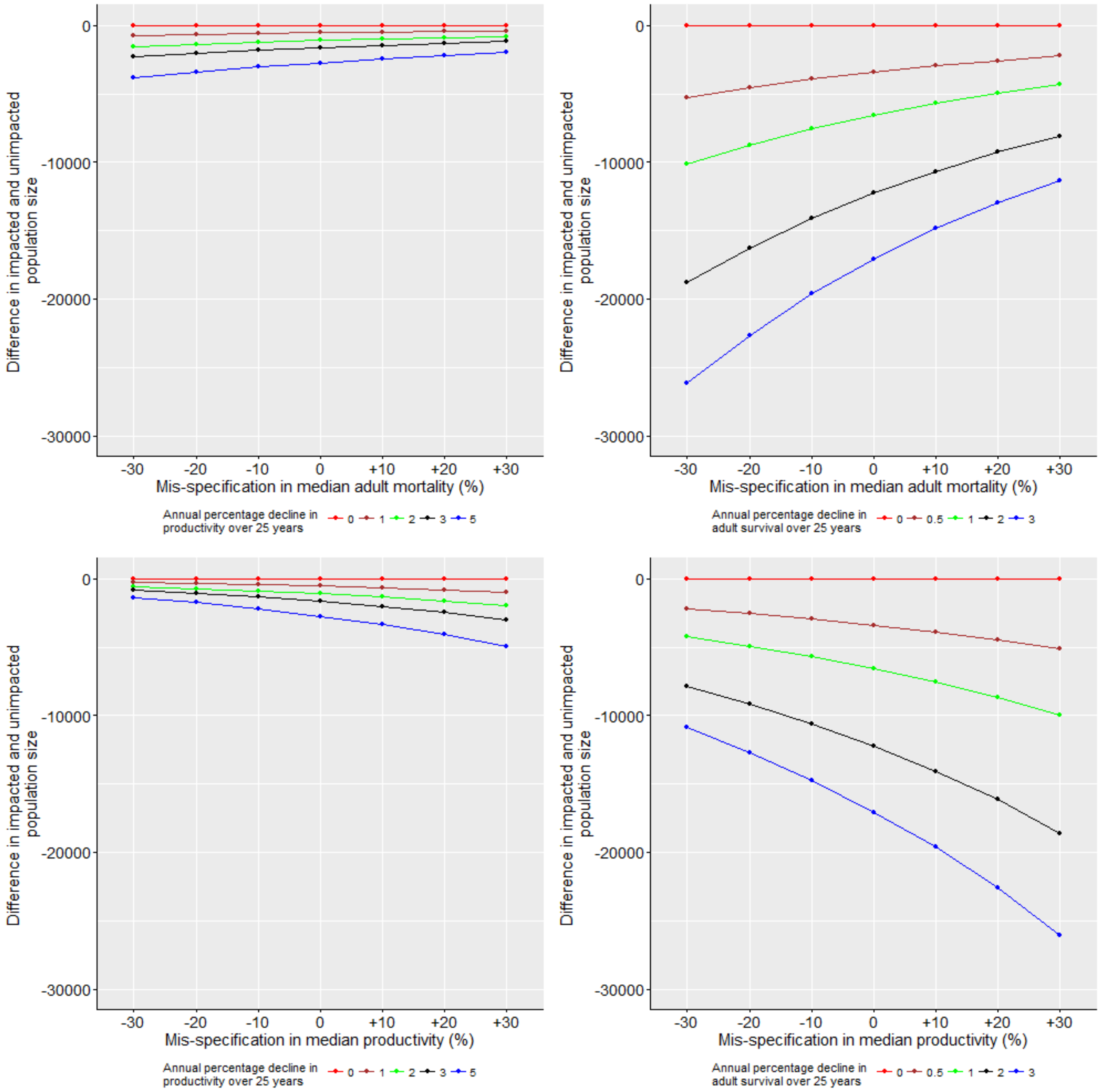


Figure A2.6e. PVA Metric E1 for St Abb’s Guillemots – probability of population decline greater than 10% from 2016-2041.

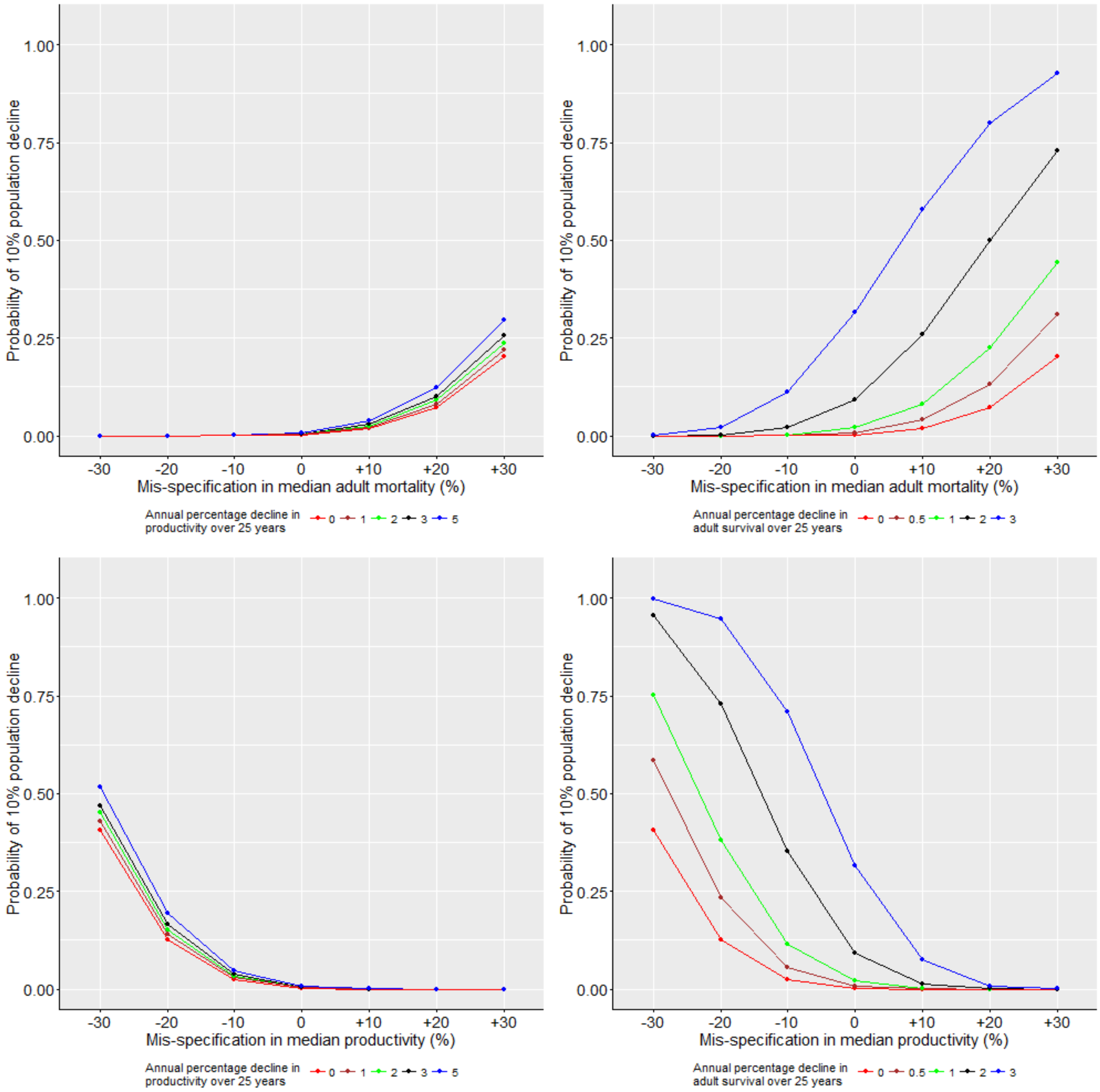


Figure A2.6f. PVA Metric E2 for St Abb's Guillemots – probability of population decline greater than 25% from 2016-2041.

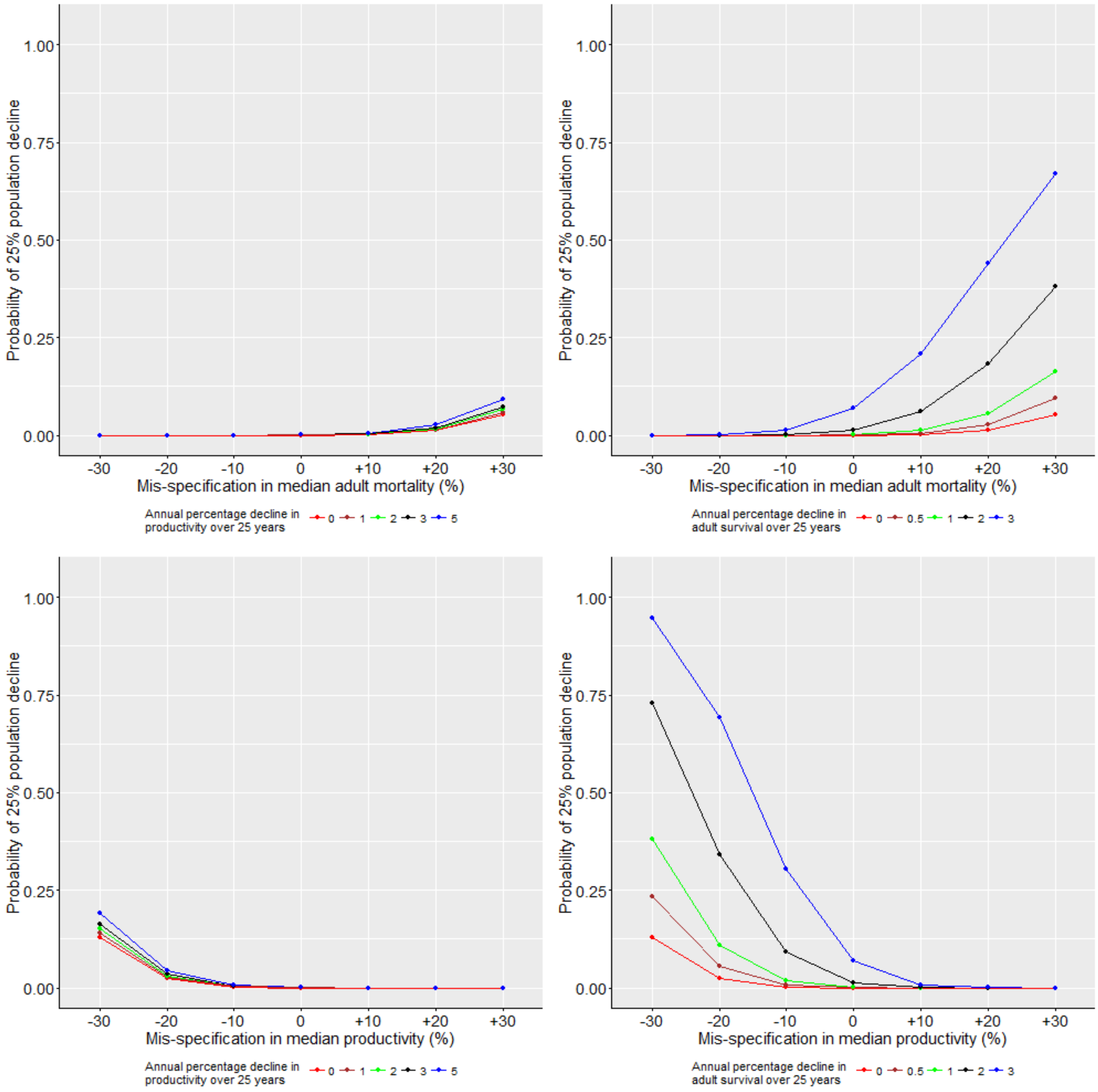


Figure A2.6g. PVA Metric E3 for St Abb's Guillemots – probability of population decline greater than 50% from 2016-2041.

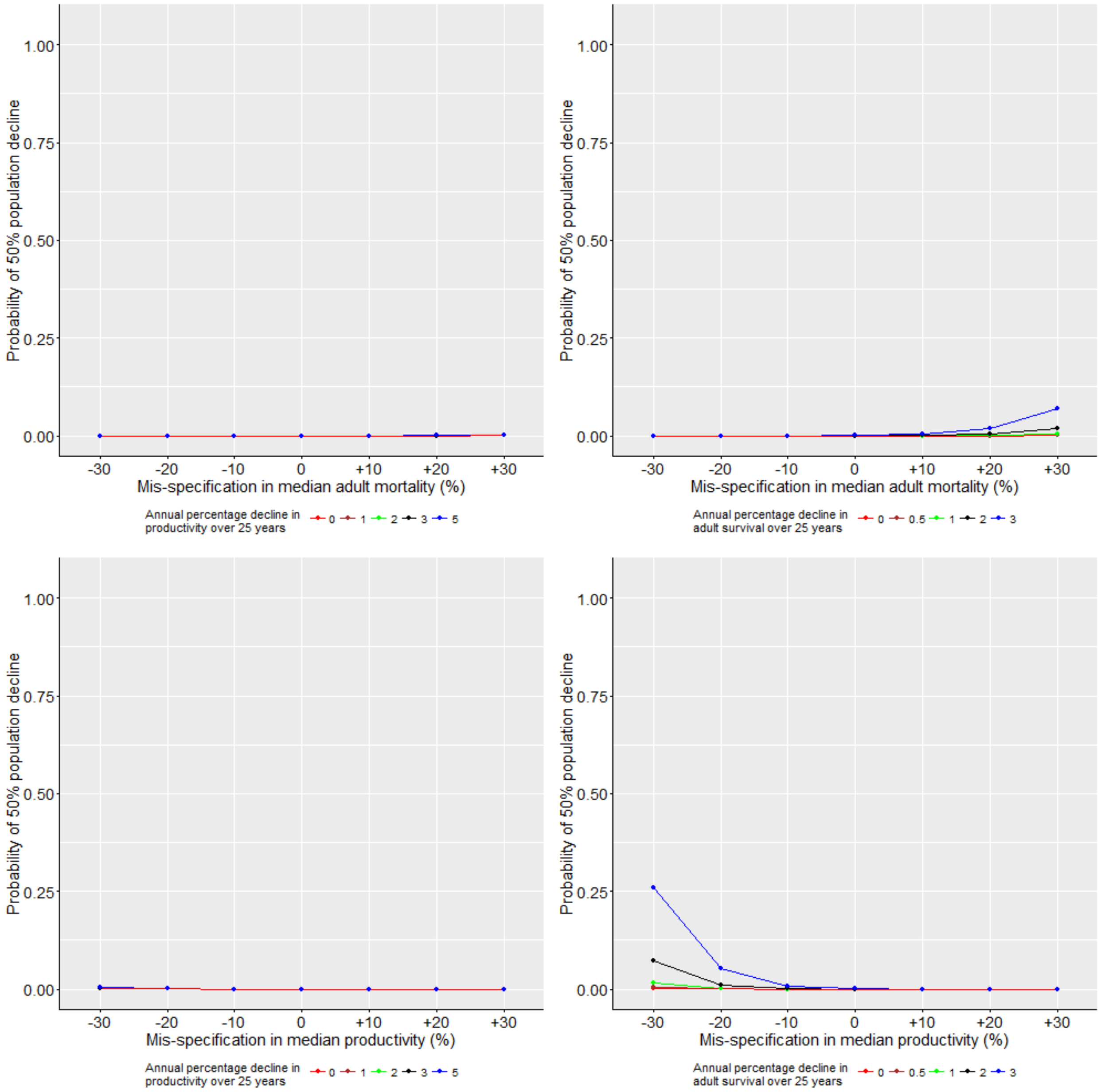
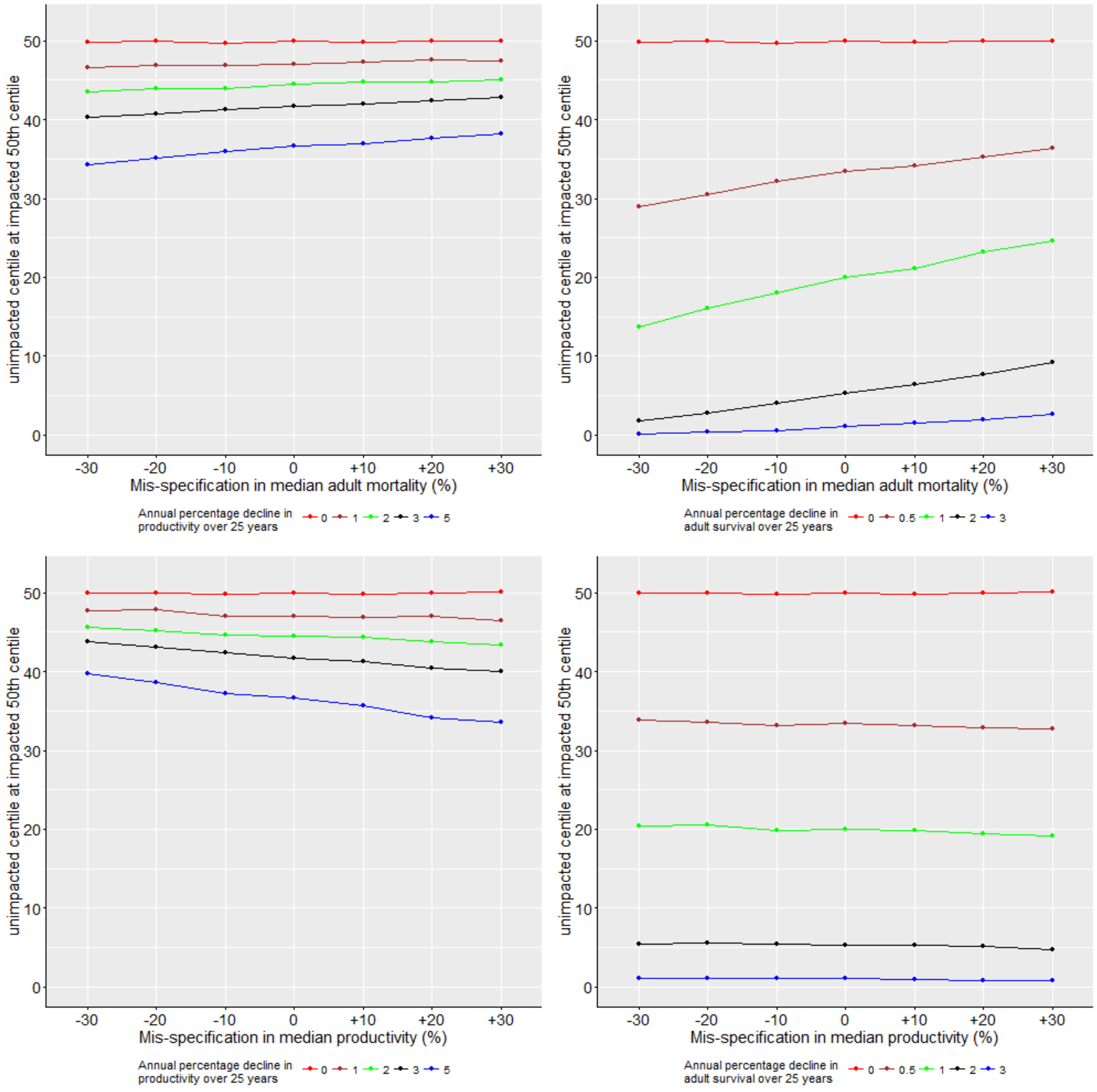


Figure A2.6h. PVA Metric F for St Abb’s Guillemots – centile from un-impacted population size equal to the 50th centile of the impacted population size, at 2041.



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.

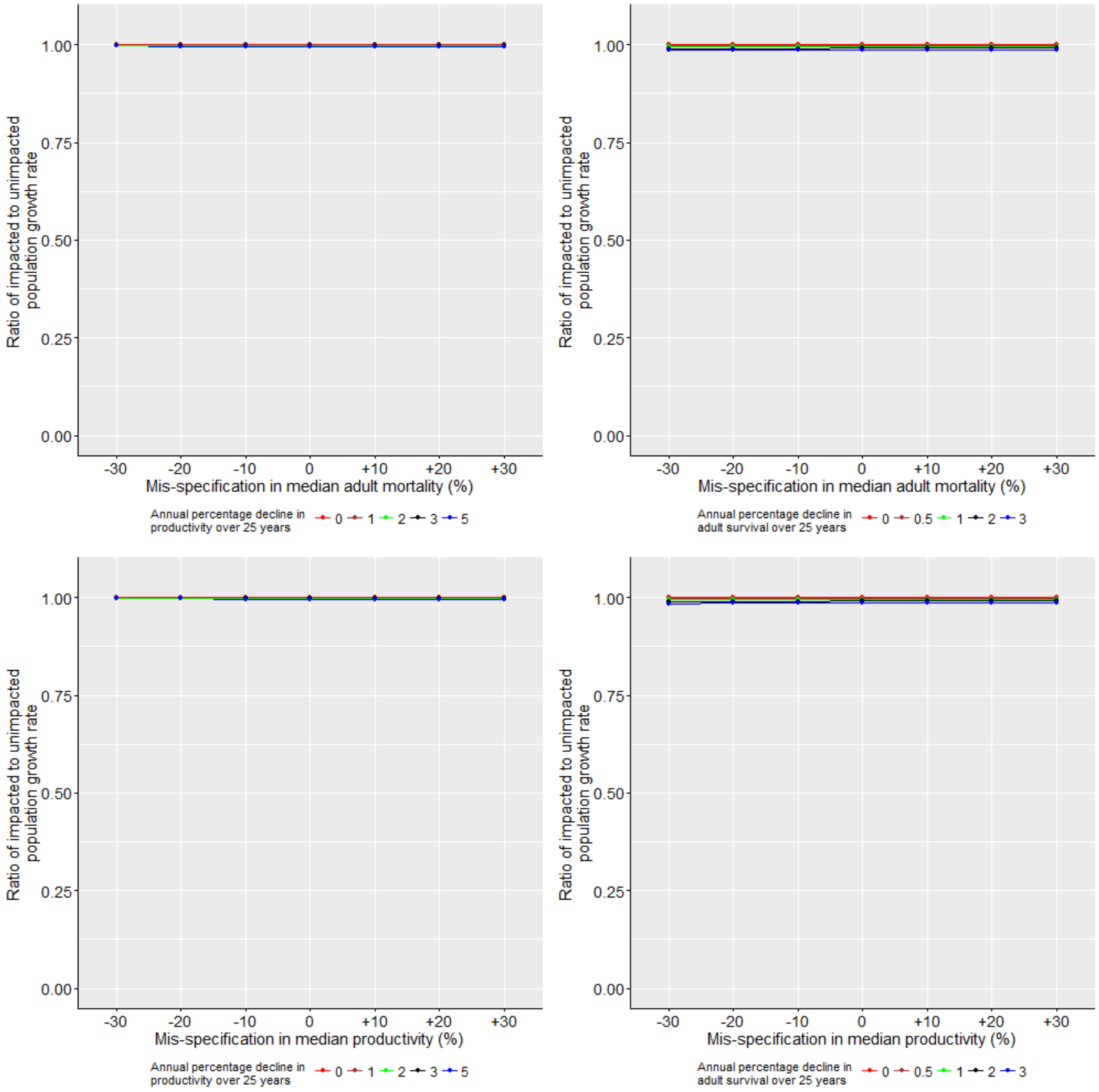


Figure A2.7b. PVA Metric B for Fowlsheugh Guillemots – ratio of population size at 2041, comparing impacted population vs. un-impacted population.

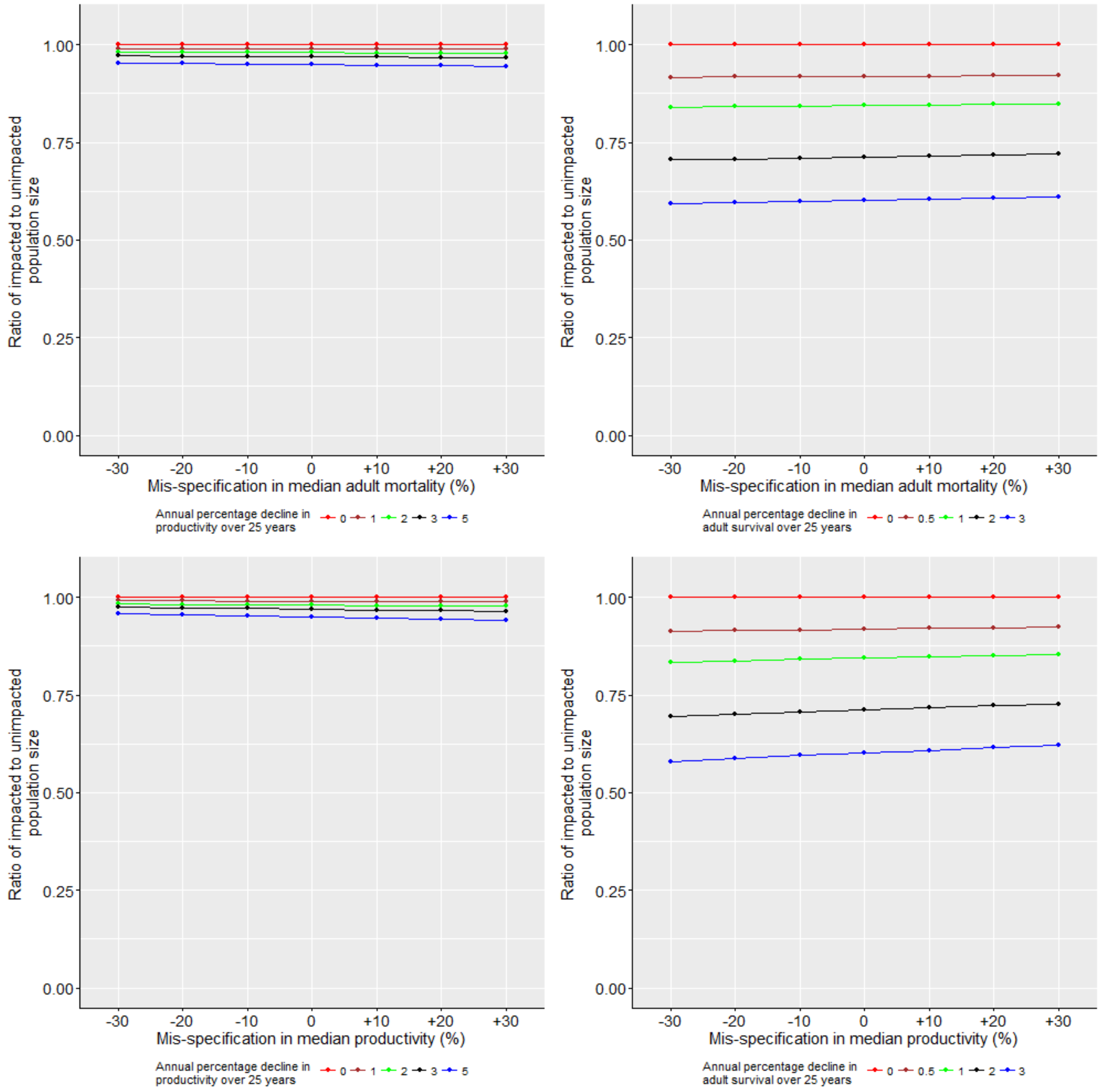


Figure A2.7c. PVA Metric C for Fowlsheugh Guillemots – difference in population growth rate from 2016-2041, comparing impacted population vs. un-impacted population.

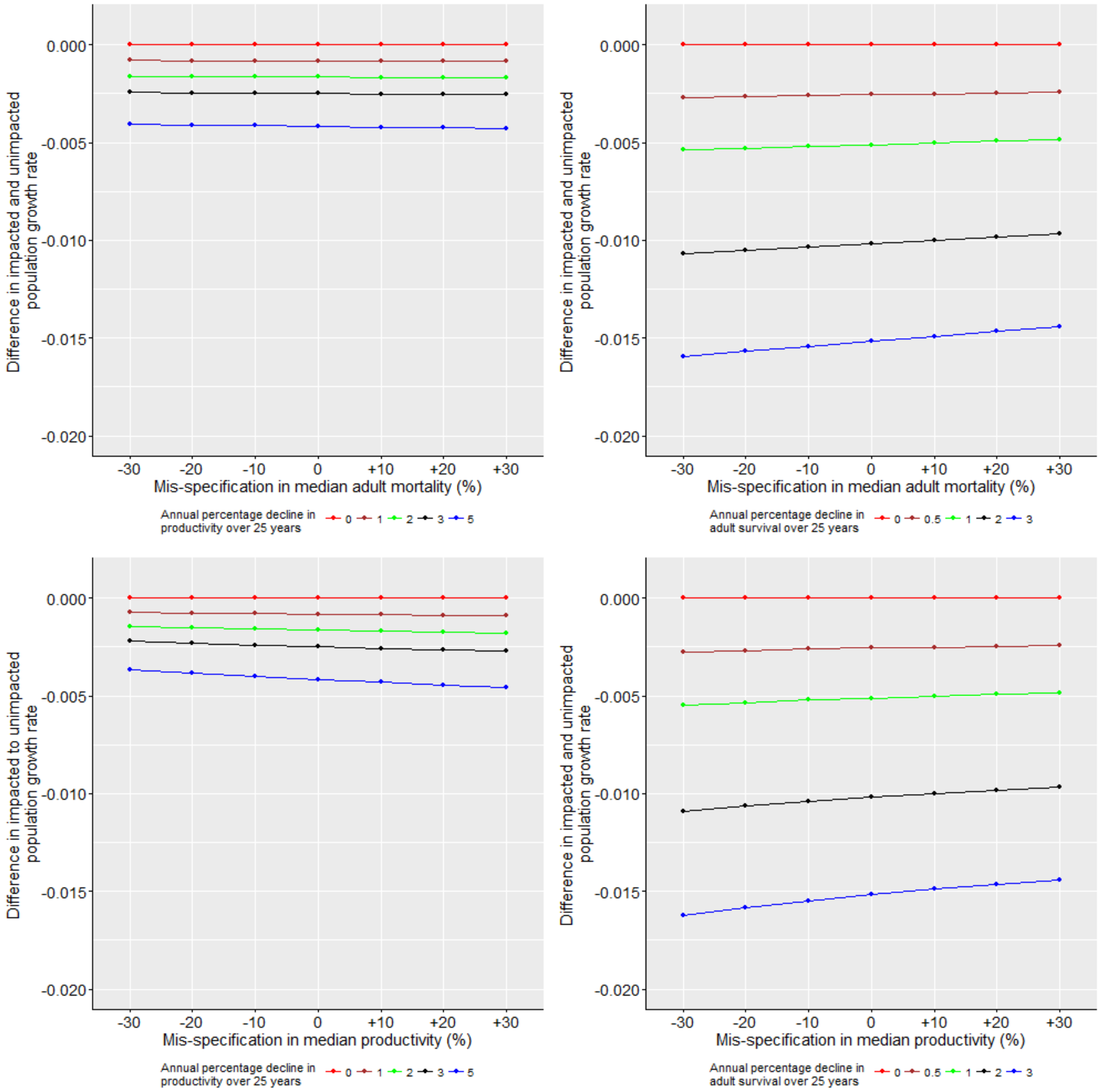


Figure A2.7d. PVA Metric D for Fowlsheugh Guillemots – difference in population size at 2041, comparing impacted population vs. un-impacted population.

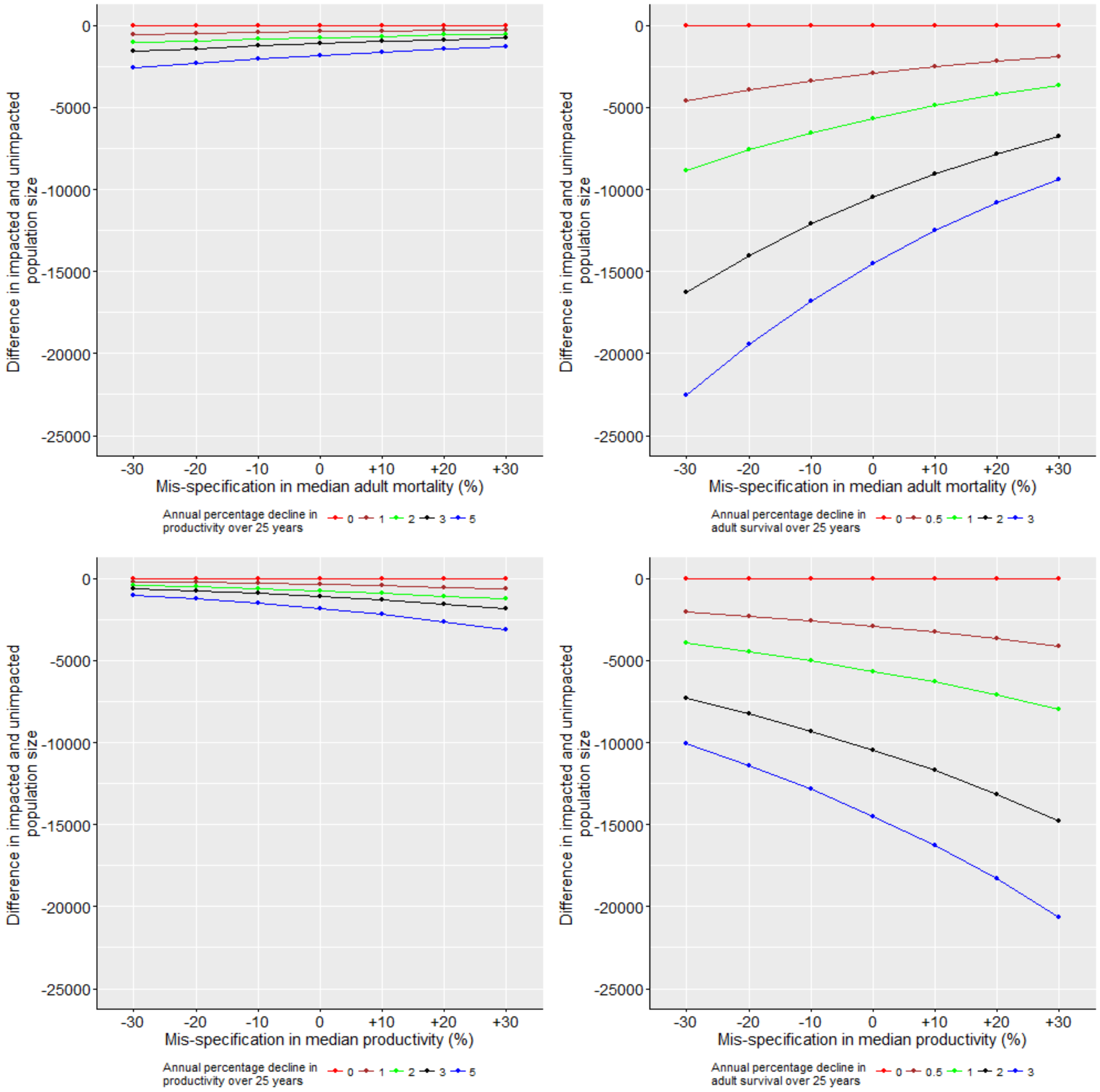


Figure A2.7e. PVA Metric E1 for Fowlsheugh Guillemots – probability of population decline greater than 10% from 2016-2041.

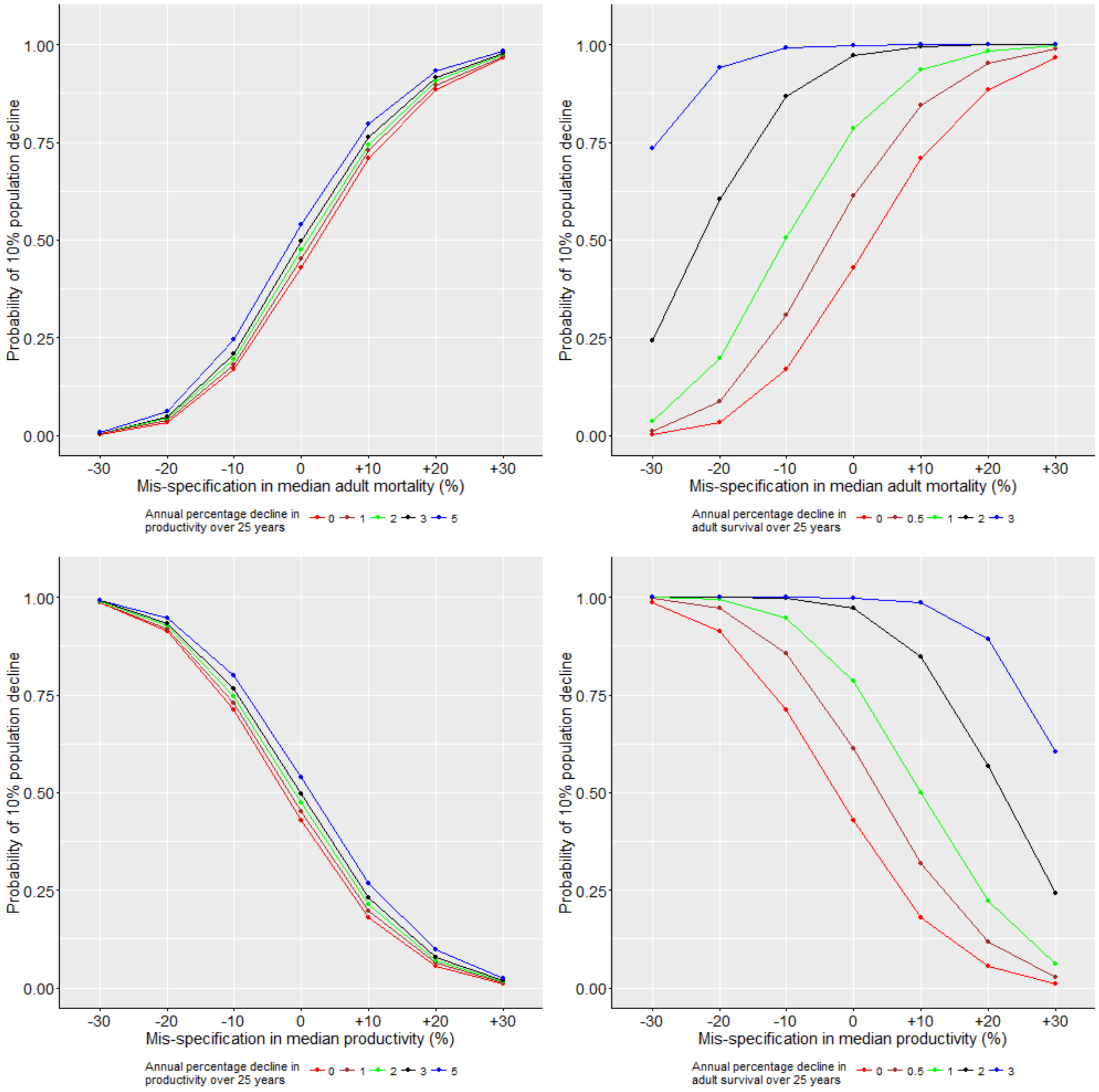


Figure A2.7f. PVA Metric E2 for Fowlsheugh Guillemots – probability of population decline greater than 25% from 2016-2041.

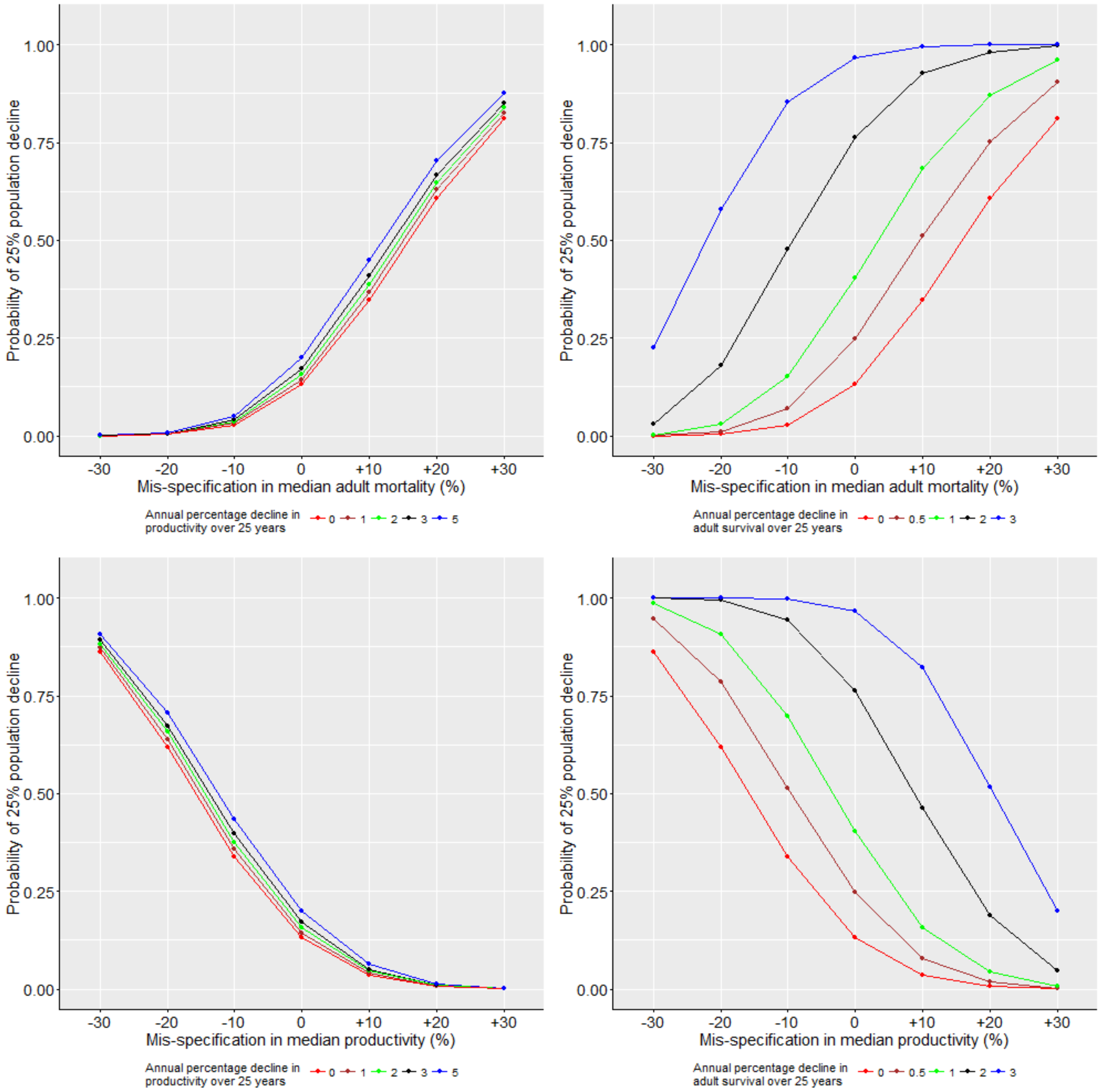


Figure A2.7g. PVA Metric E3 for Fowlsheugh Guillemots – probability of population decline greater than 50% from 2016-2041.

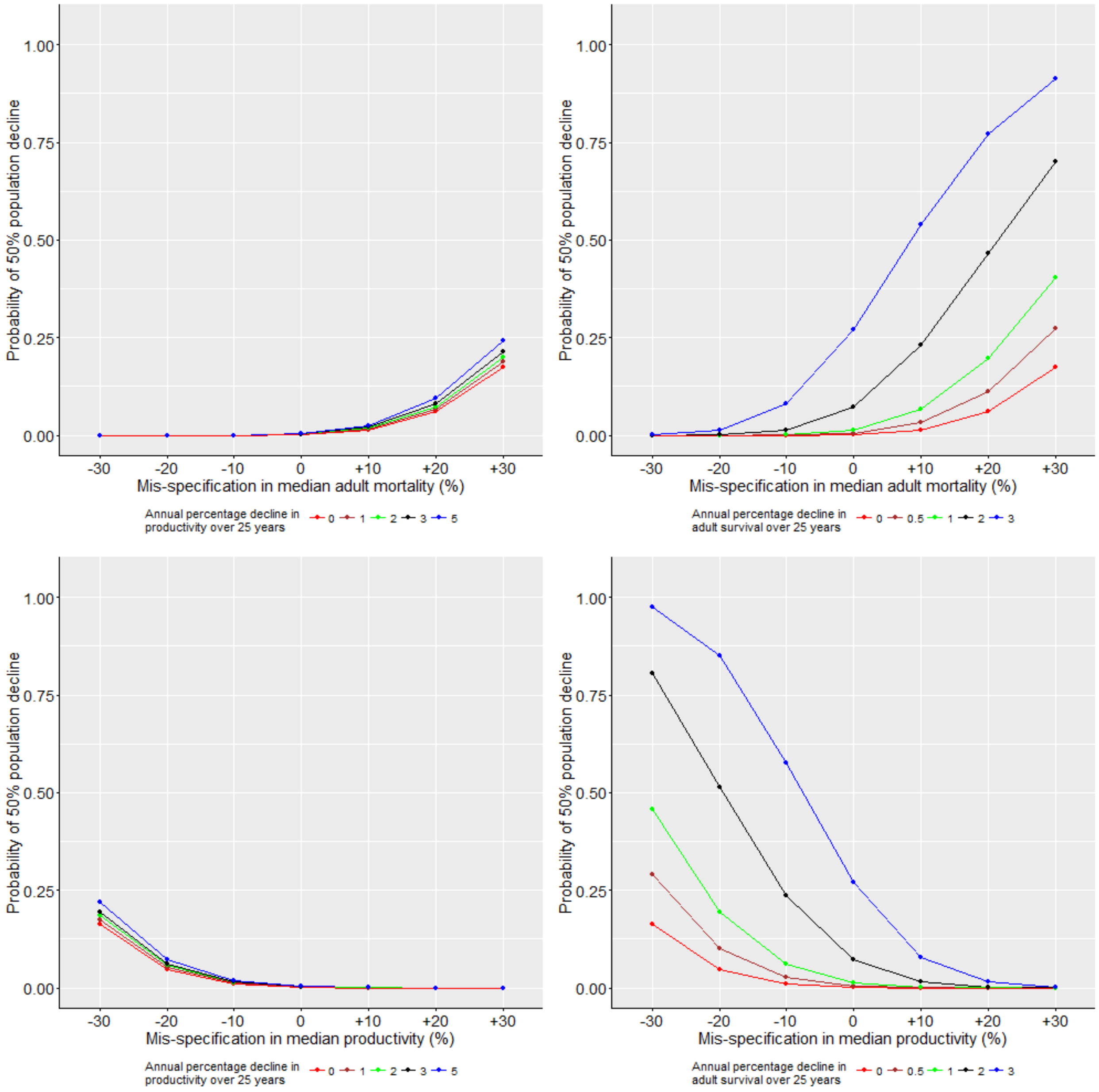
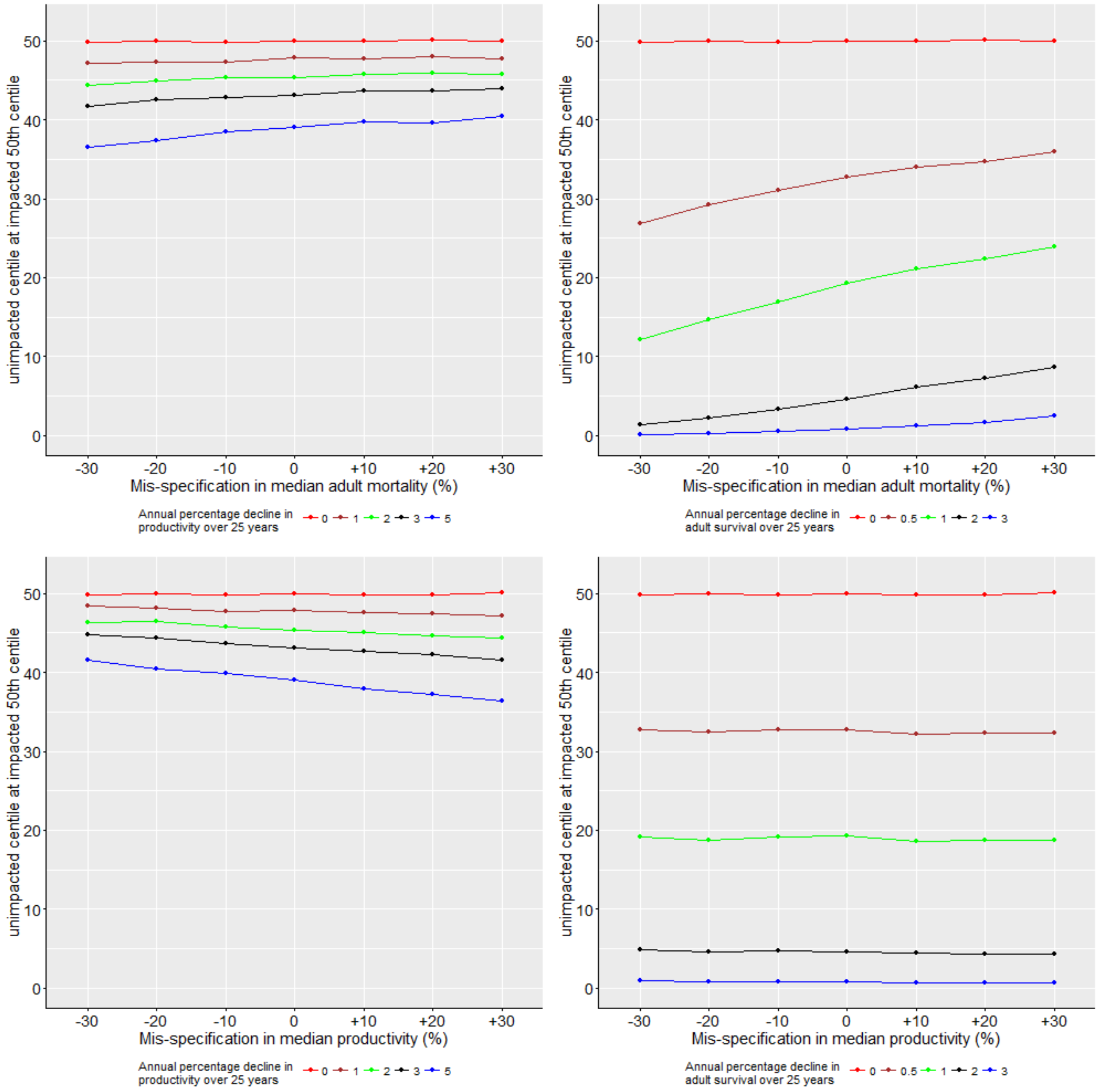


Figure A2.7h. PVA Metric F for Fowlsheugh Guillemots – centile from un-impacted population size equal to the 50th centile of the impacted population size, at 2041.



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.

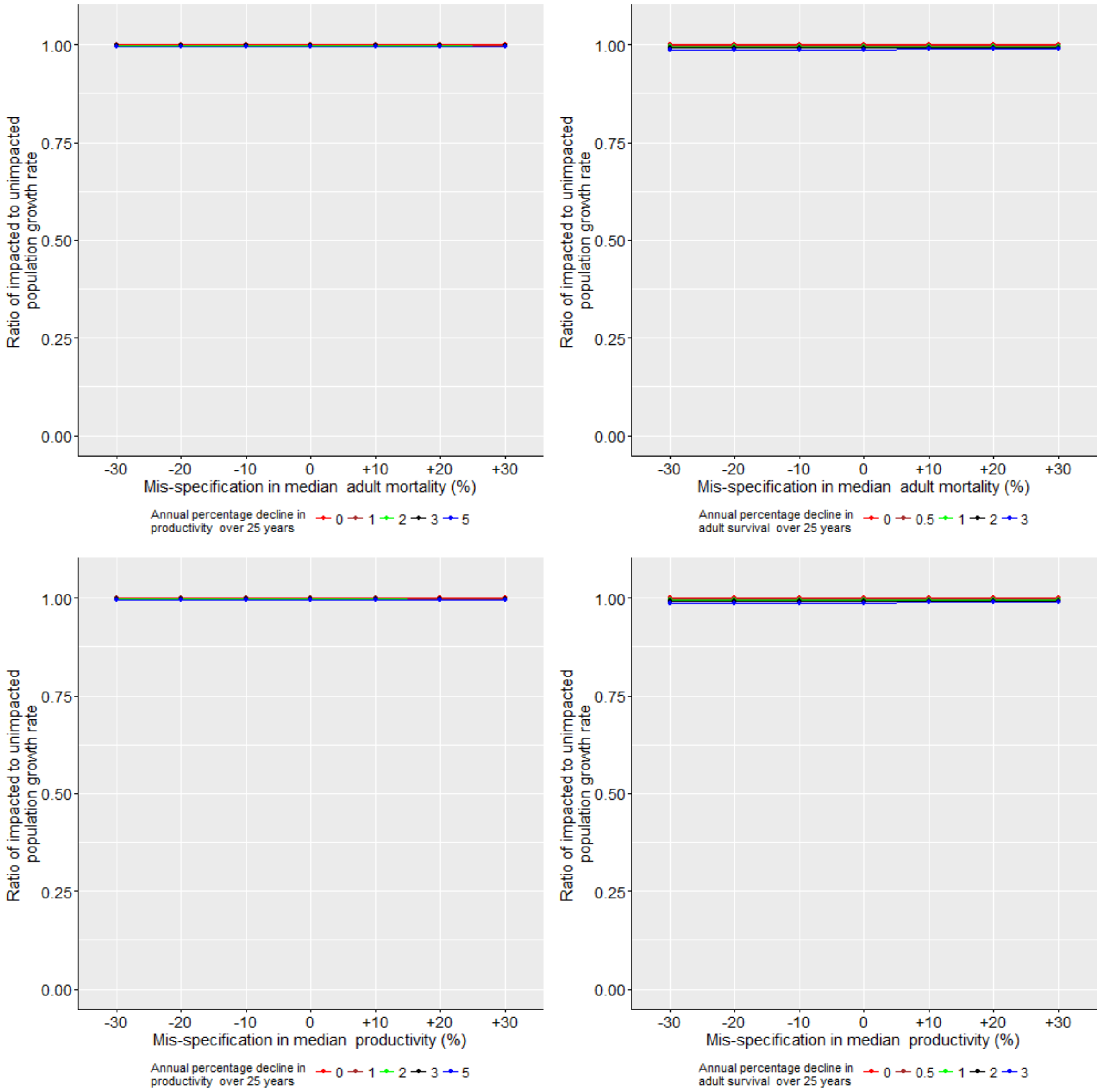


Figure A2.8b. PVA Metric B for Buchan Ness Guillemots – ratio of population size at 2041, comparing impacted population vs. un-impacted population.

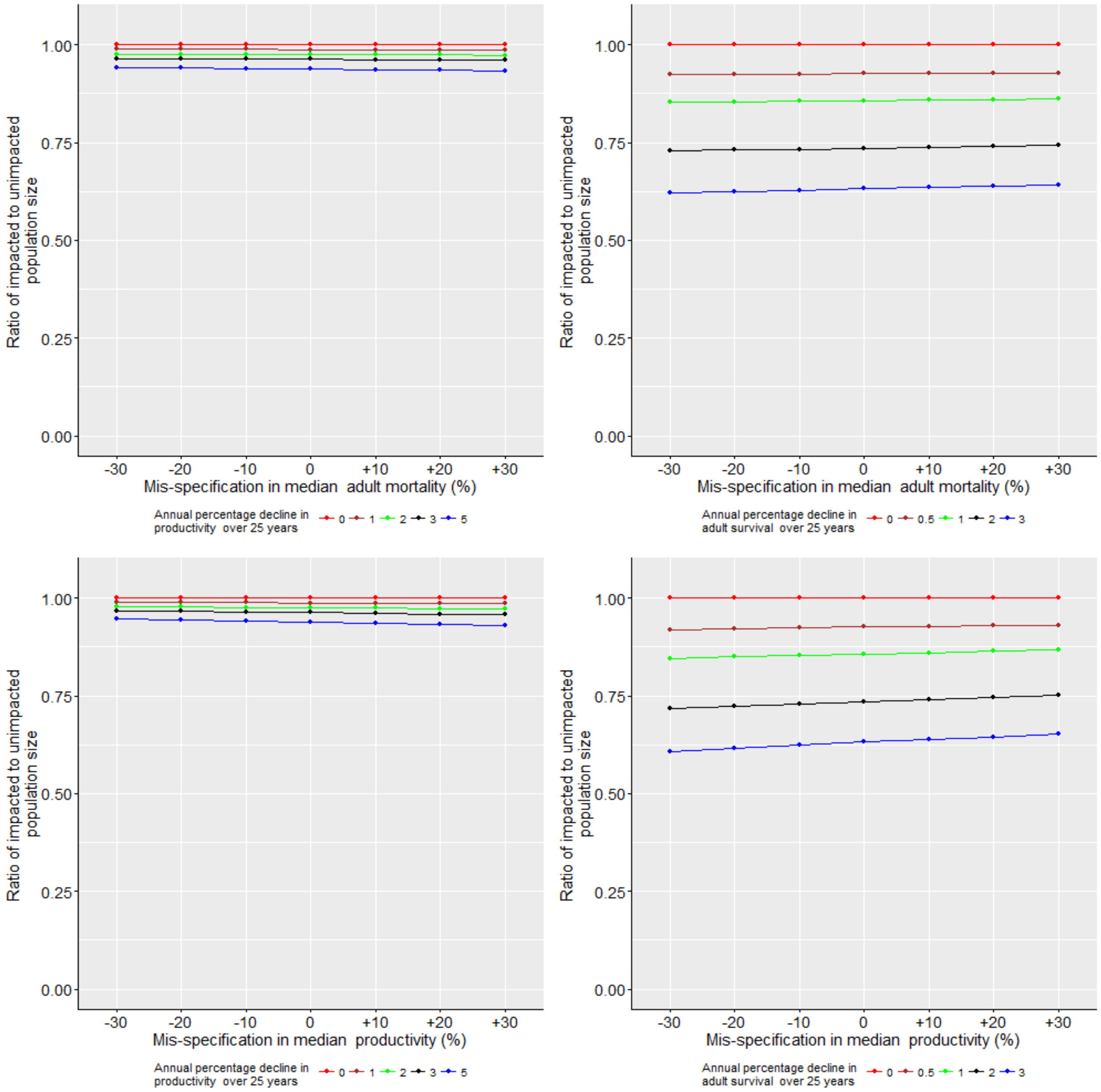


Figure A2.8c. PVA Metric C for Buchan Ness Guillemots – difference in population growth rate from 2016-2041, comparing impacted population vs. un-impacted population.

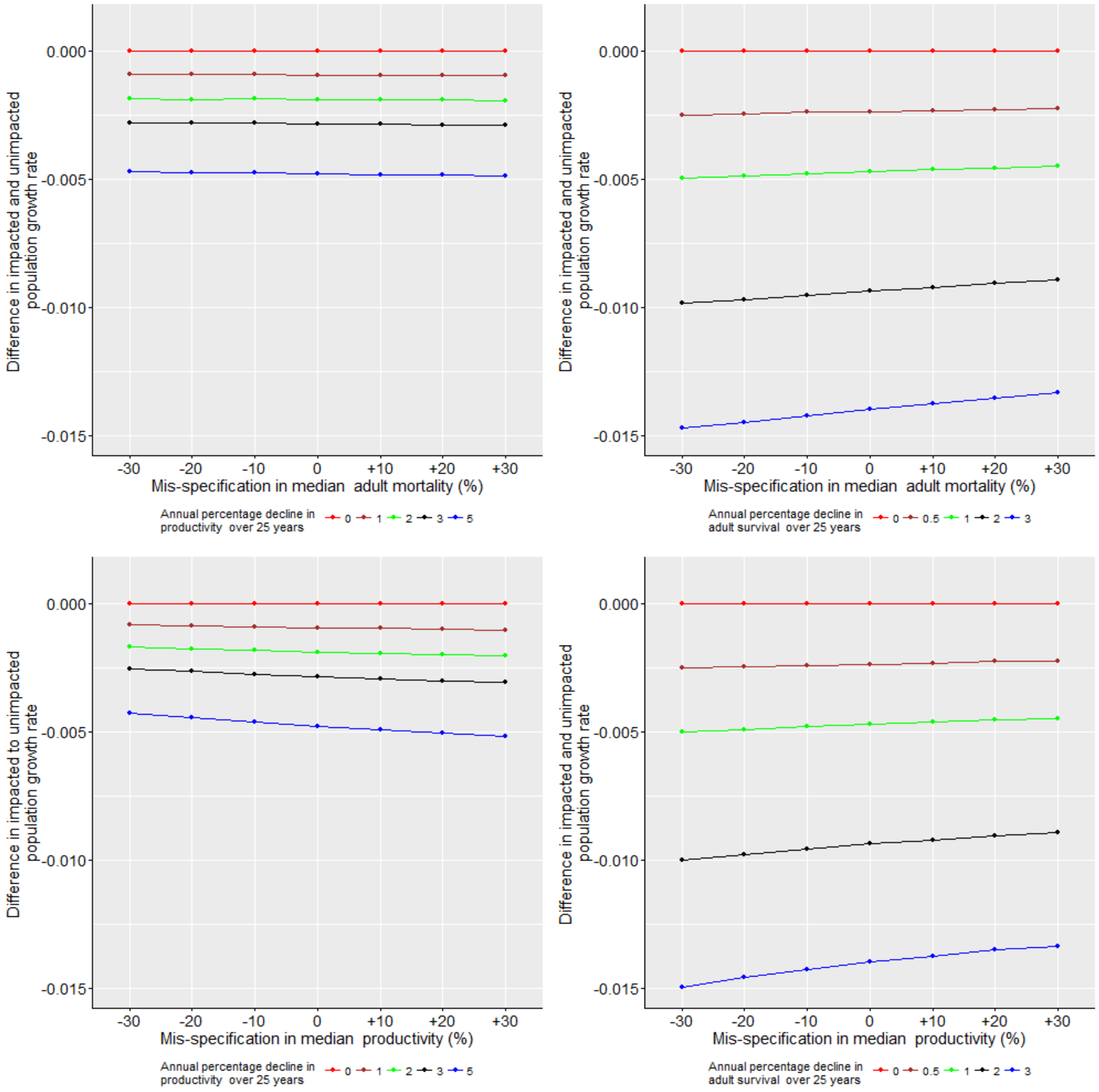


Figure A2.8d. PVA Metric D for Buchan Ness Guillemots – difference in population size at 2041, comparing impacted population vs. un-impacted population.

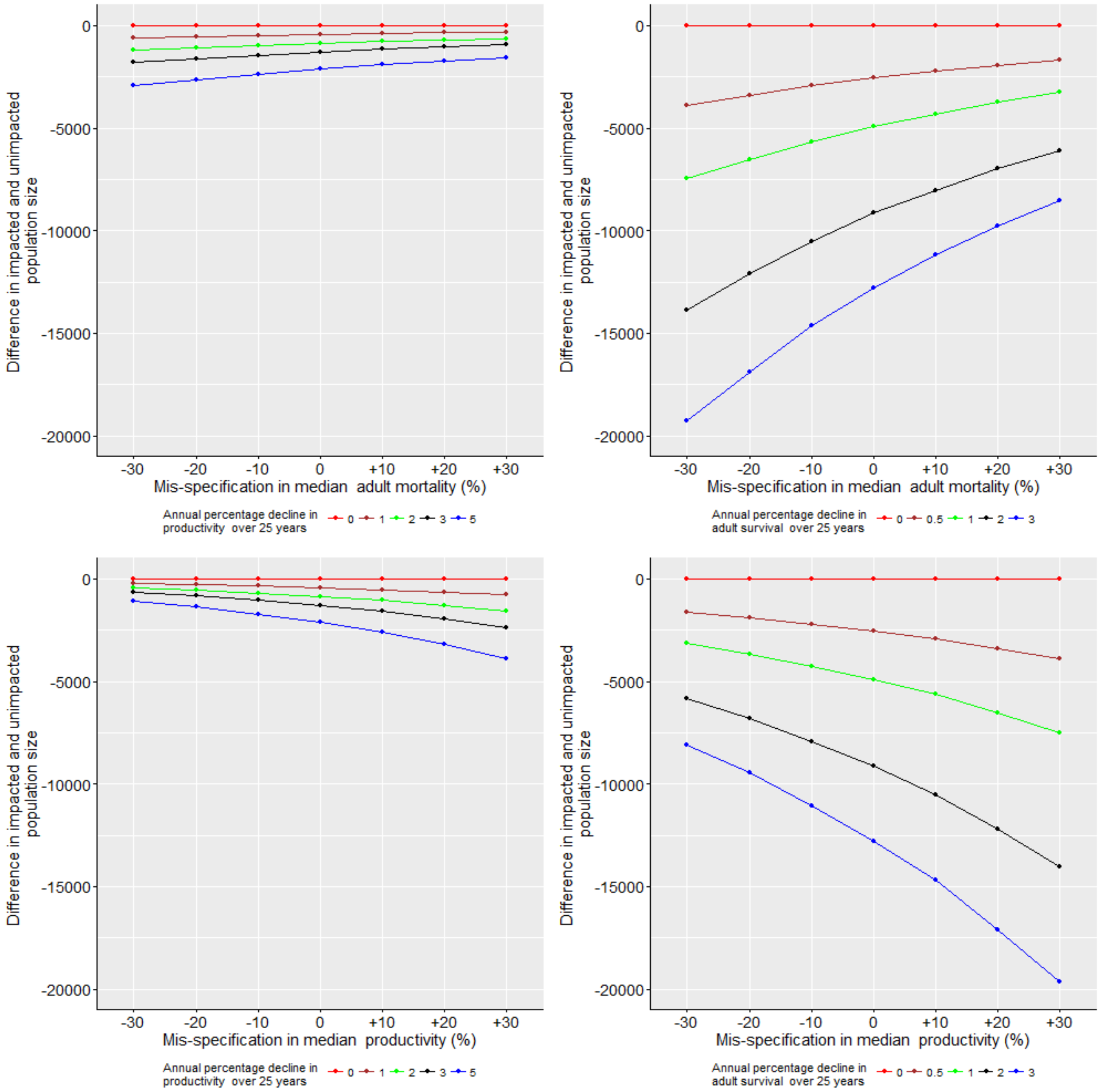


Figure A2.8e. PVA Metric E1 for Buchan Ness Guillemots – probability of population decline greater than 10% from 2016-2041.

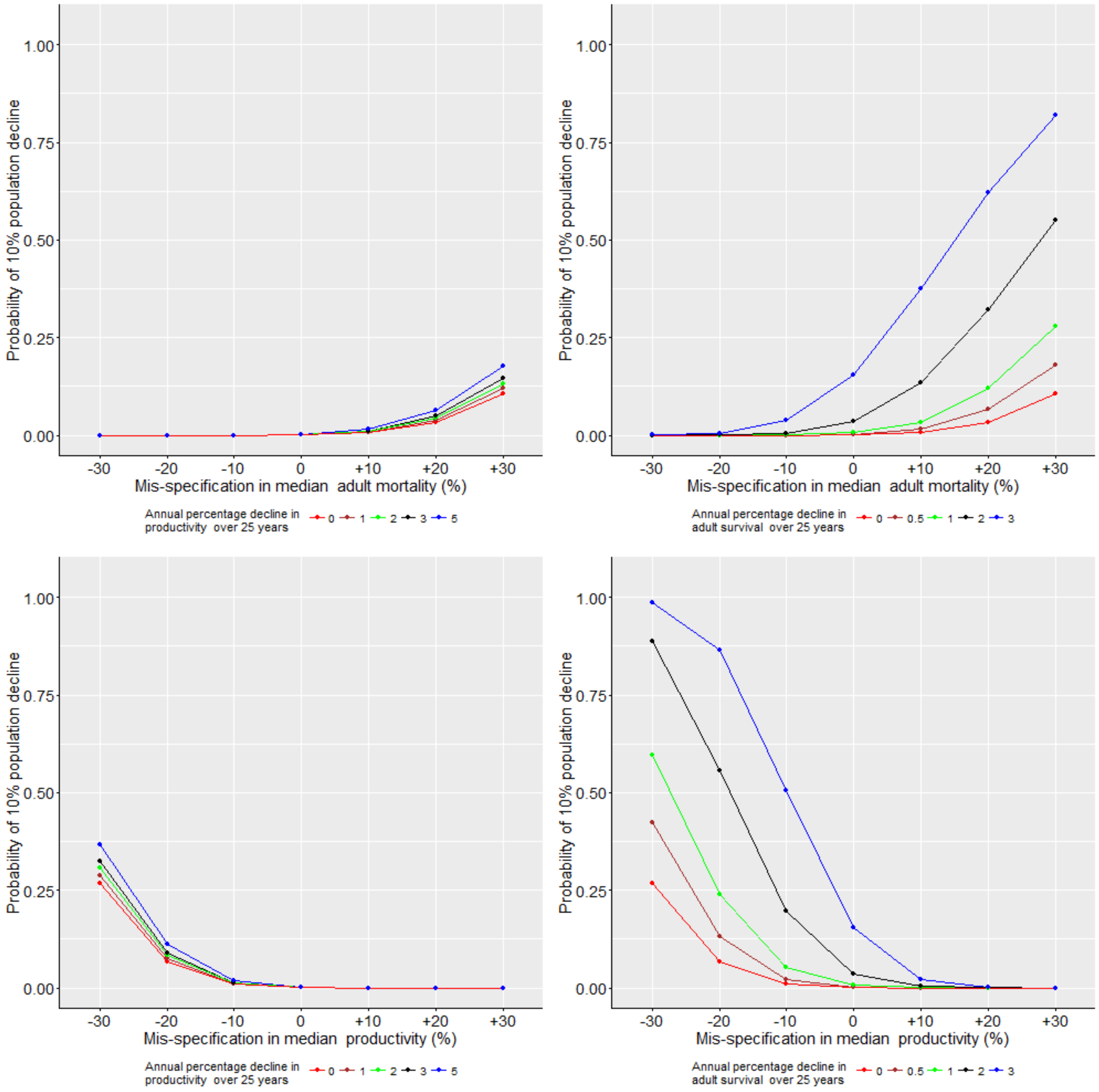


Figure A2.8f. PVA Metric E2 for Buchan Ness Guillemots – probability of population decline greater than 25% from 2016-2041.

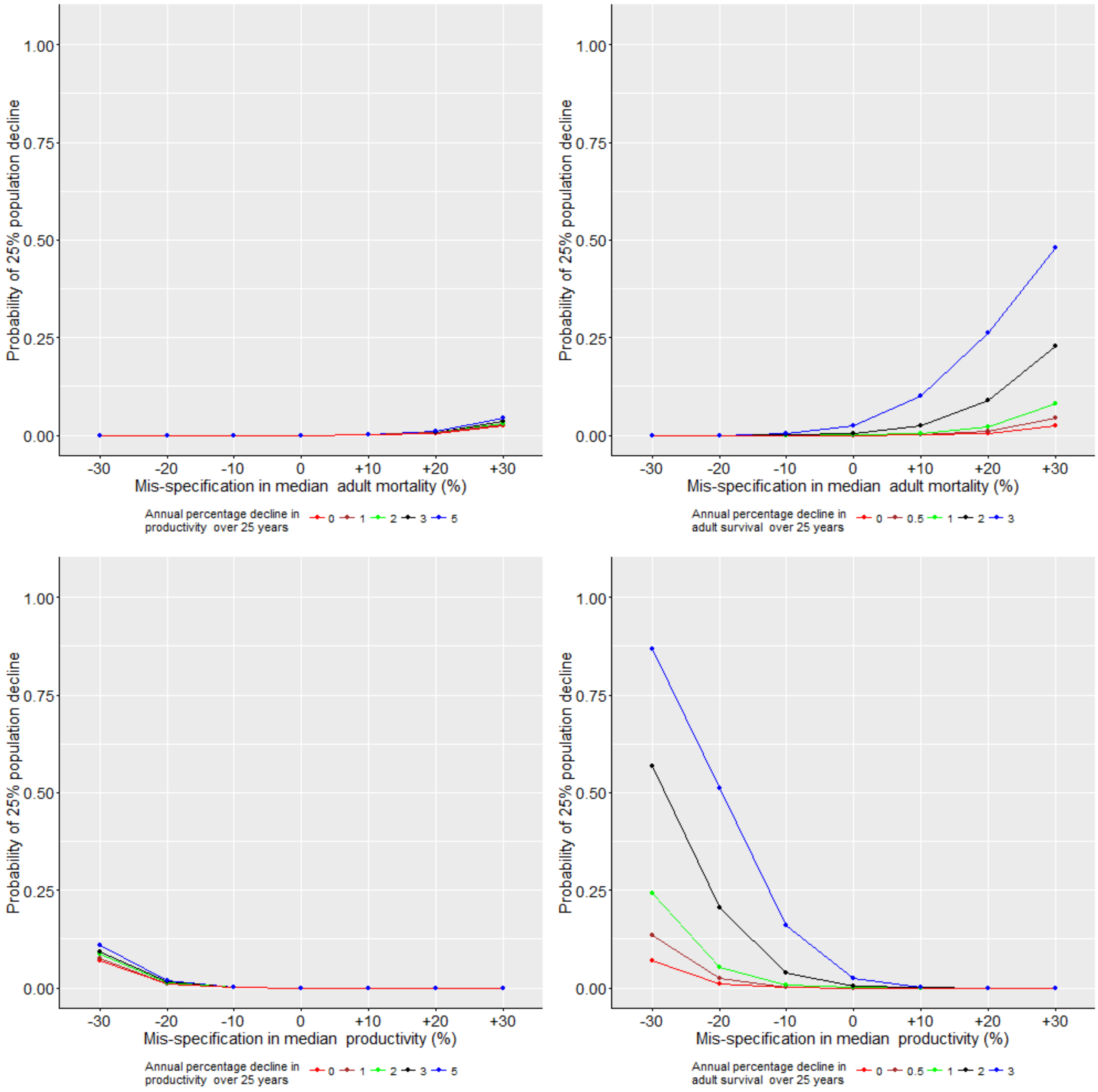


Figure A2.8g. PVA Metric E3 for Buchan Ness Guillemots – probability of population decline greater than 50% from 2016-2041.

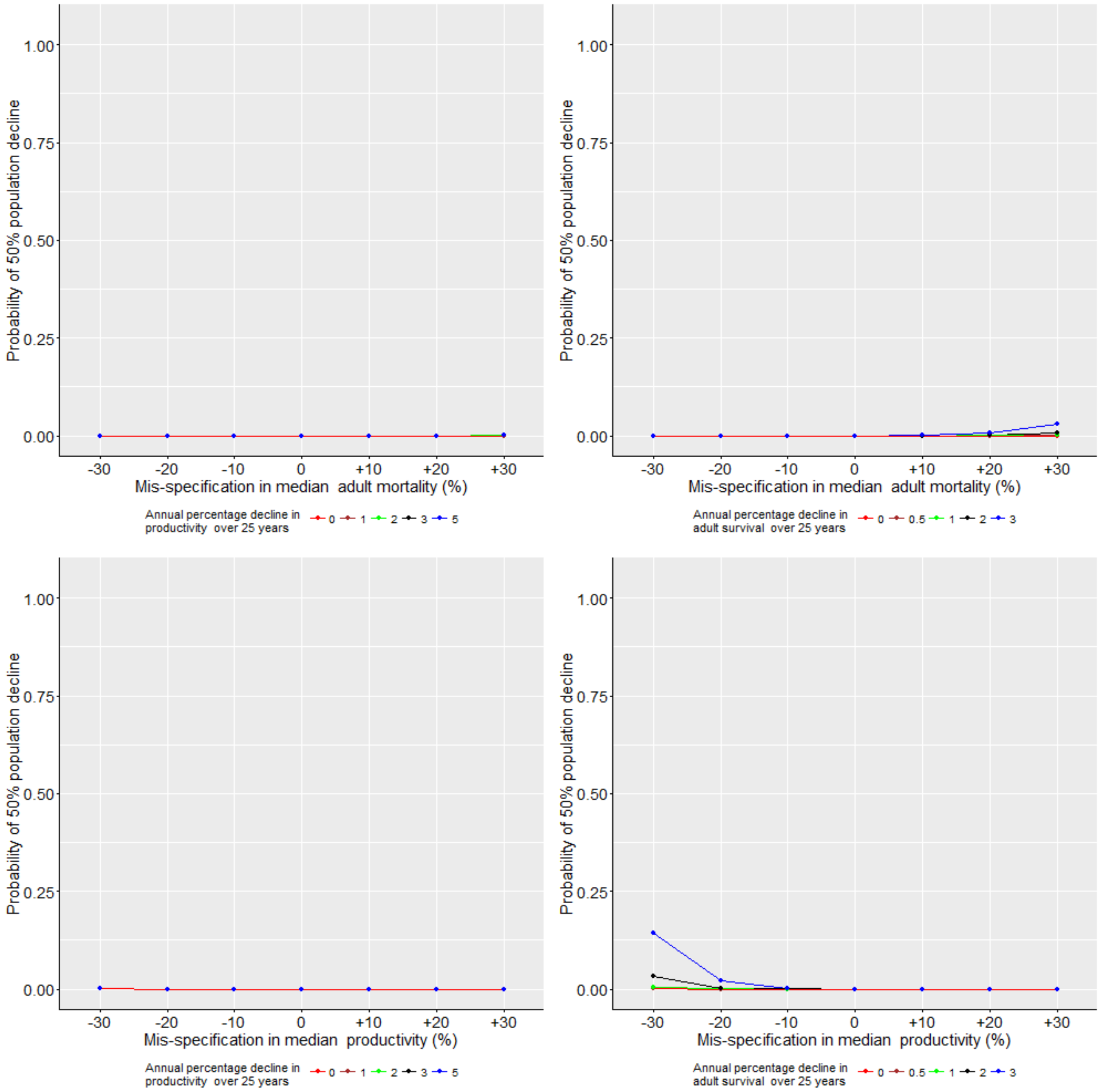
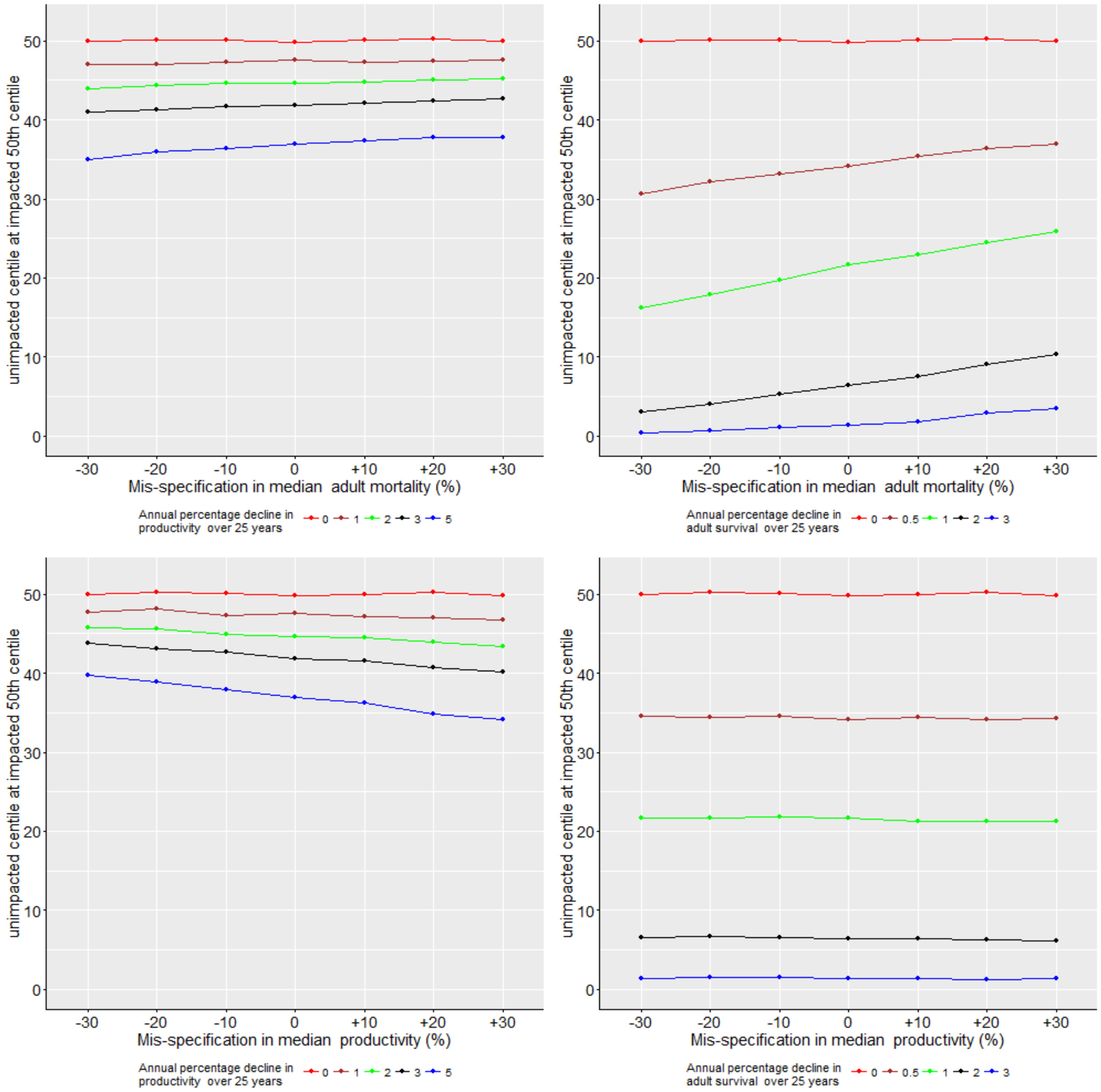


Figure A2.8h. PVA Metric F for Buchan Ness Guillemots – centile from un-impacted population size equal to the 50th centile of the impacted population size, at 2041.



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.

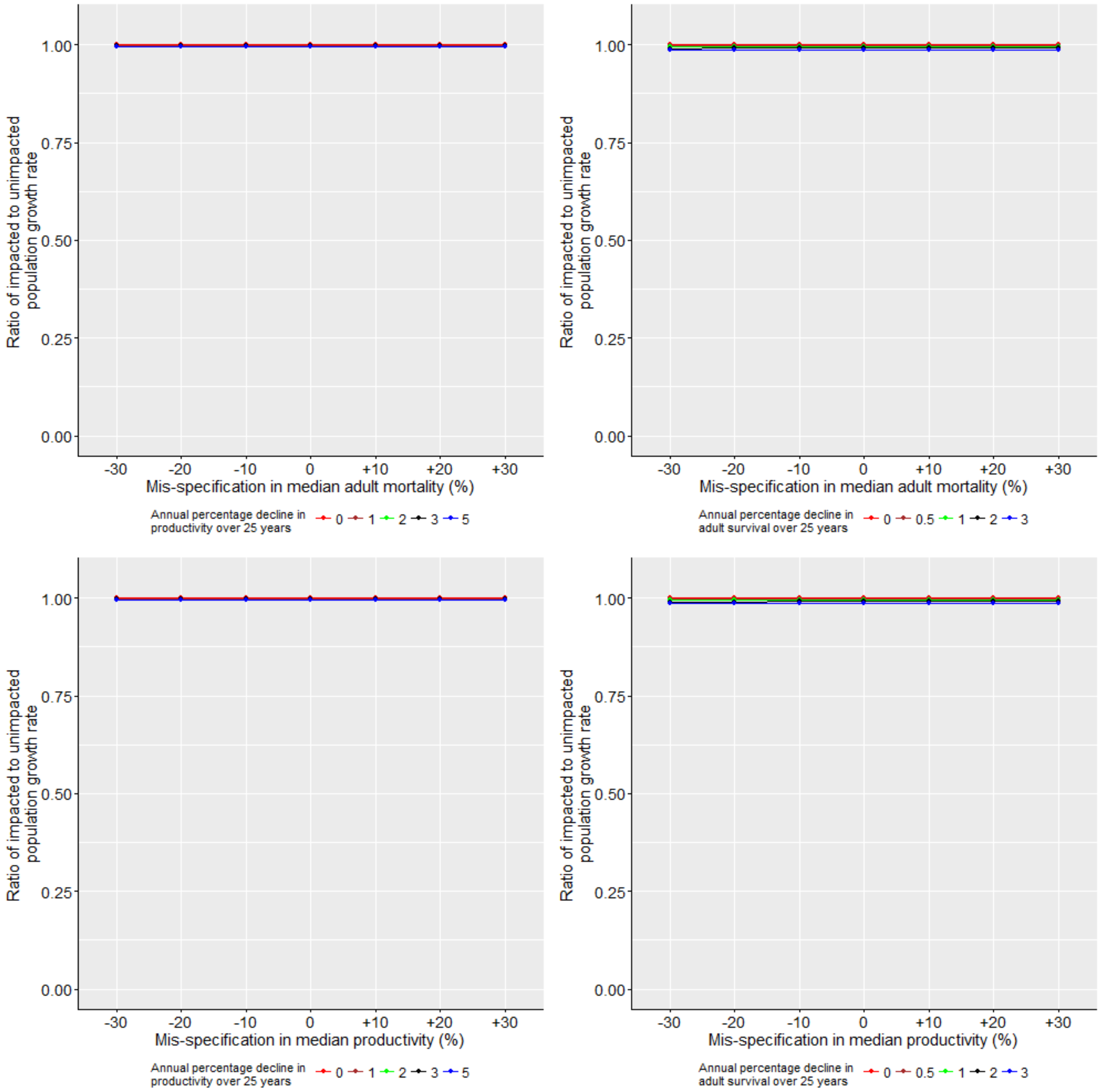


Figure A2.9b. PVA Metric B for Forth Razorbills – ratio of population size at 2041, comparing impacted population vs. un-impacted population.

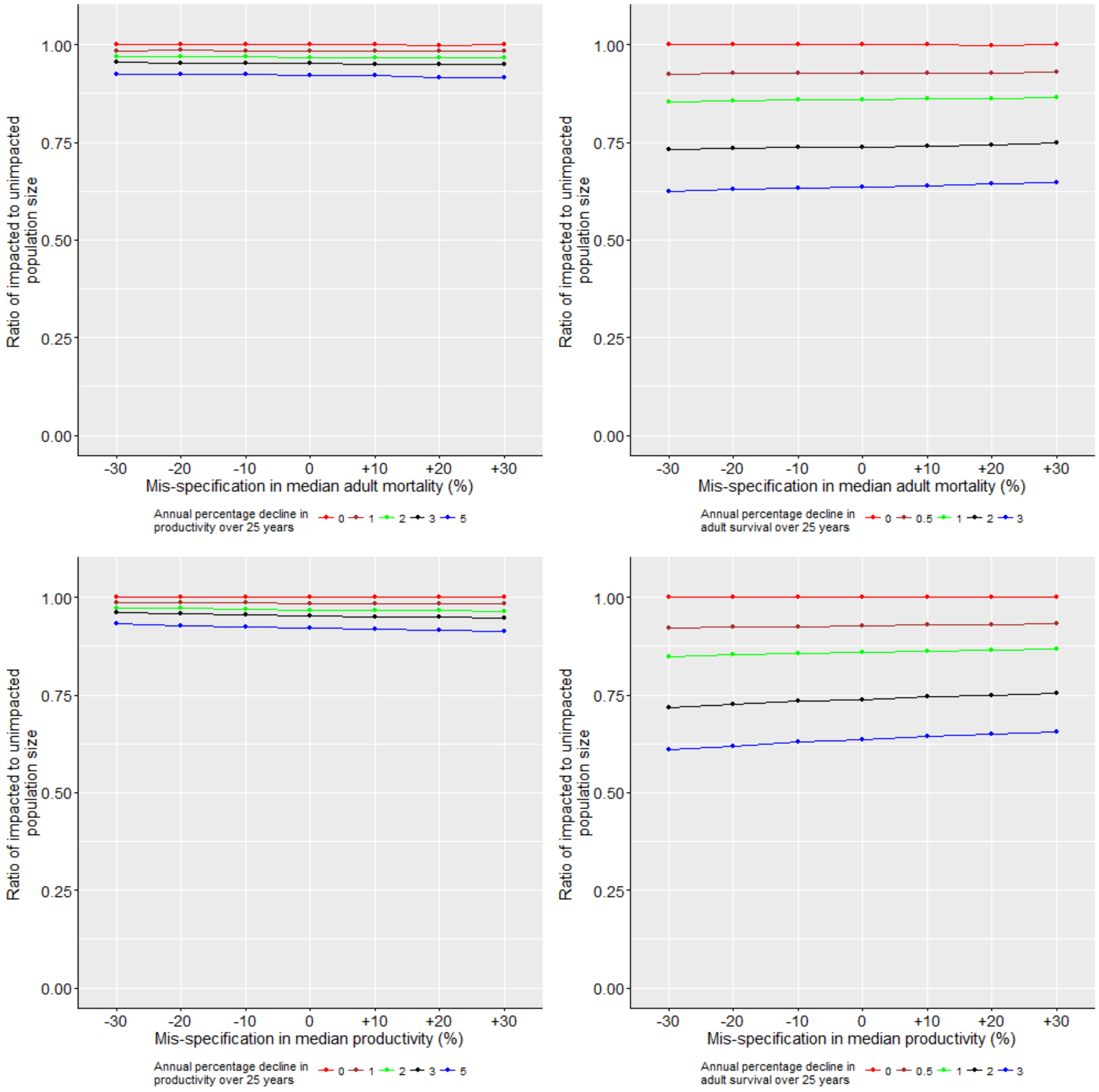


Figure A2.9c. PVA Metric C for Forth Razorbills – difference in population growth rate from 2016-2041, comparing impacted population vs. un-impacted population.

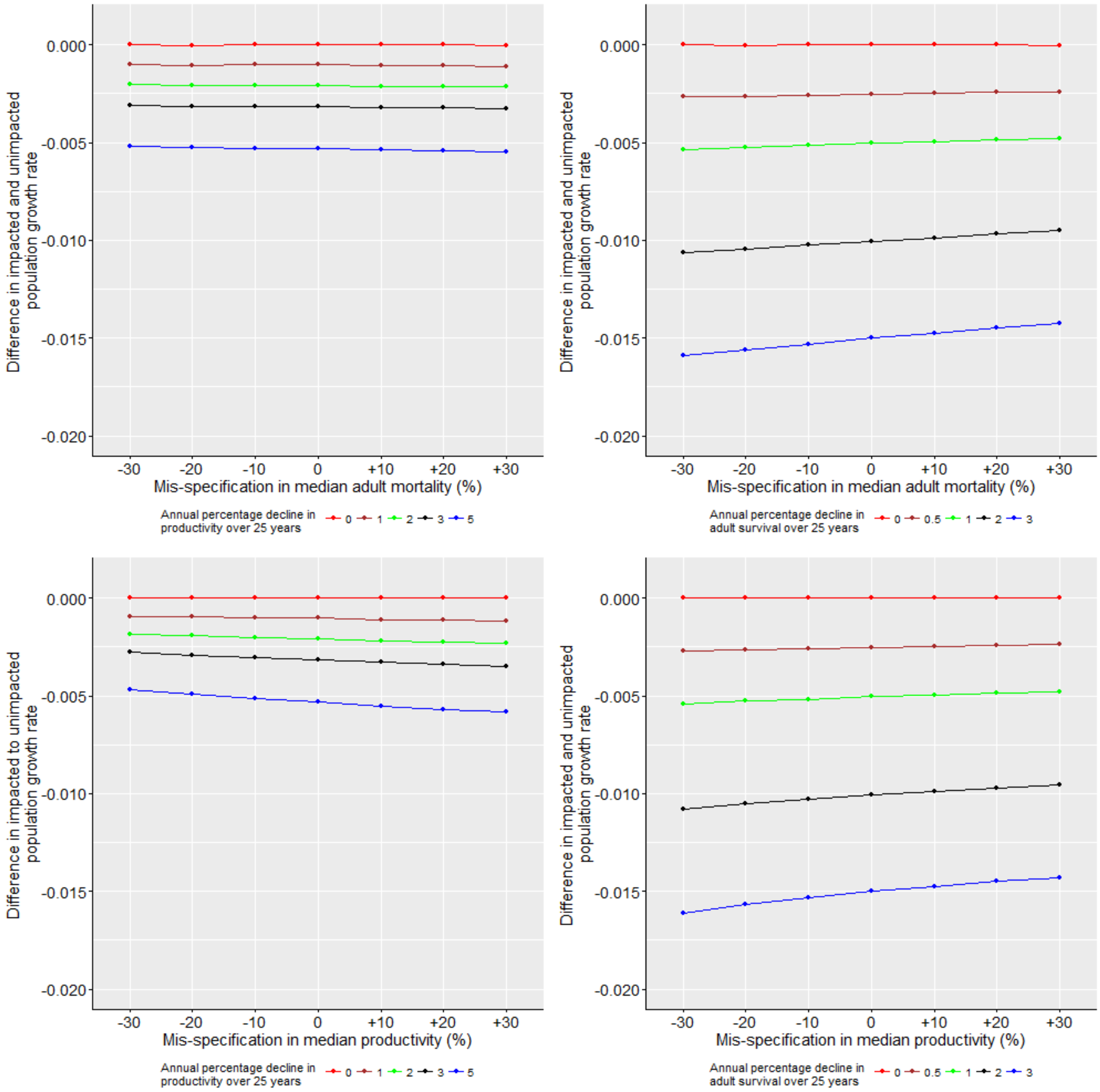


Figure A2.9d. PVA Metric D for Forth Razorbills – difference in population size at 2041, comparing impacted population vs. un-impacted population.

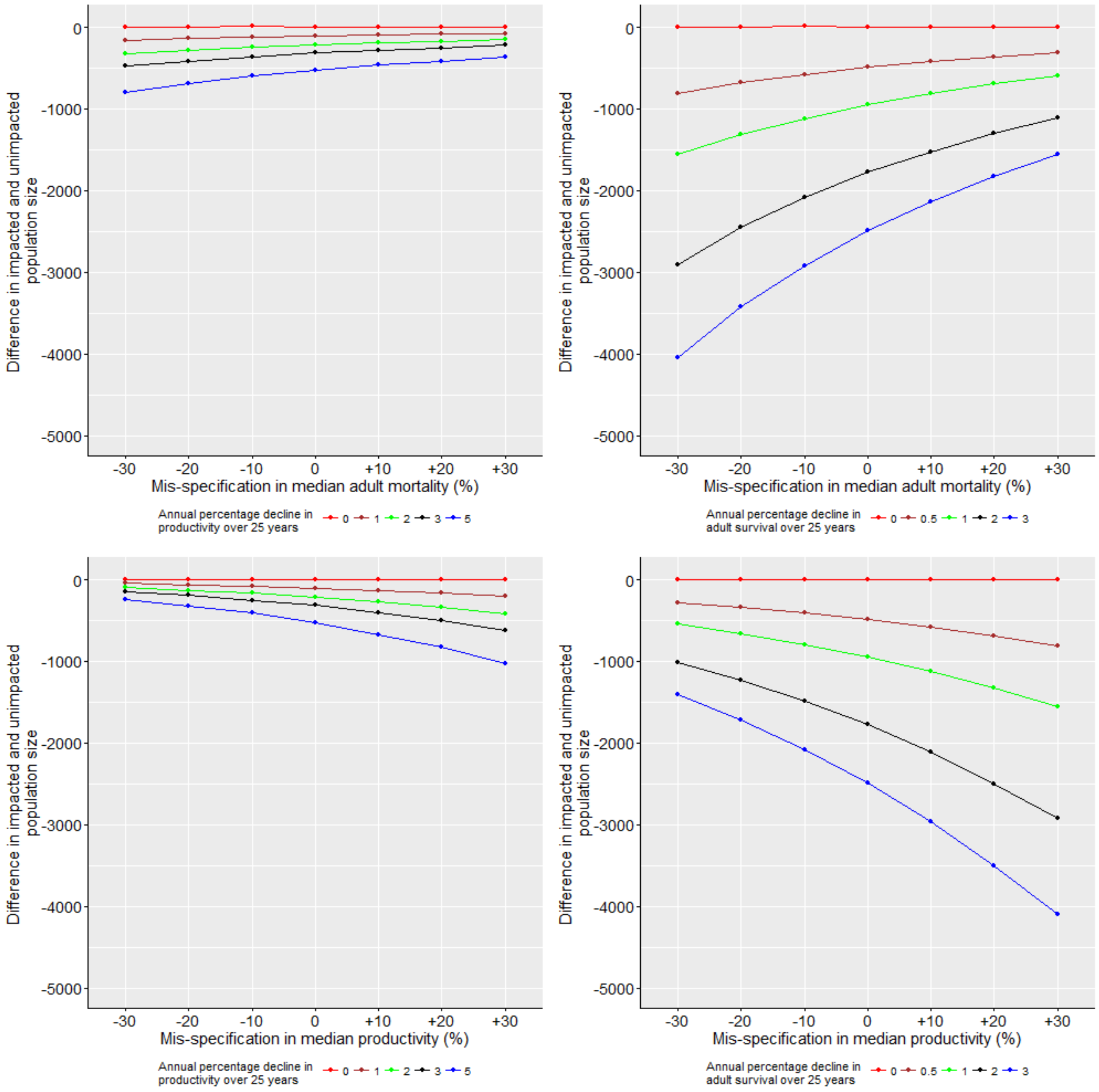


Figure A2.9e. PVA Metric E1 for Forth Razorbills – probability of population decline greater than 10% from 2016-2041.

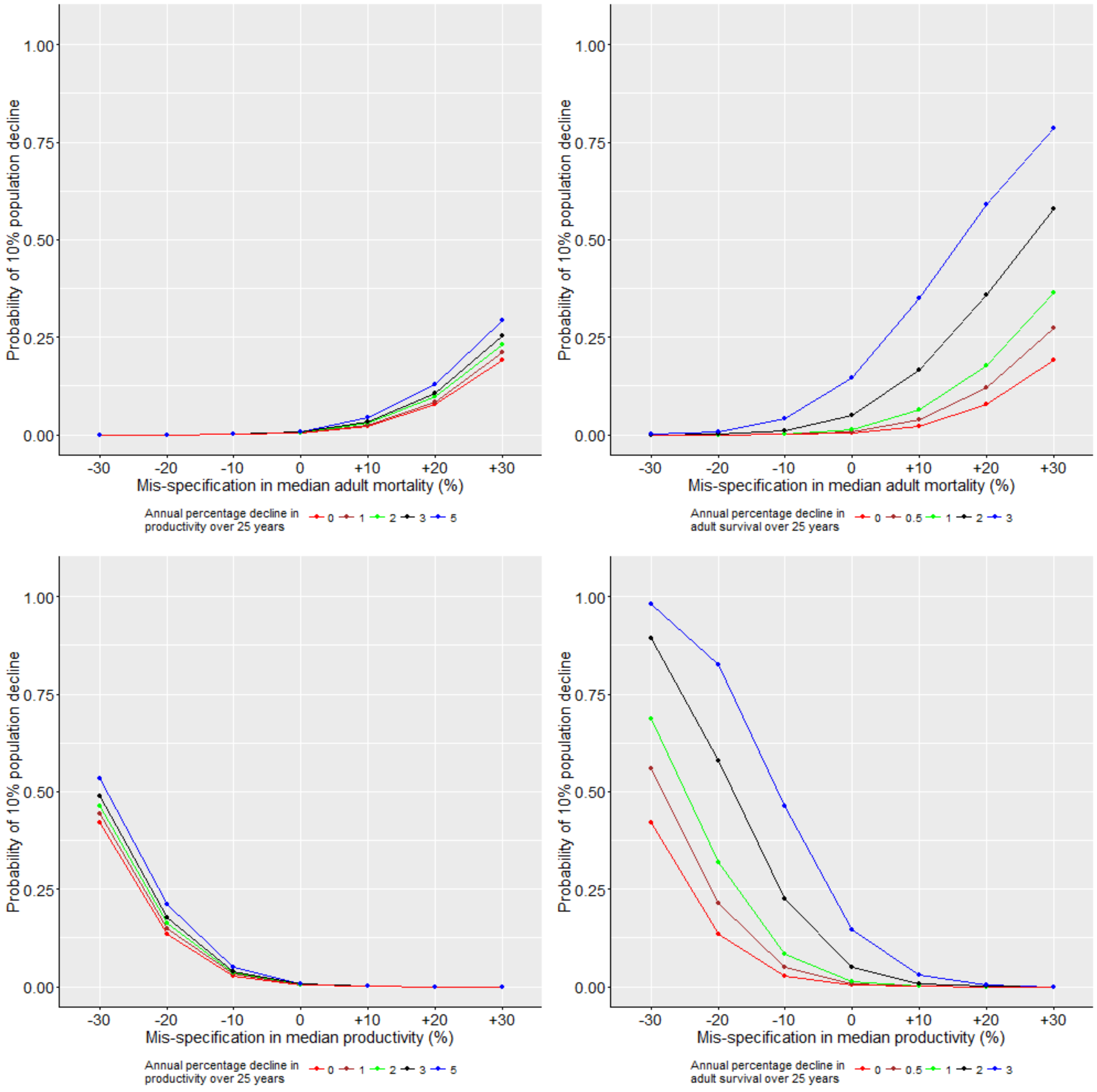


Figure A2.9f. PVA Metric E2 for Forth Razorbills – probability of population decline greater than 25% from 2016-2041.

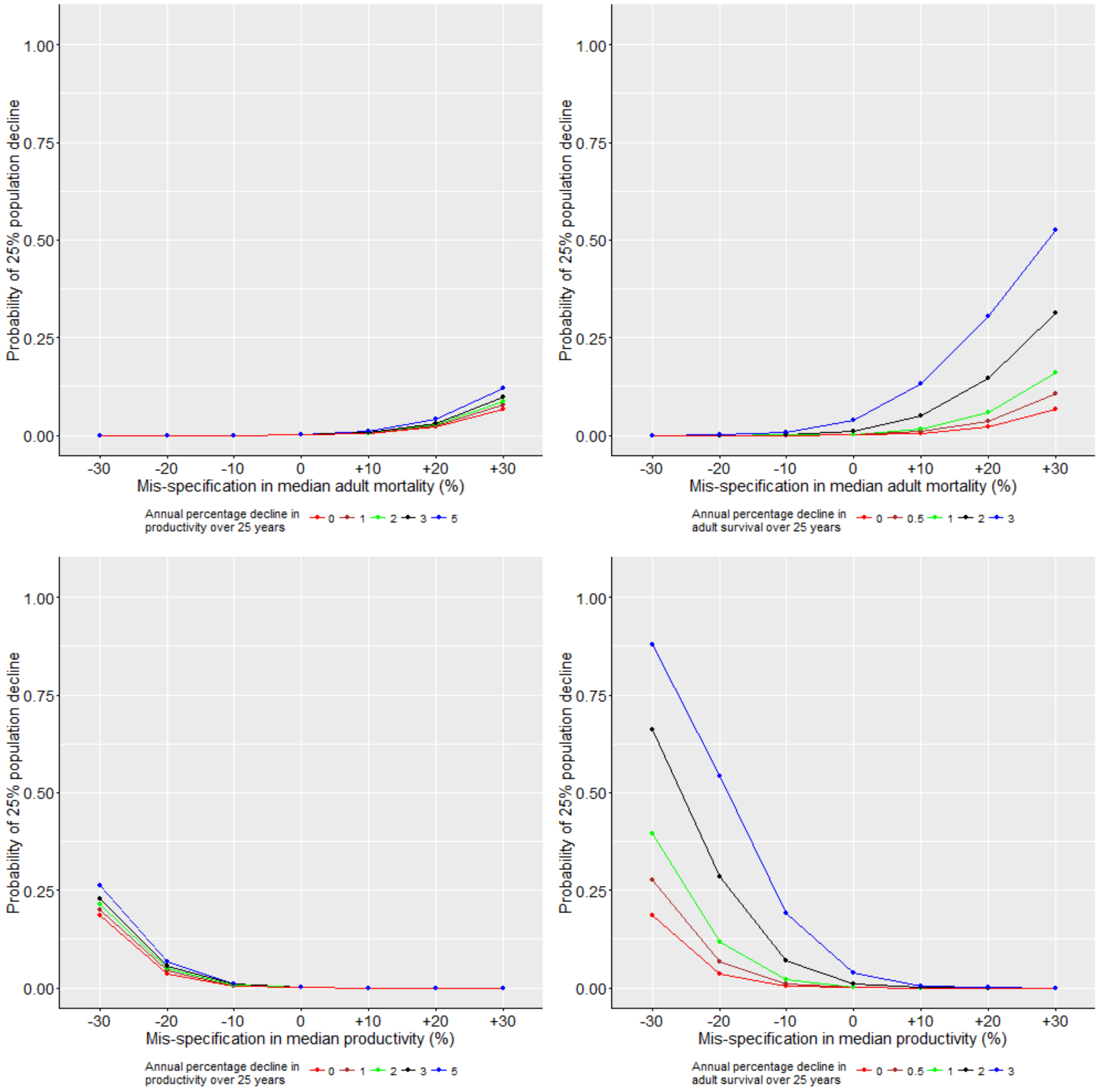


Figure A2.9g. PVA Metric E3 for Forth Razorbills – probability of population decline greater than 50% from 2016-2041.

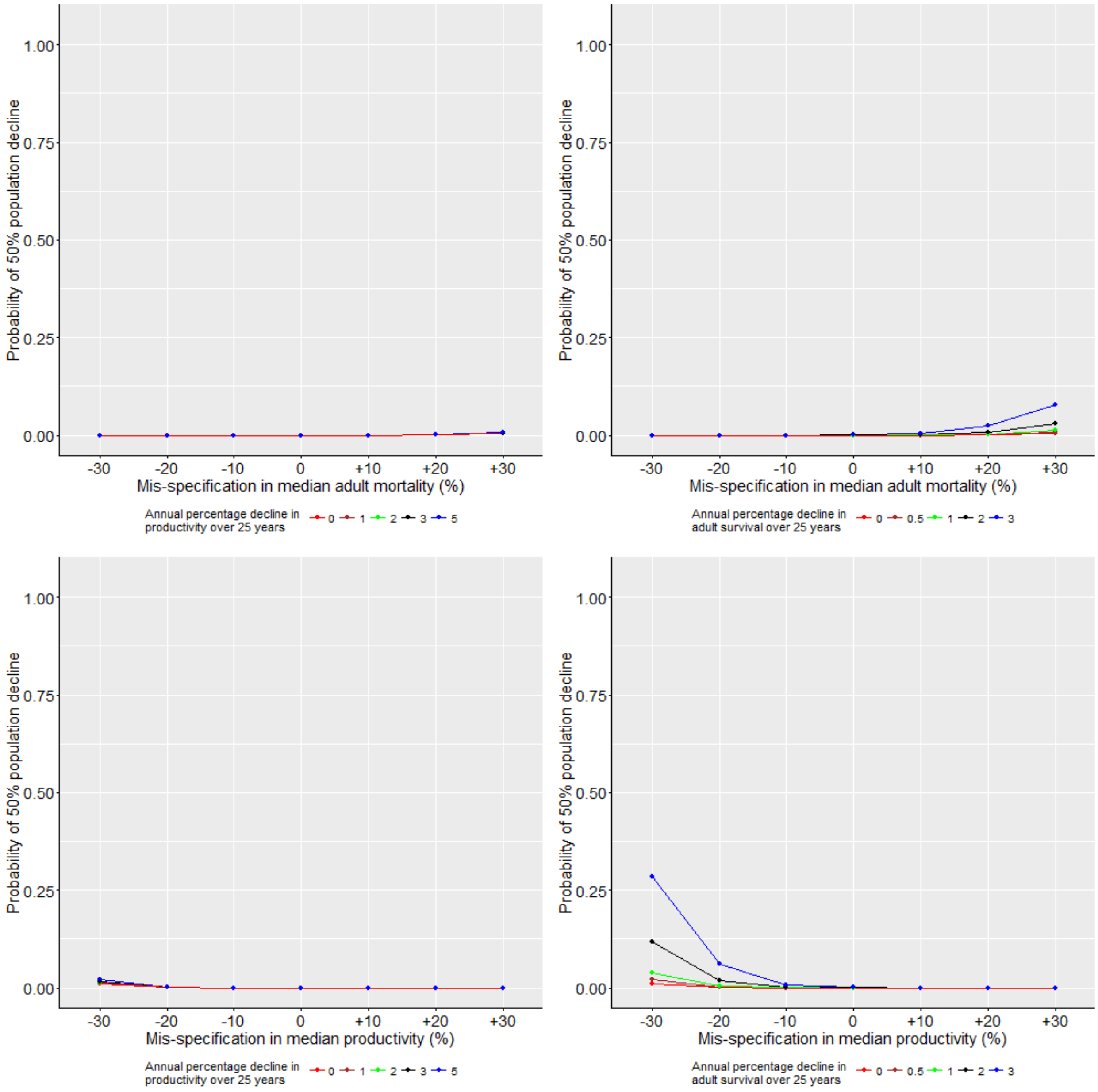
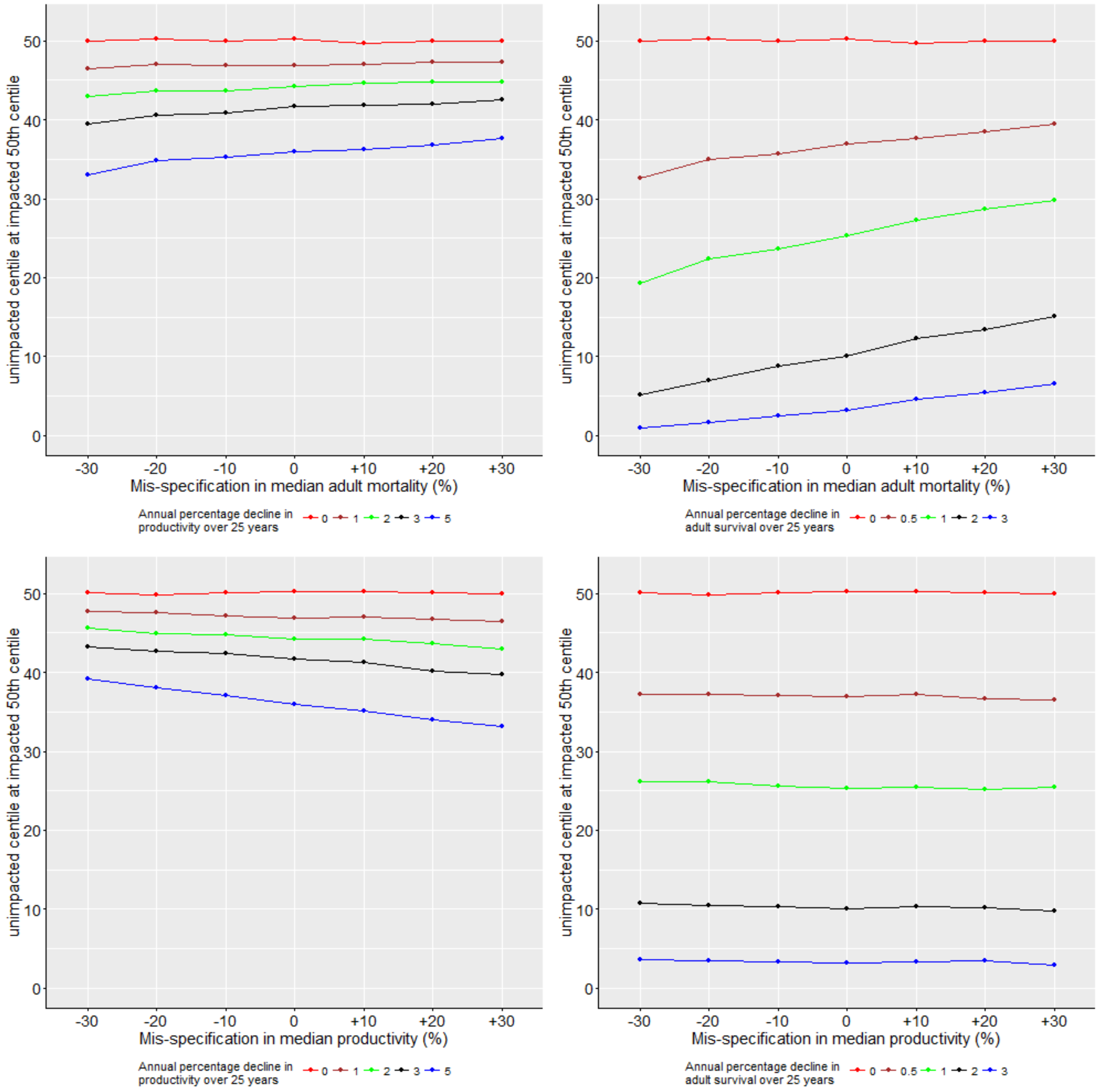


Figure A2.9h. PVA Metric F for Forth Razorbills – centile from un-impacted population size equal to the 50th centile of the impacted population size, at 2041.



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.

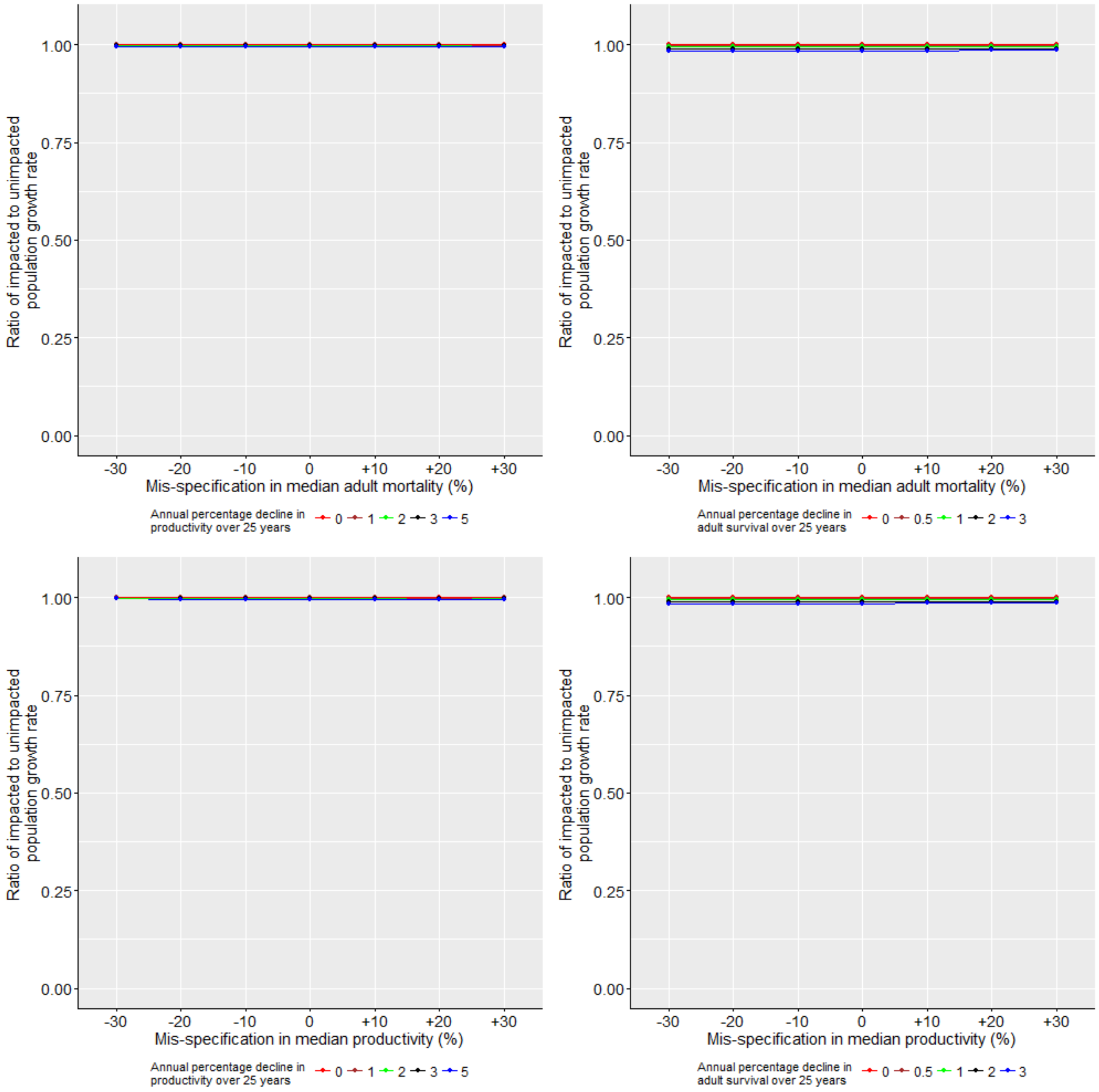


Figure A2.10b. PVA Metric B for St Abb’s Razorbills – ratio of population size at 2041, comparing impacted population vs. un-impacted population.

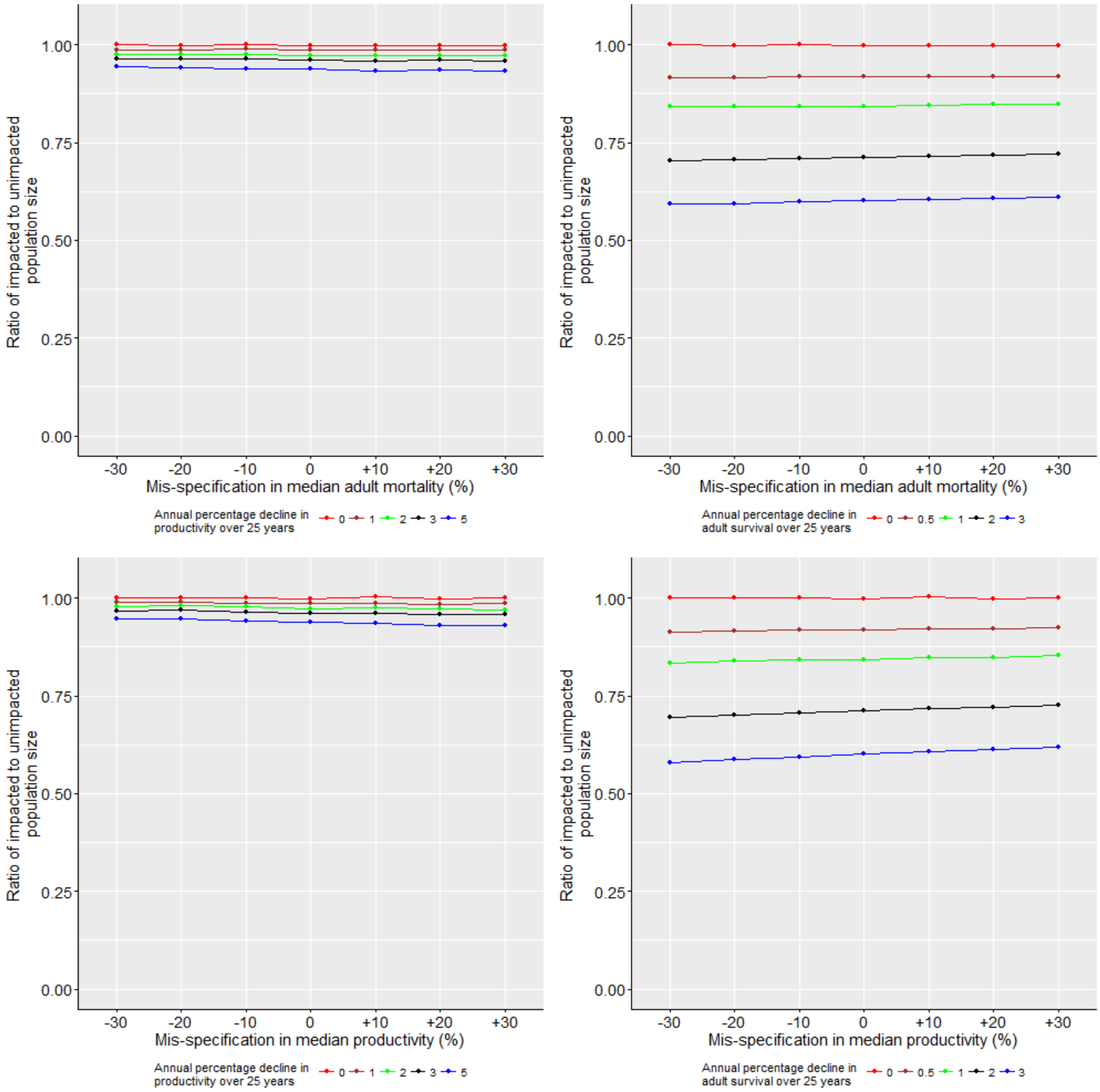


Figure A2.10c. PVA Metric C for St Abb's Razorbills – difference in population growth rate from 2016-2041, comparing impacted population vs. un-impacted population.

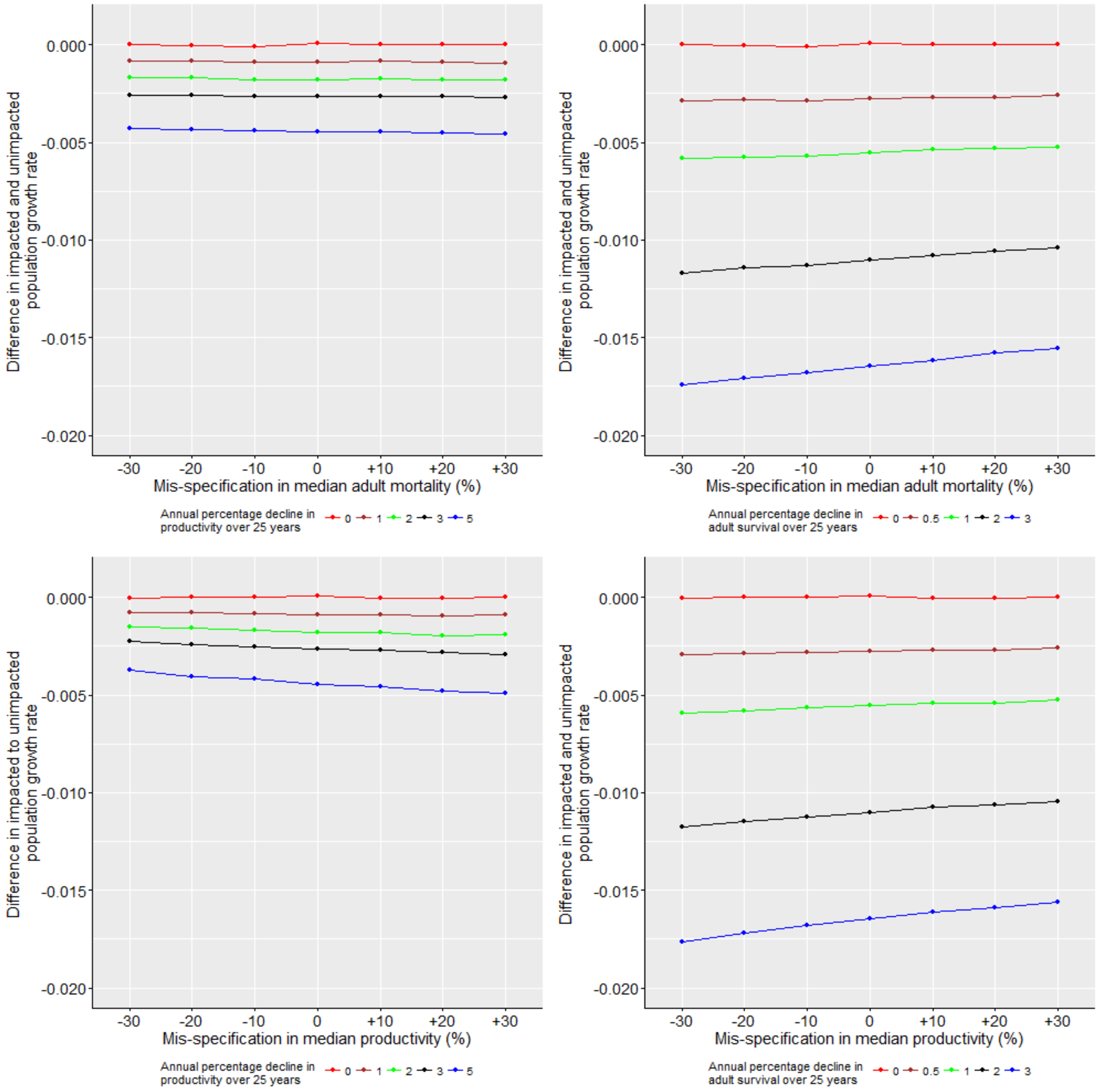


Figure A2.10d. PVA Metric D for St Abb's Razorbills – difference in population size at 2041, comparing impacted population vs. un-impacted population.

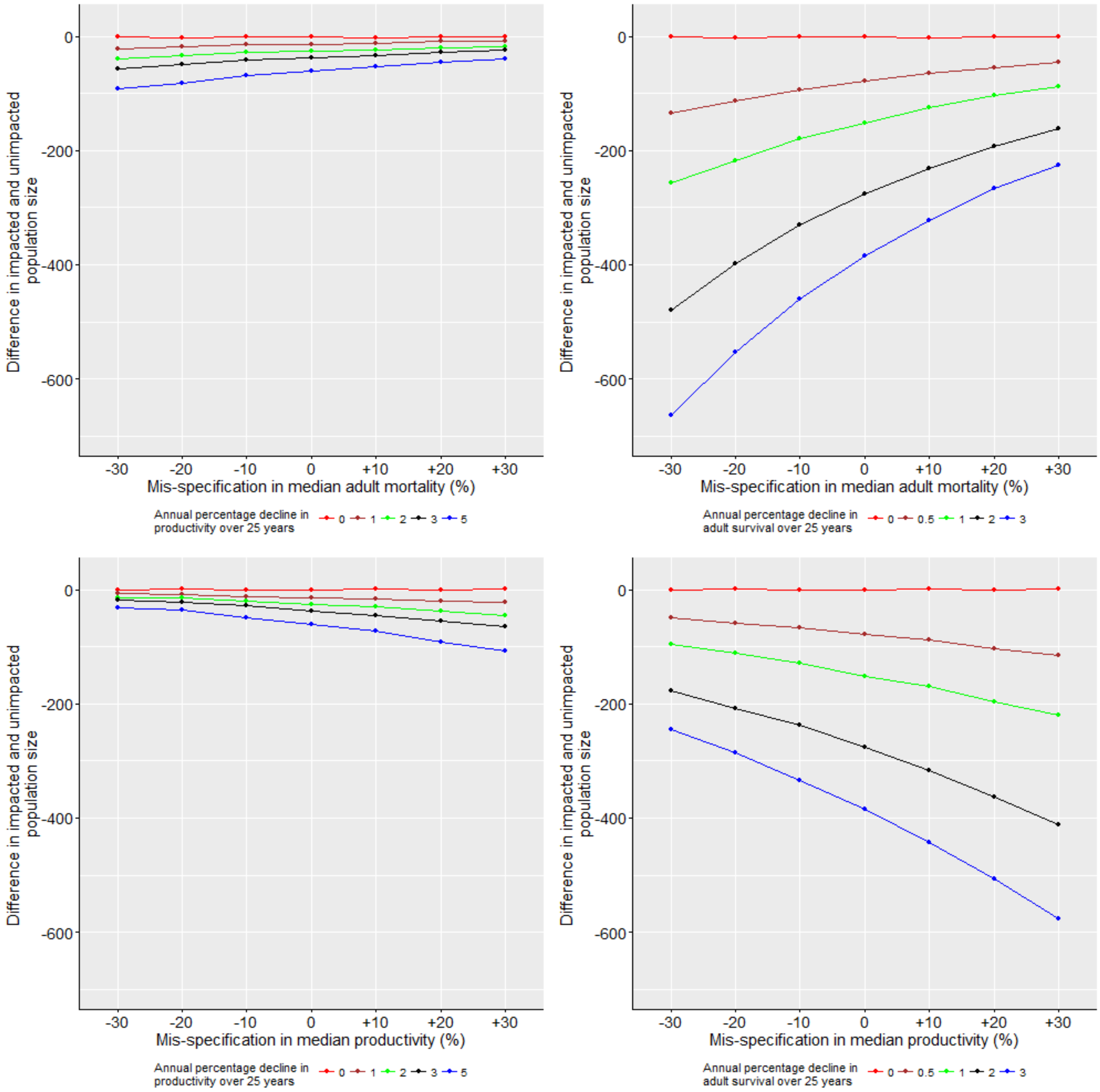


Figure A2.10e. PVA Metric E1 for St Abb's Razorbills – probability of population decline greater than 10% from 2016-2041.

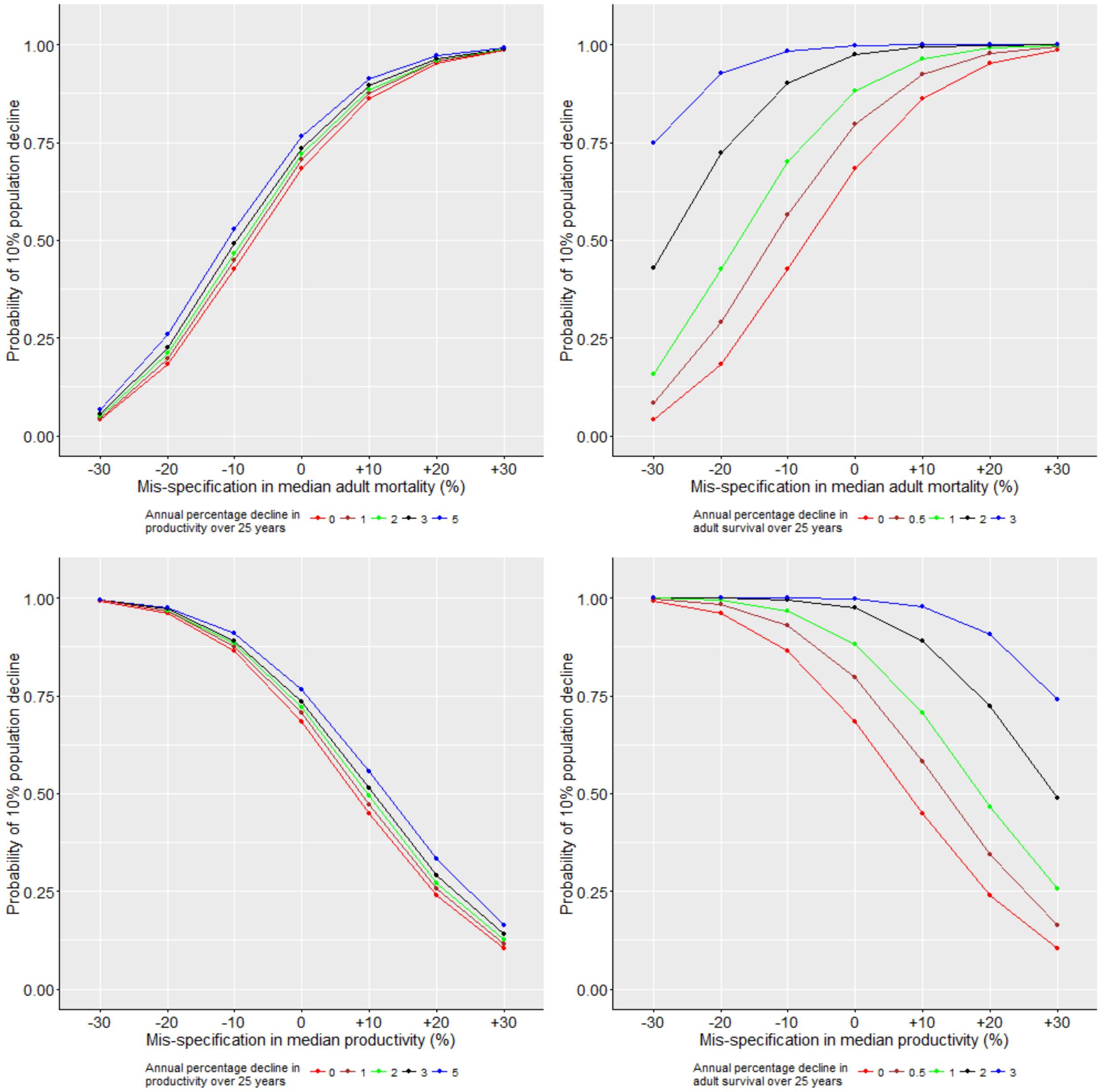


Figure A2.10f. PVA Metric E2 for St Abb’s Razorbills – probability of population decline greater than 25% from 2016-2041.

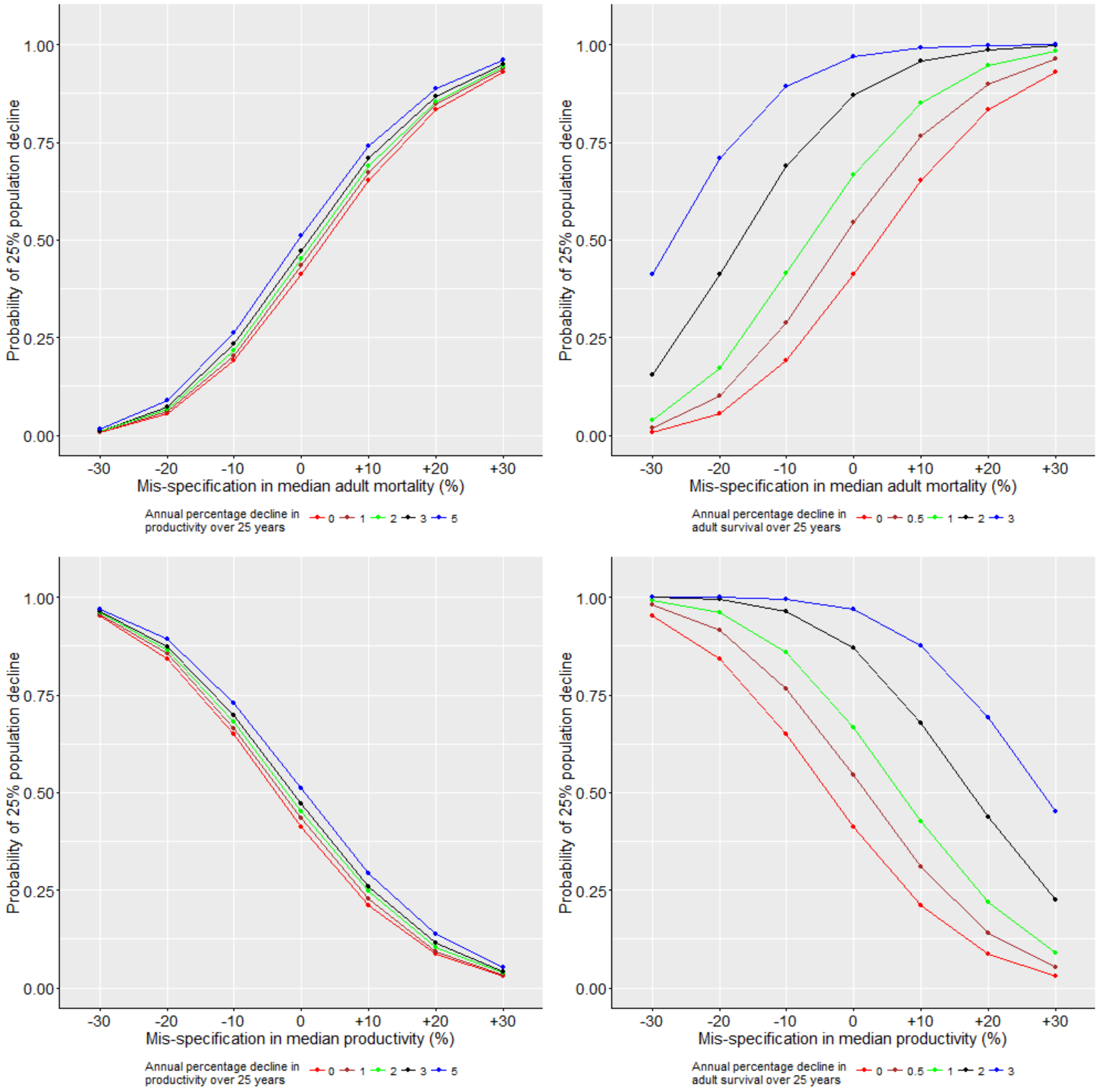


Figure A2.10g. PVA Metric E3 for St Abb’s Razorbills – probability of population decline greater than 50% from 2016-2041.

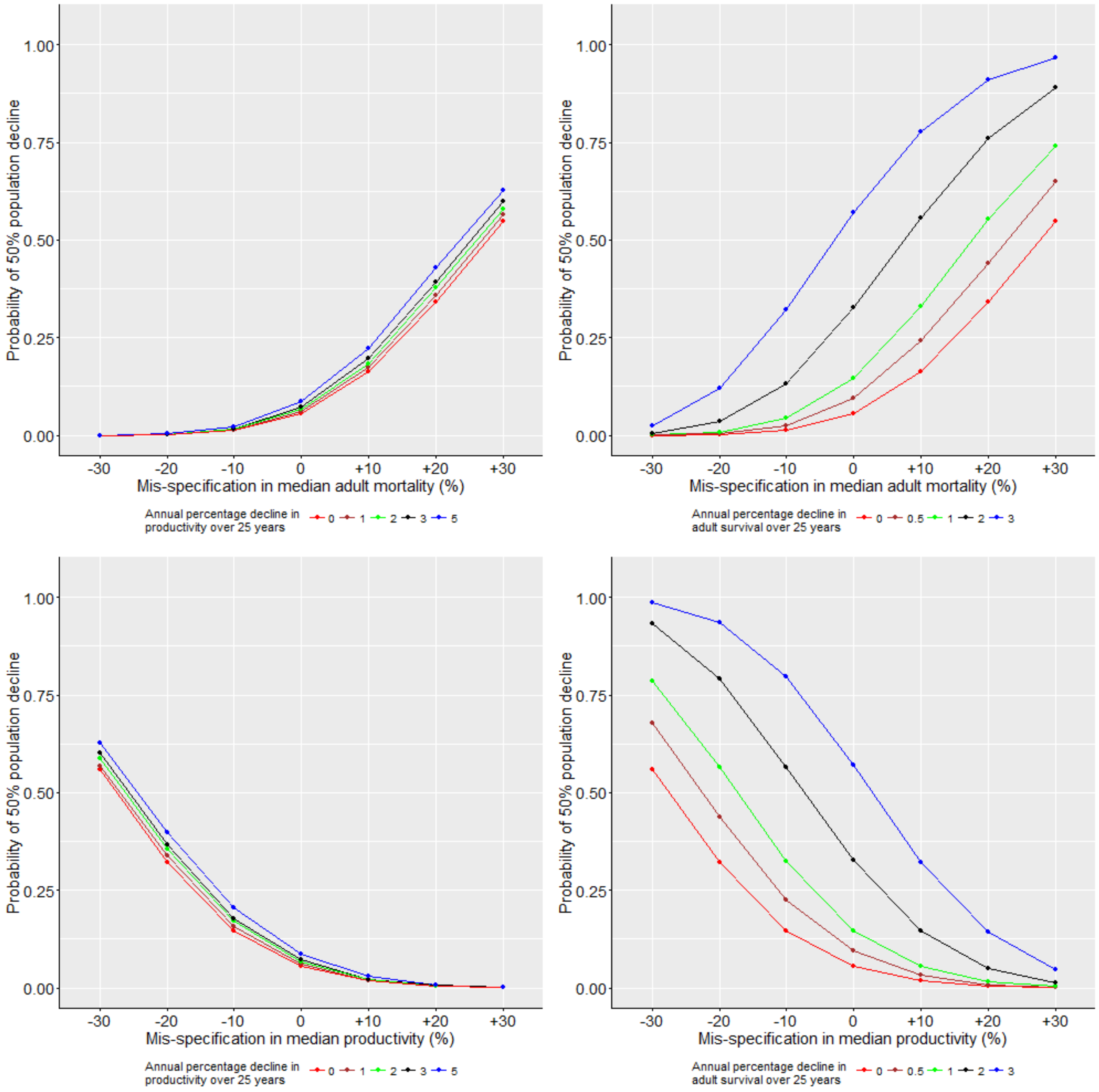
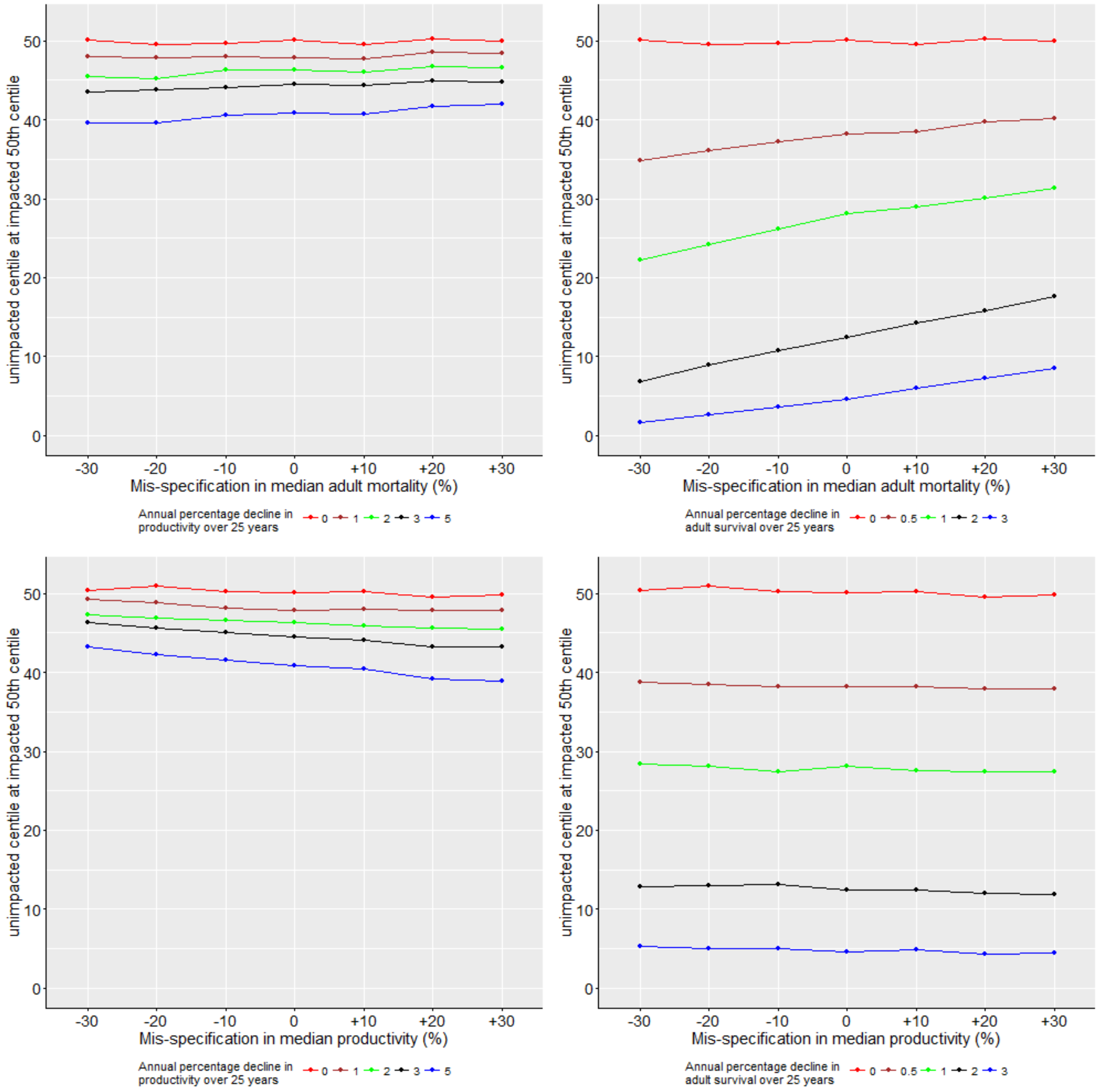


Figure A2.10h. PVA Metric F for St Abb’s Razorbills – centile from un-impacted population size equal to the 50th centile of the impacted population size, at 2041.



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.

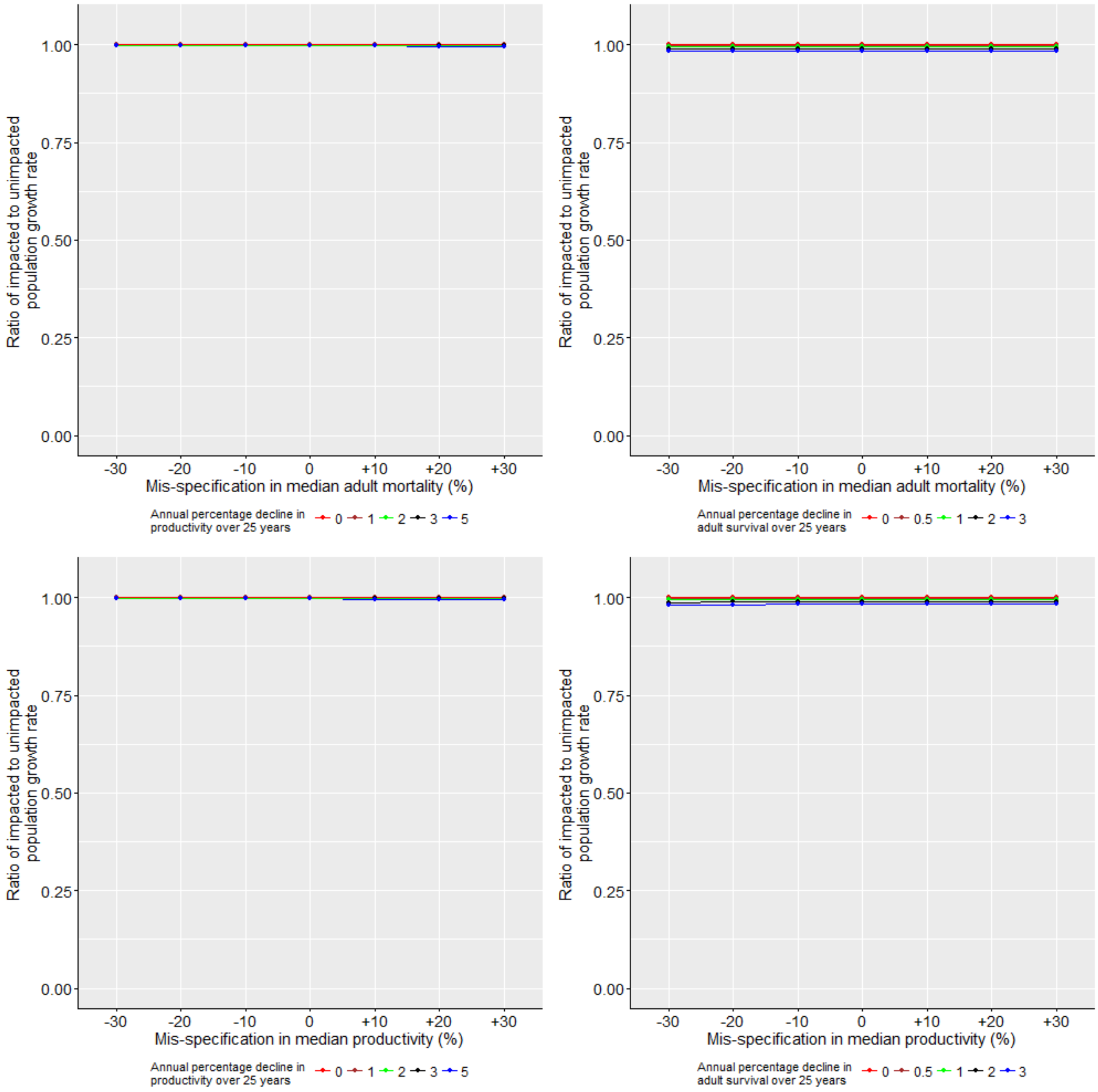


Figure A2.11b. PVA Metric B for Fowlsheugh Razorbills – ratio of population size at 2041, comparing impacted population vs. un-impacted population.

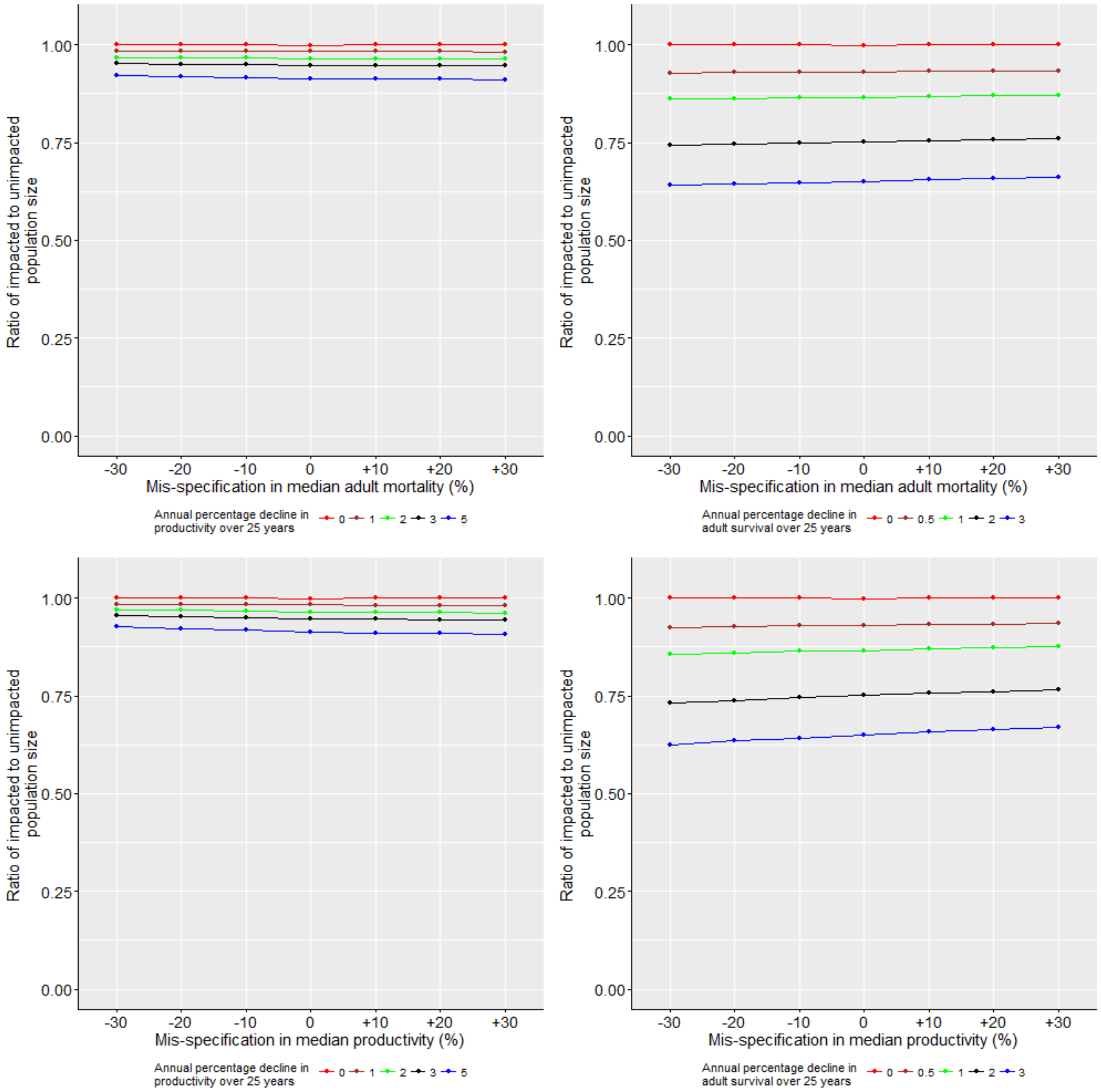


Figure A2.11c. PVA Metric C for Fowlsheugh Razorbills – difference in population growth rate from 2016-2041, comparing impacted population vs. un-impacted population.

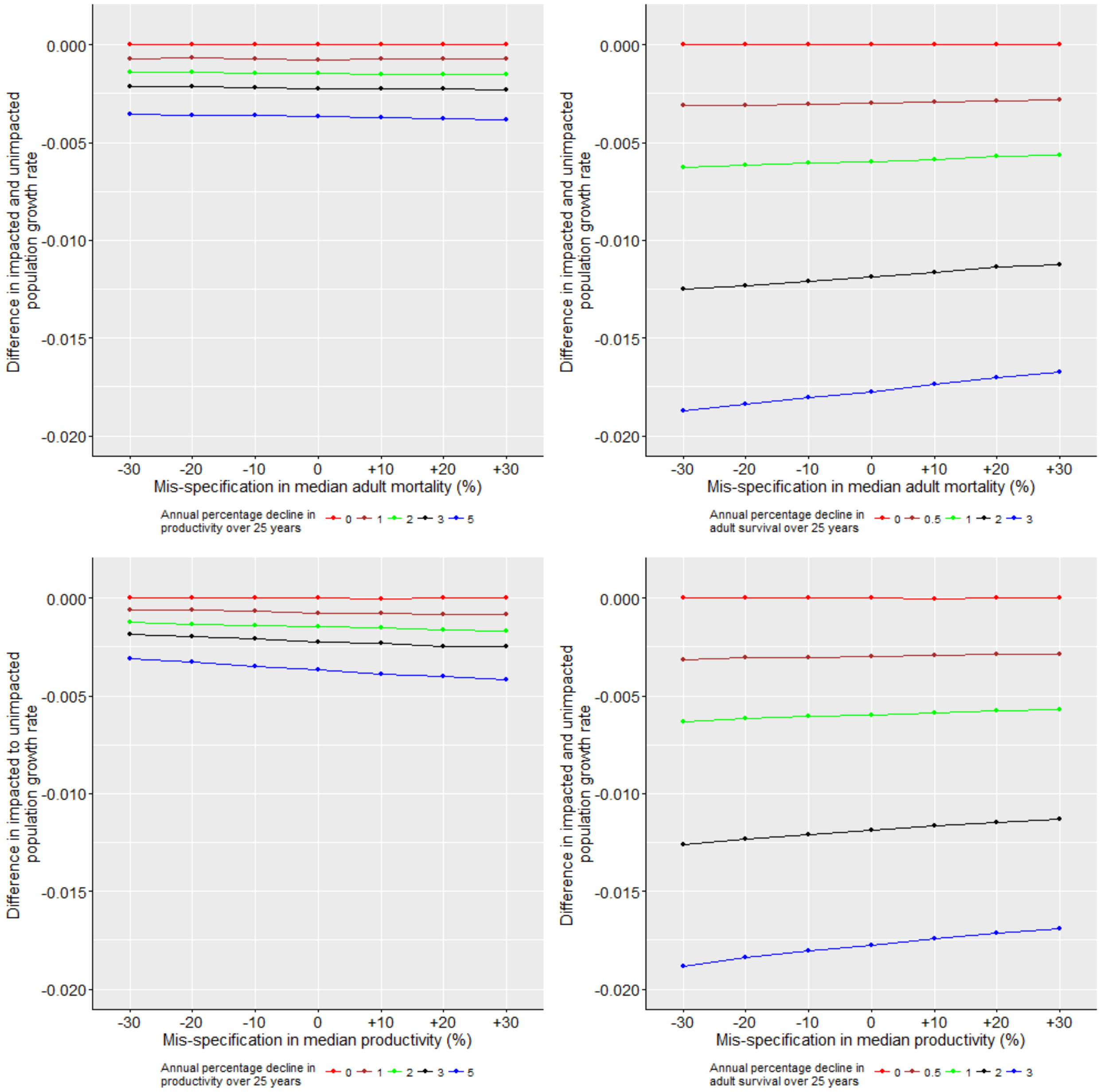


Figure A2.11d. PVA Metric D for Fowlsheugh Razorbills – difference in population size at 2041, comparing impacted population vs. un-impacted population.

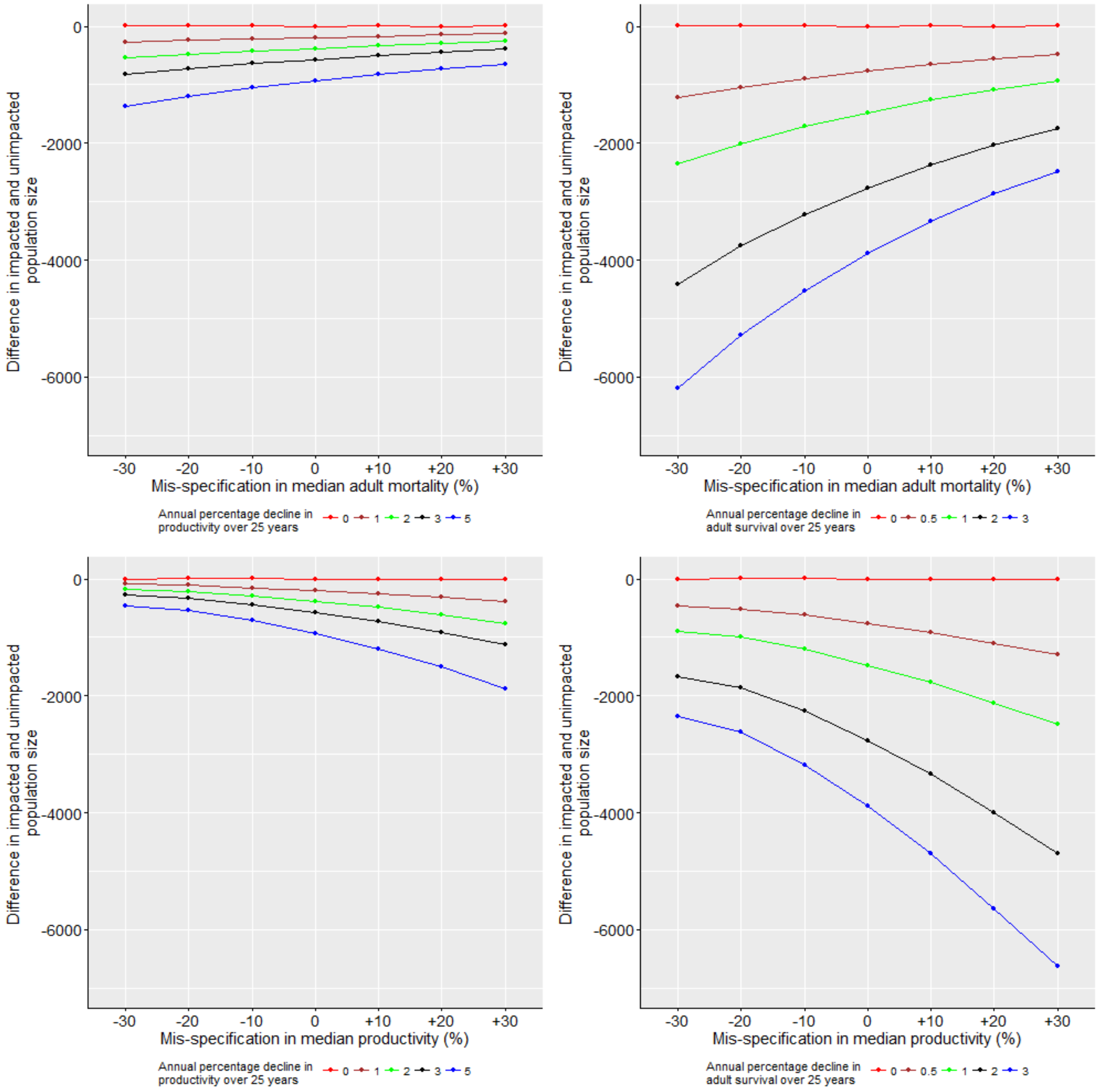


Figure A2.11e. PVA Metric E1 for Fowlsheugh Razorbills – probability of population decline greater than 10% from 2016-2041.

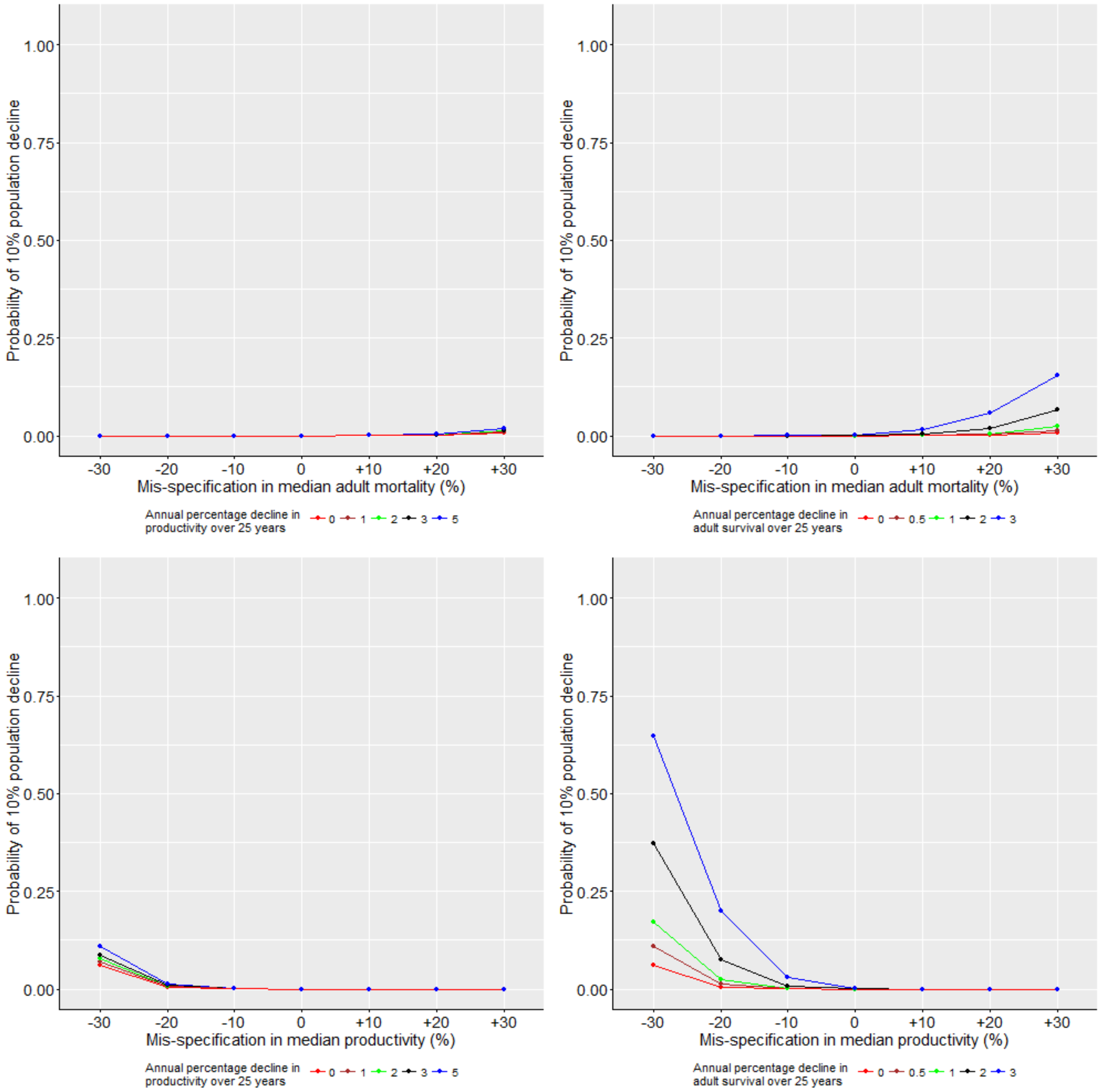


Figure A2.11f. PVA Metric E2 for Fowlsheugh Razorbills – probability of population decline greater than 25% from 2016-2041.

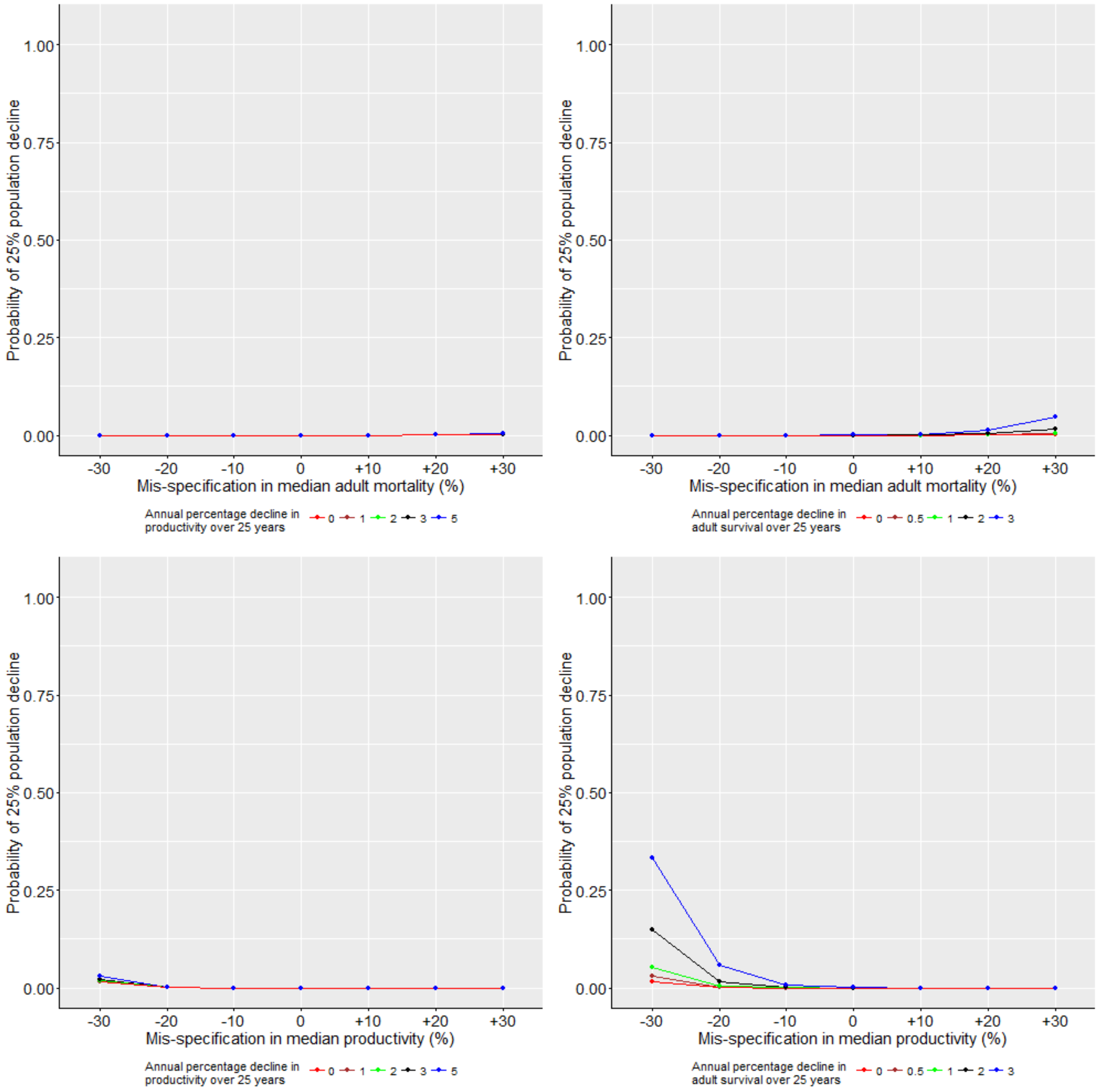


Figure A2.11g. PVA Metric E3 for Fowlsheugh Razorbills – probability of population decline greater than 50% from 2016-2041.

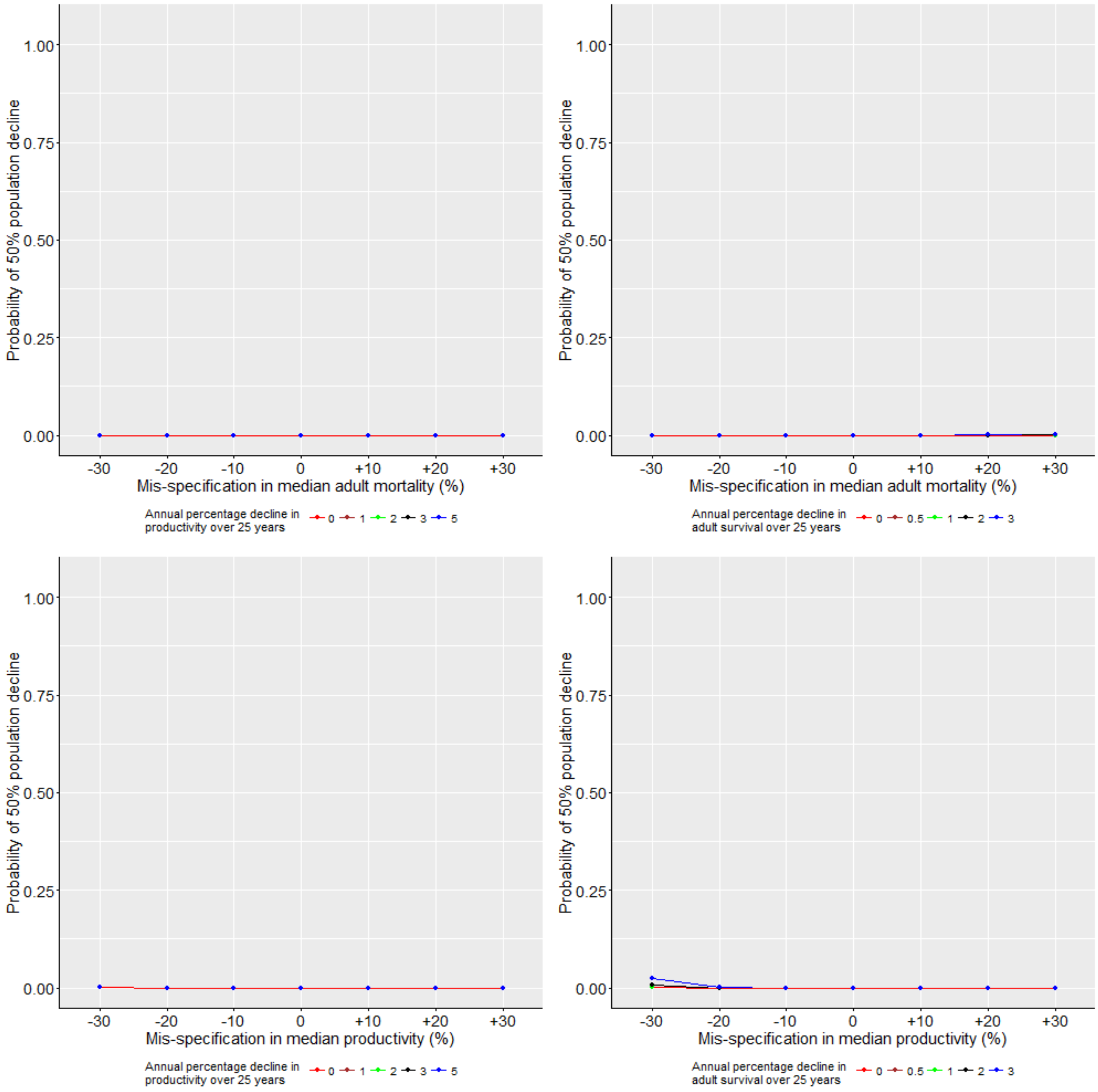
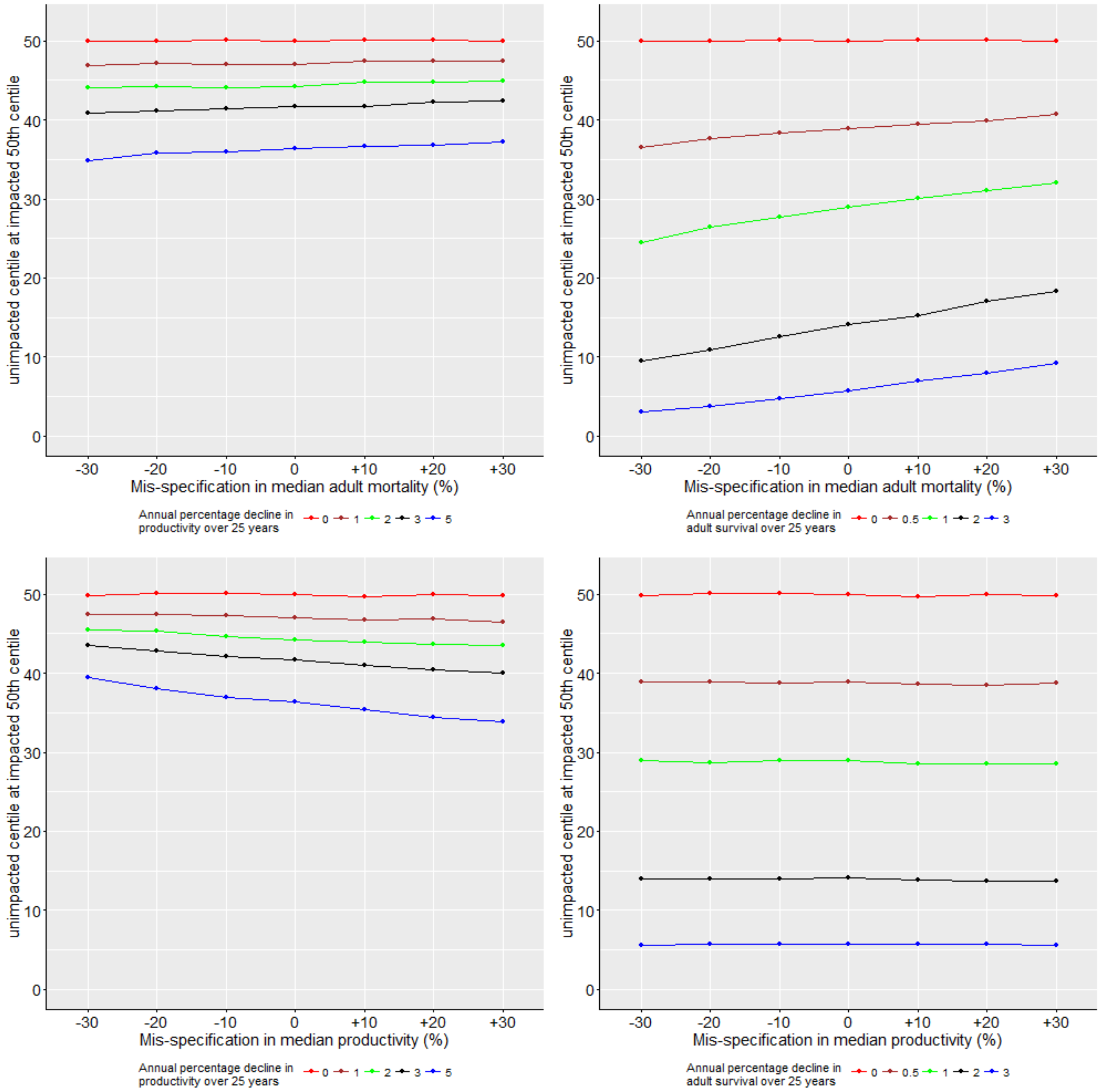


Figure A2.11h. PVA Metric F for Fowlsheugh Razorbills – centile from un-impacted population size equal to the 50th centile of the impacted population size, at 2041.



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.

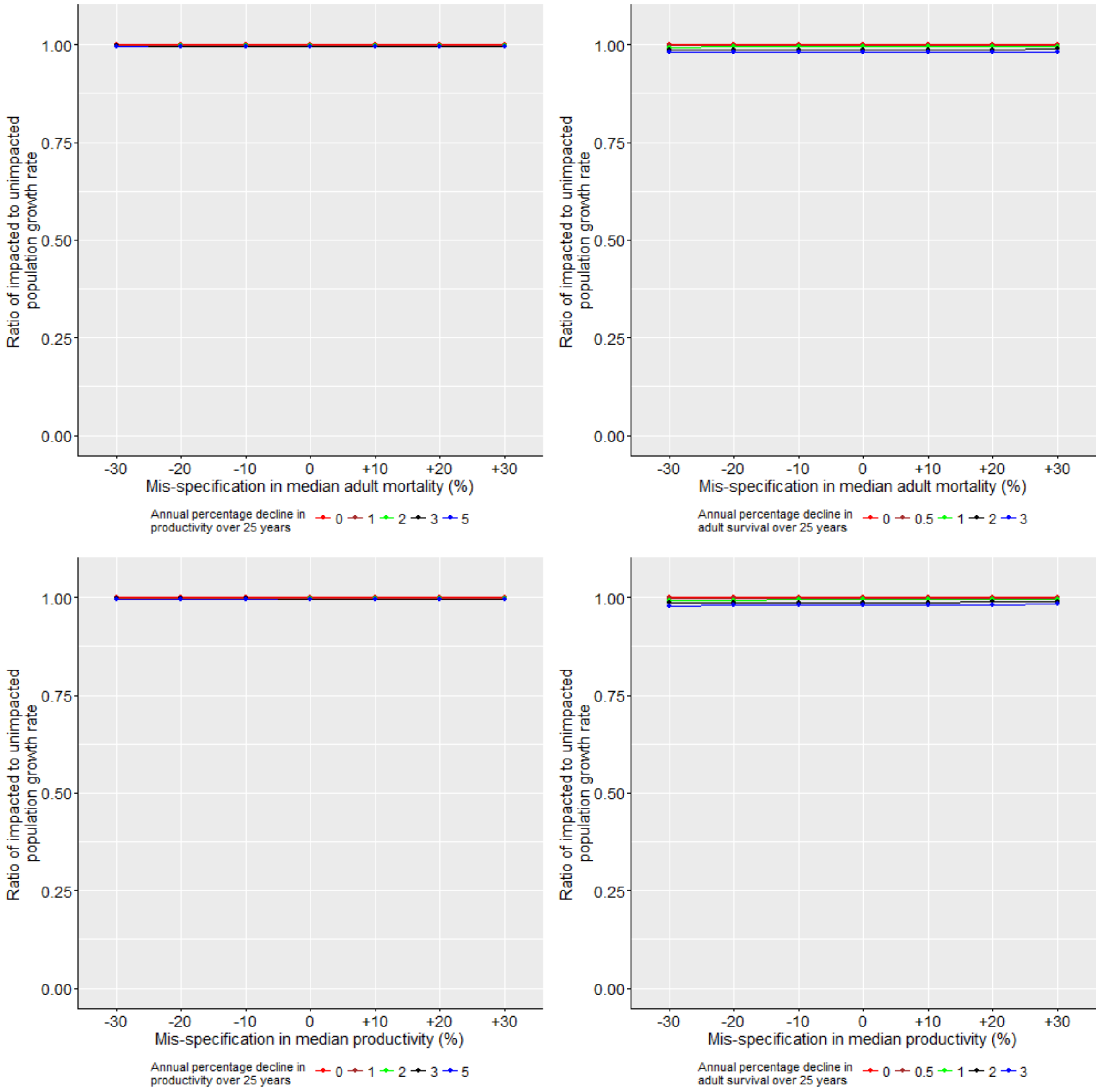


Figure A2.12b. PVA Metric B for Forth Shags – ratio of population size at 2041, comparing impacted population vs. un-impacted population.

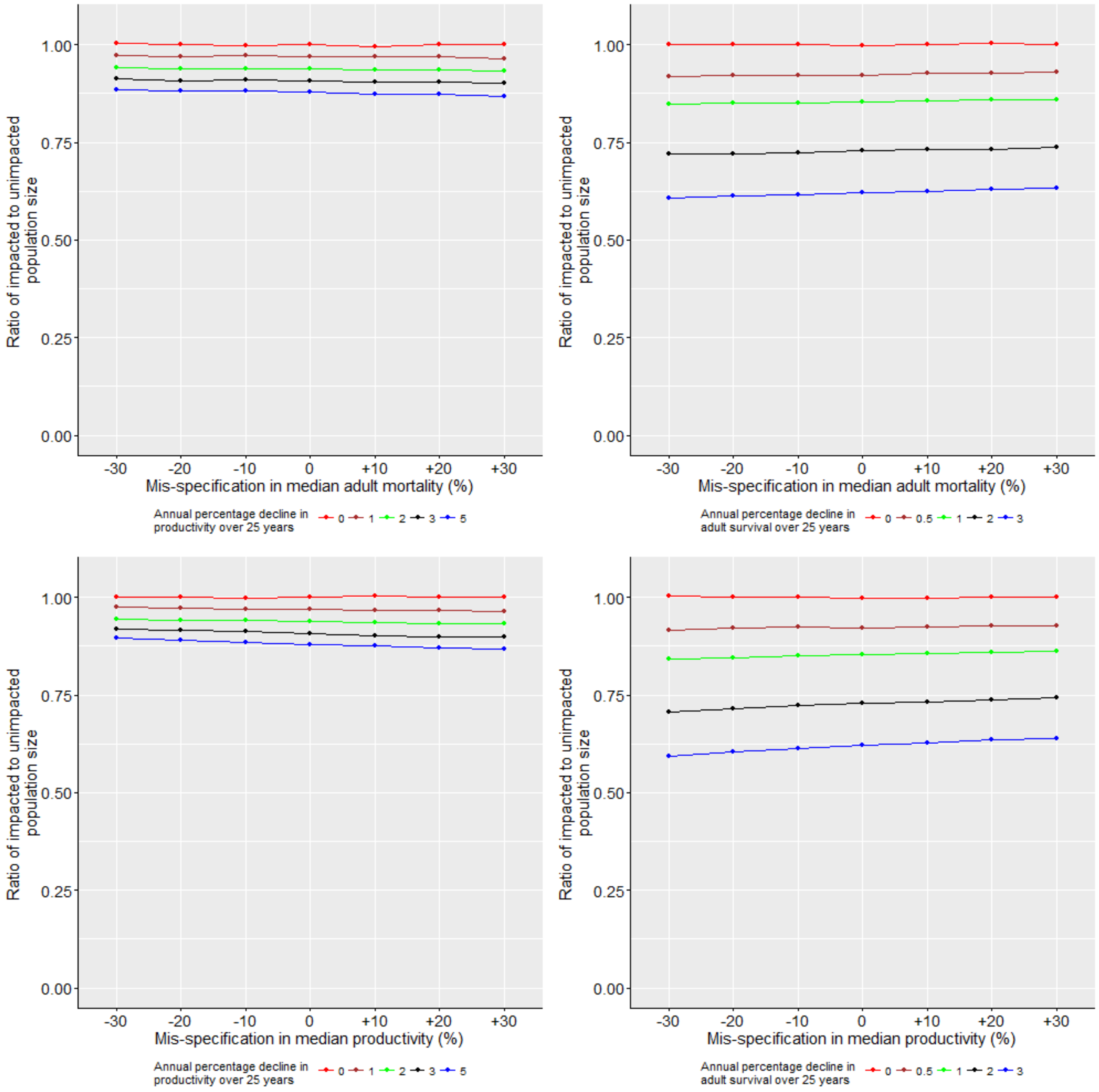


Figure A2.12c. PVA Metric C for Forth Shags – difference in population growth rate from 2016-2041, comparing impacted population vs. un-impacted population.

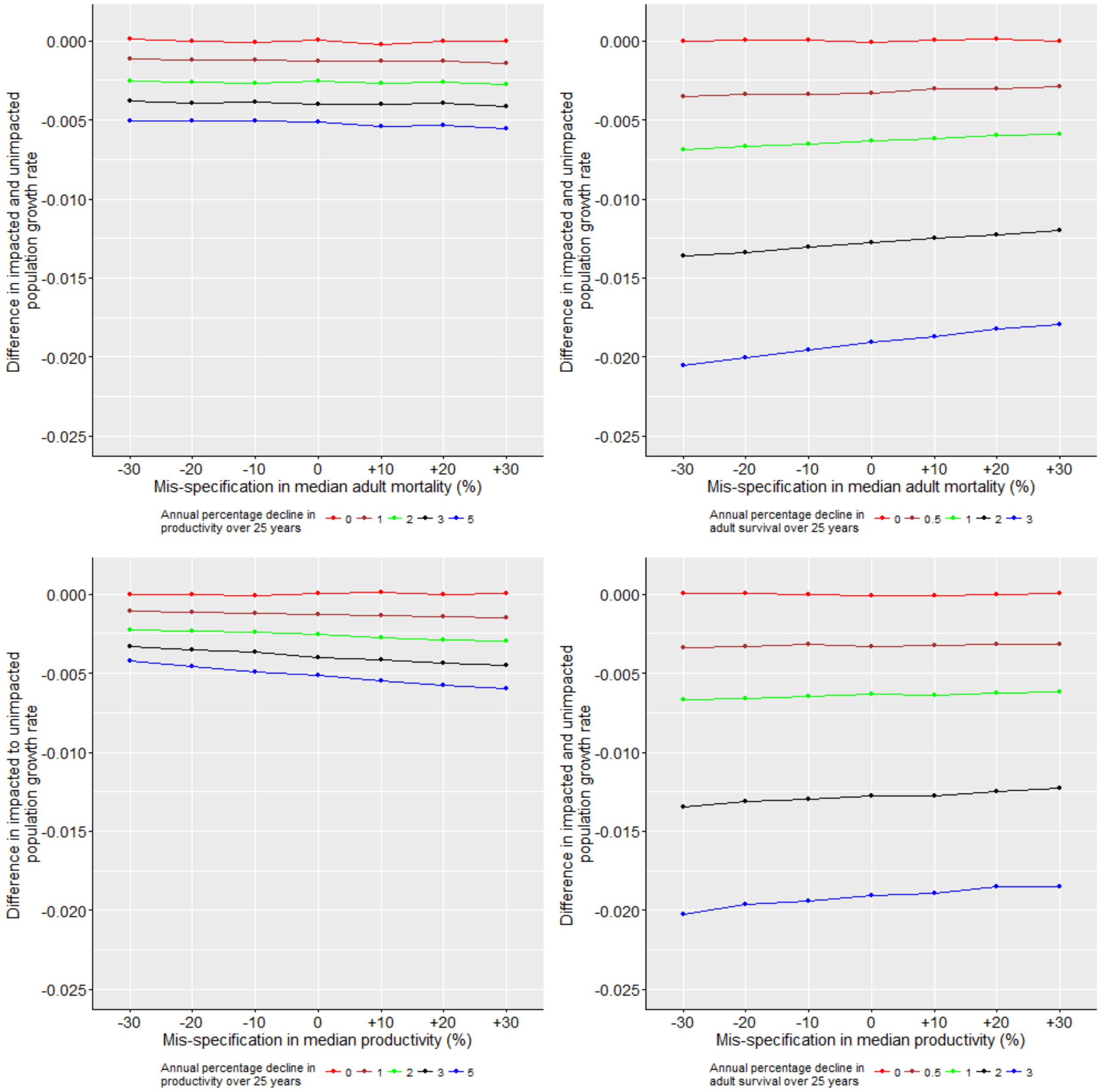


Figure A2.12d. PVA Metric D for Forth Shags – difference in population size at 2041, comparing impacted population vs. un-impacted population.

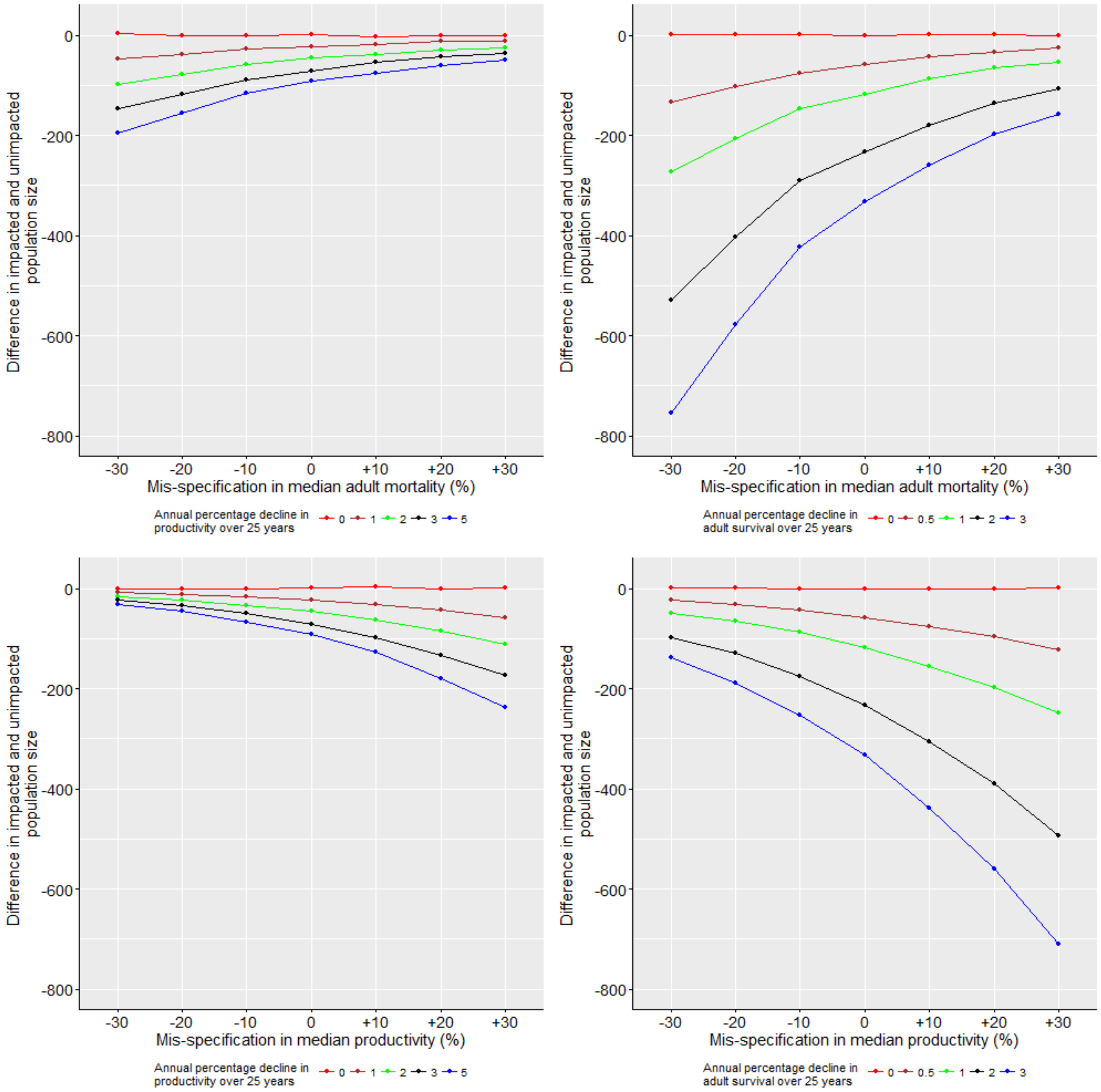


Figure A2.12e. PVA Metric E1 for Forth Shags – probability of population decline greater than 10% from 2016-2041.

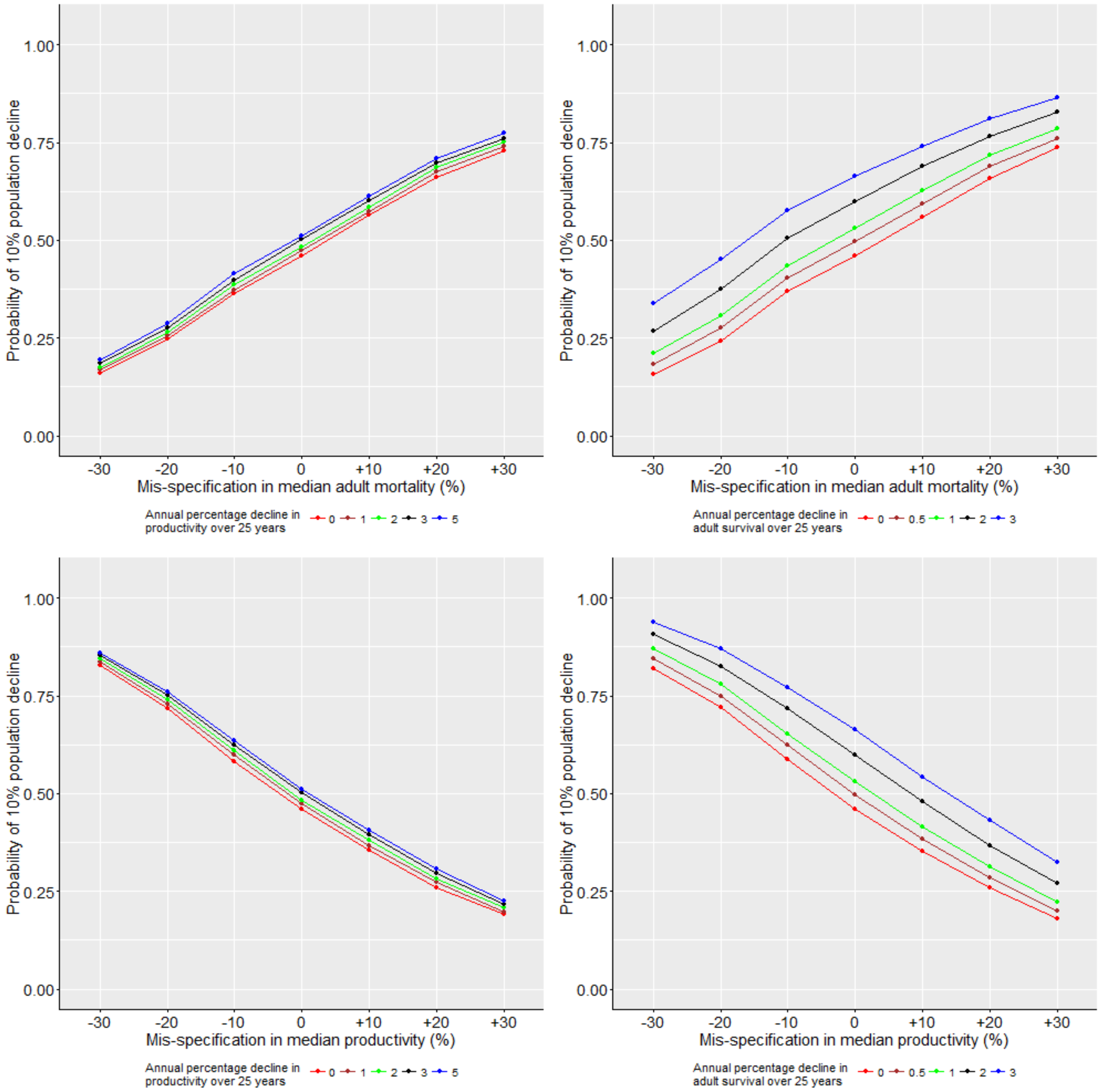


Figure A2.12f. PVA Metric E2 for Forth Shags – probability of population decline greater than 25% from 2016-2041.

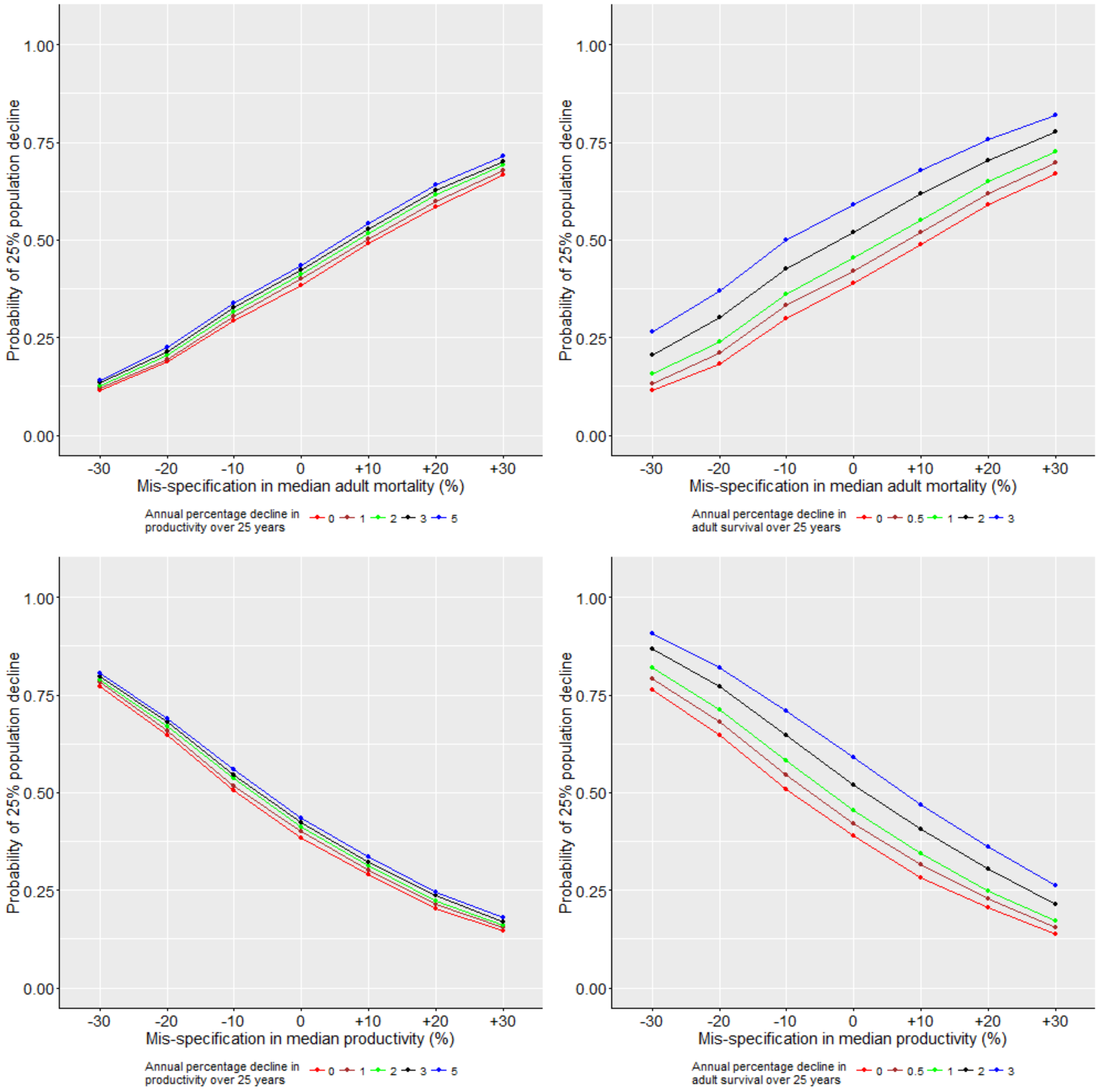


Figure A2.12g. PVA Metric E3 for Forth Shags – probability of population decline greater than 50% from 2016-2041.

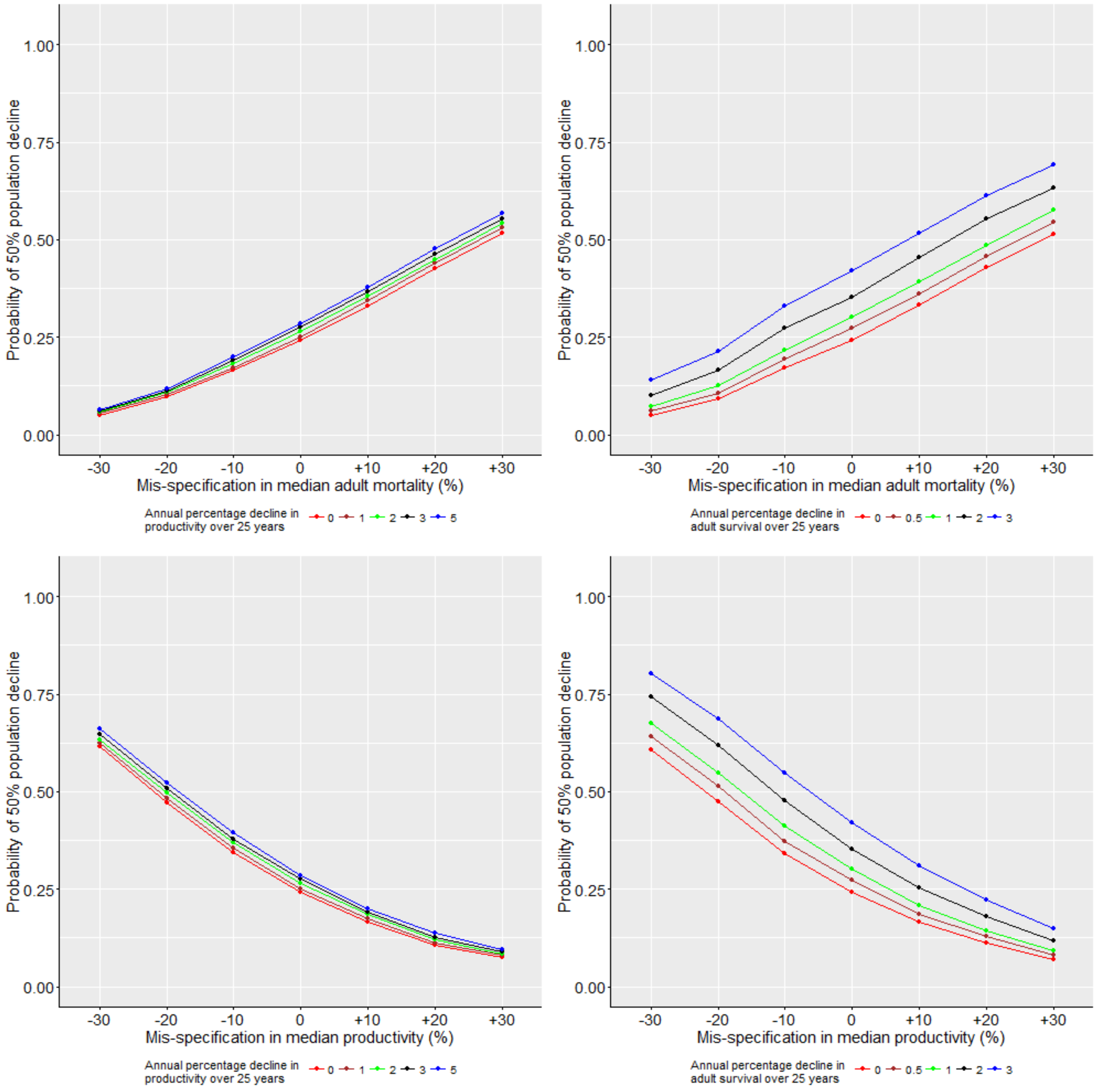
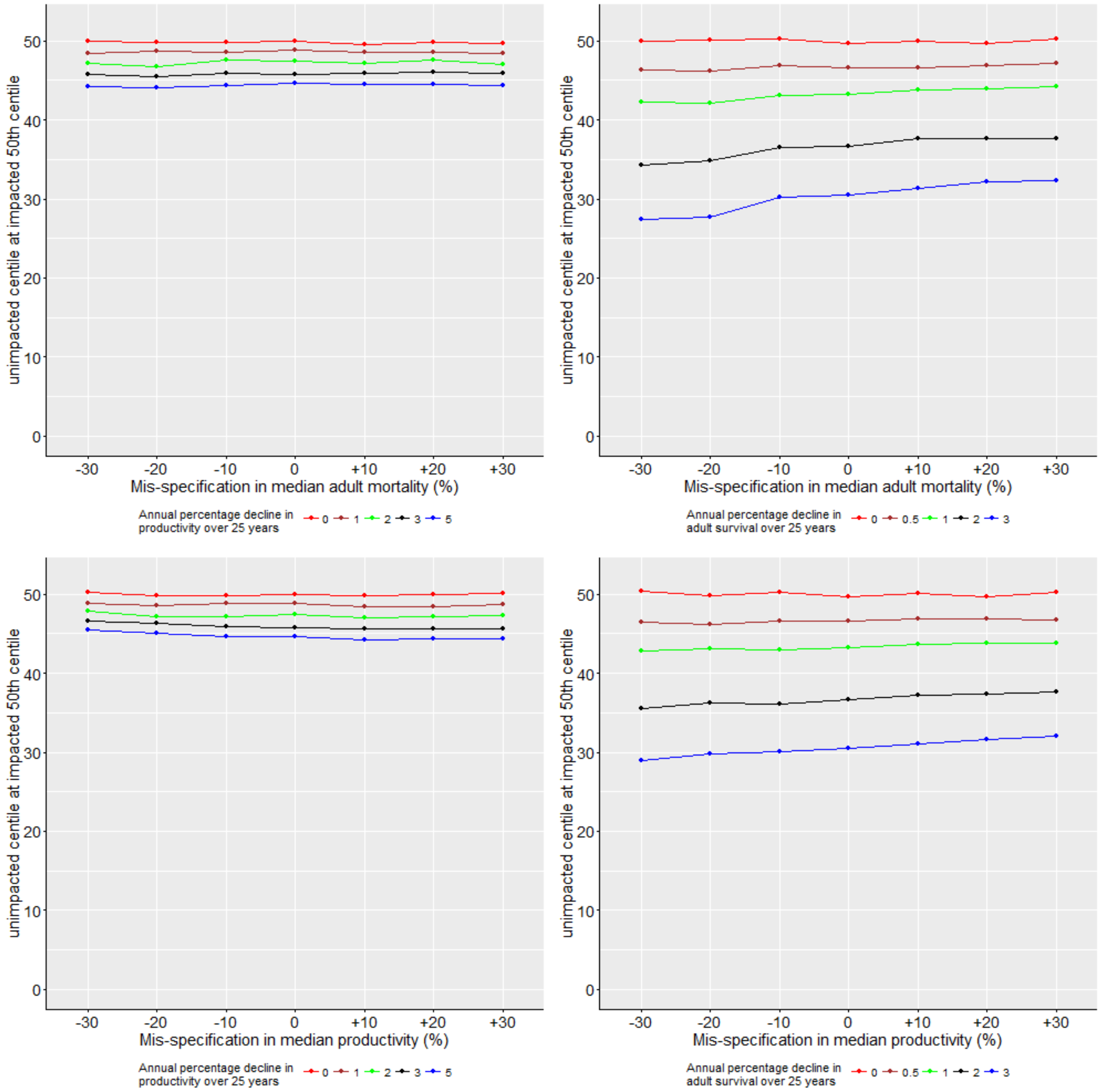


Figure A2.12h. PVA Metric F for Forth Shags – centile from un-impacted population size equal to the 50th centile of the impacted population size, at 2041.



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.

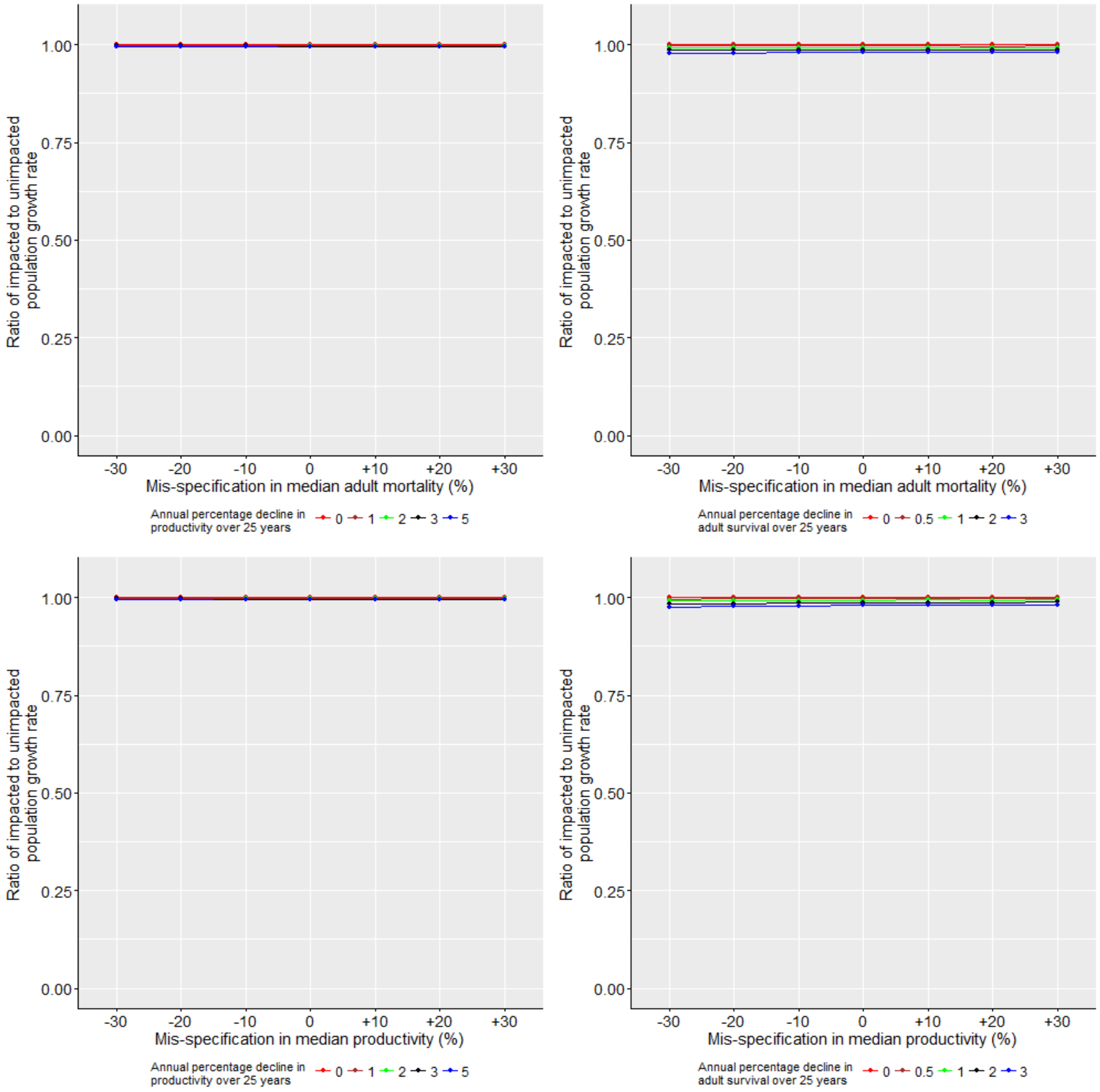


Figure A2.13b. PVA Metric B for St Abb's Shags – ratio of population size at 2041, comparing impacted population vs. un-impacted population.

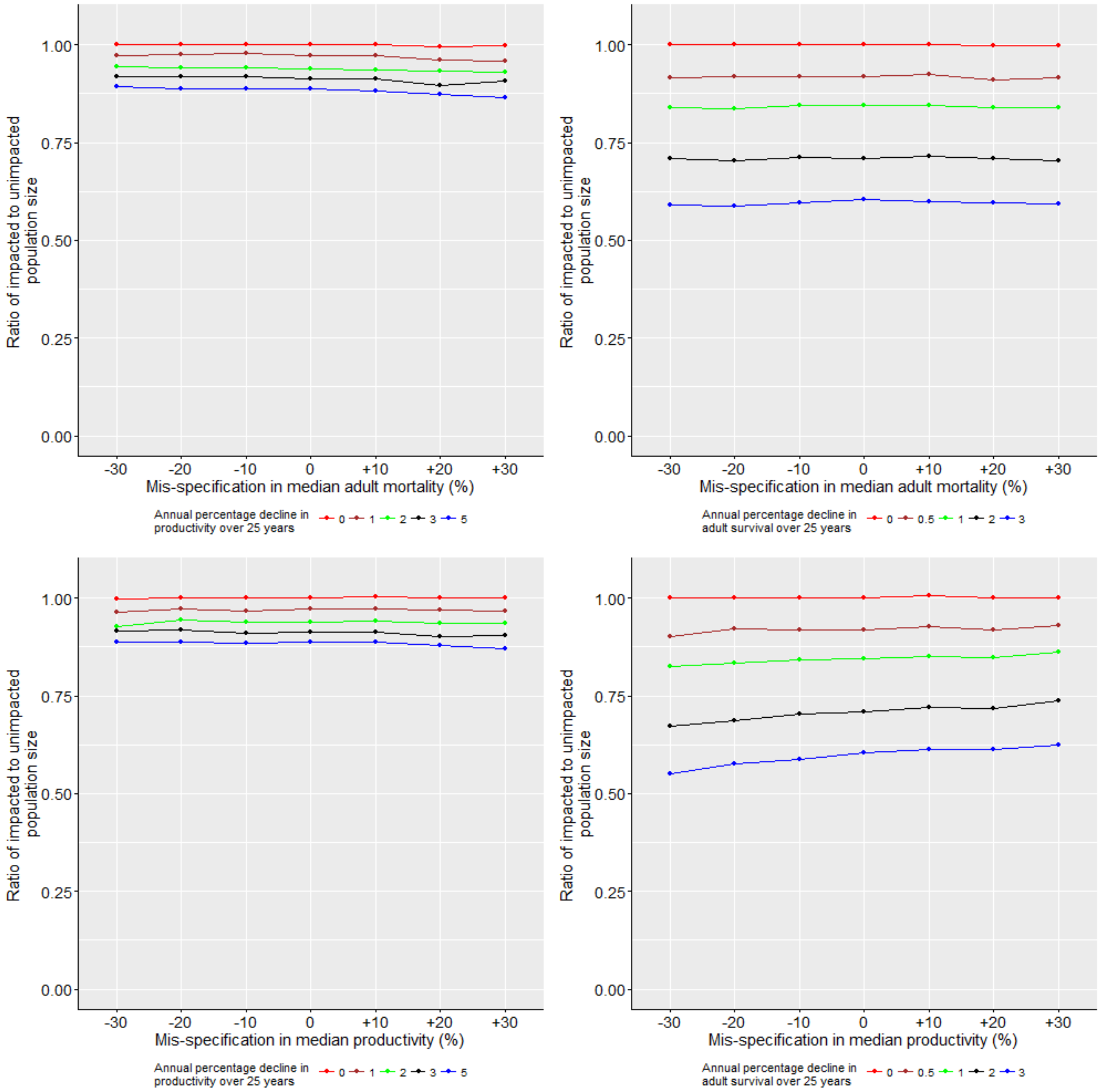


Figure A2.13c. PVA Metric C for St Abb's Shags – difference in population growth rate from 2016-2041, comparing impacted population vs. un-impacted population.

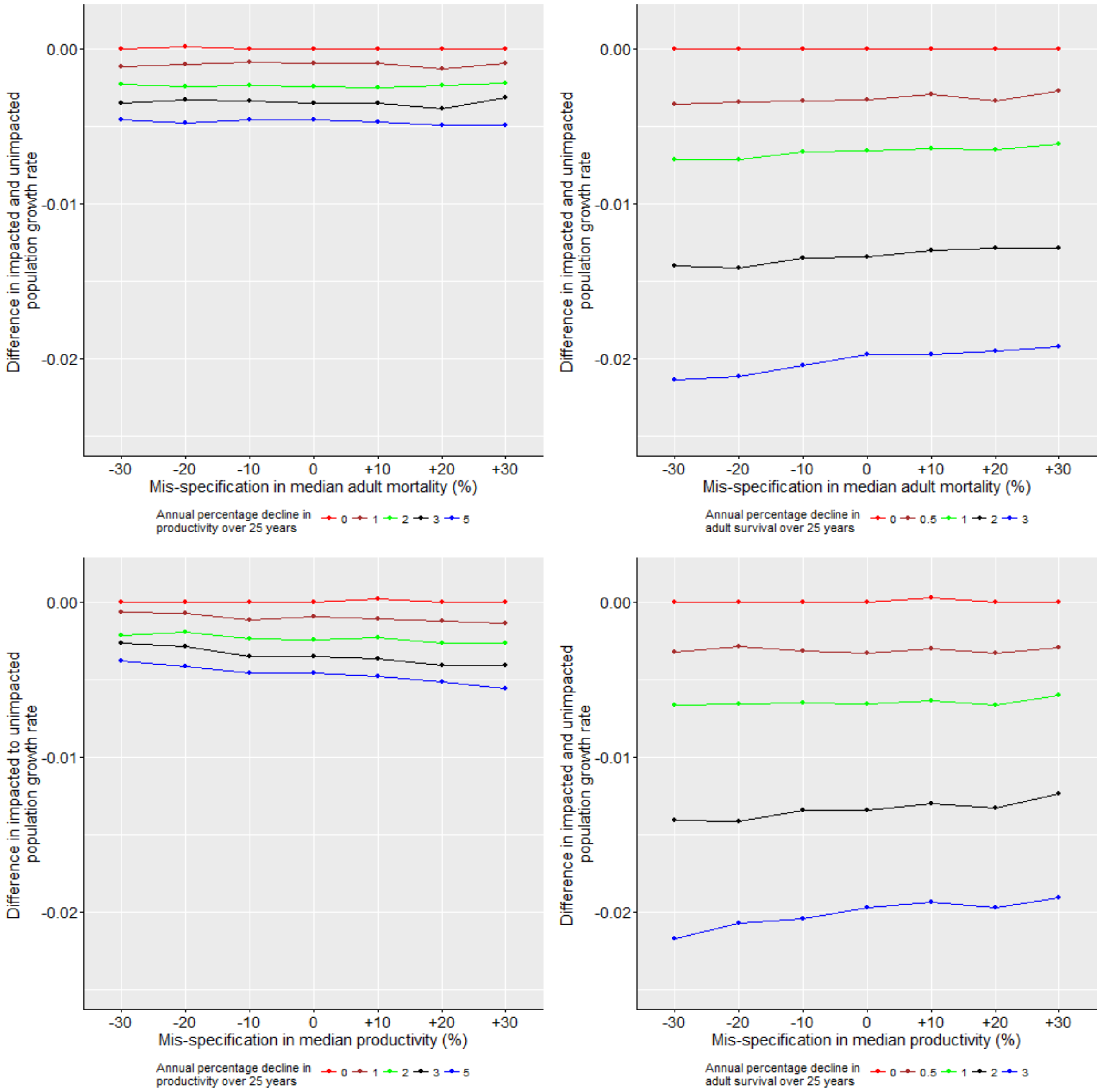


Figure A2.13d. PVA Metric D for St Abb’s Shags – difference in population size at 2041, comparing impacted population vs. un-impacted population.

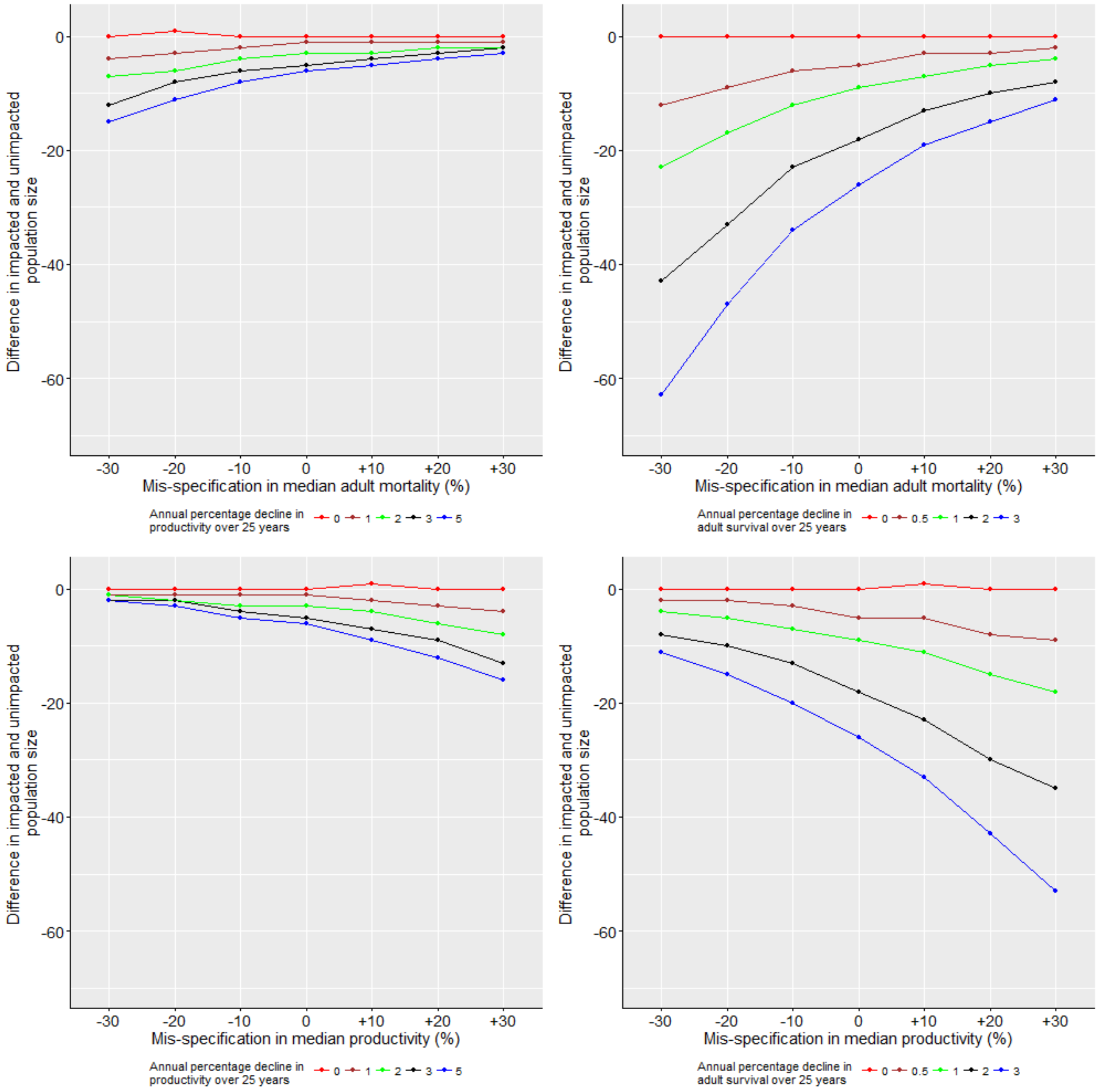


Figure A2.13e. PVA Metric E1 for St Abb's Shags – probability of population decline greater than 10% from 2016-2041.

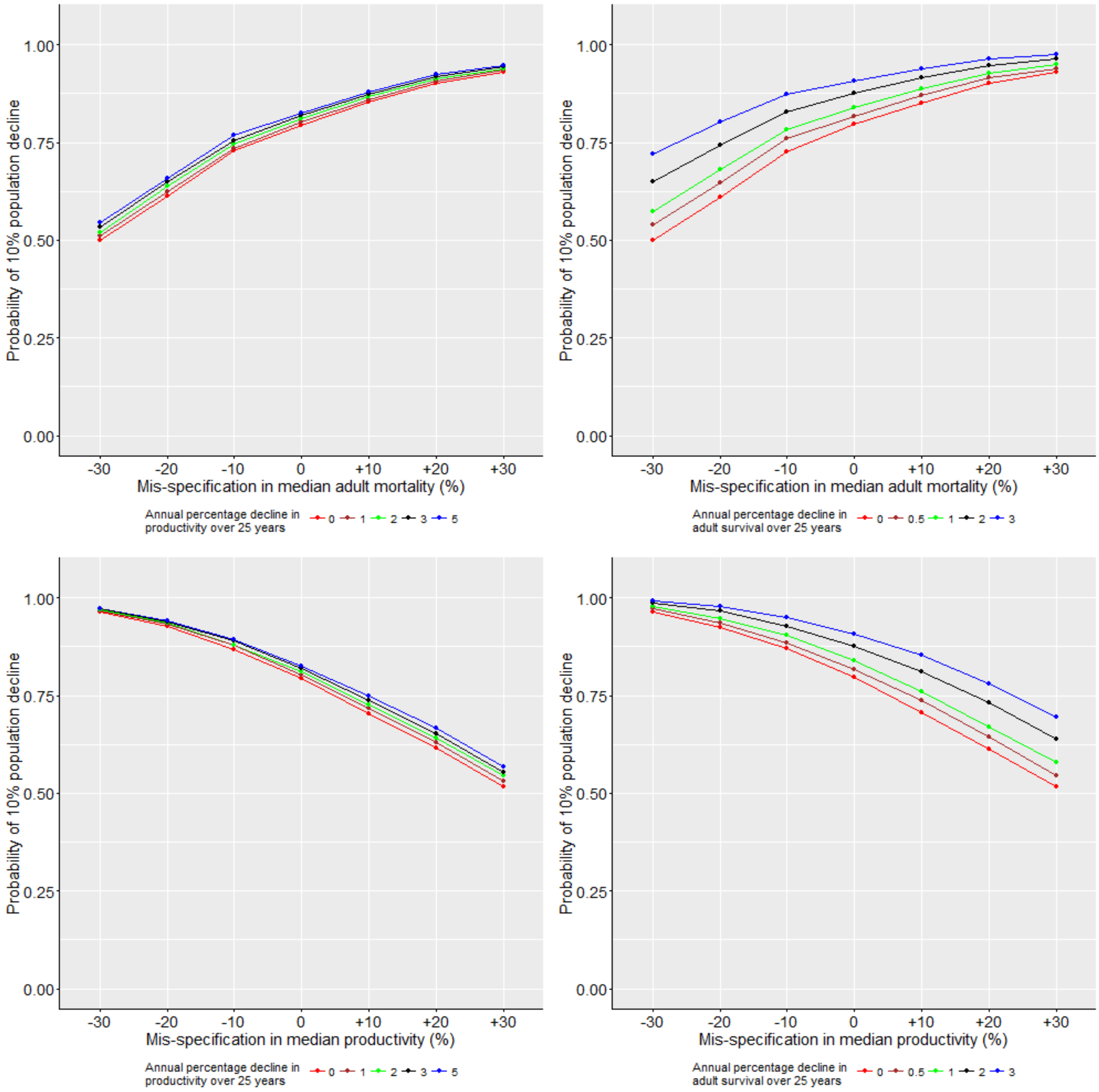


Figure A2.13f. PVA Metric E2 for St Abb’s Shags – probability of population decline greater than 25% from 2016-2041.

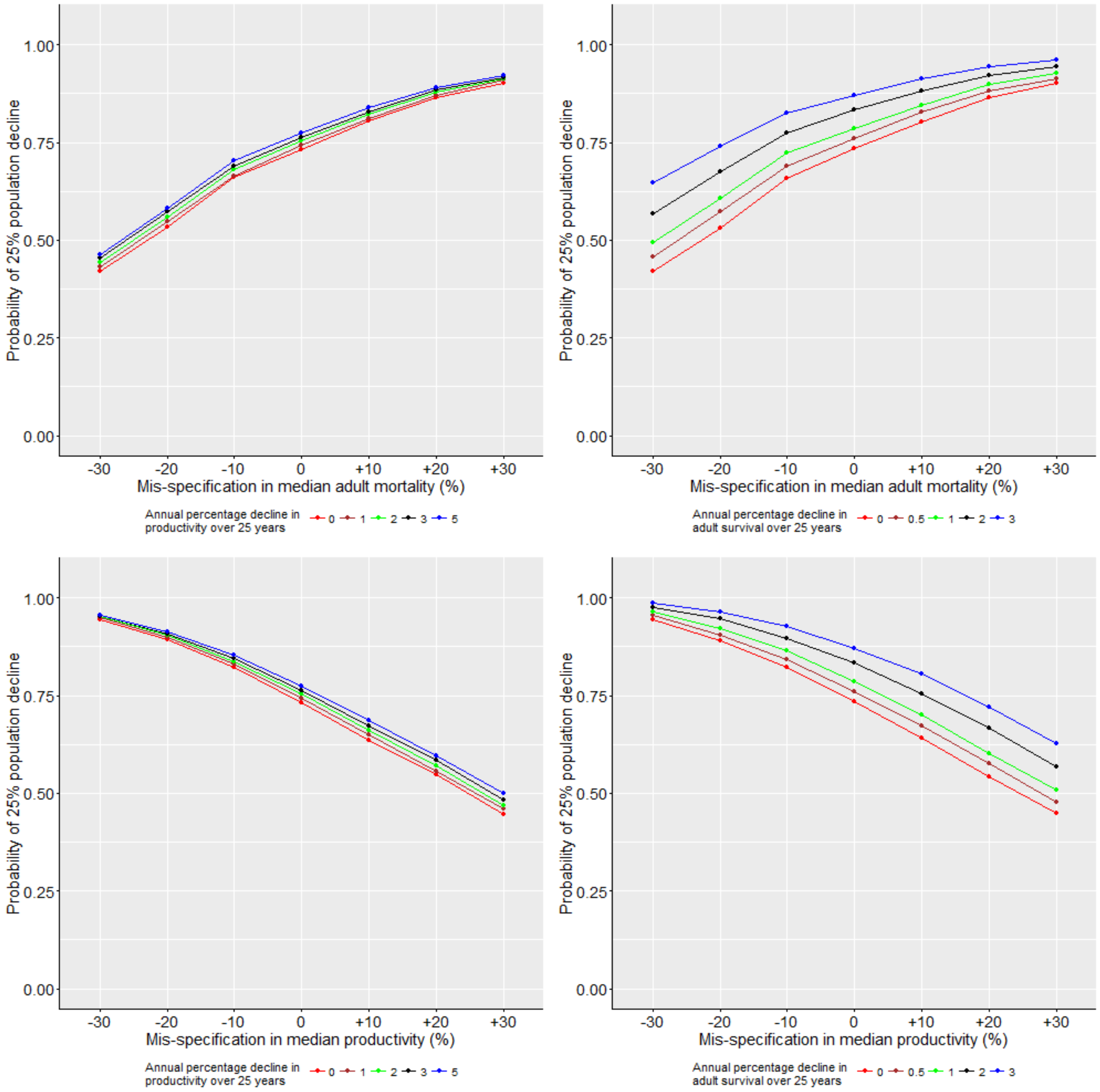


Figure A2.13g. PVA Metric E3 for St Abb’s Shags – probability of population decline greater than 50% from 2016-2041.

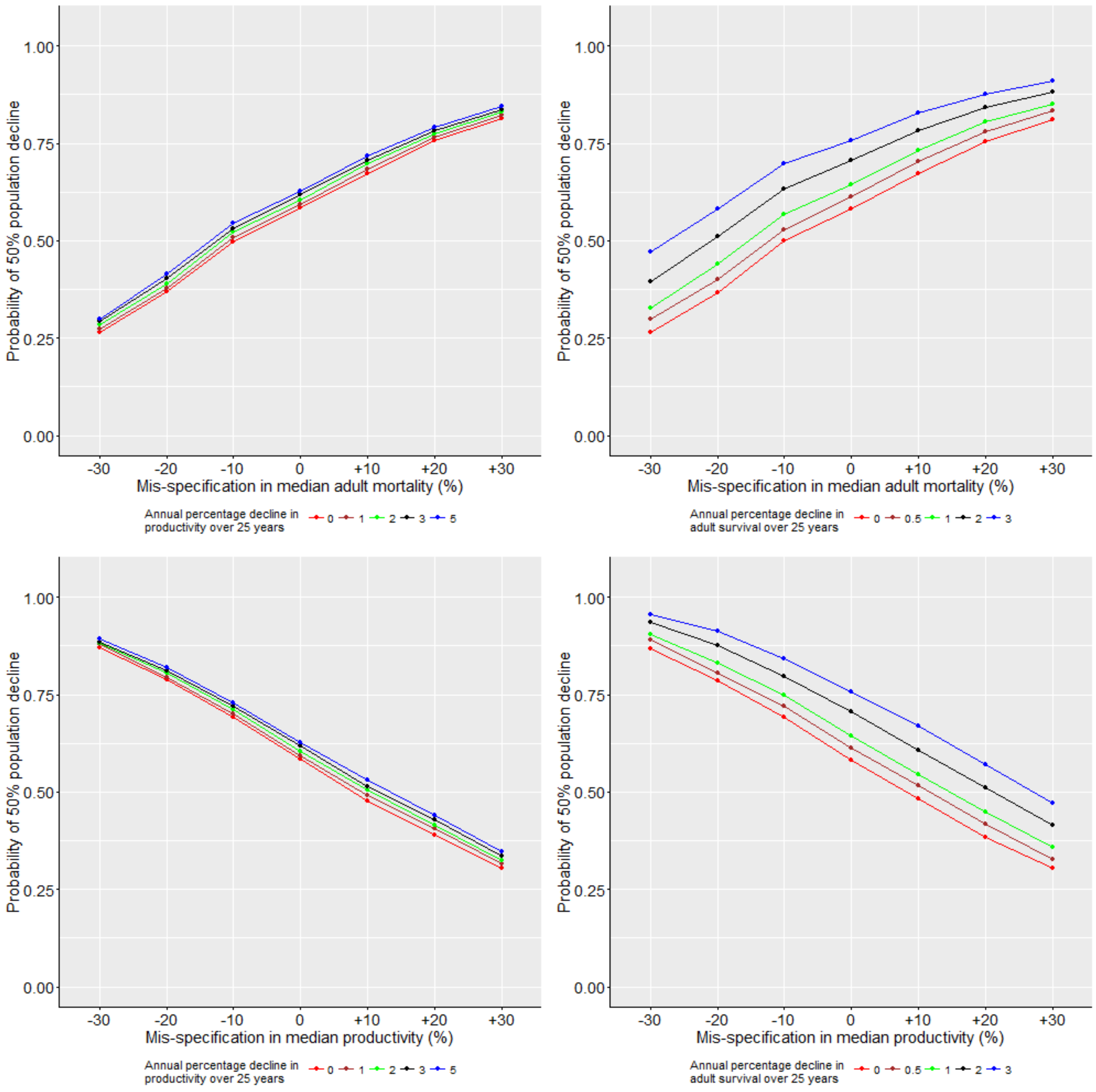
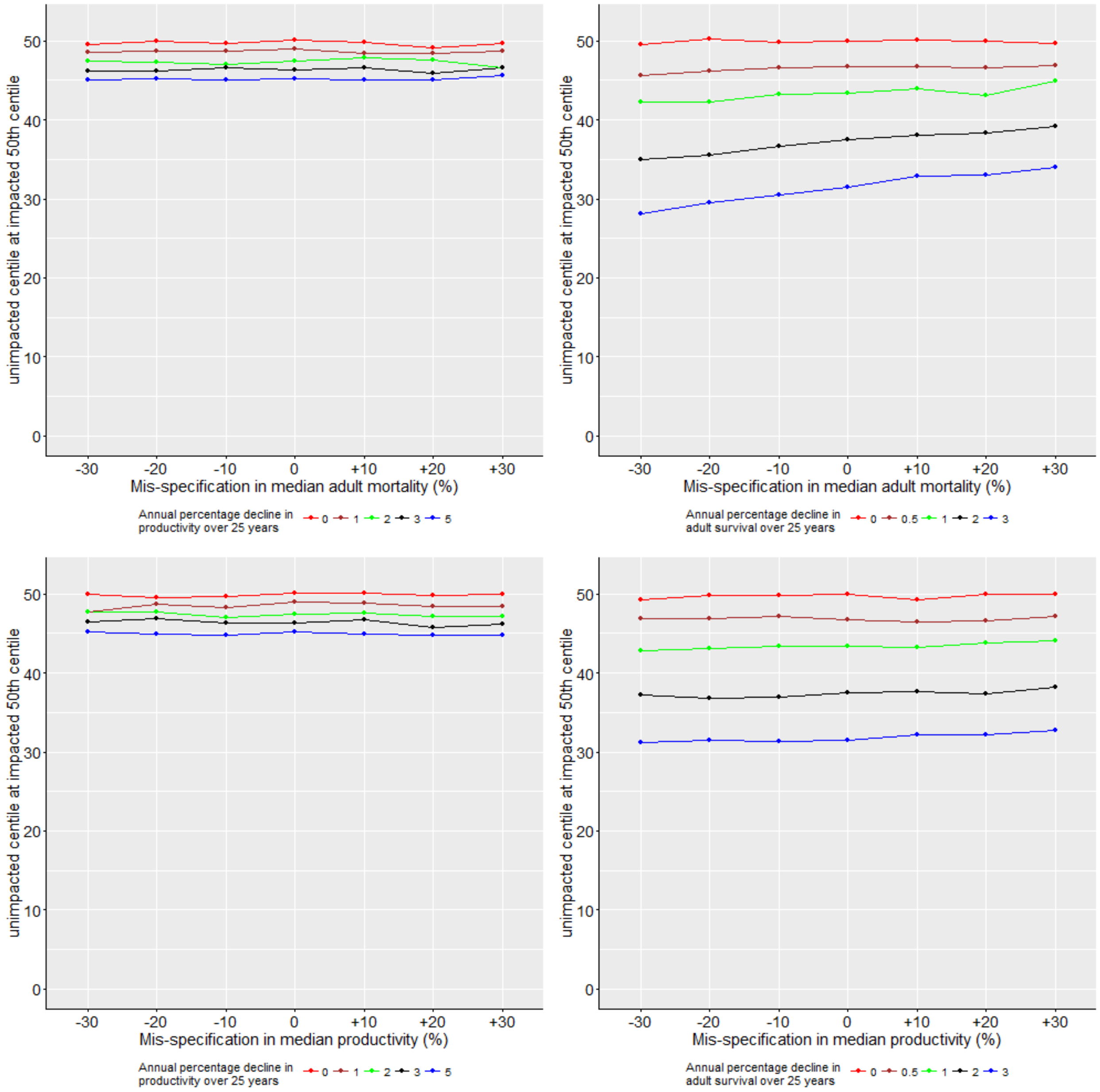


Figure A2.13h. PVA Metric F for St Abb’s Shags – centile from un-impacted population size equal to the 50th centile of the impacted population size, at 2041.



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Collision fatality of raptors in wind farms does not depend on raptor abundance

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Summary

1. The number of wind farms is increasing worldwide. Despite their purported environmental benefits, wind energy developments are not without potential adverse impacts on the environment, and the current pace and scale of development proposals, combined with a poor understanding of their impacts, is a cause for concern.

2. Avian mortality through collision with moving rotor blades is one of the main adverse impacts of wind farms, yet long-term studies are rare. We analyse bird fatalities in relation to bird abundance, and test several factors which have been hypothesized to be associated with bird mortality.

3. Bird abundance was compared with collision fatality records to identify species-specific death risk. Failure time analysis incorporated censored mortality data in which the time of event occurrence (collision) was not known. The analysis was used to test null hypotheses of homogeneity in avian mortality distribution according to several factors.

4. There was no clear relationship between species mortality and species abundance, although all large-bird collision victims were raptors and griffon vulture *Gyps fulvus* was most frequently killed. Bird mortality and bird abundance varied markedly among seasons, but mortality was not highest in the season with highest bird abundance. Mortality rates of griffon vultures did not differ significantly between years.

5. Bird collision probability depended on species, turbine height (taller = more victims) and elevation above sea level (higher = more victims), implicating species-specific and topographic factors in collision mortality. There was no evidence of an association between collision probability and turbine type or the position of a turbine in a row.

6. *Synthesis and applications*. Bird abundance and bird mortality through collision with wind turbines were not closely related; this result challenges a frequent assumption of wind-farm assessment studies. Griffon vulture was the most frequently killed species, and species-specific flight behaviour was implicated. Vultures collided more often when uplift wind conditions were poor, such as on gentle slopes, when thermals were weak, and when turbines were taller at higher elevations. New wind installations and/or repowering of older wind farms with griffon vulture populations nearby, should avoid turbines on the top of hills with gentle slopes.

Key-words: bird abundance, censored data, collision risk, failure time analysis, *Gyps fulvus*, Tarifa, wind energy, wind farm

Introduction

The use of wind as a renewable energy source has been increasing in many countries. Despite the obvious benefit of wind turbines as a clean energy source, it is known that wind farms can have adverse effects on birds, notably fatality through collision with rotating turbine rotor blades (e.g. Langston & Pullan 2003). At the current level of development, wind turbines have been estimated to comprise less than 0·01% of the total annual avian mortality from human-caused sources in the USA (Erickson *et al.* 2002). Although such analyses do not acknowledge that some bird species may be affected more by wind turbines than other anthropogenic mortality sources, at least one study has concluded that wind

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Table 1. Turbine characteristics at EEE and PESUR wind farms

	MADE AE-23	ECOTÉCNICA 20/150	AWP 56–100	MADE AE-20
Wind farm	EEE	EEE	PESUR	PESUR
Power (kW)	180	150	100	150
Blades	3	3	3	3
Rotor diameter (m)	23	20	10 or 18	20
Tower height (m)	28	28	18 or 36	21–28
Tower type	Tubular	Tubular	Lattice	Tubular
Rotor velocity (r.p.m.)	43	51	72	46
Speed (m s^{-1})	4–28	4–25	5–20	5–25
No.	16	50	156	34

turbines, when properly planned, should have minimal impact in comparison with other factors (Fielding, Whitfield & McLeod 2006). For many bird species, turbine collision is not as serious a source of mortality as other factors, such as highways (Fajardo *et al*. 1998), power lines (Ferrer, de la Riva & Castroviejo 1991; Janss & Ferrer 1998), radio/television towers (Stahlecker 1979; Smith 1985), glass windows (Klem 1990), and due to human activities such as poisoning (Harmata *et al*. 1999) and illegal shooting (Villafuerte, Viñuela & Blanco 1998). Nevertheless, the potential for wind farms to cause problems for bird populations should not be underestimated (Hunt 2002; Madders & Whitfield 2006), and the coexistence of birds and wind farms would be enhanced by a more detailed understanding of the factors involved in influencing collision fatality (Barrios & Rodríguez 2004).

A major difficulty in assessing the mortality impact of wind farms on bird populations is the apparent paucity of information from long-term studies at operational wind farms. Despite the existence of numerous studies in the 'grey' literature (Orloff & Flannery 1992, 1993; Hunt *et al*. 1995; Howell 1997; Hunt 1998; Morrison *et al*. 1998; Erickson *et al*. 2001; Kerlinger 2002), relatively little material on wind-farm impacts has been published in the peer-reviewed literature (Musters, Noordervliet & Terkeus 1996; Osborn *et al*. 2000; Johnson *et al*. 2002, 2004; Barrios & Rodríguez 2004; Garthe & Hüppop 2004; Lucas, Janss & Ferrer 2004, 2005; Chamberlain *et al*. 2006; Larsen & Guillemette 2007). Furthermore, study methods vary greatly, as do their results, and although more than 1 year of data may be needed to obtain robust estimates of fatality rates (Smallwood & Thelander 2004), long-term studies are extremely rare. Here we examine long-term avian fatalities in wind farms. We analyse 10 years of bird fatality sampling at two wind farms in Tarifa (Cadiz, Spain) in relation to bird abundance, and test several factors which have been hypothesized to be associated with bird mortality.

Materials and methods

STUDY AREA

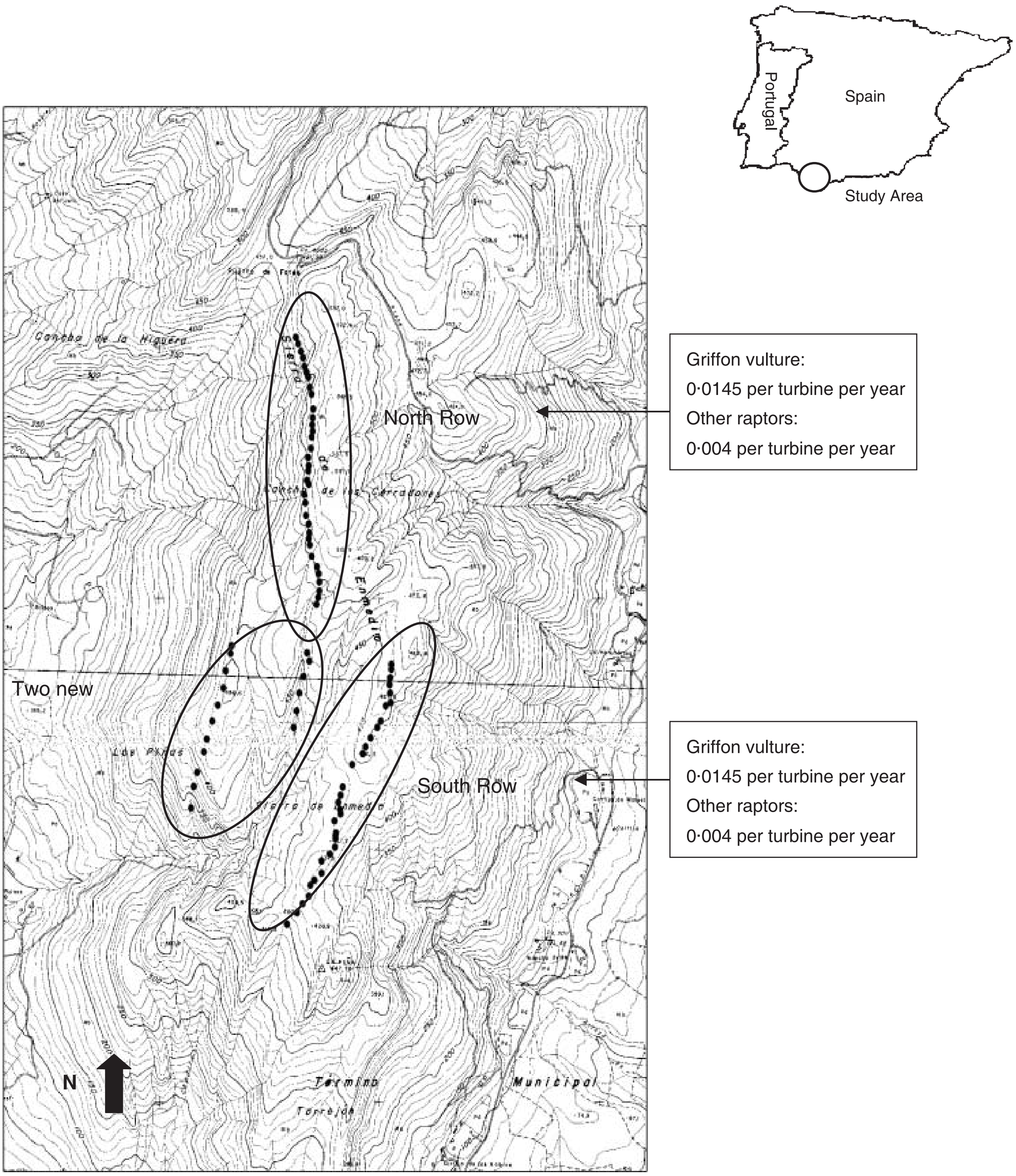
The study wind farms, called EEE and PESUR, were located in Tarifa, Campo de Gibraltar area, Andalusia region, southern Spain (30STF590000–30STE610950) (see also Barrios & Rodríguez 2004). The study area consists of a series of mountain ranges [maximum altitude 820 m above sea level (a.s.l.)] running north–south and reaching the northern shore of the Strait of Gibraltar. The vegetation is characterized by brushwood and scattered trees (*Quercus suber*, *Q. rotundifolia*) on the mountain ridges, and pasture land used for cattle grazing predominating in the lower areas. Easterly winds prevail.

The EEE wind farm is situated along the Sierra de Enmedio mountain ridge (550–650 m a.s.l.) (Fig. 1). During our study, it comprised 66 wind turbines of two models and all rotors were oriented windward (Table 1). Two new rows of turbines were constructed in 1998 and were not included in our study.

The PESUR wind farm is situated in the Dehesa de los Zorrillos, on hills with maximum peaks of 250 m a.s.l. (Fig. 2). It comprises 190 wind turbines with three different designs, and all rotors are orientated leeward to the wind (Table 1). The AWP models (see Table 1) make up a 'wind wall' configuration (Orloff & Flannery 1992) consisting of wind turbines closely aligned with each other but with alternating tower heights.

We used the distance D, defined as 2·5 times the turbine rotor diameter, between a turbine and the nearest other turbine, to distinguish between different rows within each wind farm (if a turbine was > D away from its neighbour, it was classed as being in a different row or 'string'). D is the optimal distance to maximize wind energy capture as used by computation research into wind energy (Grady, Hussaini & Abdullah 2005). On this basis, two different turbine rows were distinguished in EEE (called North and South), whereas PESUR consisted of seven rows: Castro (26 turbines), Alba (14 turbines), Poblana (21 turbines), Piedracana (30 turbines), Tesoro (33 turbines), Bujo (42 turbines) and Zorrillos (24 turbines). All rows were orientated north to south except the Zorrillos row, which was orientated north-west to south-east.

Cliff-nesting species such as griffon vultures *Gyps fulvus*, common kestrels *Falco tinnunculus*, Bonelli's eagles *Hieraaetus fasciatus*, peregrine falcons *Falco peregrinus*, Eurasian eagle owls *Bubo bubo*, as well as a forest species, short-toed snake eagle *Circaetus gallicus*, are characteristic breeding or resident birds of prey in the study area with Bonelli's eagle and peregrine falcon classed as endangered in Spain (Madroño, González & Atienza 2004). Besides supporting an important breeding and resident bird community, the Strait of Gibraltar is one of the most important migration routes for Paleartic birds (Bernis 1980; Finlayson 1992; Bildstein & Zalles 2000). During migration, thousands of soaring species pass through the study area, including European honey buzzards *Pernis apivorus*, booted eagles *Hieraaetus pennatus*, black kites *Milvus migrans*, white storks *Ciconia ciconia* and short-toed snake eagles (bird names follow Gill & Wright 2006).



Griffon vulture:
0·0145 per turbine per year
Other raptors:
0·004 per turbine per year

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Fig. 1. EEE wind-farm map (scale 1:10 000) with the North and the South rows and the two new rows (not included in this study) with mortality rates for both griffon vultures and other raptors in each row. Small map of Spain show the study area at a national scale.

FIELD METHODS AND ANALYTICAL APPROACH

We used avian fatality data collected between November 1993 and June 2003 by the Department of Cadiz of the Andalusian Environmental Ministry in the wind resource areas, comprising records of dead birds collected during research studies and by maintenance personnel at the farms. The searches for collision victims were not standardized during the study period, but occurred approximately once per week. Gauthreaux (1996) suggested that searches for bird fatalities should cover a circular area around each turbine, but because all turbines in our study area were arranged in strings, the most efficient search method was to walk transects or drive unpaved roads along the strings (see also Smallwood & Thelander 2004).

For reasons beyond our control, the search of all turbine strings was not carried out at standardized intervals throughout the study period. However, nearly all data on dead birds were recorded on a standard data sheet including date, species, turbine identity, etc., (Morrison & Sinclair 1998). Each recorded fatality was associated with a carcass that was clearly attributable to a turbine collision rather than any other cause, and that did not share a body part with

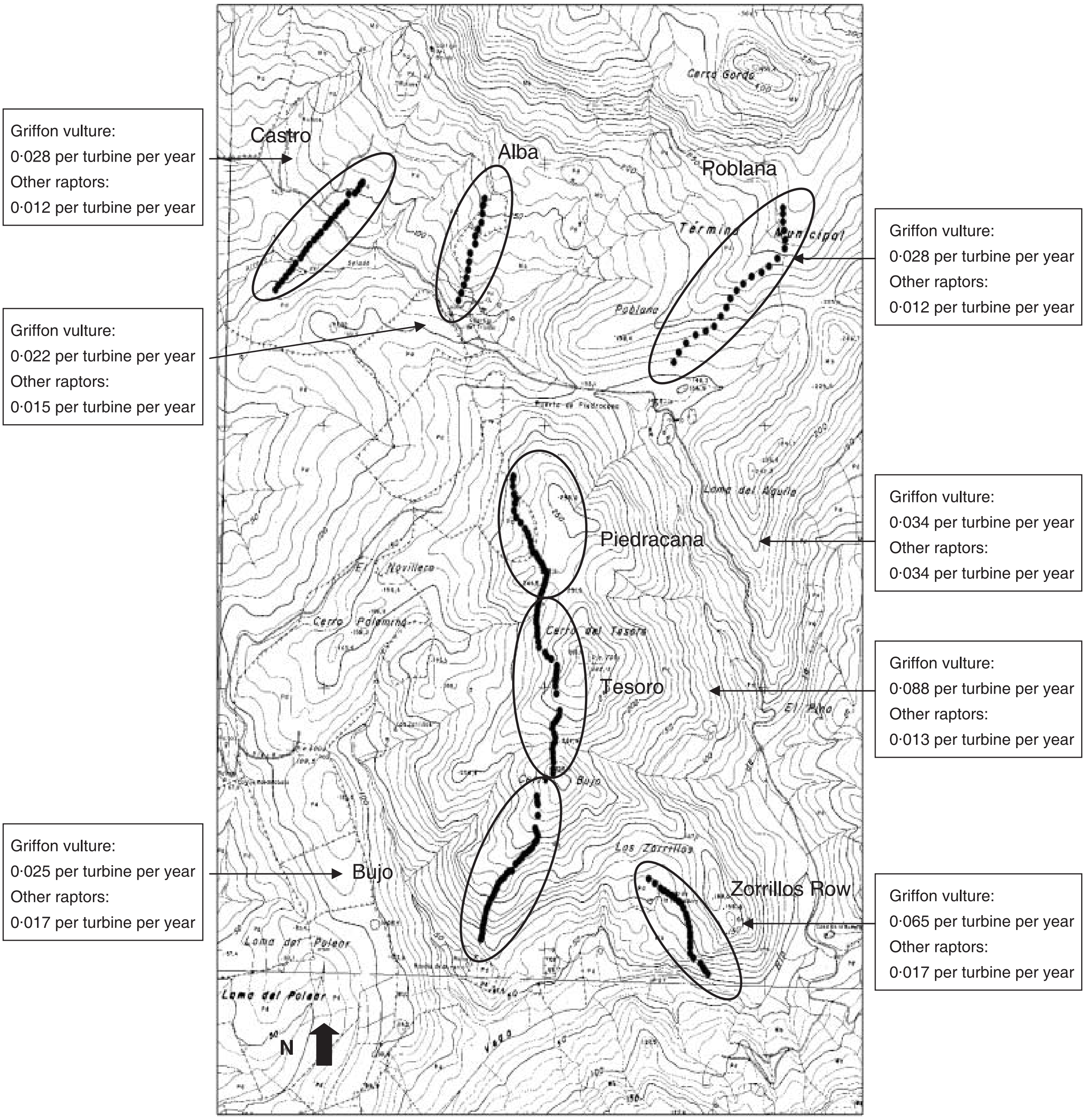

Fig. 2. PESUR wind-farm map (scale 1:10 000) with mortality rates for both griffon vultures and other raptors in each row.

contemporaneous remains. From our previous experience and other evidence at our study sites (Barrios & Rodríguez 2004), we assumed that all dead birds the size of black kite or larger were found. The carcasses of such large birds were not lost to scavengers before searches, and were readily detected by human observers. Although decomposition occurred over time, remains were still present for months to years: a period much longer than any inter-search interval. Smaller species were not included in our study (see Results), and therefore, it was not necessary to apply corrections to account for search biases on mortality (Gauthreaux 1996). We were confident that search protocols produced no spatial biases, due to similar search regimes between and within wind farms and to the longevity of carcass presence. Potential temporal biases were minimized by classing fatality events in broad seasonal or annual categories and by failure time analysis (see below).

To determine the abundance and composition of the local bird community, bird observation surveys were conducted at EEE (2000–2001) and PESUR (2000–2002) during four periods of the year: pre-breeding (mid-January to mid-April), breeding (mid-April to mid-July), post-breeding (mid-July to mid-October) and winter (mid-October to mid-January). At EEE, counts were made over 150 h, and at PESUR wind farm over 250 h. Observations were made from fixed points within 200 m of turbines and during each survey, the number of each species of bird that crossed the rows within 250 m of a turbine was recorded. Each observation lasted 1 h. Survey effort was not equal across seasons; therefore, bird abundance was averaged per season and the mean values used to give the relative abundance of each species in a year.

Failure time analysis measures the length of time from an arbitrary starting point until the first observed 'event' and compares the distributions of the time intervals for each event occurrence (Muenchow 1986): in our analysis an event was the collision of a bird at a wind farm. Failure time analysis accommodates ‘censored’ data in which an event was not observed, perhaps because the study ended before the event happened. For these censored data points, the actual time of occurrence of the event is unknown, for example, when no fatalities are recorded at a turbine (Muenchow 1986).

Several authors have proposed that features of turbine design or location may increase collision risk (e.g. Orloff & Flannery 1992, 1993; Hunt 2002; Percival 2003; Smallwood & Thelander 2004). Therefore, we recorded: (i) tower design (tubular or lattice steel tower), (iii) turbine hub height, (iii) row, (iv) turbine position within the row (end or mid-row), and (v) elevation above sea level based on topographic maps.

STATISTICAL METHODS

We used non-parametric statistics for those variables that did not fit a normal distribution. Post-hoc power analysis was used to determine whether it was appropriate to pool bird mortality data from wind farms. A given alpha value (0·05), sample sizes and effect size were used to obtain a power value higher than 0·8 (Thomas 1997).

We used bird fatality events to assess the factors associated with collisions, so that survival time in failure time analysis was calculated as the time taken for a bird collision (event) to occur and was used as an explanatory variable in a proportional hazard (Cox) regression. The Cox regression for censored data was tested for effects of turbine characteristics (height, tower model, elevation above sea level, row identity, and the effects of turbine position within the row) and for differences in collision rates between griffon vultures and all other species combined. This analysis was stratified by the two types of turbine (lattice and tubular).

Statistica 6·0 and SPSS 13 were used to perform all statistical procedures and we used an alpha value of $P = 0{\cdot}05$ to assess significance of results. GPower 3·0·8 was used to perform power analysis (Faul *et al.* 2007).

Results

BIRD MORTALITY

In the EEE wind farm, a total of 26 dead birds of four raptor species were found during the study period (Table 2). Griffon vulture mortality rate was 0·03 dead birds per turbine per year. The total raptor mortality rate was 2·69 dead birds per year or 0·04 dead birds per turbine per year.

In the PESUR wind farm a total of 125 dead birds from eight raptor species was recorded across the study period (Table 2). Griffon vulture mortality rate was 0·05 dead birds per turbine per year and for all raptors the mortality rate was 12·93 dead birds per year or 0·07 dead birds per turbine per year.

No statistical differences in the number of dead birds per turbine per year were detected between the two wind farms for griffon vultures (Mann-Whitney test, $Z = -1{\cdot}043$, $n = 10$, $P = 0{\cdot}297$) or for other raptor species combined (Mann–Whitney test, $Z = -1{\cdot}650$, $n = 10$, $P = 0{\cdot}099$). The power of the test was high for griffon vultures ($\alpha = 0{\cdot}05$, power = 0·95, effect size = 0·898), indicating that it was appropriate to combine results from wind farms. The power of the test for other raptors was lower ($\alpha = 0{\cdot}05$, power = 0·7, effect size = 0·97), but to avoid further weakening any subsequent within-site tests, we chose to combine mortality data from EEE and PESUR in subsequent analyses.

Statistically significant differences in the number of dead griffon vultures per day were detected between seasons (Kruskal–Wallis test, $\chi^2 = 34{\cdot}272$, d.f. = 3, $P < 0{\cdot}001$), due to a greater number of dead vultures in winter than in other periods (Mann–Whitney tests, $n = 10$, $Z \leq -3{\cdot}088$, $P \leq 0{\cdot}002$ for all three comparisons; Fig. 3). No significant differences in vulture mortality were evident between years (Kruskal–Wallis Test, $\chi^2 = 12{\cdot}220$, d.f. = 9, $P = 0{\cdot}271$). Significant differences in the number of dead raptors other than griffons were detected between seasons (Kruskal–Wallis Test, $\chi^2 = 12{\cdot}718$, d.f. = 3, $P = 0{\cdot}005$) with more dead birds found in winter than during other periods (Mann–Whitney tests, $n = 10$, $Z \leq -2{\cdot}694$, $P \leq 0{\cdot}007$ for all three comparisons; Fig. 3).

RAPTOR ABUNDANCE

A total of 1314 raptors crossed the two wind farms during the observation periods (Table 3). Statistically significant differences

Table 2. Number and species of dead birds in EEE and in PESUR wind farms during the study period (9·67 years), species mortality rates (n° dead birds/turbine/year) and species’ relative abundance. – = no data as a nocturnal species

	EEE wind farm			PESUR wind farm		
Species	No. of dead birds	Mortality rate	Relative abundance (%)	No. of dead birds	Mortality rate	Relative abundance (%)
Gyps fulvus	20	0·0313	89·73	91	0·0495	57·17
Circaetus gallicus	3	0·0047	0·81	4	0·0022	4·64
Bubo bubo	2	0·0031	–	5	0·0027	–
Neophron percnopterus	1	0·0016	1·08	0	0	0·63
Falco tinnunculus	0	0	3·78	19	0·0103	3·77
Falco naumanni	0	0	0	3	0·0016	0
Pernis apivorus	0	0	0	1	0·0005	8·44
Hieraaetus pennatus	0	0	0·81	1	0·0005	2·21
Milvus migrans	0	0	3·78	1	0·0005	14·66
Ciconia ciconia	0	0	0	0	0	10·55
Total	26	0·0407	100	125	0·0680	100

Table 3. Abundance (number of birds per hour) of birds recorded in the two wind farms. Number of birds are given in brackets. Number of 1-h observations = 90, 102, 101 and 102 for winter, pre-breeding, breeding and post-breeding periods, respectively

	EEE wind farm		PESUR wind farm	
Season	Bird abundance	Griffon vulture abundance	Bird abundance	Griffon vulture abundance
Winter	0·43 (17)	0·43 (17)	2·28 (118)	1·30 (69)
Pre-breeding	6·52 (215)	5·67 (187)	8·06 (548)	3·99 (271)
Breeding	0·67 (24)	0·56 (19)	1·88 (120)	0·42 (27)
Post-breeding	3·45 (115)	3·27 (108)	2·38 (162)	1·30 (157)
Overall	2·61 (371)	2·34 (331)	3·78 (948)	2·08 (524)

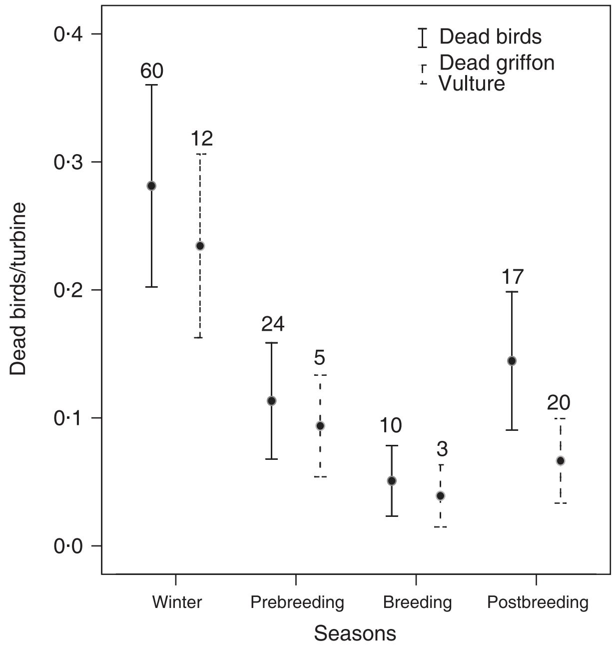

Fig. 3. Mean (± standard error bars) number of dead birds per turbine and dead griffon vultures per turbine according to season. Numbers above points give the number of fatalities for griffon vulture and other birds combined (151 total fatalities).

in the number of griffon vultures per observation were detected between seasons (Kruskal–Wallis Test, χ^2 = 61·108, d.f. = 3, P < 0·001), due to a higher abundance pre-breeding than in other periods (Mann–Whitney tests, $Z \leq -4{\cdot}569$, P < 0·001 for all three comparisons). The same seasonal differences were detected in the number of raptors other than griffon vultures (Kruskal–Wallis Test, χ^2 = 67·007, d.f. = 3, P < 0·001), again due to a higher abundance during the pre-breeding period than in other periods (Mann-Whitney tests, $Z \leq -4{\cdot}655$, P < 0·001 for all three comparisons).

Taking species as replicates, and excluding Eurasian eagle owl because it is largely nocturnal and *Falco* species because of their small size, there was no correlation between a species' mortality and its abundance at EEE (Spearman's ρ = 0·342, n = 5, P = 0·573) or PESUR (Spearman's ρ = 0·449, n = 7, P = 0·312) (Table 2). Including *Falco* species made no substantive difference to the results (EEE: Spearman's ρ = 0·188, n = 6, P = 0·722; PESUR: Spearman's ρ = 0·179, n = 9, P = 0·645).

SPATIAL MORTALITY DISTRIBUTION

No significant differences in mortality rates were detected between the two rows at the EEE wind farm for griffon vultures (χ^2 = 0·018, d.f. = 1, P = 0·892) or for all other raptors combined (χ^2 = 0·023, d.f. = 1, P = 0·877). In contrast, significant differences were detected among the seven rows at the PESUR wind farm for both griffon vultures (χ^2 = 23·866, d.f. = 6, P < 0·001) and for other raptors combined (χ^2 = 17·867, d.f. = 6, P = 0·006). These mortality differences were largely due to most dead birds being detected in Piedracana (28 total dead birds: 20 griffon vultures, 8 others raptors) and Tesoro (32 dead birds: 22 griffon vultures, 10 others raptors) rows.

A stratified proportional hazard (Cox) regression for censored data was conducted to analyse the effect of turbine characteristics on bird collisions. The model (n = 231, uncensored data 53·25%, log-likelihood of final solution = –512·236, null model = –577·539, χ^2 = 130·605, d.f. = 5, P < 0·001) included species (B = 2·23, P < 0·001, with griffon vulture the most likely to collide), turbine height (B = 0·420, P = 0·039) and elevation above sea level (B = 0·005, P = 0·011). The taller the height of the turbines and the higher their elevation above sea level, the shorter the time to a bird collision.

Discussion

Avian mortality rates have been presented for several wind farms around the world (Orloff & Flannery 1993; Dirksen, Winden & Spaans 1998; Barrios & Rodríguez 2004). Like most studies (Erickson *et al.* 2001; Percival 2003; Drewitt & Langston 2006), our mortality rates per turbine were relatively low. However, historical data have not been considered previously and, to our knowledge, our study is the first to analyse long-term samples of bird mortality. A weakness of this study is that it lacked a single protocol for carcass searches, and achieving better consistency in sampling should be an important goal for future work. Nonetheless, the apparently low scavenging rate and the consequent persistence of carcasses around wind farms, coupled with the long timeframe over which carcass searches were conducted, yielded important insights into the effects of wind farms on birds. These fall into three groups: (i) the validity of mortality estimates derived from short-term studies; (ii) the common assumption that more bird activity in the environs of a wind farm will result in more mortality; and (iii) the suggestion that physical characteristics of turbines and their location with respect to other turbines affects the collision risk that they pose to birds.

No indication of a change in mortality rates across the study period was found, suggesting that there were no long-term temporal changes in birds' reactions to the wind farms (e.g. habituation) (see also Stewart, Pullin & Coles 2004; Hötker, Thomsen & Köster 2006).

Bird mortality and bird abundance varied markedly between seasons. Although numbers of dead birds, and especially dead griffon vultures, were higher during winter, bird abundance, and especially griffon vulture abundance, was higher during the pre-breeding season. This is not consistent with the proposal of Barrios & Rodríguez (2004) that bird mortality increases with bird density but supports the results reported by Fernley, Lowther & Whitfield (2006) and Whitfield & Madders (2006) of no relationship between collision mortality and abundance. It is frequently assumed that collision mortality should increase with bird abundance because more birds are 'available' to collide (e.g. Langston & Pullan 2003; Smallwood & Thelander 2004), but our study adds to mounting evidence that such an assumption may be too simplistic. This result has important implications when attempting to predict the impacts of wind- farm proposals. For example, a direct positive relationship between mortality and abundance is an implicit assumption of predictive collision risk models (CRMs) (e.g. Band, Madders & Whitfield 2007). If this assumption is wrong, the utility of current CRMs as predictive tools is severely weakened.

We suggest that others factors, related to species-specific flight behaviour, weather, and topography around the wind farm, might be equally or more important in explaining differences in mortality rates. The different vulnerability of species to collision with turbines is well known and has been linked to species-specific flight behaviour (Orloff & Flannery 1993; Thelander, Smallwood & Rugge 2003; Barrios & Rodríguez 2004; Drewitt & Langston 2006).

High wing loading is associated with low manoeuvrability in flight and a low capability for powered flight is typical of some soaring birds like griffon vultures (Tucker 1971). This relationship has been linked with an elevated risk of collision with objects other than turbine blades (Pennycuick 1975; Janss 2000). With only weak-powered flight, griffon vultures rely heavily on wind for flying (Pennycuick 1975) and to lift them above turbines, whereas other species can use powered flight to avoid collisions with turbine blades. This increases their risk of collision with turbine blades compared with species that have a greater capability for powered flight. Winds that provide lift and assist griffon vultures in cross-country soaring flights will come from two main sources: declivity updrafts from wind deflected upwards by ground slopes, and thermals (Pennycuick 1998). We expect, therefore, that collisions will be more likely when uplift winds are weaker. Several lines of evidence from Tarifa support this idea.

All else being equal, more lift is required by a griffon vulture to fly over a taller turbine at a higher elevation and we found that such turbines killed more vultures compared to shorter turbines at lower elevations. Vulture mortality was also greatest in winter, when thermal updrafts are less common due to lower soil temperatures and lower insolation. Updrafts from gentle slopes are weaker than those from steeper slopes, and so turbines situated on the top of gentle slopes should pose a greater risk to vultures than those atop steep slopes. Piedracana and Tesoro are long turbine rows situated on the gentlest slopes within PESUR (see Fig. 2) and griffon vulture mortality rates were higher here than at other rows in PESUR. Moreover, the slopes surrounding EEE were steeper than those at PESUR and vulture mortality was 1·6 times greater at PESUR than EEE. Declivity updrafts will also be weaker when the speed of the deflected wind is lower and Barrios & Rodríguez (2004) recorded more griffon vulture deaths when winds were light (< 8 m s^{-1}).

Failure time analyses are not commonly used with ecological data (Muenchow 1986; Pyke & Thompson 1986) but the incorporation of censored data in this type of analysis adds more information when addressing ecological questions. Our failure time analysis model enabled us to assess the influence of different factors on the probability of bird collisions. Orloff & Flannery (1992, 1993; although see Smallwood & Thelander 2004) considered lattice towers more dangerous to raptors, but we found no evidence to support this supposition. The effect of a turbine's position in a row on collision has also been the subject of several investigations (e.g. Smallwood & Thelander 2004), but we found that this factor was not influential at Tarifa.

SYNTHESIS AND APPLICATIONS

Mortality rates per turbine were relatively low in this study, and we found no indication of a change in mortality rates across the study period. Griffon vulture was the species most affected by collision mortality. However, collision mortality did not simply increase with abundance, either between raptor species or between seasons for griffon vultures. Therefore, when attempting to predict the impacts of a wind-farm proposal, it is inadequate to assume that collision mortality will increase with bird abundance. Rather, we propose that differences in mortality are equally or more likely to be related to species-specific flight behaviour and morphology, weather and topography around the wind farm. Several features of griffon vulture mortality at Tarifa were consistent with this hypothesis.

It is difficult to make general recommendations for measures to minimize collision with wind turbines. However, repowering of older wind farms, such as those at Tarifa, could provide an opportunity to study such mitigation measures. Reducing the number of turbines and avoiding locations on the top of hills with gentle slopes should decrease rates of bird mortality, but the collision problem will not be eliminated by these measures (Janss 2000). A greater understanding of the mechanisms involved in influencing collision risk for different species, especially the interaction between bird flight behaviour, topography and weather, is essential if the situation is to be managed effectively. As the number of wind farms proliferates, the need for effective mitigation measures becomes increasingly important.

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1 Population Viability Analysis of the North Norfolk Sandwich tern (*Sterna sandvicensis*) Population

Final Report July 2009

Executive summary

1.1.1 A bespoke population viability analysis of the North Norfolk Sandwich tern (*Sterna sandvicensis*) population was undertaken at the request of Natural England, to understand potential impacts of estimated collision at Centrica's Docking Shoal and Race Bank proposed offshore wind farms in the Greater Wash, with a third consented site, Sheringham Shoal (SCIRA) included in background mortality.

1.1.2 The model developed specifically for the purpose, ViaPop, is an individual-based model with the potential to understand the relative effects of different population parameters (ie sensitivity analysis) at greater depth and sophistication than an off-the-shelf package. Population parameters were determined directly from the data on the relatively well-studied north Norfolk colonies coupled with estimates of some parameters from the literature, with estimates of survival rate determined by a specific analysis of British Trust for Ornithology ring recovery data. A project board of experts in the field approved the use of the different parameters used in ViaPop.

1.1.3 Analysis of available data extending back in time to the inception of Sandwich tern colonies in north Norfolk in 1920 showed that a rapid period of increase was followed by relative stability from the 1970s onwards. There is considerable inter-annual variation in the occupancy of the two principal colonies, Scolt Head and Blakeney Point. The factors responsible for colony switching are not known but may include changing prey availability in different parts of The Wash in different years, the effects of kleptoparasitism by black-headed gulls (*Larus ridibundus*) as well as predation and disturbance. Predators, particularly foxes, may have large effects on colony productivity in some years, as may poor weather. Such catastrophic events were incorporated into modelling.

1.1.4 An initial assessment of the performance of ViaPop confirmed that the outputs broadly mirrored those observed in reality. Owing to data limitations (and as requested by Natural England (NE)) it was assumed that the population is closed to immigration and emigration, even though this is unlikely in reality. In fact, there is limited evidence to suggest that north Norfolk operates as a source population for other colonies in north-west Europe.

1.1.5 A total of 35 different model scenarios were undertaken, with sensitivity analysis exploring the effects of adult and juvenile survival and population productivity on population trends, as well as the population effects of predicted wind farm mortality at both the proposed Docking Shoal and Race Bank Offshore Wind farms separately and in combination under different combinations of number and size of turbines, and a range of avoidance rates (95%, 98%, 99%, 99.5% and 99.6%).

1.1.6 The significance of effects was judged in statistical terms. This may not relate to the approach to determining the significance of any effect on the integrity of the North Norfolk Coast Special Protection Area, in which the Sandwich tern population is a cited feature. Determination of any threshold levels, which may be linked to a proportion of the population affected, is the remit of Natural England and its advisors and no further comment is made in this report.

1.1.7 Overall, in relation to the potential effects of predicted collision at Docking Shoal and Race Bank it was concluded that:

- the outcome is very sensitive to the avoidance rate chosen. At avoidance rates of 99.5% or higher, significant impacts are not predicted under any wind farm scenario unless background survival or productivity levels alter significantly over time
- in isolation, Docking Shoal is predicted to have a population impact at an avoidance rate of 99% which is of statistical significance in some years but not others, with statistically significant declines predicted at avoidance rates of 98% or lower
- Race Bank is predicted to be unlikely to cause statistically significant population declines in isolation unless avoidance rates are as low as 95%
- there is potential for statistically significant detrimental effects to the population if both wind farms operate simultaneously, assuming avoidance rates of 99% or lower
- using passage rate estimates from the foraging simulation model, which only relates to breeding birds in the breeding season (May to July inclusive) suggests statistically significant impacts of both sites combined at 98% avoidance but not 99% avoidance
- configurations of larger and fewer turbines reduce predicted population impacts, but not to any significant degree. The most likely scenario of 3.6 MW turbines is preferentially discussed
- populations are substantially more vulnerable to catastrophic events, reductions in productivity and elevations in 'natural' mortality when wind farms are operational

Background Information

1.1.8 Sandwich terns (*Sterna sandvicensis*), within two colonies at Scolt Head and Blakeney Point National Nature Reserves (NNRs), are one of the qualifying features of the North Norfolk Coast Special Protection Area (SPA). The SPA citation records 3,700 pairs accounting for at least 24.7% (5 year mean for 1994 – 1998) of the Great Britain breeding population. More recent estimates in Seabird 2000 (Mitchell *et al* 2004) suggest this is now higher at approximately 40%. The population is known to interchange between Scolt Head and Blakeney Point in different years with one or other becoming the most important in the

UK. For example, Scolt Head held 4,200 pairs in 1998 – 2002 when Blakeney Point supported just 75 pairs. The reverse was true in 1992, when Blakeney Point supported 4,000 pairs and Scolt Head had 280 pairs. From 1986 – 2004, the Scolt Head/Blakeney Point colony averaged 3,741 pairs (Centrica Energy 2008). Therefore, the Scolt Head/Blakeney Point complex may support 5 – 6% of the European breeding population of 69,000 – 79,000 pairs (BirdLife International 2004) and 2.3 – 2.8% of the world population of 160,000 – 170,000 pairs (Mitchell *et al* 2004). With a recent moderate decline in the European population that supports >50% of its global population, Sandwich tern is evaluated as Depleted and of unfavourable conservation status (SPEC 2) in Europe (BirdLife International 2004).

1.1.9 As a result of the presence of this internationally important breeding population, Sandwich tern has emerged as the key sensitive receptor in relation to the potential impacts of the offshore wind farms (OWF) within the Greater Wash Round 2 strategic development area. Currently, two Round 1 sites (Lynn and Inner Dowsing Offshore Wind Farms (OWFs)) are in operation within the Greater Wash, two sites have been consented (Sheringham Shoal and Lincs), with a further three in planning (Docking Shoal, Race Bank and Dudgeon) and a further one (Triton Knoll) under development. Figure 1.1 shows the location of the main Sandwich tern colonies in relation to the Greater Wash offshore wind farm projects included in the modelling exercise.

1.1.10 The very high sensitivity of the species contributes to the potential for predicted significant impacts, especially in relation to collision risk. In relation to the potential for significant impacts at Sheringham Shoal OWF an Appropriate Assessment (AA) was triggered, and this has also been requested for Docking Shoal and Race Bank OWFs following the recent submission of their Environmental Statements (ESs) (Centrica Energy 2008, 2009).

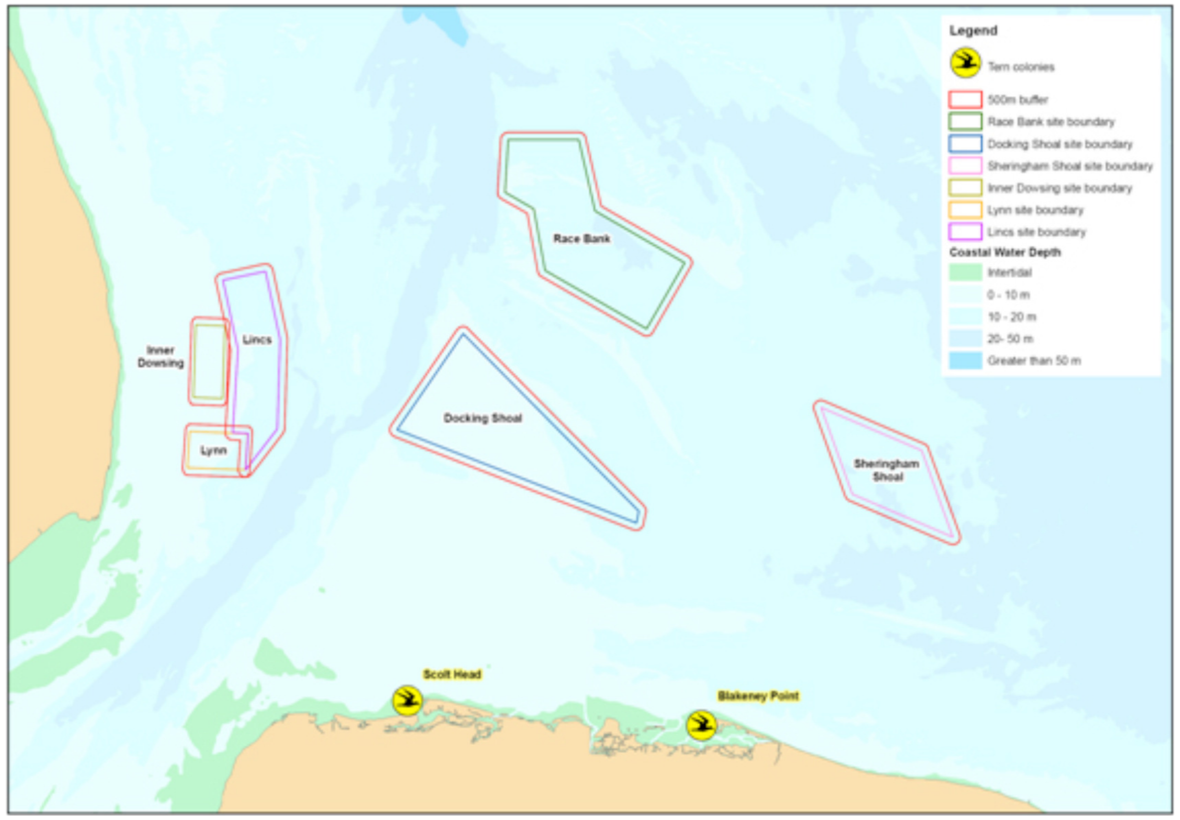


Figure 1.1 The location of the Sandwich tern colonies at Scolt Head and Blakeney Point in relation to the offshore wind farm projects included in the population modelling exercise and other consented or operational sites in the Greater Wash

1.1.11 The proposed Docking Shoal OWF is in relative proximity to the colony at Scolt Head and relatively high numbers of terns were recorded on the site, particularly in the first season of data gathering for the ES. In contrast with some other OWFs also in relative proximity to tern colonies and demonstrating the commitment of Centrica Energy/AMEC to a rigorous impact assessment process, a significant amount of additional work was undertaken. This included increasing the frequency of boat-based surveys in the tern breeding season (May to August), extending the monitoring period to three years for Docking Shoal and undertaking an intense programme of species-specific monitoring at both Docking Shoal and Race Bank. The latter comprised:

- tracking individual terns from both the Scolt Head and Blakeney Point colonies to assess foraging movements and patterns
- detailed recording of flight lines and passage rates to/from colonies
- developing a simulation foraging model to determine the relative use of different OWFs and other areas
- sampling the prey resource at Race Bank and Docking Shoal relative to inshore waters during the breeding season

1.1.12 Many aspects involved the research and development of novel techniques and means of handling data. Some of this work has now been extended in a further COWRIE funded project in 2008/2009 (Perrow *et al* 2009).

1.1.13 In contrast to Sheringham Shoal, Natural England requested further work for Docking Shoal and Race Bank before an AA, specifically focussed on whether population viability analysis (PVA) could be used to help predict the population impact of particular collision rates. This stemmed from the report by the British Trust for Ornithology (BTO) on the topic (Maclean *et al* 2007). An evaluation of whether PVA was a useful option was undertaken by ECON and colleagues under the project title 'Scoping work for Population Viability Analysis for Sandwich terns in the North Norfolk Coast SPA'. This had the following outputs:

- generated the best available estimates of various population parameters of Sandwich terns for inclusion into PVA
- discussed these parameters with representatives of statutory (NE, BERR (now DECC) and Joint Nature Conservation Committee (JNCC)) and non-statutory organisations RSPB, the respective developers and other associated parties at a PVA workshop on 11th December 2007
- briefly evaluated relevant PVA software and demonstrated the likely outputs of PVA to initiate discussion of its value as a tool to assess the impact of additional OWF mortality upon Sandwich terns

1.1.14 Although the databases supplied by the scoping work provided a good basis for future modelling, through discussion, it became clear that just a few key parameters such as adult survival, age and survival to first breeding and chick productivity are likely to 'drive' population trends, with the range of other factors populating the different available 'off the shelf' models (eg senescence leading to reduced productivity with age) likely to be of lesser importance.

1.1.15 Whilst there is a lack of definitive, specific information (in Europe, let alone within the UK or colony-based) on adult survival rate for example, the PVA workshop nevertheless came to the conclusion that PVA was the only tool available to evaluate the potential impact of collision mortality as a result of the Greater Wash OWFs. In the absence of definite information, 'best available' estimates that were acceptable to all concerned would be substituted. The decision to use one estimate over another could depend on expert opinion, which was available in project board meetings of the proposed project, and for which sensitivity analysis could be undertaken.

1.1.16 In the event, the opportunity to derive a further more specific and potentially more accurate estimate of adult survival rate became available through specific analysis of the BTO's ringing scheme and use of a software package, MARK (Robinson 2008). This approach built on previous attempts to derive survival estimates from ring recovery data, including for terns (Green *et al* 1990) and the use of MARK in population modelling (eg Freeman *et al* 2007). Wernham *et al* (2002) report that 151,985 Sandwich terns had been ringed in UK with 4,230 recoveries, and it was hoped a sufficient database from ring recoveries would be available to derive reasonable estimates of annual survival.

Introduction

1.1.17 The discussion generated from scoping (see above) outlined the fundamental advantages of ‘individual-based’ models (eg VORTEX - Lacy *et al* 1993) rather than the matrix-based approach (ie a mathematical matrix describes the population behaviour of the cohort) employed in many available software packages (eg INMAT - Mills and Smouse 1994, RAMAS – Inouye 1994 and ULM - Legendre and Clobert 1995). The value of models based on individuals and individual behaviour is well documented (eg Sutherland 1996, Grimm 1999, Brook *et al* 2000), giving benefits of:

- accurate representation of demographic processes avoiding the ‘smoothing’ simplification employed by alternative methods, which inevitably ‘iron out’ important stochastic effects
- the ability to scale-up observed individual behaviour to population levels. It is thus reasoned that it is preferable to adopt an individual-based modelling framework in any analysis that seeks to consider risks to exploited or impacted populations. Indeed, the use of a matrix-based approach undermines the precautionary approach. This key consideration appears to be absent from a recent review of the application of viability analysis to bird populations impacted by wind farms (Maclean *et al* 2007)

1.1.18 However, most ‘off the shelf’ software including VORTEX, adopts a ‘black-box’ approach providing an output with extremely limited scope to understand how this was generated and how different parameters may interact. Insight into such behaviour is likely to be crucial in determining the range of natural variation and ultimately the relative impact of collision compared to natural mortality, especially in relation to Sandwich tern, a species known to exhibit the most dramatic population fluctuations of any seabird in the British Isles (Ratcliffe 2004). It was argued that the modelling the north Norfolk Sandwich tern population required greater depth and sophistication than an off-the-shelf package could provide and as a result, the concept of an individual-based, custom-built (bespoke) model was introduced.

1.1.19 Development of an individual-based simulation model was thus the focus of the work conducted here, with this following the broad framework of Ulbrich and Kayser (2004) with appropriate species-specific adjustments. What was thought to be of critical importance to the value of the exercise particularly in relation to the lack of ‘definitive’ estimates, was the ability to understand the behaviour of particular parameters during modelling. This could be readily achieved in the bespoke model, allowing full sensitivity analyses of the tested parameters. If time was available, a comparison of outputs with the available software, VORTEX (Lacy *et al* (1993)) which was used to demonstrate the principles of PVA at the workshop, was to be undertaken.

1.1.20 A further significant advantage of the bespoke model approach is that it was possible to readily incorporate aspects of the Sandwich tern life history which do not fit with the assumptions of off-the-shelf models. This includes periodic predation by red fox (*Vulpes vulpes*) often the main cause of mortality of adult Sandwich terns during the summer months (Ratcliffe 2004). Foxes as well as stoats (*Mustela erminea*) and large gulls (herring (*Larus argentatus*) and great black-backed (*L. marinus*) gulls) also cause disturbance which can lead to abandonment. Moreover, severe weather limiting foraging opportunity and potentially causing food limitation for chicks was also thought to occur periodically, along with more direct mortality of storm events on chicks. For example, >1,000 chicks were found dead after storms on June 19th and June 23rd/24th in 2004 (NNNS 2005). Some understanding of the likely relative impact of predation and weather-related mortality compared to collision risk was seen to be crucial to this project. After all, reduction of predation through more effective protection and wardening is a tangible means of reducing adult mortality and/or raising chick productivity.

Methodology

Meetings and Consultations

1.1.21 A series of steering group meetings including representatives from ECON Ltd, AMEC and Centrica were undertaken during the course of the project and consultation with NE, JNCC and RSPB was undertaken throughout. Following consultation, the comments of JNCC, NE and RSPB were included in the project. A preliminary set of population parameters was also sent around the group for comment. These were also sent to the leading authorities on Sandwich terns, Eric Stienen (see Stienen *et al* 2001, 2002ab, Stienen 2006) and Joris Everaert (Everaert and Stienen 2006), who have also been working on the Zeebrugge Wind Farm in Belgium upon which most of the current thinking in relation to the collision risk of OWFs on breeding terns (Sandwich, common (*Sterna hirundo*) and little (*Sternula albifrons*)) is based (see Everaert and Stienen 2006, Stienen *et al* 2008). No comments were received on the population parameters to be included within the model and thus no changes were made.

1.1.22 A working version of the model was provided at a meeting on 24th September 2008, with a more complete version and preliminary outputs presented on 15th December 2008. Draft output of all model scenarios was supplied on 15th February 2009, with a final draft report delivered in April 2009.

Data Gathering

1.1.23 Both Scolt Head and Blakeney Point colonies have been the subject of conservation and research interest for approximately 100 years, with wardens in place for much of that period, relating to the interests of The National Trust at Blakeney Point and the Nature Conservancy Council (NCC) (becoming English Nature and thence Natural England) in relation to Scolt Head. Data specifically relating to the number of breeding pairs, productivity (number of chicks fledged) and any specific events at the colony have generally been documented annually in the *Transactions of the Norfolk and Naturalists Society* (with The Norfolk Naturalists Trust and Norfolk Ornithologists Union in different periods) latterly known specifically as the *Norfolk Bird and Mammal Report*. A complete copy of the Transactions from 1875 is held in the Norwich Heritage Library at the Norwich Forum, with a further set from 1966 to the present day (with some omissions) held in the ECON Ltd office. All records of breeding terns at both colonies were compiled from these sources for the purposes of this project. The only years there is no mention of Sandwich terns in these reports is 1930 – 1932, 1948 and 1951 – 1952 for both colonies and 1979 in relation to Scolt Head ie <7% of the potential database since the colonisation of Norfolk by breeding Sandwich terns.

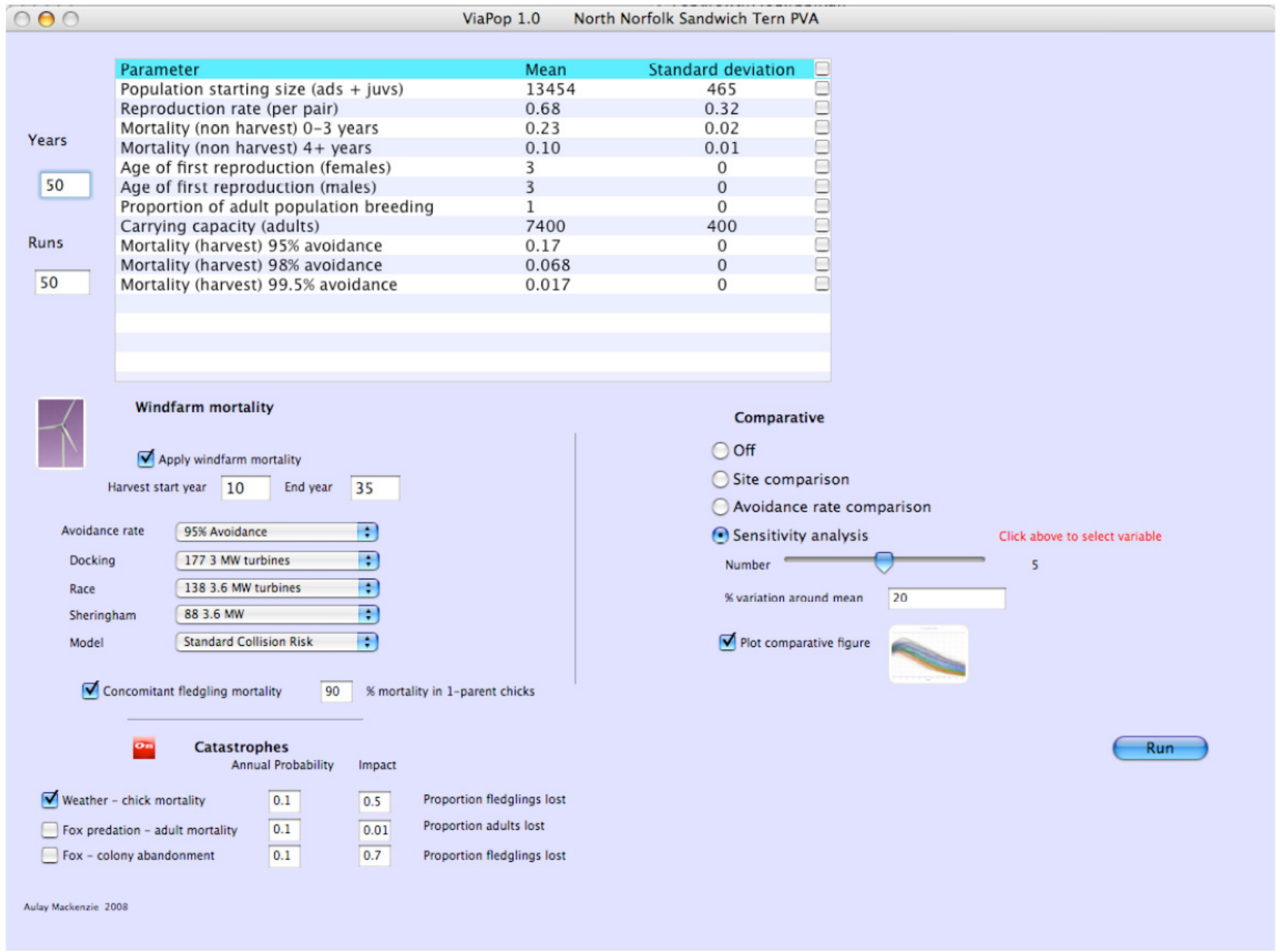
1.1.24 In addition, data from the literature on population parameters was systematically searched. In truth, much information was already held by ECON Ltd. in relation to the work undertaken on Sandwich terns in relation to The Wash OWFs (eg SCIRA Offshore Energy 2006, Centrica Energy 2008, 2009). The annual reports produced by the JNCC on seabird breeding success (eg Mavor *et al* 2006) were also consulted as were more general sources of information with specific chapters on Sandwich terns, such as the *Birds in England* (Brown and Grice 2005) and *The Birds of Norfolk* (Taylor *et al* 1999).

The Modelling Approach

1.1.25 The model developed by Aulay Mackenzie at the University of Essex, named ViaPop, is an individual-based population viability analysis simulation, which has been specifically customised around the population biology of Sandwich terns, with particular reference to the north Norfolk population. ViaPop is compiled in REALbasic on Apple Macintosh and controlled through a graphical user interface (Figure 1.2). The model assesses population change given a set of input parameters, and can incorporate sensitivity analysis of any variable. Output is in graphical and raw data form. No calculation of putative extinction probabilities is made, partly as this is a potentially misleading and error-prone approach.

1.1.26 Within the model, individuals are explicitly represented, with age and gender characteristics. This individual representation avoids structural over-simplification and allows stochastic processes to act at an individual level, features which assist in greater population realism. Population processes are simulated as discrete sequential events (see Figure 1.3), which act in a probabilistic manner on individuals (or pairs in the case of breeding).

1.1.27 Mortality and fecundity parameters are allocated to individuals from Gaussian distributions. Mortality rates are age-class specific, with two age classes specified ie adults and juveniles (incorporating juveniles of age 0 and immature birds to age <3) allowing for higher rates to be specified in pre-reproductive individuals. In the basic model, the background mortality rate for adults (in the absence of wind farm collisions or natural catastrophes) expressed as the mean likelihood of survival for adult individuals in any given year is 0.90, with a standard deviation of 0.01. For juveniles, background survival rate is 0.77 with a standard deviation of 0.02. Both values were derived by Robinson (2008). Senescence is not incorporated, so fecundity is independent of age class after individuals reach reproductive age, which is not an unreasonable assumption in a long-lived bird species with a maximum age of 30 years (and 9 months) (www.bto.org.uk)[1].



ViaPop 1.0 North Norfolk Sandwich Tern PVA

Years 50

Runs 50

Parameter	Mean	Standard deviation
Population starting size (ads + juvs)	13454	465
Reproduction rate (per pair)	0.68	0.32
Mortality (non harvest) 0–3 years	0.23	0.02
Mortality (non harvest) 4+ years	0.10	0.01
Age of first reproduction (females)	3	0
Age of first reproduction (males)	3	0
Proportion of adult population breeding	1	0
Carrying capacity (adults)	7400	400
Mortality (harvest) 95% avoidance	0.17	0
Mortality (harvest) 98% avoidance	0.068	0
Mortality (harvest) 99.5% avoidance	0.017	0

Windfarm mortality

☑ Apply windfarm mortality

Harvest start year 10 End year 35

Avoidance rate 95% Avoidance

Docking 177 3 MW turbines

Race 138 3.6 MW turbines

Sheringham 88 3.6 MW

Model Standard Collision Risk

☑ Concomitant fledgling mortality 90 % mortality in 1-parent chicks

Catastrophes

	Annual Probability	Impact	
☑ Weather – chick mortality	0.1	0.5	Proportion fledglings lost
☐ Fox predation – adult mortality	0.1	0.01	Proportion adults lost
☐ Fox – colony abandonment	0.1	0.7	Proportion fledglings lost

Aulay Mackenzie 2008

Comparative

○ Off

○ Site comparison

○ Avoidance rate comparison

● Sensitivity analysis

Click above to select variable

Number 5

% variation around mean 20

☑ Plot comparative figure

Figure 1.2 Screen shot of parameter selection window in ViaPop

1.1.28 Genetic elements are not explicitly represented, so any impacts of inbreeding in small populations are not incorporated. Population productivity becomes density dependent only when the population size exceeds carrying capacity. Carrying capacity was generally set close to the starting population size (Table 1.1), as it is assumed that the north Norfolk population is currently close to carrying capacity, given that the population has not increased in recent years despite apparently high productivity (Taylor *et al* 1999). Density dependence acts as a decay function on the gross reproduction above the carrying capacity, which is allowed to vary on an annual basis. Density dependence impacts on production but not on adult survival, which is a reasonable assumption in a highly mobile species with low adult mortality rates. The initial annual age distribution structure in each run was set at equilibrium state to avoid transient population size effects caused by age structure fluxes. The basic dataset of population parameters used in ViaPop is presented in Table 1.1.

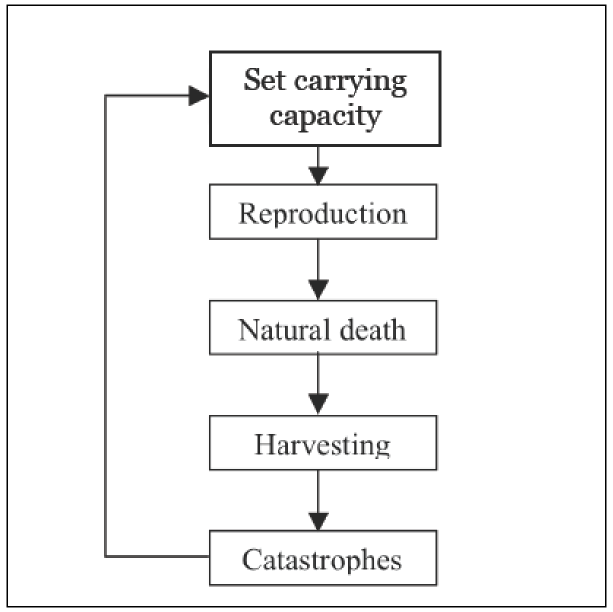


Figure 1.3 Annual life cycle of Sandwich terns as utilised in ViaPop

1.1.29 Periodic catastrophic events involving reproductive failure or elevated mortality represent a well-known feature of Sandwich tern breeding biology. In order to explore the potential for additive effects of these events in relation to wind farm collision mortality, specific forms of catastrophe were modelled separately as sporadic events occurring at a defined annual probability and with a defined average impact. The frequency and intensity of specific catastrophes were broadly determined from the database compiled from historic information relating to the Norfolk colony from 1920 to 2007 (see paragraph 1.1.21). These were:

- predator (fox) induced adult mortality - annual probability 0.1, impact of 1% loss of adults
- predator (fox) induced colony abandonment - annual probability 0.1, impact of 70% loss of eggs/chicks
- weather-induced chick mortality - annual probability of 0.1, impact of 50% loss of fledglings

Table 1.1 Baseline parameters for the north Norfolk Sandwich tern population incorporated into ViaPop

Parameter	Sub-parameter	Value	Source and notes
Initial population	Size (number of individuals and pairs)	7400 ± 250 ind (3700 pairs)	Mean population size from colony counts (1970–2007)
	Sex ratio	0.5	Assumed
	Age distribution	As determined by mortality rate	
Mortality (except "harvest")	Adult mortality (individual^{-1-} year^{-1}) [excludes catastrophes]	10% ± 1%	Robinson (2008)
	Juvenile (individual^{-1-} year^{-1})	23% ± 2%	Robinson (2008)
	Sex specific rate	Equality	Assumed
Mortality through "harvest"	Rate (mortality adult individual^{-1-} year^{-1})	Variable (max. 0.067, min. 0.017	According to site and avoidance rate (Table 1.2)

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[1] Note that birds of >15 years could not be incorporated into BTO survival estimates as a temporal limitation of the available ringing data.

Parameter	Sub-parameter	Value	Source and notes
	Start year and end year	Year 15, Year 50	Allows equilibration followed by maximum life of OWF
	Sex specific bias	No	
	Avoidance rates	95%, 98%, 99% 99.5%, 99.6%	From likely minimum to maximum values
Catastrophes (probability of occurring, impact)	Fox predation – adult survival (numbers removed)	P=0.1, lost=0.01	Broadly derived from specific North Norfolk data (1920-2007)
	Fox predation – colony abandonment (proportion abandoning)	P=0.1, lost=0.7	Broadly derived from specific North Norfolk data (1920-2007)
	Weather – fledging survival (proportion lost)	P=0.1, lost=0.5	Broadly derived from specific North Norfolk data (1920-2007)
Metapopulation	Metapopulation links	None incorporated	
Reproductive system and rates	Monogamy/ other	Monogamy	
	Age of first reproduction: females	3 yrs	BWPi (2004)
	Age of first reproduction: males	3 yrs	BWPi (2004)
	Maximum progeny/female/year	2	Stienen (2006) 1.6 eggs on average at Griend, 2% fledge 2 chicks; no data from North Norfolk
	Distribution of progeny/female/year	Not highly skewed	
	Sex ratio at birth	0.5	
	Proportion of adult females breeding	100%	Non-breeders excluded from model
	Density dependent impact on proportion of adult females breeding	Independent: 100% breeding	
	Allee effect in operation?	Yes	
Reproduction rate	Mean chicks fledged/female/ year	0.623	Derived from North Norfolk data 1966-2007
Carrying capacity	Adult population	7400 ± 250 ind.	

1.1.30 Such frequency and intensity is in keeping with the fact that over the years for which data are available, reproductive success was below 0.25 chicks per pair in 30.7% of years at Blakeney Point, and 27.2% of years at Scolt Head. In other words, productivity was less than a third of the mean value in a third of the years since the population has stabilised. However, as Sandwich terns do not generally have the option of raising more than one chick (just 2% of two-egg clutches at Griend in the Netherlands fledge a second chick – Stienen 2006), reproductive rates show relatively little fluctuation in the face of catastrophes.

1.1.31 The likelihood of wind farm collision (such mortality is generally termed ‘harvesting’ in PVA models) for each individual was determined specifically for each of Docking Shoal and Race Bank OWFs separately and in-combination on the basis of predicted collision risk estimates derived from boat-based surveys for different avoidance rates (95%, 98%, 99%, 99.5% and 99.6%) and presented in the respective ESs (Centrica Energy 2008, 2009) (Table 1.2). Collision risk estimates from boat-based surveys incorporate observed passage rates and flight height proportions in each case, assuming random movement of all individuals. A range of possible combinations of numbers and size of turbines at both Docking Shoal and Race Bank (Table 1.2) were varied in different model scenarios. As individual terns do not return to colonies until they reach age 3, individuals aged from one to two years old are assumed not to be at risk from wind farm collision in ViaPop.

Table 1.2 Summary of collision rate predictions for Docking Shoal OWF, Race Bank OWF and Sheringham Shoal OWF

Site	Turbine layout (number and size)	Survey method	Collision risk factor (%)	Rates of avoidance (%)					
				0	95	98	99	99.5	99.6
Docking Shoal	138 x 3.6 MW	Boat-based[1]	10.3	10,815	541	216	108	54	43.3
		Foraging model[2]	10.3	4515	226	90	45	22.5	18
	105 x 5 MW	Boat-based	9.7	8,682	434	174	87	43	34.7
	87 x 6 MW	Boat-based[1]	9.3	7,741	387	155	77	39	31
Race Bank	138 x 3.6 MW	Boat-based[1]	10.3	6777	339	136	68	34	27
		Foraging model[2]	10.3	1225	61	25	12	6	5
	105 x 5 MW	Boat-based	9.6	5418	271	108	54	27	22
	87 x 6 MW	Boat-based	9.3	4814	241	96	48	24	19
Sheringham Shoal	108 x 3 MW	Boat-based	14.3	1150	58	23	1 2	6	5

[1] Luuk Folkerts Ecofys March 2007 Collision risk modelling for Sandwich terns final- PWNDNL062805-2.pdf

[2] Luuk Folkerts -Ecofys Memo Collision Risk Analysis 20080716.pdf

1.1.32 Moreover, to understand the movements of breeding Sandwich terns in relation to potential wind farm sites, a bespoke individual-based foraging model was developed (Centrica Energy 2008 and 2009). The model is based on observations of breeding adults leaving colony sites, according to bearing specific distance functions broadly derived from tracking of individual birds in a rigid-hulled inflatable boat (RIB). The resulting model can generate predictions of the rates of Sandwich tern passage, as foraging end points and flyovers for any given area of sea within the region. Modelled passage rates tend to be relatively lower than those observed on boat-based surveys (Centrica Energy 2008 and 2009), which accords with the fact that the model does not account for migrating or dispersing individuals from outside of the colony, which may be relatively numerous immediately prior to and especially immedately after the breeding season. The predictions of the foraging model are thought to offer a relatively meaningful representation of potential impacts on the breeding population. The collision risk estimates derived from passage rates from the model were also used in ViaPop simulations.

1.1.33 In all cases, the losses predicted to occur at Sheringham Shoal OWF were incorporated as additional background mortality (Table 1.2). Model scenarios were run for 10 years to establish trends before wind farm mortality (as harvesting) was turned on and run for 25 years, the putative life of the turbines, before harvesting was again turned off to record the response of the population. A total of 50 simulation runs were made for each scenario. Only in Figure 1.5 are all outputs shown. Thereafter, figures show mean population trends together with ±1 Standard Error (SE) bars derived from all 50 runs. Where standard errors do not overlap, differences between trend lines may be interpreted as being statistically significant. For ease of reference, a summary of the extensive comparison between different scenarios is presented in Table 1.3.

Table 1.3 Summary of modelling scenarios undertaken using ViaPop

No.	Figure no.	Parameter variation from default set
1	5	None, default set, no harvesting
2	6	No harvest + catastrophes
3	7	Sensitivity analysis: Reproduction rate (20% variation, maximum 0.74, no harvesting)
4	8	Sensitivity analysis: adult mortality (10% variation around mean, no harvesting)
5	9	Sensitivity analysis: juvenile mortality (20% variation around mean, no harvesting)
6	10	Harvesting, Docking 3.6 MW, avoidance rate comparison
7	11	Harvesting, Race 3.6 MW, avoidance rate comparison
8	12	Site comparison with harvesting at Race and Docking 3.6 MW, 95% avoidance
9	13	Site comparison with harvesting at Race and Docking 3.6 MW, 98% avoidance
10	14	Site comparison with harvesting at Race and Docking 3.6 MW, 95% avoidance + catastrophes
11	15	Site comparison with harvesting at Race and Docking 3.6 MW, 98% avoidance + catastrophes
12	16	Harvesting, Docking 3.6 MW, and Race 3.6 MW, avoidance rate comparison
13	17	Harvesting, Docking 3.6 MW, Race 3.6 MW + fox-induced mortality, avoidance rate comparison.
14	18	Harvesting, Docking 3.6 MW, Race 3.6 MW + all catastrophes, avoidance rate comparison
15	19	Sensitivity analysis: Reproduction rate (20% variation around mean, maximum 0.93, harvesting: Race 3.6 MW, avoidance rate 98%).
16	20	Sensitivity analysis: Reproduction rate (20% variation around mean, maximum 0.74, harvesting: Docking 3.6 MW, avoidance rate 98%)
17	21	Sensitivity analysis: Reproduction rate (50% variation around mean, maximum 0.74, harvesting: Race 3.6 MW, avoidance rate 98%)
18	22	Sensitivity analysis: Reproduction rate (20% variation around mean, maximum 0.93, harvesting: Race 3.6 MW, avoidance rate 99%).
19	23	Sensitivity analysis: Reproduction rate (20% variation around mean, maximum 0.93, harvesting: Docking 3.6 MW, avoidance rate 99%).
20	24	Sensitivity analysis: adult mortality (10% variation around mean, harvesting Race 3.6 MW, avoidance rate 98%)
21	25	Sensitivity analysis: adult mortality (10% variation around mean, harvesting Docking 3.6 MW, avoidance rate 98%)
22	26	Sensitivity analysis: adult mortality (10% variation around mean, harvesting Race 3.6 MW, avoidance rate 99%)
23	27	Sensitivity analysis: adult mortality (10% variation around mean, harvesting Docking 3.6 MW, avoidance rate 99%)
24	28	Sensitivity analysis: juvenile mortality (20% variation around mean, harvesting Race 3.6 MW, avoidance rate 98%)
25	29	Sensitivity analysis: juvenile mortality (20% variation around mean, harvesting Docking 3.6 MW, avoidance rate 98%)
26	30	Sensitivity analysis: juvenile mortality (20% variation around mean, harvesting Race 3.6 MW, avoidance rate 99%)
27	31	Sensitivity analysis: juvenile mortality (20% variation around mean, harvesting Docking 3.6 MW, avoidance rate 99%)
28	32	Sensitivity analysis –reduction of mean production and survival parameters by 10%. Harvesting, Docking 3.6 MW and Race 3.6 MW, avoidance comparison)
29	33	Sensitivity analysis – vulnerability (Reduction of mean production and survival parameters by 10% + catastrophes. Harvesting, Docking 3.6 MW and Race 3.6 MW, avoidance rate comparison)
30	34	Harvesting, Docking 3.0 MW, and Race 3.0 MW, avoidance rate comparison
31	35	Harvesting, Docking 3.6 MW, and Race 3.6 MW, avoidance rate comparison
32	36	Harvesting, Docking 5 MW, and Race 5 MW, avoidance rate comparison
33	37	Harvesting, Docking 6 MW, and Race 6 MW, avoidance rate comparison
34	38	Foraging model rates. Harvesting, Docking 3.6 MW and Race 3.6 MW, avoidance rate comparison
35	39	Foraging model rates, comparing harvesting at Race and Docking 3.6 MW, 95% avoidance rate

Results

The North Norfolk Sandwich Tern Colonies

1.1.34 In the nineteenth century, numbers of Sandwich terns breeding in Britain were considerably lower than the present day. In 1920, the total British population was thought to be as low as 2,000 pairs (Brown and Grice 2005). Since that time, there has been an upward trend in population size. Prior to 1900, Sandwich terns were irregular visitors to Norfolk, with the first confirmed breeding occurring in 1920 (at Blakeney Point, Taylor *et al* 1999). Although Blakeney Point was initially the principal breeding site, breeding also occurred at Scolt Head on a periodic basis from 1923 onwards.

1.1.35 Following initial colonisation, the Norfolk population grew rapidly to a level of more than 1,000 pairs in 1929 (Figure 1.4). This speed of expansion suggests that the newly established population continued to receive immigrants from outside the county. The origin of this influx is difficult to determine, but probably involved birds from colonies in Kent, Essex and Suffolk, all of which had been abandoned in preceeding years (Brown and Grice 2005). Influxes may have also occurred from colonies further afield. This seems likely given that the rate of increase in Norfolk was extremely high relative to the size of the British breeding population at the time.

1.1.36 After the initial period of growth, the Norfolk population remained relatively stable for the following 20 years, with numbers peaking at around 2,000 breeding pairs. From 1962 onwards, a second significant period of growth occurred, with numbers building to 4,057 pairs in 1970. A number of other sites were also used in this period with Salthouse supporting up to 1,000 pairs especially in the mid to late 1930s. This site was last used in 1944 by 20 pairs. In 1959 and 1960, 400-600 pairs used a site near Warham. Thereafter, Stiffkey Binks, located within approximately 3 km of Blakeney Point, which had only been used periodically since the late 1960s became important, supporting 350 pairs in 1972. The site unexpectedly became the principal colony in 1973 with 3,500 pairs fledging 3,000 chicks. Nesting also occurred in 1974 and 1977, but not again until 2001-2004 inclusive with a maximum of 800 pairs. Scroby Sands in east Norfolk at around 77 km from Blakeney Point was periodically used by up to 400 pairs from 1949 – 1976, after which the bank became submerged at high water.

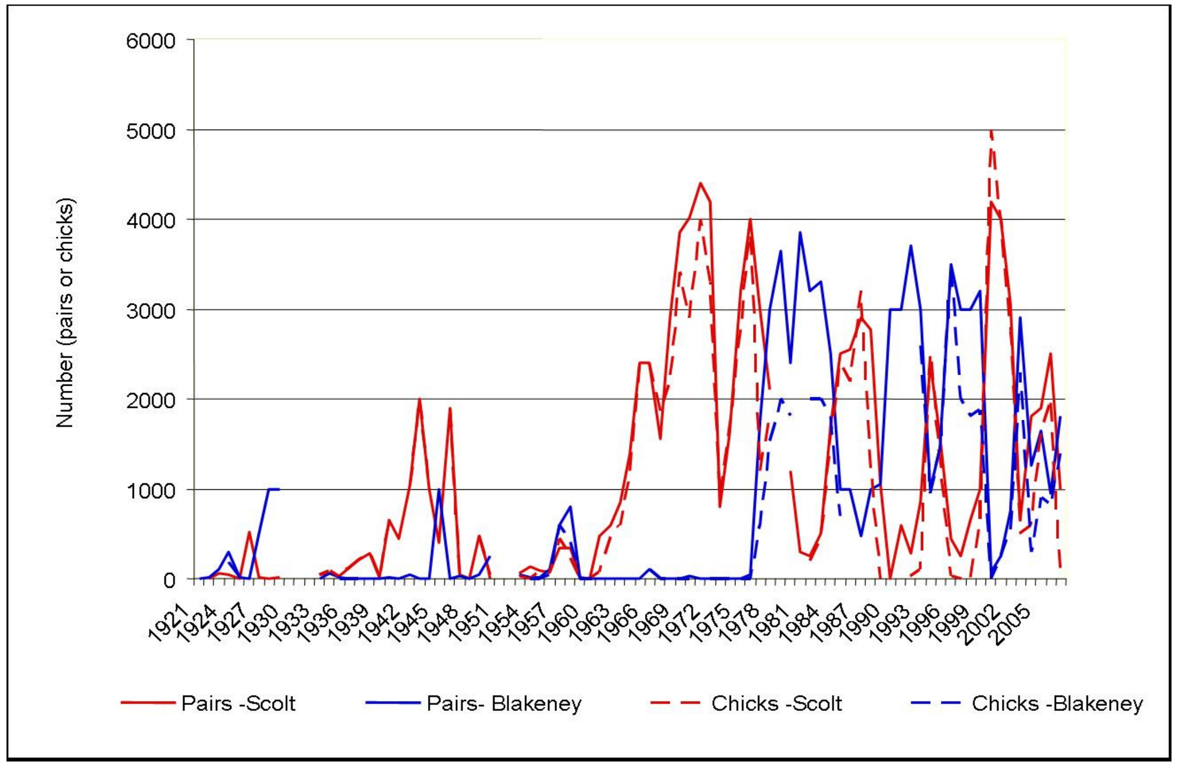

Figure 1.4 Annual numbers of breeding pairs and chicks raised at the two principal Sandwich tern breeding sites in Norfolk since colonisation of the county in 1920

1.1.37 Within the North Norfolk population and as depicted in Figure 1.4, fluctuations in breeding numbers at the two major sites, Scolt Head and Blakeney Point have always been closely correlated, strongly indicating a pattern of alternation between the sites by the same breeding population. The factors responsible for colony switching are not known but may include changing prey availability in different parts of The Wash in different years, the effects of kleptoparasitism by black-headed gulls (see Stienen 2006) as well as predation and disturbance. Predators (particularly foxes) and poor weather may have catastrophic effects on colony productivity in some years. For example, around 1,000 chicks died after storms in 2004 (see 1.1.20 above).

1.1.38 Given the rapidity of growth up to the 1970s it is likely that much of this population increase related to immigration from colonies elsewhere, including from other countries. Ring-recoveries offer some evidence of inter-colony movement. Of the large numbers of individuals ringed as chicks in Britain and Ireland, 567 have subsequently been recovered at locations more than 20 km from their natal site during the breeding season (birds recovered at breeding age). This suggests a relatively high rate of breeding dispersal, with some individuals recruiting into colonies as far away as the Danube Delta, southern Spain and southern Italy (Wernham *et al* 2002).

1.1.39 This raises the possibility that patterns occurring at other colonies in the metapopulation could be reflected in north Norfolk. For example, numbers of birds in the Netherlands fell from around 16,000 pairs at the beginning of the 20th century to just a few hundred by 1910 (Stienen 2006), largely as a result of human persecution. This, as well as movement from within the UK (see above) may have influenced the colonisation of Norfolk, although by the time the Norfolk colonies were established, numbers in the Netherlands were again on the increase. The population in the Netherlands then crashed from a maximum of 46,000 pairs in the 1950s to just 875 pairs by 1965, which was attributed to organochlorine pollution in the Rhine. Although high rates of mortality were noted for both chicks and adults during this period (Stienen 2006), it is also likely that large numbers of individuals abandoned the region entirely. This seems likely to have been reflected (again) as immigration into the Norfolk population which increased (and stabilised) in the 1970s onwards, as well as into breeding colonies elsewhere such as in Germany and France (Mitchell *et al* 2004).

1.1.40 Given this potential level of interconnectivity, the predicted impacts of wind farm development on the Norfolk breeding population should also be considered at the wider metapopulation scale. Connectivity between sub-populations (for example those in Norfolk and the Netherlands) presents a significant complication when trying to understand the effects of wind farm mortality on population dynamics. Unfortunately, data on inter-colony dispersal rates for Sandwich tern are relatively scant, and as such it is impossible to incorporate this parameter into population viability analysis with any level of confidence. The ViaPop model therefore assumes a closed local breeding population without immigration. As such, the model represents a simplification of the likely real situation, although this does not necessarily imply that model predictions will be inaccurate in terms of population impacts. Rather, it should be noted that any effects of collision mortality on the local population might be difficult to detect in the field if they are buffered or masked by immigration from colonies elsewhere.

Outputs of ViaPop

In the Absence of Wind Farm Collision

1.1.41 The first test of the model was to determine whether it realistically simulates the Norfolk Sandwich tern population in the absence of wind farm collision or natural catastrophes. Based on starting parameters derived from the available literature, the outputs of the model show variation both between years and between runs (ie repeats of each simulation). In all cases, however, the model predicts a stable population over the entire simulation period, with numbers of individuals remaining close to carrying capacity (Figure 1.5). The model thus appears to provide a meaningful representation of the population at equilibrium state, agreeing fairly closely with recent actual population trends.

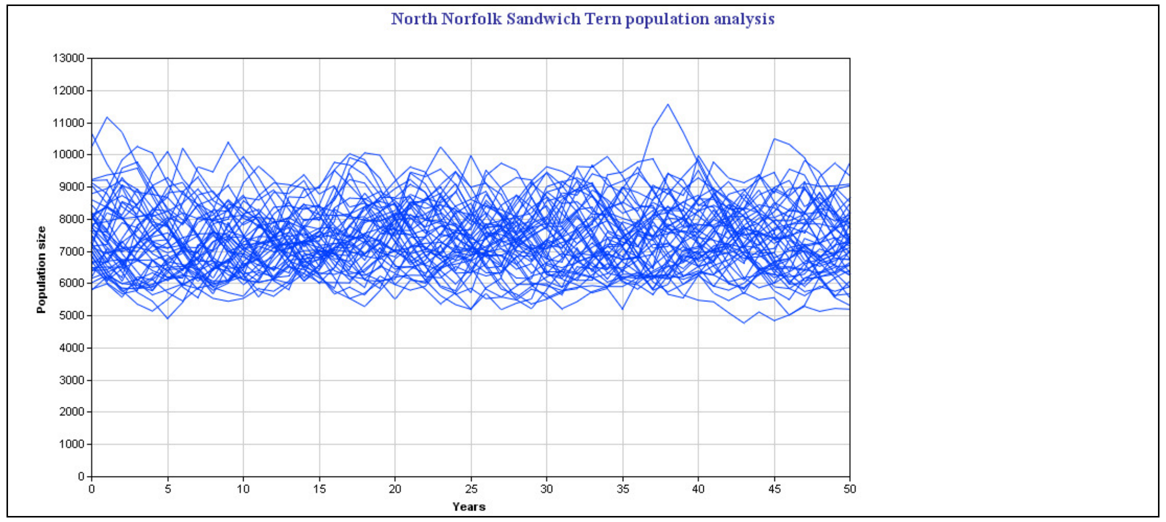

Figure 1.5 Outputs of 50 simulation runs for a scenario incorporating background variation in reproductive success and age-specific survival rates, but excluding the effects of natural catastrophes and wind farm collision

1.1.42 When natural catastrophes are incorporated into the model, the magnitude of variation within and between runs is slightly increased, but overall population stability remains similar to that simulated in the basic model (Figure 1.6). This outcome suggests that periodic catastrophes of the magnitude assessed here (see 2.3) do not have the potential to cause population declines in the long term. Again, this is tentatively supported by empirical evidence, given that breeding numbers in the north Norfolk population have appeared to remain relatively stable despite reasonably frequent catastrophic events.

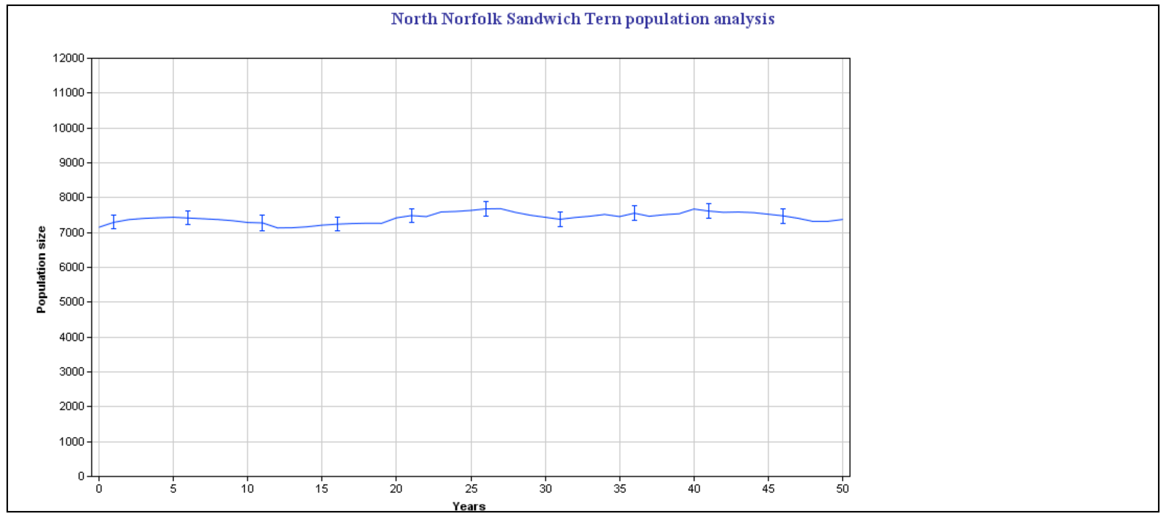

Figure 1.6 Mean (±1 SE) population trend for simulations incorporating background productivity variables together with periodic natural catastrophes.

1.1.43 In order to understand the relative influence of background variation in reproductive success and survival rates on population trends, sensitivity analysis was performed on the basic model, both with and without natural catastrophes. This involved re-running the simulation repeatedly with slight variations made to each parameter in question.

1.1.44 Simulations were run over a range of values for reproductive success representing 20% variation around the mean (with a maximum of 0.74 chicks per pair) (Figure 1.7). These showed that if reproductive success was to decline by 20% (to 0.50 chicks per pair per year), the population would very gradually decrease, becoming significantly different from a population generated from the mean value. At a slightly higher reproductive rate of 0.56 chicks per pair per year, the population would again be fairly stable, but with fluctuation, so that in some years no difference in population size compared to that generated by the mean value could be detected.

1.1.45 It should also be noted that increases in breeding productivity of up to 20% do not result in overall increases in population size. This is because the starting population is already assumed to be at carrying capacity, such that any significant increases in population size are kept in check in the model by density dependent mortality, maintaining the population size around the original level. In the natural situation, birds in such a population may simply emigrate to other colonies, with the Norfolk population operating as a ‘source’ population.

1.1.46 Sensitivity analysis was also performed on adult survival rate (10% variation around mean) and juvenile survival rate (20% variation around mean). These analyses showed that populations would remain stable across the simulation period despite fluctuations of similar magnitude to the variation in the parameter (Figures 1.8 and 1.9).

1.1.47 Overall, the basic model predicts stability for the population on the basis of what is known of the relevant parameters for Sandwich terns in the north Norfolk population. Current evidence suggests that this prediction is broadly realistic, implying that the validity of the model is sufficient to make further exploration worthwhile.

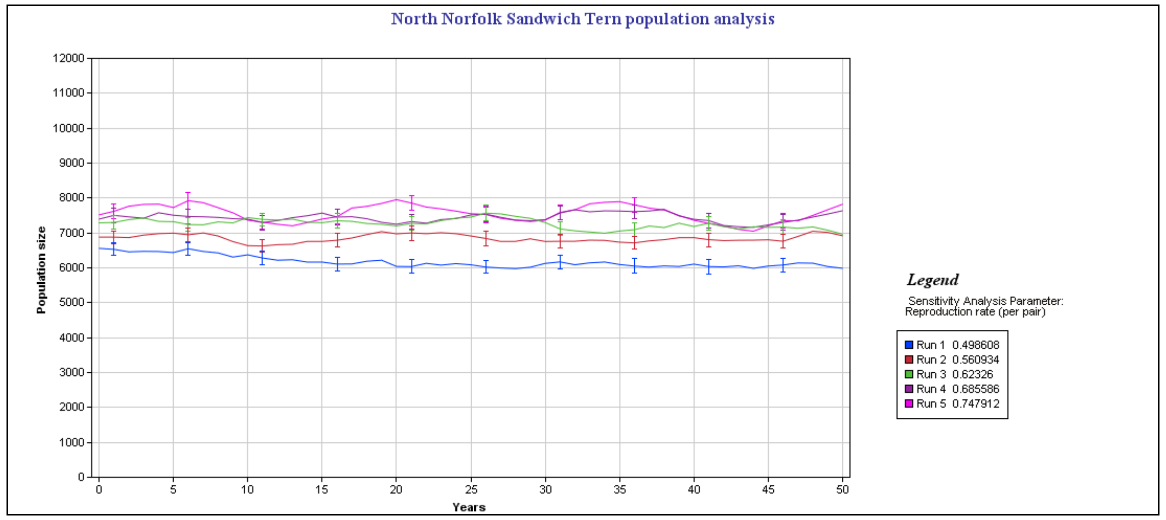

Figure 1.7 Mean (±1 SE) population trends for simulations in which reproductive rate (ie number of chicks raised per pair per year) is varied up to 20% around the mean value derived from field studies

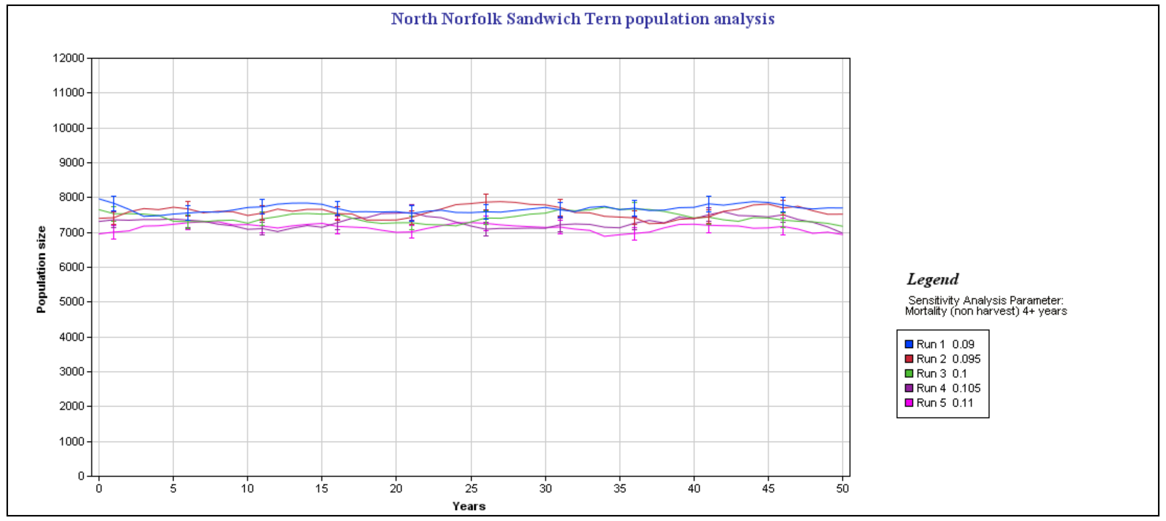

Figure 1.8 Comparison of mean (±1SE) population trends for simulations in which adult survival is varied up to 10% around the mean value derived from field studies

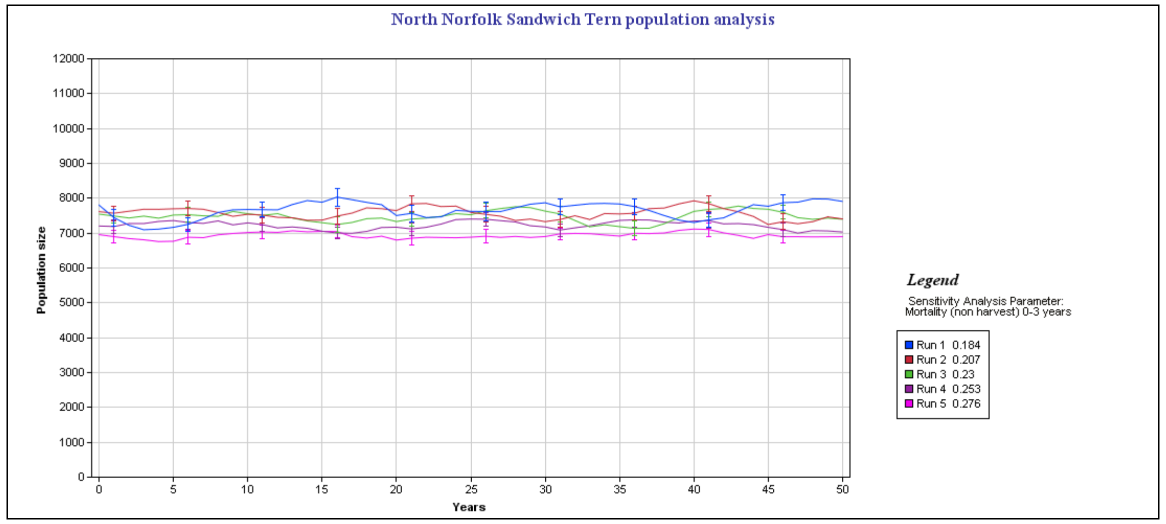

Figure 1.9 Comparison of mean (±1SE) population trends for simulations in which juvenile survival is varied up to 20% around the mean value estimated from field studies

Incorporating Wind Farm Collision

1.1.48 Simulations were run for a variety of scenarios including the operation of either or both proposed wind farms at Docking Shoal and Race Bank, together with a range of potential avoidance rates. Simulation runs were repeated across various wind farm layout scenarios involving different numbers and sizes of turbines. The results of simulation will be discussed for each modelled variable in turn: wind farm site, avoidance rate and wind farm layout. For all scenarios, proportionate rates of decline were calculated by assessing changes in mean population sizes across all model runs during the harvest period. Mean trends are shown in figures together with standard error bars. Where standard error bars overlap, trend lines are not considered to differ significantly from one another.

Effects of Docking Shoal in Isolation

1.1.49 Simulations were run to compare the relative impact on population trends of each proposed wind farm in isolation. Figure 1.10 shows the mean simulated population trends if Docking Shoal were to operate in isolation with a 3.6 MW turbine array, over a range of potential avoidance rates (with all other parameters set at mean levels).

1.1.50 At avoidance rates of 98% or less, mortality from Docking Shoal generates significant population declines, with a 52% decline predicted over 25 years at 95% avoidance. At an avoidance rate of 99%, mean population trends are significantly lower than those without harvest in some years but not others, indicating a slightly reduced but relatively stable population. At avoidance rates above 99%, population trends do not differ significantly from baseline conditions ie zero wind farm collision throughout the harvest period.

Effects of Race Bank in Isolation

1.1.51 Figure 1.11 shows the equivalent simulated population trends if Race Bank were to operate in isolation with a 3.6 MW turbine array. In this case, significant declines are predicted at an avoidance rate of 95%, but scenarios with avoidance rates of 98% or more do not differ significantly from those without collision. The magnitude of decline at 95% is considerably less than that predicted for Docking Shoal, with a 29% decline predicted over 25 years for Race Bank.

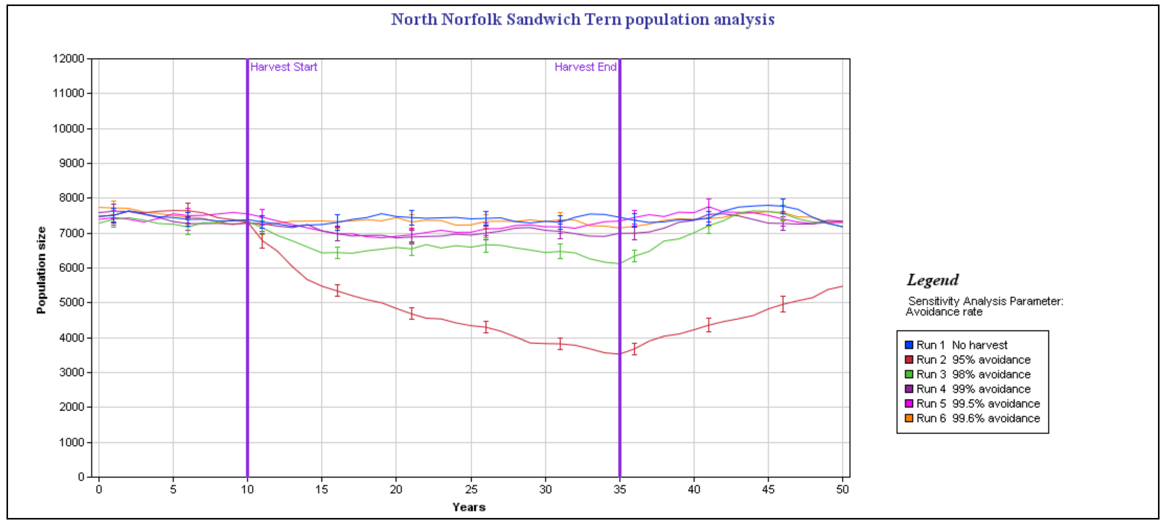

Figure 1.10 Mean (±1SE) simulated population trends in which Docking Shoal operates in isolation (3.6 MW) across a range of avoidance rates

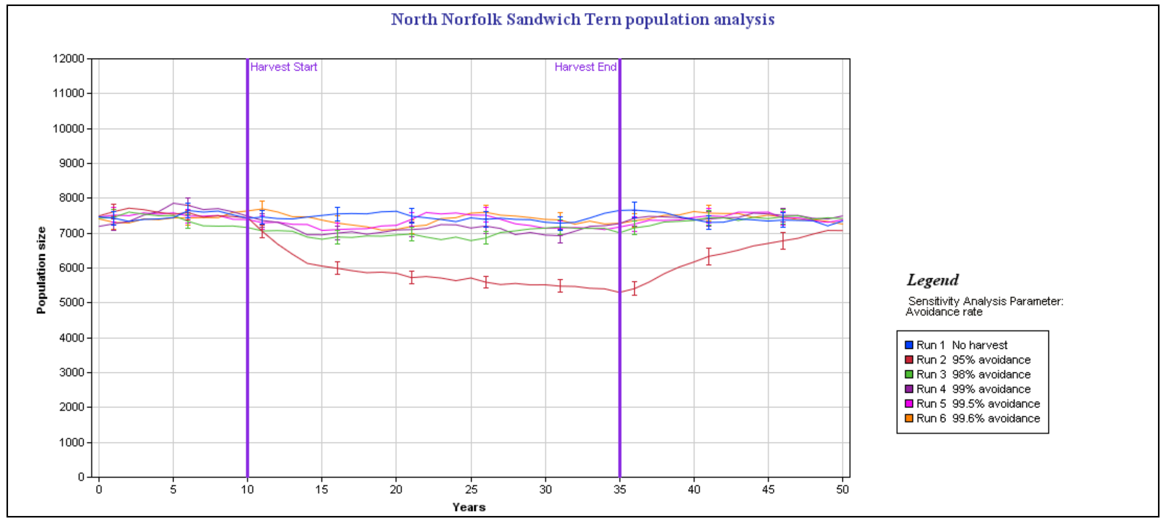

Figure 1.11 Mean (±1SE) simulated population trends in which Race Bank operates in isolation (3.6 MW) across a range of avoidance rates

Comparison of the Effects of Both Sites

1.1.52 Figure 1.12 shows a direct comparison of simulated mean population trends resulting from the separate operation of each wind farm, assuming a 95% avoidance rate and predicted mortality from Sheringham Shoal included in the baseline. As implied above, population declines resulting from collision mortality at Docking Shoal are significantly greater than those resulting from collision at Race Bank.

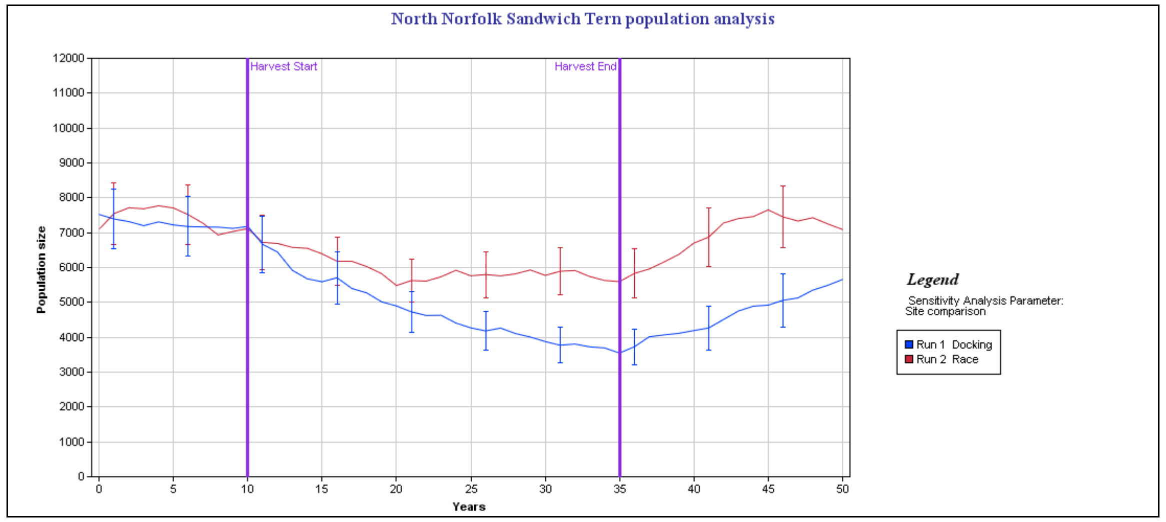

Figure 1.12 Comparison of mean (±1SE) population trends for simulations in which each wind farm operates separately assuming an avoidance rate of 95%, together with a 3.6 MW turbine array

1.1.53 A direct comparison at 98% avoidance is shown in Figure 1.13. In this case, the magnitude of difference between the two sites is considerably lower.

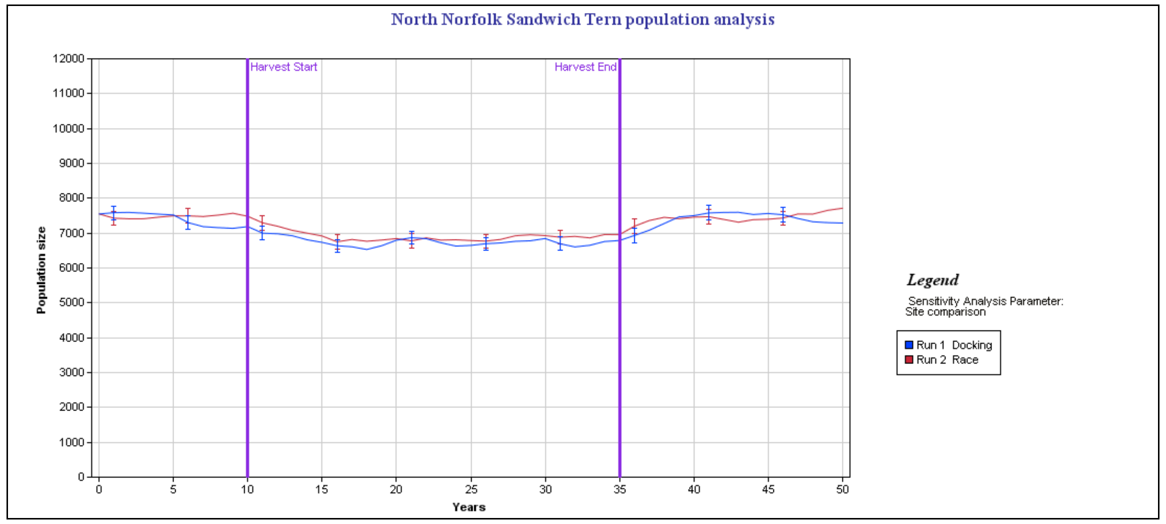

Figure 1.13 Comparison of mean (±1SE) simulated population trends for simulations in which each wind farm operates separately, assuming an avoidance rate of 98%, together with a 3.6 MW turbine array

1.1.54 The results are broadly similar if natural catastrophes are included, but with some differences dependent upon the avoidance rate. In the 95% avoidance rate simulation, the results were very similar regardless of whether natural catastrophes were included (Figure 1.14). However, in the case of the 98% avoidance rate simulations, the difference between Docking Shoal and Race Bank is slightly more significant, with a 22% decline occurring over 25 years at Docking Shoal in comparison to a 15% decline at Race Bank (Figure 1.15).

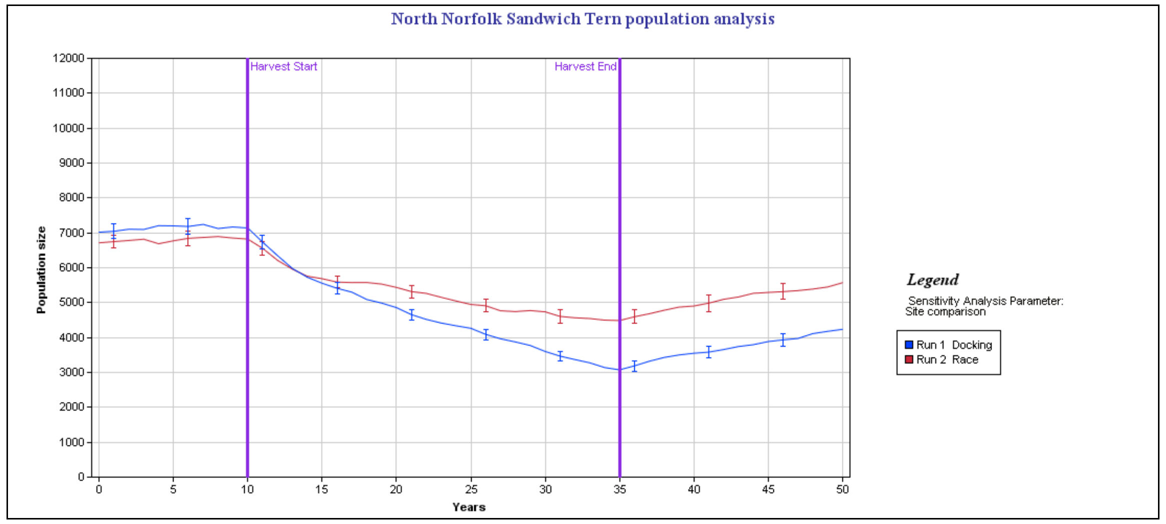

Figure 1.14 Comparison of mean (± 1SE) population trends for simulations in which each wind farm operates separately, assuming an avoidance rate of 95%, together with a 3.6 MW turbine array. Natural catastrophes are included

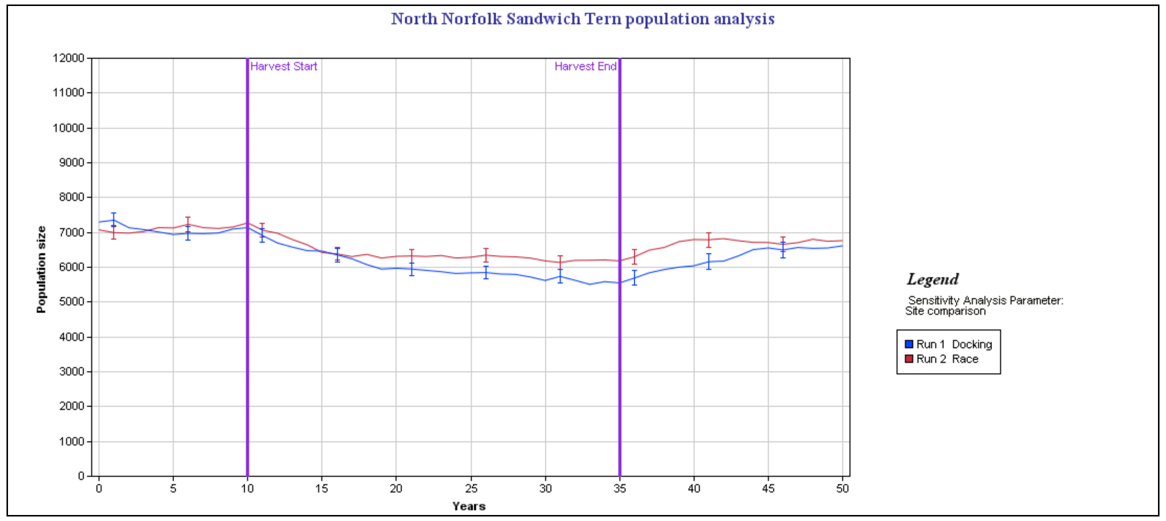

Figure 1.15 Comparison of mean (±1SE) population trends for simulations in which each wind farm operates separately, assuming an avoidance rate of 98%, together with a 3.6 MW turbine array. Natural catastrophes are included

Effect of Avoidance Rate

1.1.55 Figure 1.16 shows the mean simulated population trends for a series of potential avoidance rates. These scenarios assume that both Docking Shoal and Race Bank wind farms are operating simultaneously with 3.6 MW turbine arrays. As might be expected, avoidance rate has a significant effect on simulated population trends.

1.1.56 At an avoidance rate of 99.5% or above, simulated populations remain stable and do not differ significantly from the scenario without wind farm collision. At 99% avoidance, the population trend is significantly lower than that of the scenario without wind farm collision, but the magnitude of decline is small (5% decline over 25 years). Declines were considerably more significant at 98% avoidance (21% decline over 25 years), and greater still at 95 % avoidance (76% decline over 25 years).

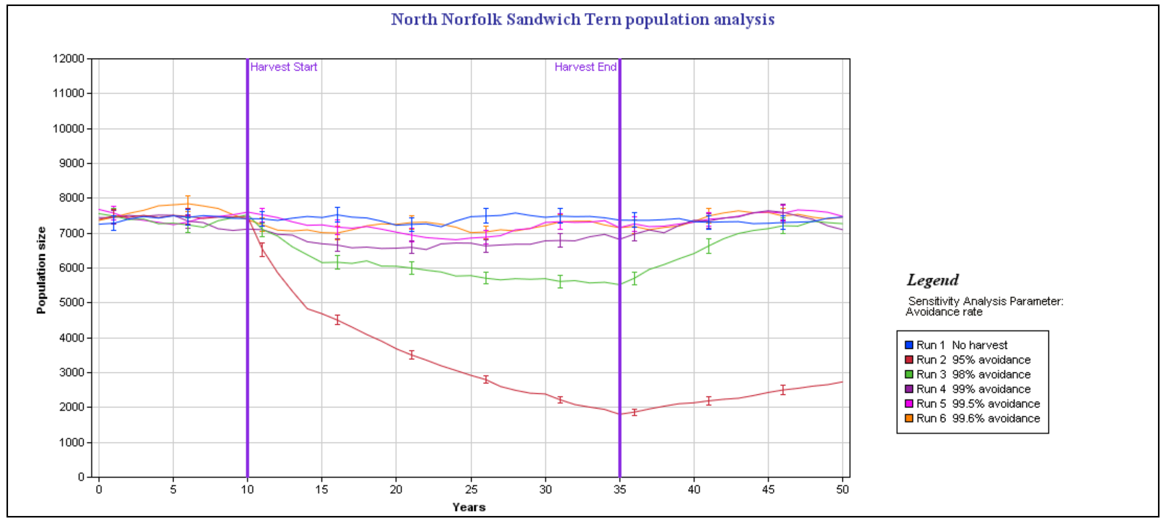

Figure 1.16 Comparison of mean (±1 SE) simulated population trends over a range of potential avoidance rates. Each simulation assumes both Race Bank and Docking Shoal wind farms are operational with 3.6 MW turbines

1.1.57 When natural catastrophes are included in model simulations, the results are broadly similar. Figure 1.17 shows the results of models incorporating fox-induced predation catastrophes, together with the effects of both wind farms operating 3.6 MW turbine arrays. In comparison to Figure 1.16 (ie without any natural catastrophes), the results are very similar, with no significant differences in trends even at high avoidance rates.

1.1.58 Figure 1.18 shows the equivalent simulation results including all natural catastrophes. Again, there is no evidence that population trends differ as a result of catastrophes at any avoidance rate, at least for the magnitude of catastrophic events considered here. Nonetheless, catastrophes of greater magnitude or frequency could conceivably have more significant impacts.

Additive Effects of Variation in Population Parameters

1.1.59 Sensitivity analysis was conducted on the impact of variation in both survival and reproductive rates on populations subjected to collision risk from each wind farm in turn.

1.1.60 Figure 1.19 shows how variation in reproductive success influences simulated population trends for a scenario involving a 3.6 MW turbine array at Race Bank with an avoidance rate of 98%. If reproductive success were reduced by either 10% or 20% below mean (ie to 0.56 or 0.50 chicks per pair per year respectively), population declines over the design-lives of the turbines are significantly steeper than at reproductive success levels at or above the mean level.

1.1.61 A 20% reduction in reproductive success (plus collision mortality) results in a population decline of 28% over 25 years, as opposed to a decline of 16% occurring at the mean reproductive success level. Similar results were found when the same sensitivity analysis was carried out for a 3.6 MW turbine array at Docking Shoal (Figure 1.20). When reproductive success was reduced by 50% for the same scenario at Race Bank, the magnitude of the resulting population decline was more dramatic, with a 65% population decline occurring over 25 years (Fig. 21).

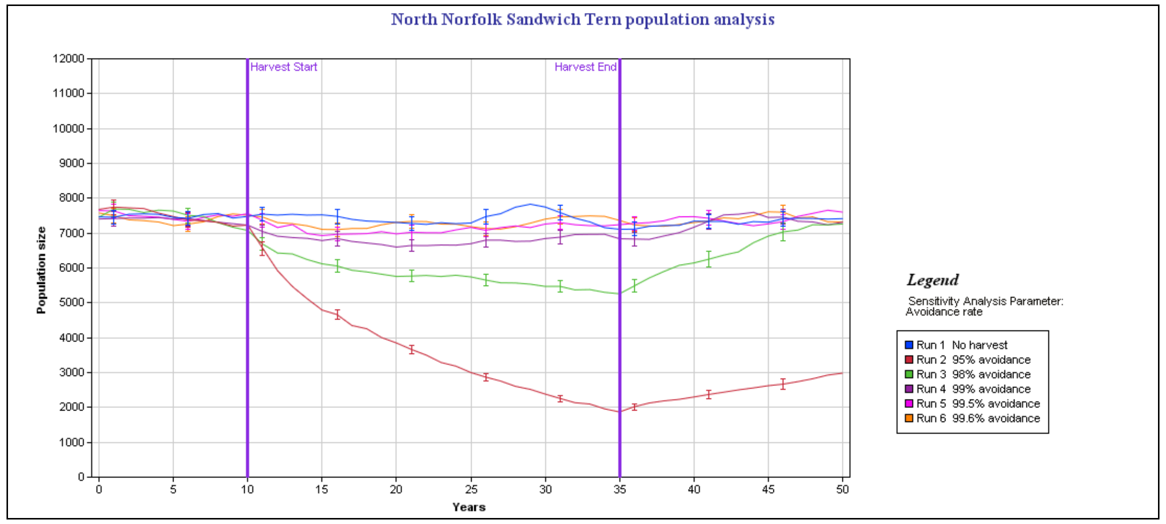

Figure 1.17 Comparison of mean (±1 SE) simulated population trends over a range of potential avoidance rates for model simulations including the effects of fox-induced predation events. Each simulation assumes both Race Bank and Docking Shoal wind farms are operational with 3.6 MW turbines

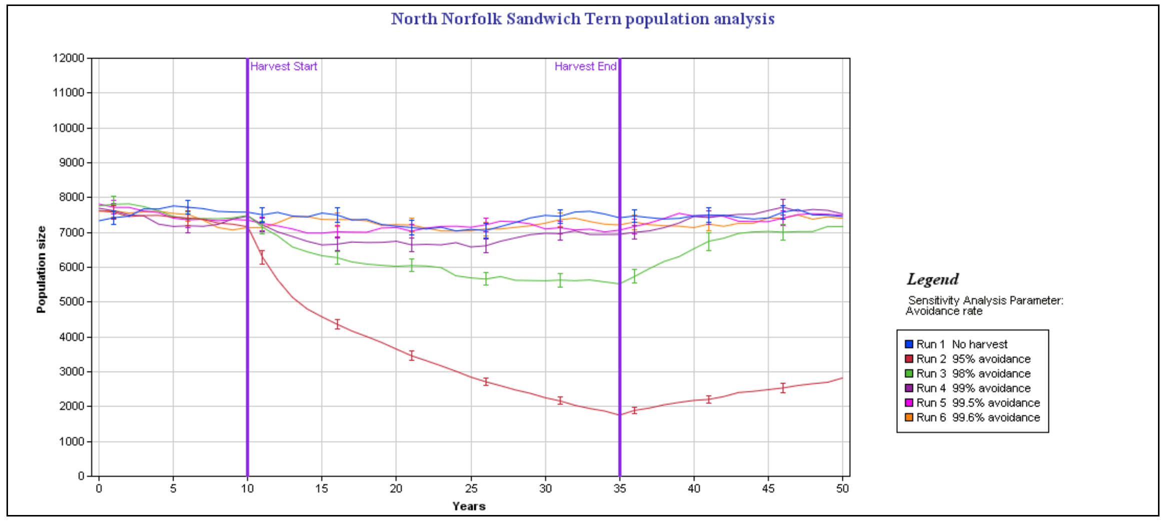

Figure 1.18 Comparison of mean (±1 SE) simulated population trends over a range of potential avoidance rates for model simulations including the effects of all simulated natural catastrophes. Each simulation assumes both Race Bank and Docking Shoal wind farms are operational with 3.6 MW turbines

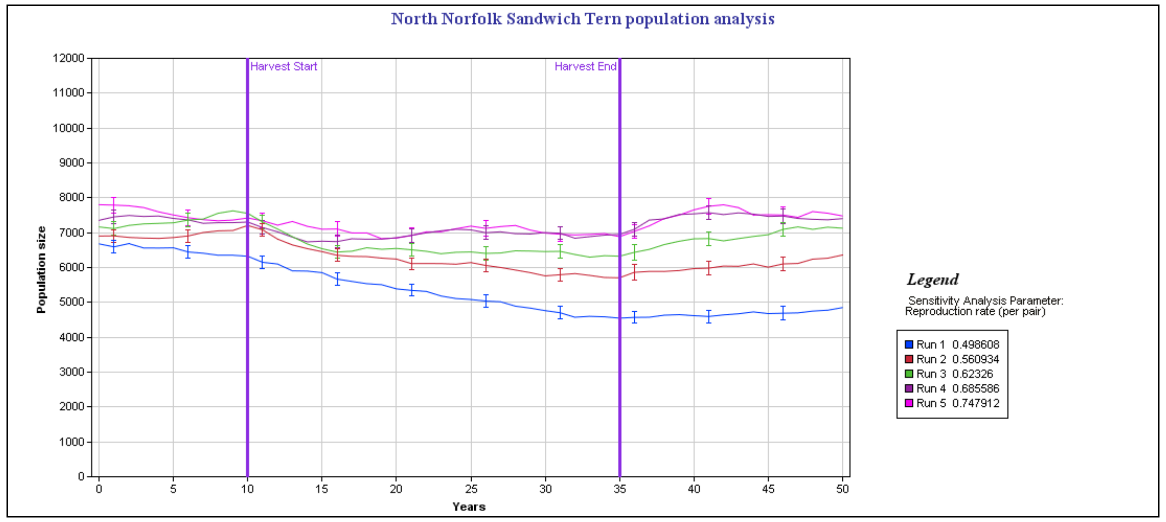

Figure 1.19 Comparison of mean (±1 SE) simulated population trends with collision mortality at Race Bank (3.6 MW) over a range of reproductive success rates, representing 20% variation around the mean value derived from field studies. Each scenario assumes an avoidance rate of 98%

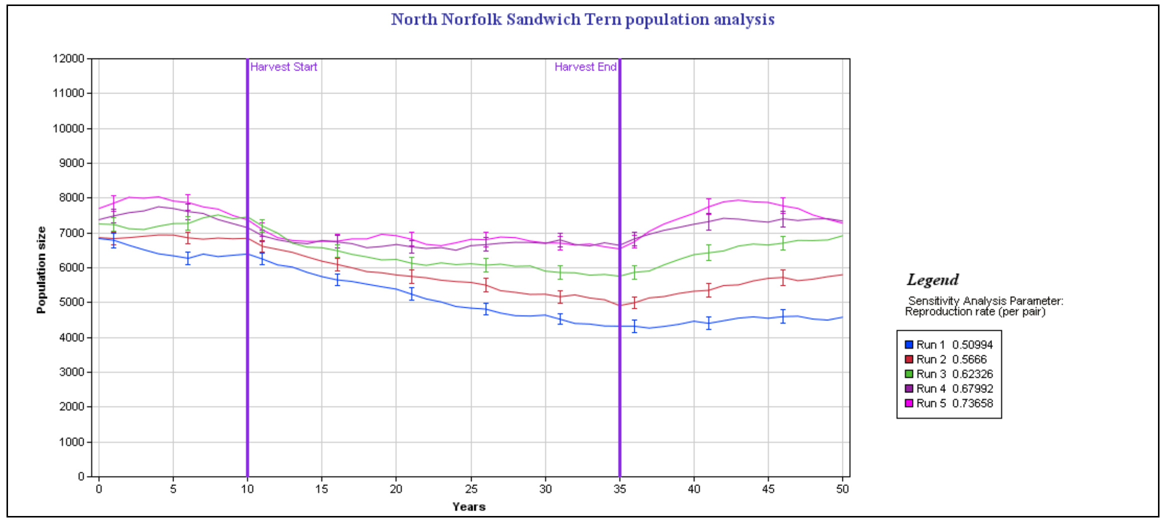

Figure 1.20 Comparison of mean (±1 SE) simulated population trends with collision mortality from Docking Shoal (3.6 MW) over a range of reproductive success rates, representing 20% variation around the mean value derived from field studies. Each scenario assumes an avoidance rate of 98%

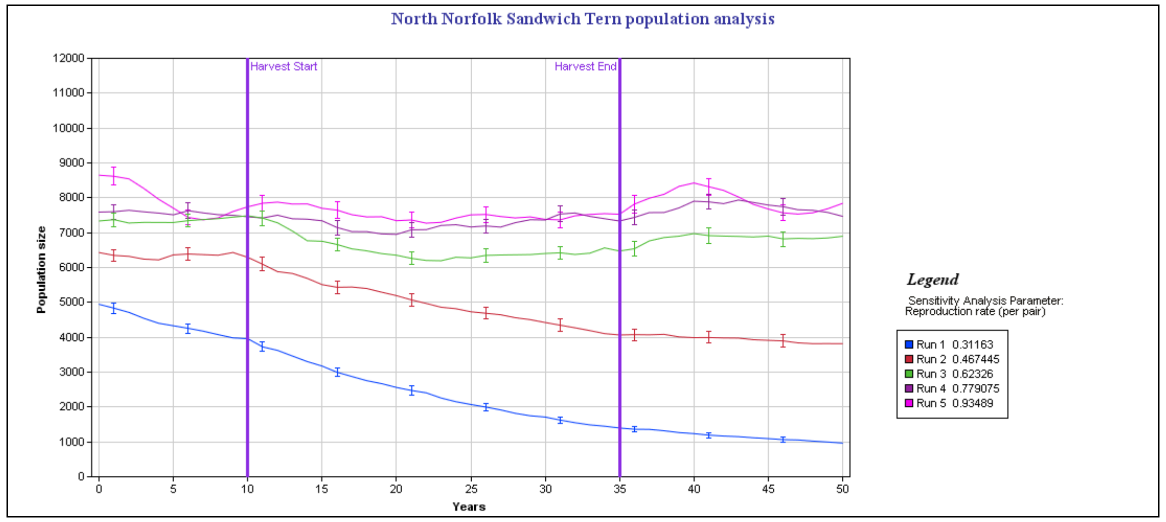

Figure 1.21 Comparison of mean (±1 SE) simulated population trends with collision mortality at Race Bank (3.6 MW) over a range of reproductive success rates representing 50% variation around the mean value derived from field studies. Each scenario assumes an avoidance rate of 98%

1.1.62 When these analyses were repeated assuming an avoidance rate of 99%, the influence of variation in reproductive success on population trends was significantly lower. For both Race Bank (Figure 1.22) and Docking Shoal (Figure 1.23), significant negative population trends were only noted when reproductive success was reduced by 20% below mean levels. These trends were shallower than the equivalent trends at 98% avoidance (Figure 1.19 and Figure 1.20), representing declines of 11.2% for Race Bank and 16.1% for Docking Shoal. A reduction of 10% in reproductive success resulted in relatively stable population trends for both sites neither of which differed significantly from those predicted at mean reproductive success levels (Figure 1.22 and Figure 1.23).

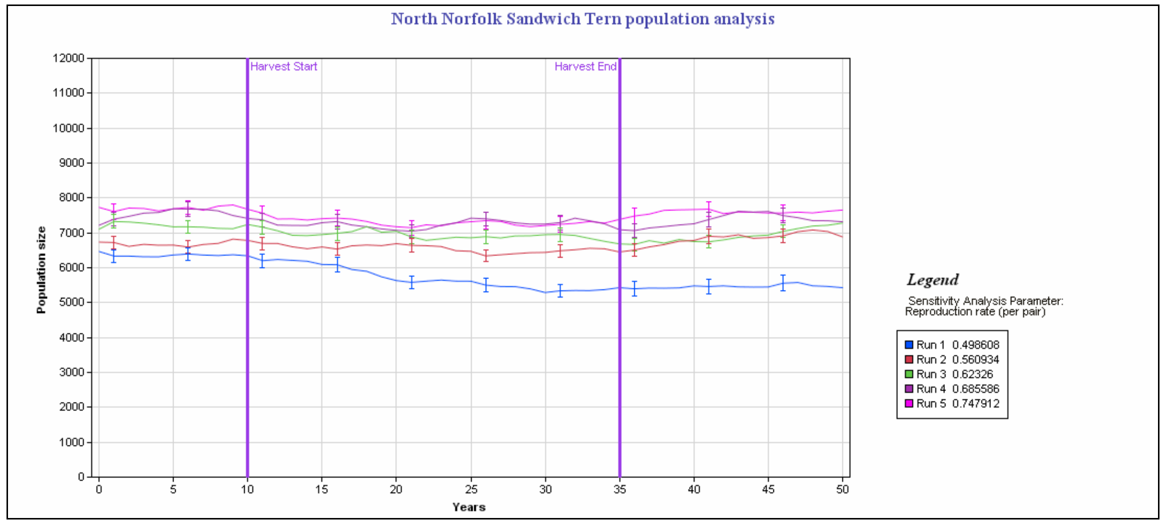

Figure 1.22 Comparison of mean (±1 SE) simulated population trends with collision mortality at Race Bank (3.6 MW) over a range of reproductive success rates, representing 20% variation around the mean value derived from field studies. Each scenario assumes an avoidance rate of 99%

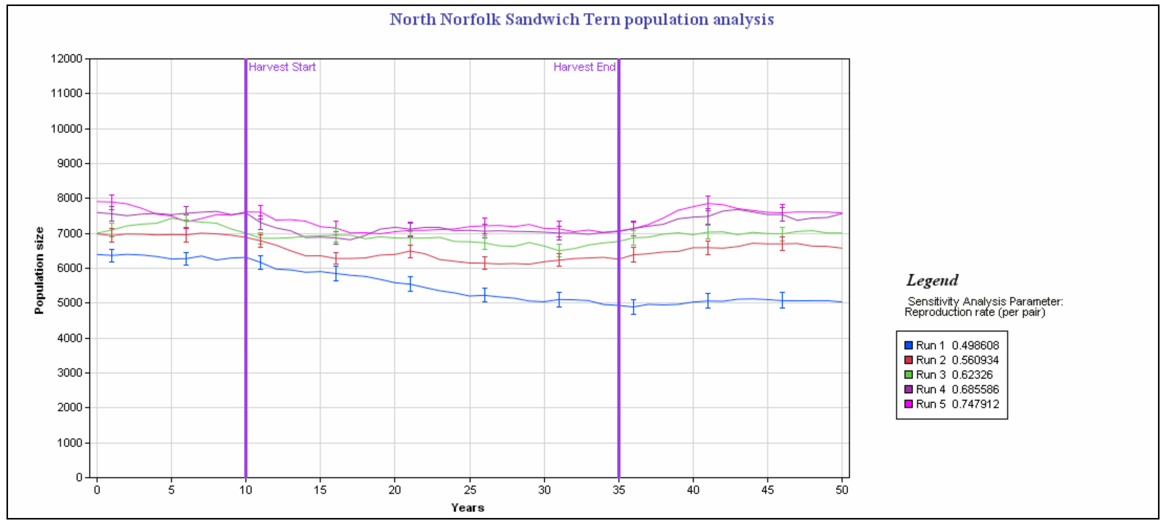

Figure 1.23 Comparison of mean (±1 SE) simulated population trends with collision mortality at Docking Shoal (3.6 MW) over a range of reproductive success rates, representing 20% variation around the mean value derived from field studies. Each scenario assumes an avoidance rate of 99%

1.1.63 Sensitivity analysis was also used to assess the influence of background variation in adult and juvenile survival rates upon population trends. Variation in adult survival rate (10% around mean) was found to have negligible influence assuming a 98% avoidance rate at both Race Bank (Figure 1.24) and Docking Shoal (Figure 1.25). In both cases, no clear significant differences were found between population trends at different levels of adult survival.

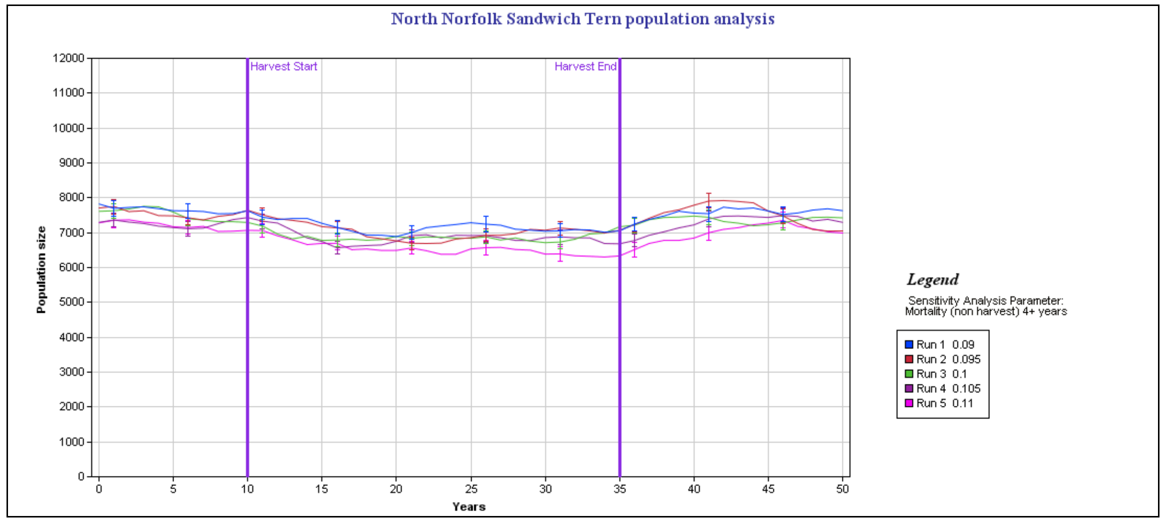

Figure 1.24 Comparison of mean (±1 SE) simulated population trends with collision mortality at Race Bank (3.6 MW) over a range of adult survival rates representing 10% variation around the mean value derived from field studies. Each scenario assumes an avoidance rate of 98%

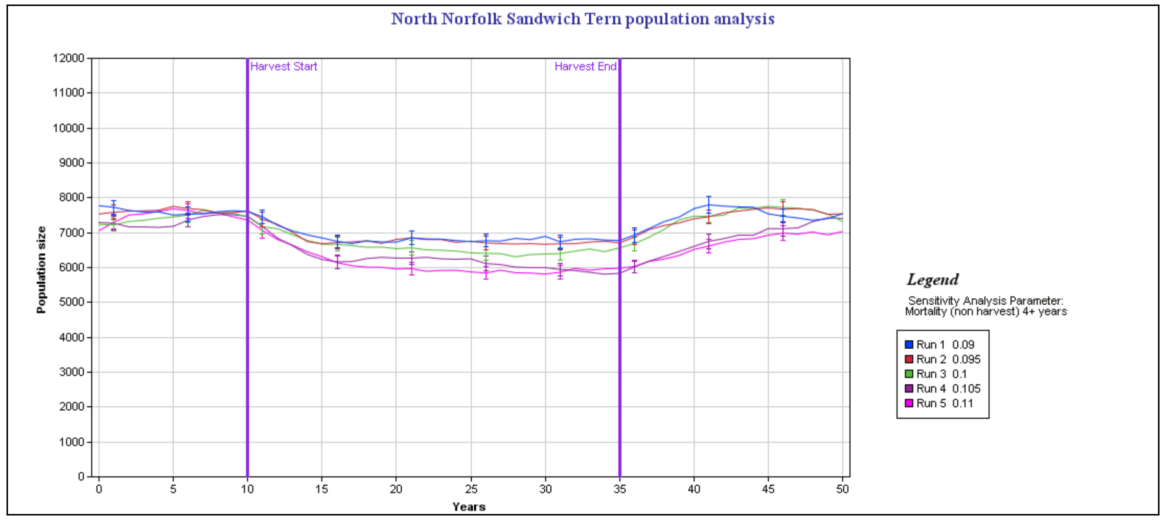

Figure 1.25 Comparison of mean (±1 SE) simulated population trends with collision mortality at Docking Shoal (3.6 MW) over a range of adult survival rates representing 10% variation around the mean value derived from field studies. Each scenario assumes an avoidance rate of 98%

1.1.64 Population sensitivity to variation in adult survival was even lower when analyses were repeated assuming an avoidance rate of 99%. There were no significant differences between population trends across the range of simulated adult survival levels, either from Race Bank (Figure 1.26) or Docking Shoal (Figure 1.27), with all trends being relatively stable across the simulation period.

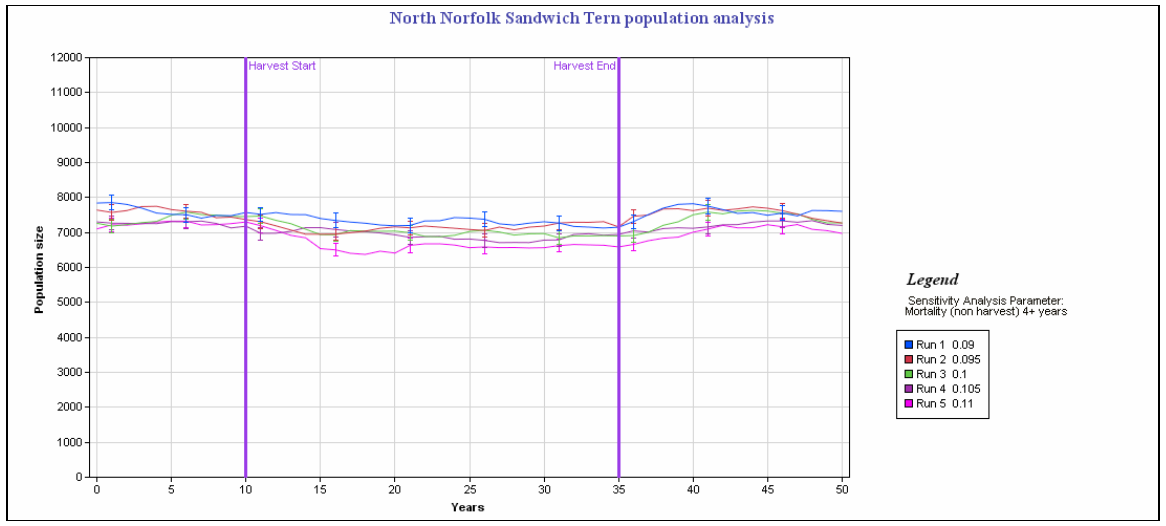

Figure 1.26 Comparison of mean (±1 SE) simulated population trends with collision mortality at Race Bank (3.6 MW) over a range of adult survival rates representing 10% variation around the mean value derived from field studies. Each scenario assumes an avoidance rate of 99%

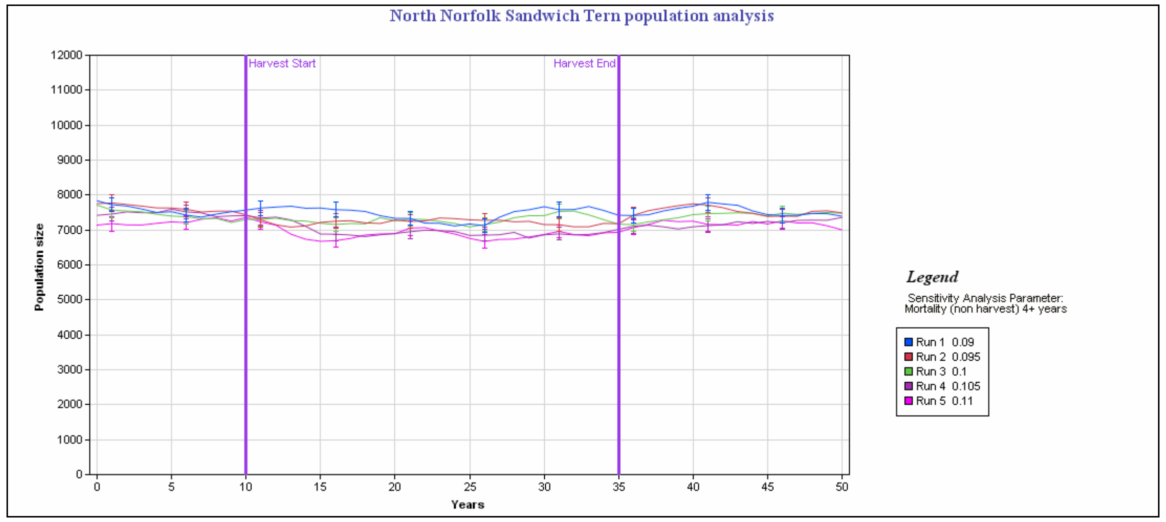

Figure 1.27 Comparison of mean (±1 SE) simulated population trends with collision mortality at Docking Shoal (3.6 MW) over a range of adult survival rates representing 10% variation around the mean value derived from field studies. Each scenario assumes an avoidance rate of 99%

1.1.65 Variation in juvenile survival rate (20% around mean) had a more noticeable effect on population trends, with reductions of both 10% and 20% from average juvenile survival resulting in significant population declines in simulations for both Race Bank (Figure 1.28) and Docking Shoal (Figure 1.29) at avoidance rates of 98%. For Docking Shoal, a decline of 50% occurred over 25 years at a background juvenile survival rate 20% lower than the mean, as opposed to a decline of 24% occurring at the mean rate.

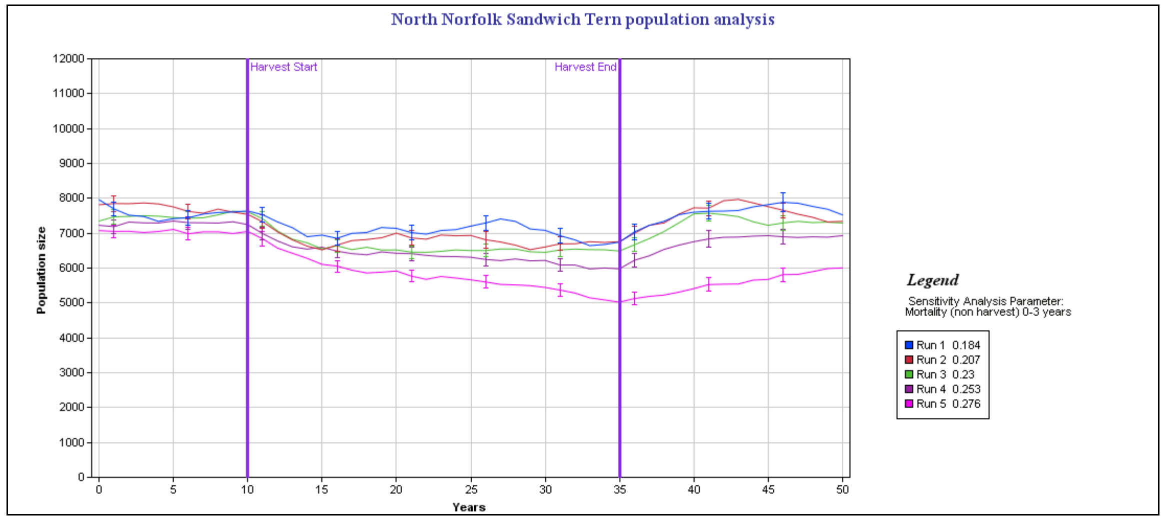

Figure 1.28 Comparison of mean (±1 SE) simulated population trends with collision mortality at Race Bank (3.6 MW) over a range of juvenile survival rates representing 20% variation around the mean derived from field studies. Each scenario assumes an avoidance rate of 98%

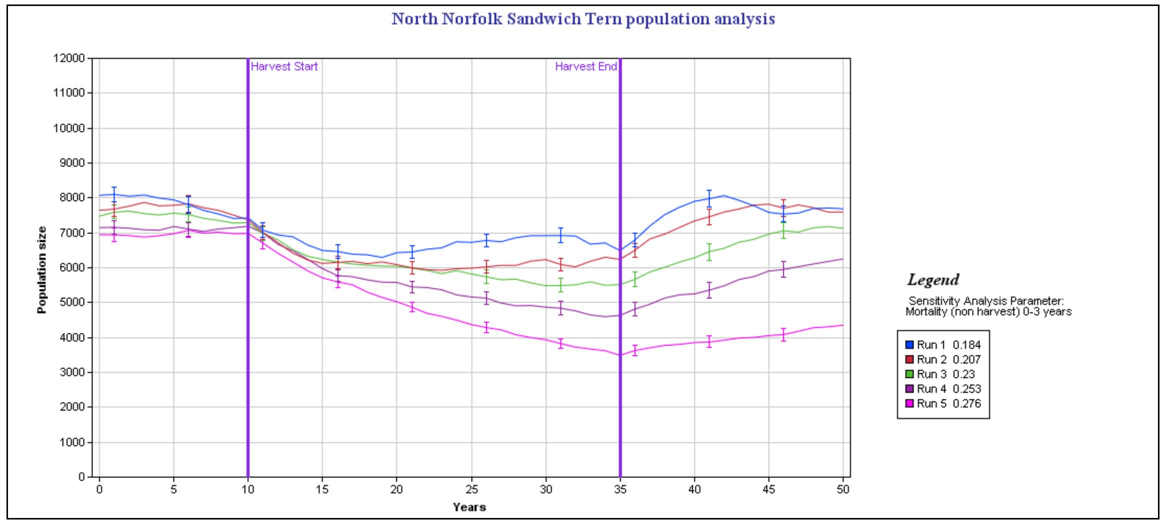

Figure 1.29 Comparison of mean (±1 SE) simulated population trends with collision mortality at Docking Shoal (3.6 MW) over a range of juvenile survival rates representing 20% variation around the mean derived from field studies. Each scenario assumes an avoidance rate of 98%

1.1.66 When these analyses were repeated using a 99% avoidance rate, the influence of variation in juvenile survival on population trends was reduced. For Race Bank, population trends for scenarios up to 20% below mean juvenile survival levels did not differ significantly from those of mean levels (Figure 1.30). For Docking Shoal, the simulated population trend for a 20% reduction in juvenile survival was slightly (but significantly) lower than that predicted at mean levels, although there was still no evidence of long term declines during the simulation period (Figure 1.31).

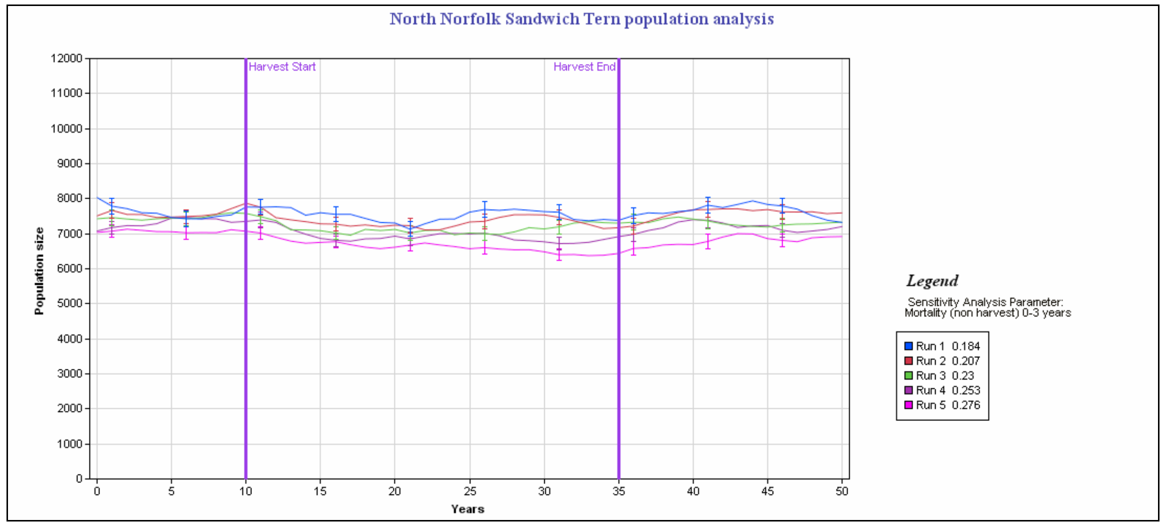

Figure 1.30 Comparison of mean (±1 SE) simulated population trends with collision mortality at Race Bank (3.6 MW) over a range of juvenile survival rates representing 20% variation around the mean derived from field studies. Each scenario assumes an avoidance rate of 99%

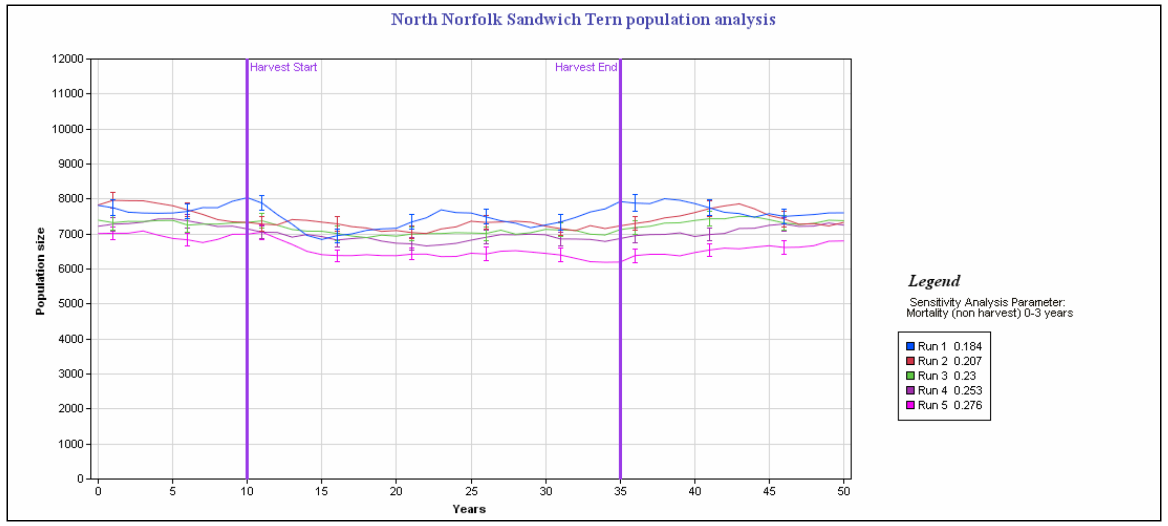

Figure 1.31 Comparison of mean (±1 SE) simulated population trends with collision mortality at Docking Shoal (3.6 MW) over a range of juvenile survival rates representing 20% variation around the mean derived from field studies. Each scenario assumes an avoidance rate of 99%

1.1.67 Simulations were also run to explore the effects of an additive scenario involving simultaneous 10% reductions in all productivity parameters (ie reproductive success, adult survival and juvenile survival). When both Race Bank and Docking Shoal wind farms were operational (3.6 MW layout), this simulation predicts that significant population declines would occur at all avoidance rates (including 99.6%) in relation to the same scenario without wind farm collision (Figure 1.32). Declines at 99.6% avoidance were relatively slight (13% over 25 years), whereas those at lower avoidance rates were considerably more significant, for example an overall decline of 50% over 25 years at a 98% avoidance rate.

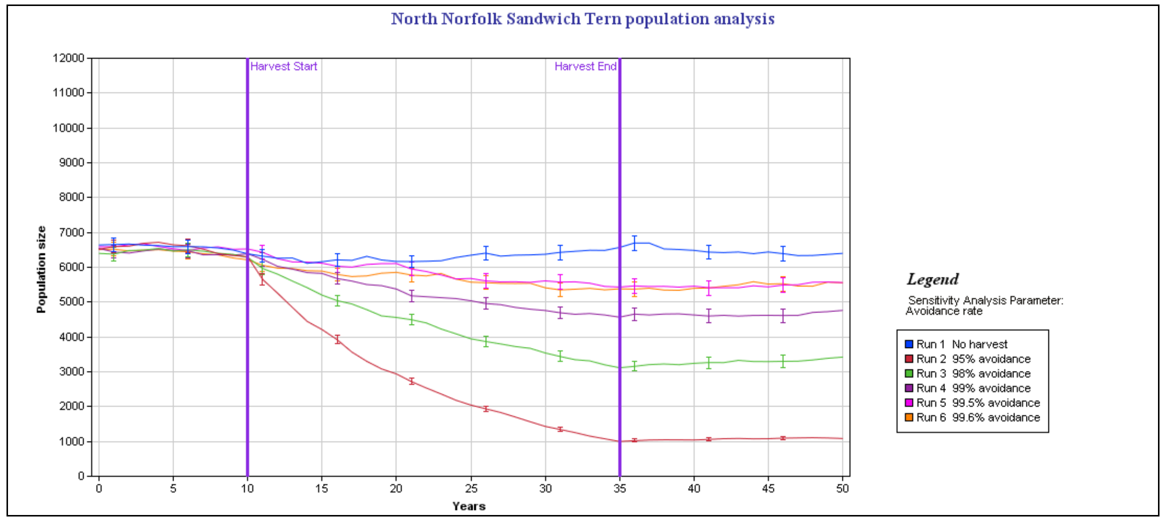

Figure 1.32 Mean (±1SE) simulated population trends if both wind farms are operational (3.6 MW) and all background productivity parameters (ie reproductive success, adult and juvenile survival) are reduced by 10%. Trends for a range of potential avoidance rates are shown

1.1.68 When natural catastrophes are included in the same simulation, the results are very similar (Figure 1.33). This again suggests that natural catastrophes of the magnitude considered here are unlikely to have a significant additive effect on population trends.

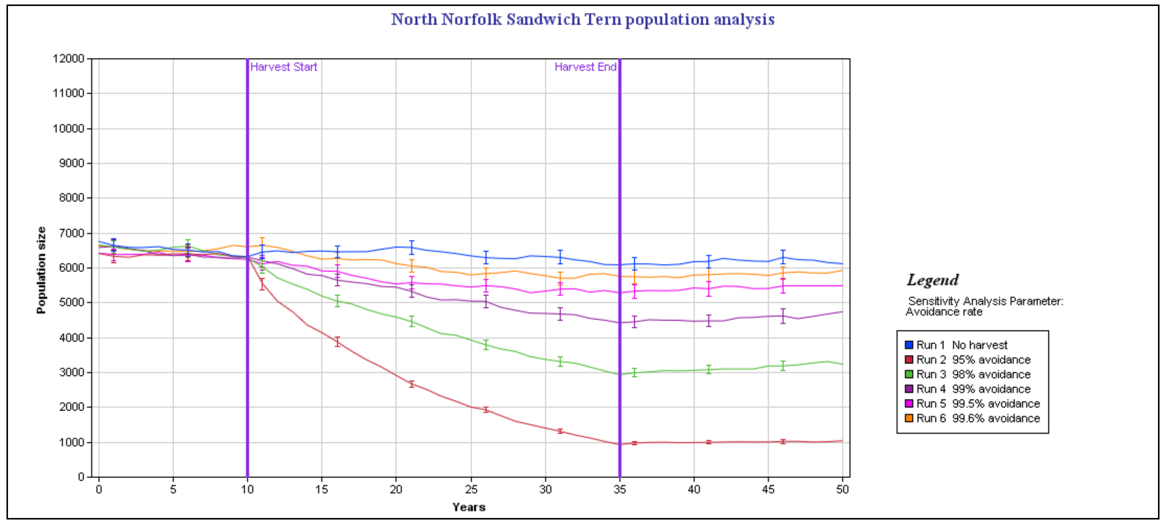

Figure 1.33 Mean (±1 SE) simulated population trends for scenarios in which both wind farms are operational (3.6 MW), all background productivity parameters (ie reproductive success, adult and juvenile survival) are reduced by 10%, and natural catastrophes are included. Trends for a range of potential avoidance rates are shown

Effect of Varying Wind Farm Layout

1.1.69 Simulations were run comparing scenarios ranging from the largest number of smallest turbines (3 MW and 3.6 MW) to fewer, larger turbines (5 MW and 6 MW) for both sites (Figures 1.34, 1.35, 1.36 and 1.37).

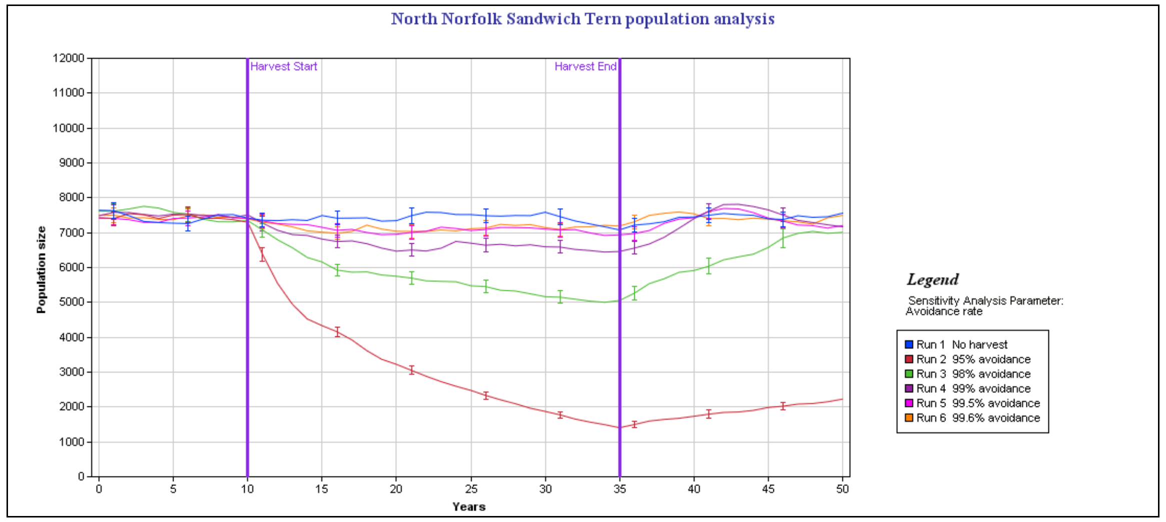

Figure 1.34 Mean (±1SE) simulated population trends for a range of potential avoidance rates, assuming both wind farms are operating with a 3 MW turbine array

1.1.70 At low avoidance rates, the impact of collision on population trends is reduced for the two wind farm layout scenarios involving fewer, larger turbines. However, the magnitude of difference between scenarios was relatively slight. For example, at 95% avoidance, the predicted population declines for the 3 MW and 3.6 MW turbine scenarios were 81% and 76% over 25 years respectively, whilst the equivalent declines for the 5.0 and 6.0MW scenarios were 65% and 62%.

1.1.71 At 99% avoidance, a slight effect of wind farm layout was evident. Under 3 MW and 3.6 MW scenarios, simulated population trends were significantly lower than those of with zero harvest across the majority of the harvest period (Figures 1.34 and 1.35). Under 5 MW and 6 MW scenarios, however, population trends did not differ significantly from those of the zero harvest scenario (Figures 1.36 and 1.37). At avoidance rates of 99.5% or above, there was no detectable effect of wind farm layout, with none of the simulated scenarios differing significantly from zero harvest.

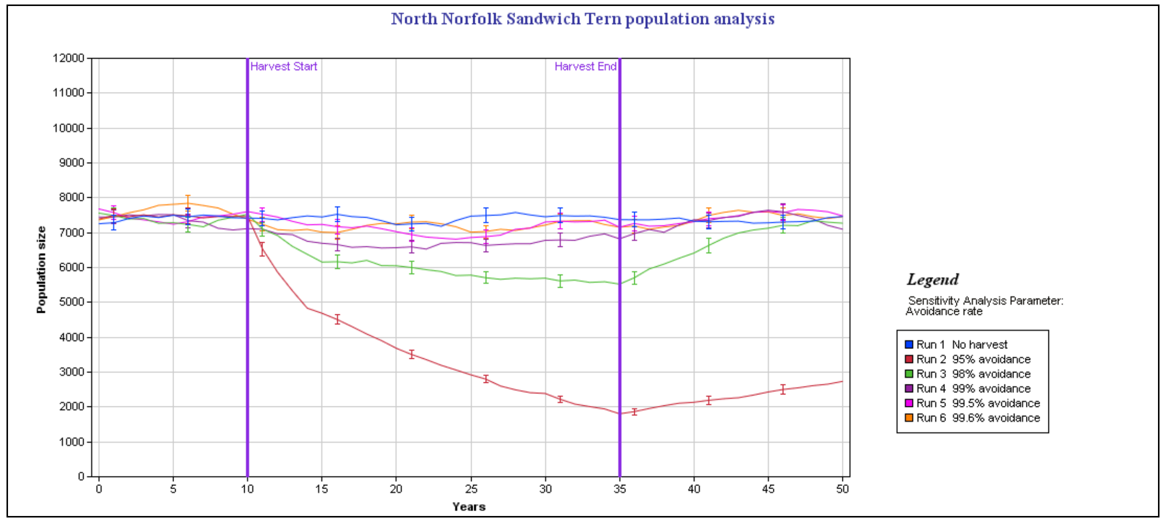

Figure 1.35 Mean (±1 SE) simulated population trends for a range of potential avoidance rates, assuming both wind farms are operating with a 3.6 MW turbine array (note this is also represented in Fig. 10)

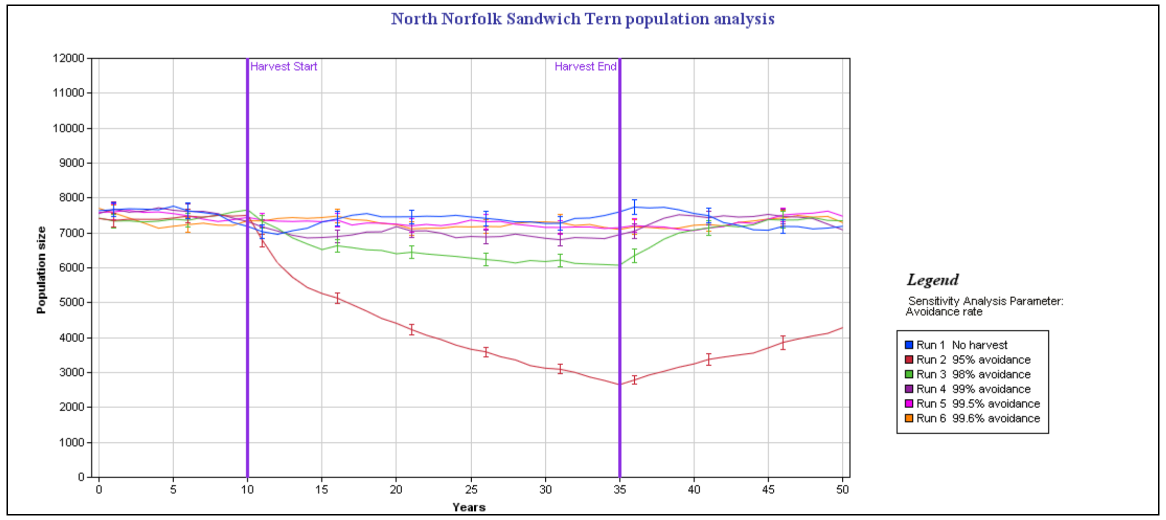

Figure 1.36 Mean (±1 SE) simulated population trends for a range of potential avoidance rates, assuming both wind farms are operating with a 5 MW turbine array

Effects of Collision Mortality Estimates Derived from the Foraging Model

1.1.72 Simulated population trends using the mortality rates predicted by the foraging model are shown in Figure 1.38. Predicted population trends are shown across a range of avoidance rates if both wind farms are operational (with 3.6 MW turbine arrays). The foraging model predicts a lower rate of collision, and hence a less significant population impact than simulations based on data from boat-surveys. Significant population declines only occur at an avoidance rate of 95%, with simulated trends being similar for all avoidance rates above 98%. Even at 95%, the magnitude of population decline is relatively slight, with a 27% decline predicted over 25 years.

1.1.73 Figure 1.39 shows a direct comparison between simulated trends based on the foraging model for each site operating in isolation with 3.6 MW turbine arrays and 95% avoidance rates. In comparison to direct comparisons derived from boat-based data (Figure 1.34), it can be seen that the difference between sites predicted by the foraging model is of a similar magnitude (Figure 1.39). Specifically, declines are predicted to be steeper if Docking Shoal were to operate in isolation than Race Bank. The trend for Race Bank predicted from the foraging model is for the population to remain relatively stable (3% decline over 25 years), unlike the significant decline predicted from boat-based data (decline of 29% in 25 years). For Docking Shoal, the foraging model does still predict an overall population decline, although this is considerably less severe (15% over 25 years) than that derived from boat-based data (56% over 25 years).

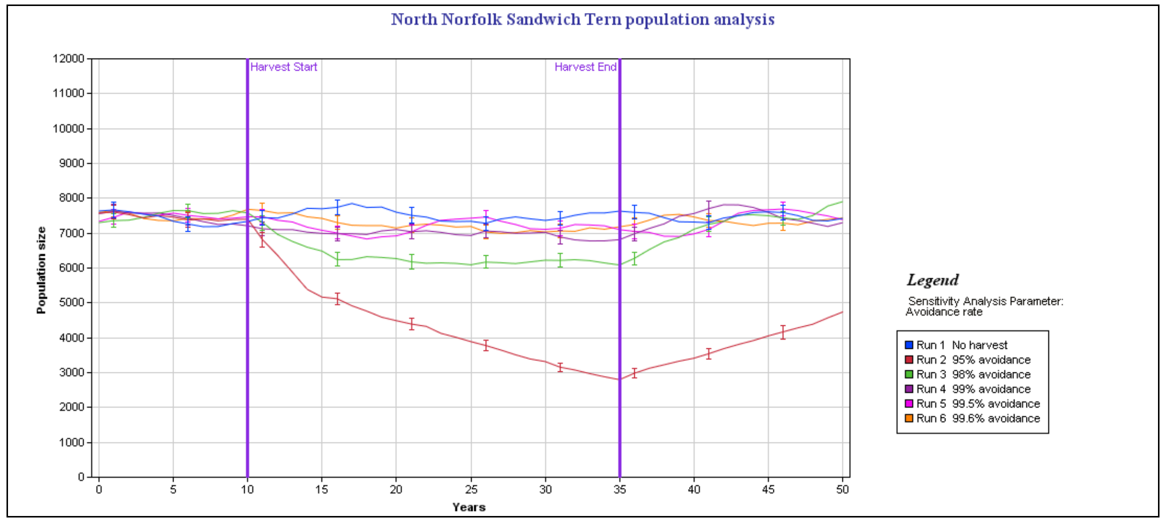

Figure 1.37 Mean (±1 SE) simulated population trends for a range of potential avoidance rates, assuming both wind farms are operating with a 6 MW turbine array

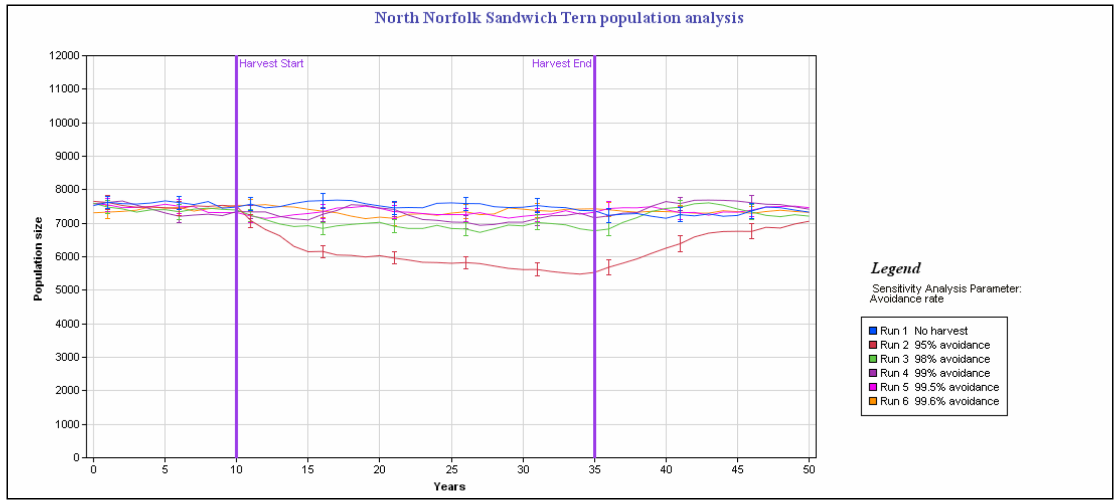

Figure 1.38 Comparison of mean (±1 SE) population trends across a range of potential avoidance rates if both wind farms are operational with 3.6 MW turbine arrays, based on passage rates derived from the foraging model

1.1.74 This stems from the relatively lower predicted rates of use of both sites than was observed from boat-based surveys, but the difference is particularly significant for Race Bank. This discrepancy is likely to relate to the presence of migrating individuals, as well as dispersive or non-breeding birds at both sites, with perhaps more being present at Race Bank due to its location further offshore. Discrepancies between foraging model predictions and observed trends are also likely to arise as a result of natural stochasticity, together with other sources of variation that were not taken into account in the foraging model, for example the many subtle environmental factors that may influence fish distribution.

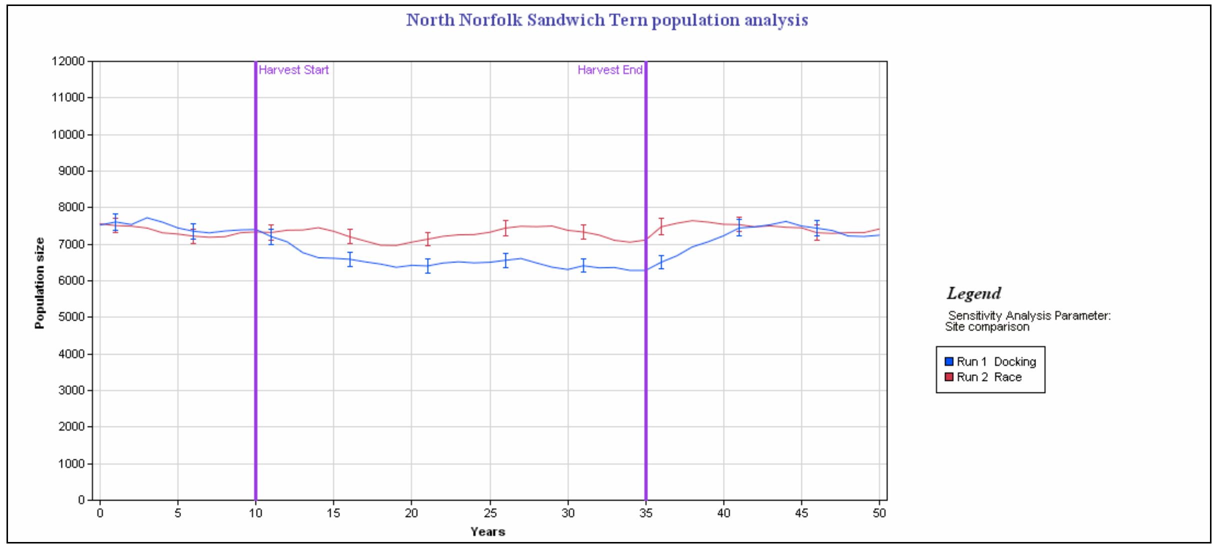

Figure 1.39 Comparison of mean (±1 SE) simulated population trends based on collision estimates derived from passage rates from the Sandwich tern foraging model, assuming each site operates in isolation, with 3.6 MW turbine arrays at a 95% avoidance rate

Discussion

Insights into Population Regulation

1.1.75 Using model parameters derived from the population and from other wider sources, ViaPop appears to produce a realistic simulation of the observed trends in the Sandwich tern population in north Norfolk, which has been relatively stable for the last 30 – 40 years. This not only increases confidence in the validity of the model for making predictions of the impacts of extraneous factors such as mortality from wind farms but also offers some insight into the factors regulating the population.

1.1.76 Modelling without additional wind farm mortality illustrated the relative insensitivity of population trends to reductions in background survival and productivity implying that the present population is currently producing more offspring than is required to maintain population stability. Given that population increases do not appear to be occurring in reality, this might in turn be taken to imply that the north Norfolk population is acting as a source for other populations in north-west European waters perhaps from France to Denmark, with perhaps particular links with populations in the Netherlands and Belgium. It is of note that in the latter, the population at Zeebrugge was only recently established in 1988, growing rapidly to 1,550 pairs in 2000 (Mitchell *et al* 2000) and although fluctuating, reached 4,067 pairs by 2004 (Everaert and Stienen, 2006). Although this is suggestive of a healthy population overall, this does not imply that further mortality within the north Norfolk population would be without impact. Rather, it suggests that negative impacts within north Norfolk may constrain the further development of populations elsewhere. It must be remembered that Sandwich tern is still classified as Depleted on a European scale and thus of unfavourable conservation status (BirdLife International 2004).

1.1.77 Moreover, the simulations suggest that any declines in background productivity levels, if they were to occur on a long-term basis, would significantly compound the effect of wind farm collision mortality. In particular, declines in either reproductive success or juvenile survival resulting from extraneous factors such as prey depletion are likely to impact cumulatively on population trends in combination with wind farm collision.

Effects of Wind Farm Mortality

The Issue of Avoidance Rate

1.1.78 Unsurprisingly, the rate at which terns are assumed to avoid collision with wind turbines has a highly significant influence on simulation outcomes. At avoidance rates of 95 – 99%, significant population scale impacts are predicted if both wind farms were to operate simultaneously, as well as for Docking Shoal in isolation. However, at avoidance rates of 99.5% or above, none of the wind farm scenarios are predicted to cause statistically significant population declines unless background survival or reproductive rates are reduced. Unfortunately, there is a lack of rigorous evidence on which to base estimates of likely avoidance rates for Sandwich terns. This issue is currently a subject of discussion between NE/JNCC and Centrica, and no further comments will be made in this report.

Comparison of the Population Effects of Individual Wind Farms

1.1.79 It is clear that the relative impacts of the two wind farm sites are considerably different, with Docking Shoal having a relatively larger impact in comparison to Race Bank. This is not unexpected given the more inshore position of the former relative to the colonies, particularly Scolt Head, but also including Blakeney Point which tracking data (see Centrica Energy 2009) shows is well within the typical range of foraging birds. In general, birds heading towards Race Bank are likely to pass through Docking Shoal if they originate from Scolt Head.

1.1.80 The operation of Docking Shoal in isolation is predicted to result in a clear reduction in population size at 98% avoidance compared to the baseline condition of no harvesting. The model also predicts a significant but lesser impact at 99% avoidance, showing significant differences in some years but not others. In contrast, for Race Bank operating in isolation, there is no effect of mortality at 99% avoidance and even for 98% avoidance there is no consistent significant difference from scenarios with no harvesting. The model therefore predicts that the operation of Race Bank in isolation is likely to have no detectable impact upon the north Norfolk Sandwich tern population, even with the Sheringham Shoal wind farm also in operation.

1.1.81 If both wind farm sites were to operate simultaneously, the model predicts consistent and significant population declines at avoidance rates of 99% or lower. Reductions in survival and productivity rates are also predicted to have significant additive effects in relation to the population effects of wind farm collision mortality, with reductions in breeding success and juvenile survival being most influential. At avoidance rates of 99% or less, reductions in reproductive success of 20% are predicted to result in significant population declines when either wind farm site operates in isolation. Reductions in adult and juvenile survival, however, are only predicted to have significant additive effects at avoidance rates of 98% unless both sites operate simultaneously, in which case an impact would be detectable at 99% avoidance.

1.1.82 Primarily, the inferences drawn in this report have been based on collision risk estimates calculated using passage rates from boat-based surveys, partly as this has allowed a consistent approach to judging the effects of different sites. However, the specific monitoring of Sandwich terns and the development of a simulation model of foraging distribution also provided a further estimate of passage rates across the sites in from which basis collision risk modeling could be undertaken. As this concentrates entirely on breeding birds from the colonies, this provides a useful measure of the risk to birds when they are breeding, as it does not include passage or non-breeding birds. It is of note that the proportion of birds predicted to forage within or pass through Sheringham Shoal is 7.9% from both colonies combined compared to 15.2% and 3.6% at Docking Shoal and Race Bank respectively (Centrica Energy 2009) suggesting greater use of Sheringham Shoal than that suggested by the respective ESs. This is a direct function of the relative proximity of Sheringham Shoal to Blakeney Point in particular.

1.1.83 As a large number of birds were encountered outside the breeding season at both sites and at Race Bank in particular, the predicted mortality from foraging model simulations is reduced to 42% and 18% of that predicted using boat-based survey data for Docking Shoal and Race Bank respectively (Folkerts 2008). In truth, it is still impossible to determine the origin of birds recorded outside the breeding season as birds with attendant juveniles remain in British waters until they leave for winter quarters mainly in September, although there may also be interchange with birds from the Continent (Wernham *et al* 2002). This means that a proportion of the birds encountered in autumn may still actually be from the north Norfolk population, although even an approximation of that proportion cannot be attempted at present. With the reduction in predicted rates of mortality from foraging model, population trends when both wind farms are operational are not predicted to differ significantly from the zero harvest at 99% avoidance, although the separation with baseline condition is still clear at an avoidance rate of 98%.

The Effects of Wind Farm Layout

1.1.84 There is little relative difference in population-scale effects of the variation in different wind farm layouts. In simulations in which both wind farms operate simultaneously, all wind farm layouts were predicted to cause significant population declines at avoidance rates of 98% or lower. At an avoidance rate of 99%, however, a slight difference between wind farm layouts was evident. Simulations in which both sites operate 3 MW and 3.6 MW turbine arrays showed population trends slightly but significantly lower than those predicted without wind farm mortality. In simulations using 5 MW and 6 MW arrays, however, population trends did not differ significantly from those without wind farm mortality. This suggests that wind farm layouts featuring fewer, larger turbines could have a positive influence on population impacts, although this influence would be very slight and only evident if realised avoidance rates were 99% or higher.

Conclusions

1.1.85 In simple terms it may be concluded that the population viability analysis of the impacts of the Docking Shoal and Race Bank offshore wind farms on the Sandwich tern population indicates:

- the outcome is very sensitive to the avoidance rate chosen. At avoidance rates of 99.5% or higher, significant impacts are not predicted under any wind farm scenario unless background survival or productivity levels alter significantly over time
- in isolation, Docking Shoal is predicted to have a population impact at an avoidance rate of 99% which is of statistical significance in some years but not others, with statistically significant declines predicted at avoidance rates of 98% or lower
- Race Bank is predicted to be unlikely to cause statistically significant population declines in isolation unless avoidance rates are as low as 95%, which is thought to be highly unlikely
- there is potential for statistically significant detrimental effects to the population if both wind farms operate simultaneously, assuming avoidance rates of 99% or lower
- using passage rate estimates from the foraging simulation model, which only relates to breeding birds in the breeding season (May to July inclusive) suggests statistically significant population impacts of both sites combined at an avoidance rate of 98% but not at 99%
- configurations of larger and fewer turbines reduce predicted population impacts, but not to any significant degree. The most likely scenario of 3.6 MW turbines is preferentially discussed
- populations are substantially more vulnerable to catastrophic events, reductions in productivity and elevations in ‘natural’ mortality when wind farms are operational

1.1.86 It is important to note that the significance of effects is judged in statistical and not biological terms. It is not known whether the effect on the integrity of the SPA, which is the test of AA, will be judged in a similar manner or some other threshold of acceptance will be determined (ie a proportion of the population affected). Determination of such a threshold is for NE and its advisors to decide and is beyond the scope of this document.

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Avian collision risk models for wind energy impact assessments

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ABSTRACT

With the increasing global development of wind energy, collision risk models (CRMs) are routinely used to assess the potential impacts of wind turbines on birds. We reviewed and compared the avian collision risk models currently available in the scientific literature, exploring aspects such as the calculation of a collision probability, inclusion of stationary components e.g. the tower, angle of approach and uncertainty. 10 models were cited in the literature and of these, all included a probability of collision of a single bird colliding with a wind turbine during passage through the rotor swept area, and the majority included a measure of the number of birds at risk. 7 out of the 10 models calculated the probability of birds colliding, whilst the remainder used a constant. We identified four approaches to calculate the probability of collision and these were used by others. 6 of the 10 models were deterministic and included the most frequently used models in the UK, with only 4 including variation or uncertainty in some way, the most recent using Bayesian methods. Despite their appeal, CRMs have their limitations and can be 'data hungry' as well as assuming much about bird movement and behaviour. As data become available, these assumptions should be tested to ensure that CRMs are functioning to adequately answer the questions posed by the wind energy sector.

1. Introduction

As wind energy developments increase globally both onshore and offshore (Lewis and Wiser, 2007; Snyder and Kaiser, 2009; Bilgili et al., 2011; Wang et al., 2012), the potential associated environmental impacts are receiving considerable attention, particularly avian impacts. Typically, wind energy developments require an environmental impact assessment to quantify the potential risk to the environment. The potential impacts of wind farms 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 avoidance responses of birds to turbines (Fox et al., 2006; Langston, 2013). Avian collision has received much attention as it is considered a very real threat to bird populations (Johnson et al., 2002; Krijgsveld et al., 2009) and a variety of methods have been developed to aid the assessment of the risk of collision. The methods can be categorised as those that measure and assess collisions empirically including direct and remote observations of bird flights in the development area (pre- and post-construction of the wind turbines) to assess flight behaviour, habitat use and flux of birds (Desholm and Kahlert, 2005; Desholm et al., 2006; Douglas et al., 2012) and corpse searches to document actual collisions (Winkelman, 1992; Huso and Dalthorp, 2014), and those which are more theoretical such as collision risk models which predict likely collisions (Holmstrom et al., 2011; Eichhorn et al., 2012; Smales et al., 2013). In addition to estimating collisions between birds and wind turbines, collision risk models (CRMs) are used in a range of other situations including marine mammals and marine renewable energy devices i.e. tidal stream turbines (Wilson et al., 2006), fish and turbines (Hammar and Ehnberg, 2013) and shipping collisions with moving and stationary objects (Montewka et al., 2010).

At their core, most avian collision risk models include a calculation of the probability of a collision occurring (assuming no evasive action or behaviour) and a measure of the number of birds within a risk window in order to estimate the likely number of collision events. 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. This therefore relies upon information on both bird and wind turbine characteristics including but not limited to bird morphometrics and flight speed, turbine rotor speed and turbine size. In addition to the probability of collision, an understanding of bird avoidance behaviour is required if realistic estimates of collision events are to be predicted. In the UK, the most frequently used avian collision risk model is commonly known as 'the Band model' (Band et al., 2007). Since its original development, it has undergone several iterations with the most recent associated with the Strategic Ornithological Support Services (SOSS) (Band, 2012a, 2012b). However, it is not the only collision risk model available to predict potential collisions of birds with wind turbines, and others are used outside of the UK and vary in their approach to assessing avian collision risk.

The aim of this review therefore is to discuss the range of avian collision risk models in order to raise awareness of those available, their

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strengths and limitations. In addition we qualitatively compare models, and highlight when it may be appropriate to use different models, as well as discussing the interpretation of results. Finally, we also suggest where future efforts should be focussed to advance collision risk modelling.

2. The collision risk models

The peer-reviewed scientific literature and the grey literature were extensively reviewed for references to avian collision models. Using Web of Science, Google and Google Scholar we searched for relevant peer-reviewed papers, reports, conference proceedings and book chapters relating to wind farms and collision risk models. The search terms used were "collision risk model or CRM or collision model" refined by "bird or avian or ornithology or ornithological" and "wind farm or wind turbine or windmill". We identified 10 distinct collision risk models referring to birds and wind turbines, the earliest dating back to 1996 (Tucker, 1996a). We defined the Band model and its various options and iterations as one model, though we will discuss the different versions below. We are aware other models are available, but following our literature review we were unable to find any documentation for these models, and were unable to contact the model developers. In this section we present brief descriptions of the collision risk models available, ordered chronologically in an effort to show the development and history of this field of research. We do not provide the fine mechanistic detail required to reproduce any single model but rather an overview of the methods available. The original intention of the project was to quantitatively compare models, but this was not possible as insufficient details were provided to do so. Although commercial confidentially is often given as the reason for a lack of detail regarding collision risk models, increased transparency would increase confidence in the final model outputs.

2.1. *Tucker (1996a, 1996b)*

Tucker (1996a) was the first to publish a complete analysis of bird–rotor collisions and went on to show how rotors could be designed so fewer birds collide for an equivalent energy generation (Tucker, 1996b). "The model analyses the motions and dimensions of both birds and propeller-type rotor blades, and predicts the probability of a collision when the bird flies through the area swept by the blades." (Tucker, 1996a). However, it does not estimate a likely number of collisions as a measure of bird density or flux through the turbine is not included. The probability of collision is calculated as a ratio of the time taken for a bird to move through the rotor swept area compared to the time taken for the turbine blades to complete a single revolution. In the model the theoretical blades are either one, or three dimensional consisting of length, chord and twist but no thickness. Collision with the static turbine tower is not considered in the calculations. The bird moves on fixed wings i.e. gliding not flapping, and is two-dimensional and rectangular with wingspan being greater than body length (Fig. 1b). It is therefore the corners of the rectangle which collide with either the leading or trailing edges the blades. The bird always moves perpendicular or parallel to the turbine rotor but flight can be parallel or oblique to the wind direction and the model can accommodate upwind or downwind flight. Avoidance behaviour of the bird is mostly not included in this model though it assumes that there is an inner radius at the turbine hub where birds will always avoid collision with the blades as it is a slow moving object.

2.2. *Band (2012a, 2012b)*

The approach was originally developed for onshore wind turbines and promoted as guidance by Scottish Natural Heritage (Scottish Natural Heritage, 2000). It has been further developed by Band et al. (2007) and more recently for application in the offshore environment by Band (2012a, 2012b). Similar to Tucker (1996a) this model is based on the probability of a turbine blade occupying the same space as a bird during the time it takes the bird to pass through the rotor swept volume of the turbine. The probability of collision relies on information about the bird (wing span, body length, flight speed, flight height, nocturnal flight activity) and the turbine (blade width, blade length, blade pitch, rotor speed, hub height, operational time). The bird is assumed to be cruciform i.e. cross-shaped (Fig. 1c), though this simplification may underestimate collision risk and the turbine blade is assumed to have a width (chord) and a pitch angle but no thickness. The model only considers flights that are parallel to the wind i.e. perpendicular to rotation of turbine and assumes that the effects of approaching the turbine at oblique angles will cancel each other out though this may underestimate collision risk (Band, 2012b). It also only considers the moving rotor excluding the stationary elements such as the tower.

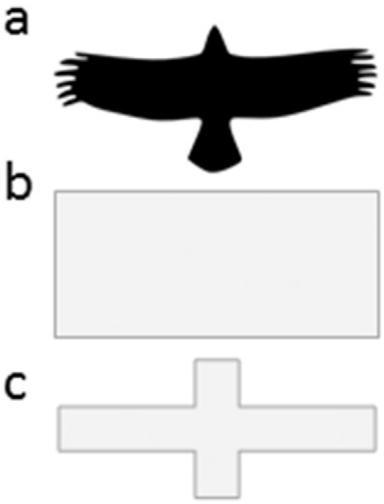

Fig. 1. Modelled representation of bird (a) as either rectangular (b) or cruciform (c).

2.2.1. *'Basic' Band model*

The approach of the original model (Band et al., 2007) had two stages for estimating the number of collisions per annum which included calculating: i) the number of birds flying through the rotor and ii) the probability of collision from a single transit of a rotor. The probability of collision is calculated at fixed intervals along the rotor blade and then averaged over the rotor swept area. The more recent offshore iteration of the model (Band, 2012b) includes a method to use boat-based survey data i.e. densities, rather than vantage point data to calculate the number of birds flying through the rotor. This modification is necessary due to the different data collection techniques applied in the onshore and offshore environments. The most recent version also includes a measure of avoidance behaviour, allowing for a proportion of birds to avoid collision.

2.2.2. *'Extended' Band model*

The extended model is built on the basic model. The basic model assumes a uniform distribution of birds across the rotor swept area of the turbine. However, it was recognised that the distribution of birds, as well as the width of the turbine, all vary with height within the rotor swept area, thus affecting the collision risk. It is not possible to consider each of these individually due to covariance, however it is possible to use flight height curves (Johnston et al., 2014) to calculate the probability of a bird flying at a particular height within the turbine rotor sweep and colliding with a turbine blade. These individual probabilities are then integrated to gain the collision integral. Although the extended Band model is considered a more realistic model than the basic model, it is potentially more sensitive to uncertainty, particularly in relation to flight height estimates (Cook et al., 2014).

2.3. McAdam (2005)

This Monte Carlo model is based on the original Band model (Scottish Natural Heritage, 2000) but includes stochastic modifications to account for variation in flight height and the effects of wind. For the height-sensitive variant of the model, it calculates the probability of a bird being struck, given that it passed through the plane of the turbine at a given height and at a distance less than the rotor length from the centre. The model also considered the effect of variation in wind on collision probability by varying bird speed and direction through the turbine. The model includes oblique angles of approach but does not take into account the variation in bird orientation relative to the turbine.

2.4. Smales et al. (2013)

The collision risk model developed by Biosis Propriety Limited has been widely used to assess wind-energy developments in Australia since 2002 (Smales et al., 2013). The model provides a predicted number of collisions between turbines and a local or migrating population of birds. It uses a deterministic approach but has the potential to be modified to accommodate Monte Carlo simulation. Unlike other models, it includes collision with static components of the wind turbine such as the tower. It also does not assume that birds always approach the turbine perpendicular to the blades or from a specific angle but rather flights can approach turbines from any direction meaning all dimensions of the turbine contribute to the area with which a flying bird might collide and the model uses a mean presented area. The model also estimates collision risk as the sum of the average number of turbines encountered per flight within a scattered wind turbine array, rather than for all turbines in an array. The average number of turbines likely to be encountered is calculated using a topological, non-affine mapping technique (Smales et al., 2013).

2.5. Bolker et al. (2014)

This collision risk model is based on the geometry of the wind farm and was developed to determine the average number of turbines encountered if birds move through a wind farm, in particular the Cape Wind project in Nantucket Sound off the coast of Massachusetts. It makes the assumption, similar to other models such as the Band model, that birds fly in straight lines, and that the probability of collision is fixed and known and that it incorporates any avoidance behaviour. It also only includes collisions with the rotors and does not include any other related infrastructure.

2.6. Desholm (2006)

The author presents a stochastic simulation model developed to estimate the number of bird fatalities at a wind farm, using the case study of the Nysted offshore wind farm in Denmark. The assessment includes variability in the input parameters of the model. The model includes information on bird migration volume, proportion of birds entering the wind farm, proportion of birds within the horizontal range of the rotor-blades, proportion of birds in the vertical range of the rotor-blades, proportion of birds trying to cross the area swept by the rotor without showing avoidance, the mean number of turbine rows passed, and also the probability of passing safely through the rotor by chance. The probability of passing safely through the rotor by chance was assumed to be fixed and taken from Tucker (1996a) rather than calculated directly. Information on wind direction was also included and influenced both the orientation of the blades (and therefore the proportion of birds in horizontal reach of the blades) and the probability of passing safely through the swept area, as it is known that the probability of collision differs upwind and downwind.

2.7. Podolsky (2008)

This model is presented in a patent document (Podolsky, 2008) and describes the bird, the turbine and the wind farm. The model follows a similar method to Band et al. (2007) and Tucker (1996a) calculating the distance travelled across the rotor disc and thus the time required which is then compared to rotor speed and the time required for a single revolution of the blades. The bird is represented as a cross with length and wingspan and uses the largest linear dimension of the bird so that the most conservative results are produced. The model considers oblique angles of approach, not only those parallel to the wind and includes a proportion of birds which avoid collision. The probability of collision is calculated for a bird colliding with a single rotor and subsequently the model calculates the probability of collision for a given row of wind turbines and for multiple rows.

2.8. Holmstrom et al. (2011)

The Hamer model presented by Holmstrom et al. (2011) is based on that of Tucker (1996a, 1996b). This model however includes oblique angles of approach flight to the wind turbine, which the authors suggests, "provides important improvements over existing models since birds' flight paths are not always dependent on wind direction" (Holmstrom et al., 2011). As with Tucker (1996a) the model only considers the calculation of collision probability for a single transit and a single turbine rotor. It only estimates collision risk for a single bird passing through the rotor plane and avoidance behaviour is also not considered within the model (apart from close to the hub) rather it should be incorporated after calculating the mathematical risk of collision.

2.9. Eichhorn et al. (2012)

The purpose of the simulation model produced by Eichhorn et al. (2012) was to determine the annual mortality of a central-place forager, the red kite *Milvus milvus*, as a function of the distance to the bird's nest and a wind turbine. The model assesses collision risk as a function of distance to turbine, nests and other parameters and combines an agent-based spatial model with a collision risk model. The agent-based model is based on movement processes and two events (collision and prey capture) and movements of individuals are based on decision rules according to habitat quality. "Collisions can occur when an individual occupies a habitat cell where a wind turbine is located, flies at the height of the rotor blades and moves through the part of the cell affected by the rotors" (Eichhorn et al., 2012). Should a bird co-occur in space with a wind turbine, the probability of collision is calculated using stage 2 of the Band model (Band et al., 2007) i.e. probability of collision from a single rotor transit. This is an example of where a component of another collision risk model has been used rather than developing another method.

2.10. U.S. Fish and Wildlife Service (2013)

The U.S. Fish and Wildlife Service have developed a collision risk model for predicting eagle fatalities that uses a Bayesian estimation framework (U.S. Fish and Wildlife Service, 2013; New et al., 2015) and is based on eagle exposure to wind turbine hazards. In this model, the total annual eagle fatalities is the product of the rate of eagle exposure to turbines, the probability that eagle exposure will result in a collision with a turbine, and an expansion factor that scales the fatality rate to a predicted number of annual fatalities. The exposure rate is the expected number of exposure events i.e. eagles present (per hour per km^2) in the area of interest. The probability of collision given exposure is based on Whitfield (2009) but any suitable data and studies could theoretically be used. Finally the expansion factor is based on information on the number of turbines and the hazardous area surrounding a turbine. The model does not include detail on the mechanisms of collision but uses the Bayesian method which allows for adaptive management and the updating of information such as actual collision events, over time, as it becomes available. This model is therefore more suitable for onshore wind farm developments where corpse searches are possible, unlike at offshore facilities.

3. Model comparisons

From the model descriptions above it can be seen that there has been a development of collision risk models over time, each with its own purpose, though some clearly influenced by others. Subsequently there is an array of models to choose from that include different components and use different approaches (Table 1). However, of the models included in this review, it is possible to say they comprise two main elements: i) the number of birds exposed to turbines and therefore at risk of collision; and ii) the probability of an individual bird colliding. Many of the differences between models relate to the former rather than the latter. Here we shall compare some of these aspects to highlight similarities and differences which may influence the applicability and suitability of a given model to a given problem.

3.1. Probability of collision

All of the models presented here incorporate a probability of collision for a single transit of a bird through a turbine rotor i.e. an interaction with a wind turbine, however there is variation in how this is estimated or included, and in the complexity of the calculations. Some of the models, for example Tucker (1996a), calculate a probability of collision that varies according to bird and turbine parameters i.e. a mechanistic model. Other, more complex, models have been developed based on Tucker (1996a) and Band (Scottish Natural Heritage, 2000) but have been adapted for a specific application. For example, Eichhorn et al. (2012) used the probability of collision component from the Band model in their individual based model of red kite foraging. In addition and by contrast, several authors (Desholm, 2006; Bolker et al., 2014) have chosen to use a constant probability of collision rather than calculate a variable collision probability. The use of a single constant probability removes the theoretical calculation of the probability of collision as presented by Tucker (1996a) for example, and the associated uncertainty of factors such as avoidance. As an alternative to this the U.S. Fish and Wildlife Service (2013) have developed a Bayesian method which does not calculate the probability of collision mechanistically and is therefore more similar to those that use a constant but by contrast, their method allows for the estimate of probability of collision to be amended and updated when information becomes available.

3.2. Bird shape

For those models that follow a mechanistic approach to including and calculating a probability of collision, they must include a simplified representation of a flying bird. Tucker (1996a) uses a rectangular representation of a bird whereas Band et al. (2007) uses a cruciform representation (Fig. 1). It is thought that both slightly underestimate the actual collision risk as a bird is larger than these representation, however the latter gives slightly lower estimates (Holmstrom et al., 2011). It is unlikely that the choice of bird shape will dramatically affect the results more than other factors.

3.3. Angle of approach

Models differ in the assumptions made about bird movement and flight, and in particular the angle of approach to the turbine. However, the collision risk is dependent on the angle of approach. Flight paths can be assumed to be perpendicular to the axis of rotation of the turbine or they can be oblique. Birds on oblique angles of approach, rather than perpendicular, are presented with a reduced cross-sectional area of risk.

Table 1
Summary of avian collision risk models.

Model name and reference	Base model	Includes avoidance behaviour	No. of turbines	Tower included	Wind speed/direction included	Oblique angles of approach	Individual or population	Onshore or offshore example	Stochastic	Model output
Band (Band, 2012a, 2012b)	–	Y	Multiple	N	N	N	Population	Offshore	N	Number of birds colliding
Tucker (Tucker, 1996a, 1996b)	–	N	Single	N	N	N	Individual	–	N	Probability of collision
Biosis (Smales et al., 2013)	–	Y	Multiple	Y	N	Y	Population	Onshore	N	Number of birds colliding
Podolsky (Podolsky, 2008)	–	Y	Multiple	Y	N	Y	Individual	Onshore	N	Probability of collision
McAdam (McAdam, 2005)	Band	N	Single	N	Speed & direction	Y	Individual	Offshore	Y	Probability of collision
Desholm (Desholm, 2006)	–	Y	Multiple	N	Direction	N	Population	Offshore	Y	Number of birds colliding
Eichhorn (Eichhorn et al., 2012)	Band	Y	Single	N	N	N	Individual	Onshore	Y	Mortality rate
Hamer (Holmstrom et al., 2011)	Tucker	N	Single	N	Speed & direction	Y	Individual	–	N	Probability of collision
Bolker (Bolker et al., 2014)	–	N	Multiple	N	N	Y	Individual	Onshore	N	Probability of collision
USFWS (U.S. Fish and Wildlife Service, 2013)	–	Y	Multiple	Not specified	N	N	Population	Onshore	Y	Number of birds colliding

However, should the bird pass through the area of risk, the time required to clear the rotor blades is extended, therefore increasing the risk of collision. Band (2012a, 2012b) assumes all flights to be perpendicular, and that oblique angles of approach will cancel out. This may not always be the case in all situations, for example it is possible that for a turbine placed near a seabird breeding colony, the birds may follow different flight paths in and out of the colony, and they might not be perpendicular to the turbines. Therefore the birds would only be approaching the turbines from a limited number of angles, and thus these would not cancel out. In addition, this does not consider the varying speed of blade approach relative to the bird movement across the rotor. Tucker (1996a) also only considers flights parallel to the wind direction i.e. perpendicular to rotor axis, for the calculations of collision risk in 3 dimensions. However, the Hamer model (Holmstrom et al., 2011) explored the importance to oblique angles of approach, building on the original analyses of Tucker (1996a). Using a case study of raptor migration, Holmstrom et al. (2011) demonstrate that the angle of approach has a significant effect on the probability of collision and thus the estimates of mortality, with estimated collision probabilities as much as 31% higher than those estimated for downwind flight. It is therefore important to ascertain which model is more appropriate for the case study and that this may be different for long distance migration than for breeding seabirds foraging from a colony.

3.4. Collisions with stationary components

Another difference between models is that the majority include only the moving rotor in the collision estimate, because it is assumed that birds will avoid non- or slow-moving parts however Smales et al. (2013) include the stationary turbine tower as well. This therefore means that the collision risk estimated using the method described in Smales et al. (2013) may be greater than estimated using other methods. There is evidence of collisions of birds with fixed structures such as communications towers and fences, as well as power lines (Bevanger, 1995; Baines and Andrew, 2003; Martin, 2012; Loss et al., 2014). It is possible that some species, when flying and foraging in open airspace e.g. offshore, plains or prairies, and if turning their head to look downwards, will have little visual coverage of what lies ahead so making them particularly vulnerable to collisions with obstacles which are built into these otherwise predictably open airspaces (Martin, 2012). Therefore for species known to be at risk of collision with these fixed structures it may be important to include them in the collision estimates as well as moving blades. Subsequently, in some circumstances it may result in species, for example tetraonids such as black grouse in the terrestrial environment or auks in the marine, being included in collision risk assessments which do not fly at heights which would put them at risk with collision with turbine blades (Johnston et al., 2014). It should be recognised that estimates are not comparable if they do not consider similar components of the turbines.

3.5. Onshore or offshore

The methods also differ in the setting for which they are developed and the questions that they were developed to address. The U.S. Fish and Wildlife Service model is a good example of this. The Bayesian method allows for estimates to be updated when information is available. In this case, carcasses provide information on the actual number of collisions. The model was developed for eagles at onshore wind farms where carcass searches are possible, however this method would be less applicable for the offshore environment, although technologies such as Thermal Animal Detection Systems (TADS), as demonstrated offshore by Desholm (2006), may make this possible. The models also vary in perspective, with the majority being turbine-based and focussing on the number of birds encountering and colliding with a turbine (Band et al., 2007; Smales et al., 2013). However, Eichhorn et al. (2012) present an agent-based model which considers collisions from the perspective of the individual and estimates the number of turbines a bird encounters.

3.6. Uncertainty

Uncertainty is a topic addressed to varying degrees within the different collision risk models. The majority of models (Tucker, 1996a; Podolsky, 2008; Holmstrom et al., 2011; Smales et al., 2013) are deterministic and do not consider data or parameter uncertainty within the model (Masden et al., 2015). The Band model is also deterministic but in the latest iteration of the model, the guidance provides a method to express the uncertainty associated with a collision estimate *post hoc* (Band, 2012b). Smales et al. (2013) is similarly deterministic however the authors suggest that it would be possible to use Monte Carlo methods to introduce stochasticity into the model. By contrast, McAdam (2005) used a Monte Carlo model to consider joint distributions of wind speed and direction and distributions of flight height to produce collision risk estimates with associated measures of uncertainty and the U.S. Fish and Wildlife Service model uses Bayesian methods which allows for the consideration of uncertainty. Including a measure of uncertainty in collision estimates moves CRM towards a risk-based framework, providing information not only on the magnitude of an event but also on the probability of occurrence.

Uncertainty may also be considered in terms of information available about the model structure i.e. model uncertainty, not only the data and model output (Masden et al., 2015). Within this review we provide a qualitative comparison and have adopted a descriptive approach because details about the model structure were often incomplete. Therefore another difference between the models is the quantity and quality of detail provided and whether the model and a detailed description have been subjected to peer-review. This does not signify or relate to the quality of the model itself, but to the information available. Models such as Tucker (1996a); Holmstrom et al. (2011) and Eichhorn et al. (2012) have been presented and published in the scientific peer-reviewed literature and Band et al. (2007) has been subjected to scrutiny (Chamberlain et al., 2006) but it is not always transparent as to the degree of scrutiny and peer-review that the models may have been exposed to. Economically important decisions are being made on the basis of the outputs from collision risk modelling, so it is important that tools used to reach these conclusions are subject to scrutiny. Transparency is vital as wind farm applications may be rejected on the basis of the predicted collision rate, for example Docking Shoal was turned down due to the collision risk estimate for Sandwich tern (Department of Energy and Climate Change, 2012). The method of estimation should therefore be clear and available to be scrutinised.

4. Limitations and assumptions

Although CRMs are a useful tool to estimate likely collisions of birds with wind turbines, and provide information on the potential environmental impacts of wind farm developments, they also have limitations and it is important that these are recognised to ensure that the data outputs are used appropriately.

Potentially the greatest limitation of CRMs is that they generally assume much about bird behaviour. For example, the majority of models assume a linear relationship between bird abundance and collision risk which may not be true for all situations (De Lucas et al., 2008); there may be interactions between topography, species-specific behaviour, turbine layout, and wind (Barrios and Rodriguez, 2004; Smallwood et al., 2009; Schaub, 2012; De Lucas et al., 2012). In addition many of the models include avoidance behaviour in the form of an avoidance rate which assumes that a certain proportion of those birds on a collision path, will take avoiding action before a collision occurs. Most models assume that avoidance behaviour is constant across all individuals within a species and this is unlikely. However, there is very limited data available on avoidance rates and estimating variability between species is difficult, least of all within a species (Cook et al., 2014).

Also some models such as Band (2012a, 2012b) assume a constant flux of birds through a wind farm. Such models assume that there are X birds within the wind farm at any given time, each of which takes Y seconds to fly through the rotor. From these data it is possible to estimate the total number of birds passing through a wind farm in a day or year based on the speed of the birds (typically mean flight speed). However, if the birds are not flying at the reported speed, or not commuting through the wind farm but instead moving tortuously within the area (Patrick et al., 2014), it is possible to over-estimate the number of birds flying through the wind farm in a given time period, and consequently elevate the total number of collisions; inflated estimates of collision rates can be of serious consequence and cost for a wind farm developer. This is likely the case also for models that use data from vantage point surveys, for example the model developed by the U.S. Fish and Wildlife Service (2013) uses data on the number of expected exposure events. It may be difficult to distinguish whether two observations of a bird are the same individual or different. This distinction is important because each bird can only collide once (if we assume collision equates to mortality) and if the number of birds using the area is overestimated, it is possible to overestimate the total number of collisions. Eichhorn et al. (2012) circumvent this limitation using an agent-based model to describe movements of individual birds through a landscape and applying a collision risk to each interaction of an individual with a wind turbine, though such a method can be computationally intensive.

Another limitation of collision risk models is that they are frequently ‘data hungry’ in situations where data availability is often limited. For example, mechanistic models such as Band (2012a, 2012b) have many input parameters relating both to the birds (flight speed, morphometrics, etc.) and the turbines (rotor speed, blade width, etc.). In relation to birds, there is still much to be learned about behaviour and therefore our knowledge of aspects required within the models, such as flight speed, is limited. This is improving with the development of biologging technologies such as miniaturised GPS tags but often data are limited to the breeding season. In addition for offshore wind, many of the turbines suggested for offshore projects are still under development and therefore only design envelopes can be provided, i.e. a range of values for any given parameter. The approach developed by U.S. Fish and Wildlife Service (2013) removes some of these data requirements by using a Bayesian framework, however it relies on the ability to collect data on actual collisions to validate the model, and thus currently limits the approach to onshore sites. Therefore a Bayesian method is unlikely to be suitable for offshore sites without development of methods to collate data on collisions and therefore a mechanistic theoretical modelling approach will be required, else a constant collision risk be applied.

Not only a lack of data on model inputs but also the opportunities for model validation or lack thereof, is a limitation of collision risk models. Collision risk models are rarely validated, but where they have been, predictions from EIA often show only a weak relationship with observed effects and predictor variables. Ferrer et al. (2012) found no relationship between variables predicting risk from EIAs and actual mortality, and only a weak relationship between mortality and the numbers of the study species crossing the area. Similarly, De Lucas et al. (2008) found a weak relationship between abundance and recorded collisions and in addition Everaert (2014) found larger gulls were more likely to collide than smaller birds. More specifically, opportunities for model validation are particularly limited for offshore wind farms because bird mortality events are more difficult to document in the offshore environment as corpses do not remain in the area. In the terrestrial environment i.e. onshore, the U.S. Fish and Wildlife Service (2013) model uses a Bayesian framework within which information can be updated. Although this is not validation, the input of empirical mortality data from corpse searches, allows for the collision estimate to be refined. However, until the collection of mortality data from offshore developments becomes standard practice, Bayesian collision risk models are unlikely to be a feasible option.

Collision risk models are a tool, but only a tool, to aid in the assessment of impacts and management of wind farm developments, and it is important to remember that “Essentially, all models are wrong, but some are useful” (Box and Draper, 1987). Due to the limitations of collision risk models discussed above and the assumptions made within each, at present the model estimates provide a means of comparison between different development or management options but the estimates should only be considered indicative and never absolute.

5. Future model developments

Since the first avian collision risk models were created (Tucker, 1996a; Scottish Natural Heritage, 2000) there have been significant developments in the wind energy industry. Wind turbines have increased in size and energy rating, wind turbine arrays or wind farms have increased in area and number, and wind energy projects have moved offshore, and may move further with the development of floating wind turbines. Collision risk modelling tools must also accommodate these developments to ensure they are fit for purpose, as well as incorporating new methods that become available. From here on in we discuss the areas where we consider that advances in collision risk modelling will likely make the greatest contribution to impact assessments.

Despite the acknowledgement that the environment is variable, the majority of collision risk models to date have been deterministic, excluding variability and/or uncertainty from calculations. 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. The incorporation of uncertainty in future models would reduce the possibility that a collision estimate was driven by the choice of a single input parameter value.

Marques et al. (2014) suggest that collision risk is related to individual bird behaviour, as well as phenology, landscape type, and weather. Collision may also depend on the spatial distribution and configuration of turbines in arrays (De Lucas et al., 2008; Ferrer et al., 2012). If such factors were incorporated into collision risk models it may be possible to move away from situations where bird survey data collected during the environmental impact assessments of developments and bird abundance are only weakly related to mortality events at wind farm sites (De Lucas et al., 2008; Ferrer et al., 2012). Systematic research into the factors affecting mortality events is much needed and with ever increasing amounts of post-construction data, this should be plausible, assuming suitable experimental and/or survey design. Improving our understanding of bird behaviour including avoidance behaviour, would also greatly improve collision risk estimates. With telemetry data becoming available from bird tracking projects it may also be possible to improve our knowledge of bird flight speeds, and potentially in relation to wind conditions. Information from tracking projects could also be used to better understand how birds use wind farms. The data would allow for the assessment of bird movements in and around wind farms and it would be possible to determine the suitability of the oft-used assumption of constant flux.

An alternative approach to the increasingly complex mechanistic or kinematic models for collision risk estimation which are known to lack validation, is to use more simpler models but within a Bayesian framework (U.S. Fish and Wildlife Service, 2013). Such models can accommodate increasing complexity, however in the absence of knowledge, these models remain simple, using empirical data on actual collision events to determine parameter values. The Bayesian framework may be progressively used more if these increasingly mechanistic models fail to deliver improvements to the collision risk estimates, and the relationships between predicted and observed mortalities remain weak.

6. Conclusions

Avian collision risk models are a valuable tool used in the impact assessment of windfarms, both on- and offshore. Although the Band model (Band et al., 2007) is the most frequently used within the UK, a variety of models have been developed and are available to use. These models vary in their suitability for different situations and circumstances, due to the specific case or development they were designed for. Therefore it is important that the most appropriate model or method is used or adapted for the question at hand, and in some situations this may not always be the most frequently used model. This is particularly important as all wind energy stakeholders (developers, consultants, regulators, advisers and conservation organisations) must have confidence in the methods used. Failure to ensure this can lead to disputes and lengthy discussions over project details, potentially causing mistrust and costly delays in the consenting process, to no benefit.

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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).

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.

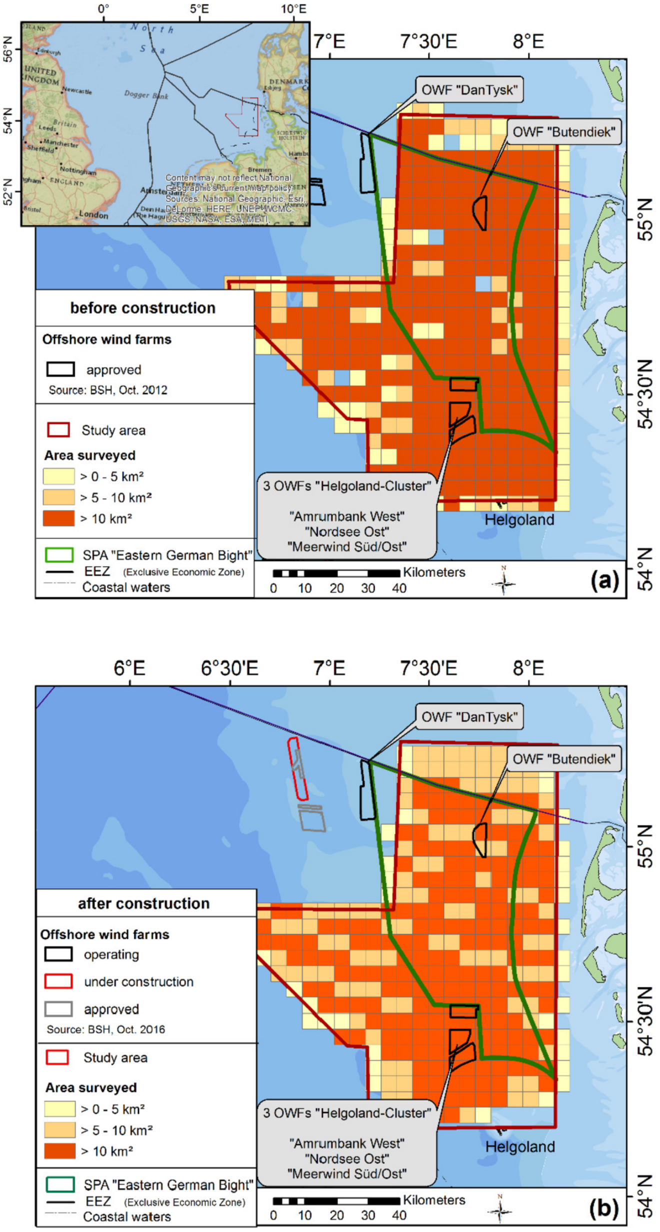

Fig. 1. Location of the study site within the south-eastern North Sea (inserted map in Fig. 1a) and in the eastern German Bight (North Sea) with locations of the different OWFs and the area surveyed for Loon abundance (yellow to red squares) across the 'Eastern German Bight' Special Protection Area (SPA; 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.)

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):

$$\log(y_{ij}) = \beta_0 + u_i + \mathrm{f(mean\_depth_j)} + \mathrm{f(dist\_coast_j)} + \mathrm{f(dist\_helgoland_j)} + \mathrm{s(latitude,longitude)} + [\mathrm{wind\ farm_j}] + \mathrm{period_j} + [\mathrm{wind\ farm_j}] \times \mathrm{period_j} + \mathrm{offset}(\log(\mathrm{area_j})) + \varepsilon_{ij} \tag{1}$$

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:

$$\log(y_{ij}) = \beta_0 + u_i + \beta_1 \mathrm{dist\_coast_j} + \mathrm{s(latitude,longitude)} + [\mathrm{wind\_farm_j}] + \mathrm{period_j} + [\mathrm{wind\_farm_j}] \times \mathrm{period_j} + \mathrm{offset}(\log(\mathrm{area_j})) + \varepsilon_{ij} \tag{2}$$

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):

$$\log(y_{ij})\beta_0 + u_i + \mathrm{f(mean\_depth_j)} + \mathrm{f(dist\_coast_j)} + \mathrm{f(dist\_helgoland_j)} + [\mathrm{wind\_farm_j}] + [\mathrm{ship\_number_j}] + \mathrm{offset}(\log(\mathrm{area_j})) + \varepsilon_{ij} \tag{3}$$

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:

$$\log(y_{ij}) = \beta_0 + u_i + \mathrm{f}(\mathrm{mean\_depth_j}) + \beta_1\mathrm{dist\_coast_j} + \mathrm{s}(\mathrm{dist\_helgoland_j}) + [\mathrm{wind\_farm_j}] + [\mathrm{ship\_number_j}] + \mathrm{offset}(\log(\mathrm{area_j})) + \varepsilon_{ij} \qquad (4)$$

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.

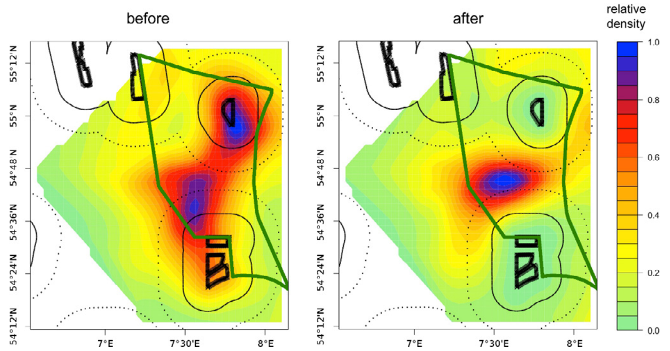

Fig. 2. Spatial density plots of predicted Loon distributions 'before' vs. 'after' the construction of OWFs, based on the BACI-GAMM. Bold black lines: OWFs; thin black lines: 10 km distance buffer; dotted black lines: 20 km distance buffer; bold green line: Special Protection Area.

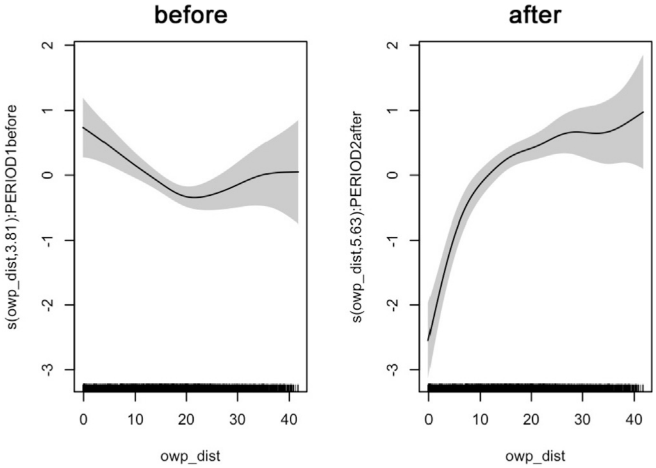

Fig. 3. Change in Loon abundance in relation to distance from the closest OWF site 'before' (left) and 'after' construction of the OWFs. Smoothed curve: predicted number of Loons at a given distance from the closest OWF; shaded area: 95% confidence interval; small lines on the x-axis: observations of 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

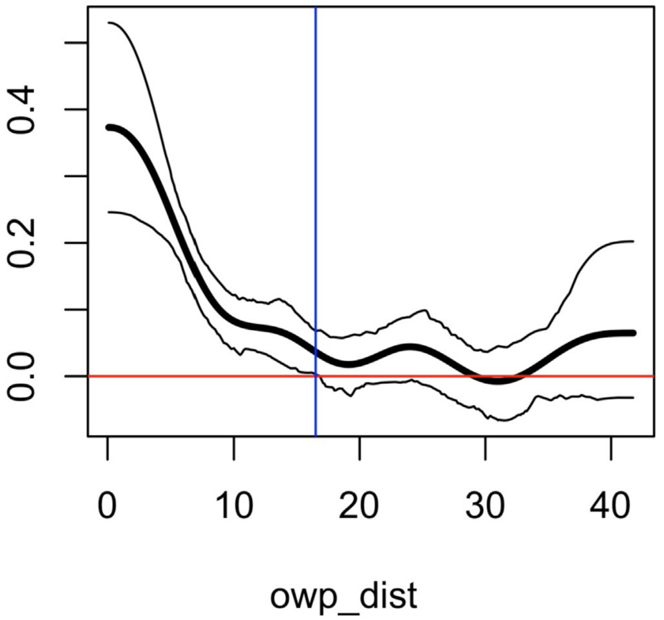

Fig. 4. First finite difference of the smooth depending on the distance from the closest OWF, partially evaluated for the 'after' period. Red line indicates a derivative of zero, blue line indicates distance at which the derivative was significant. Thick black line corresponds to the first derivative; 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.

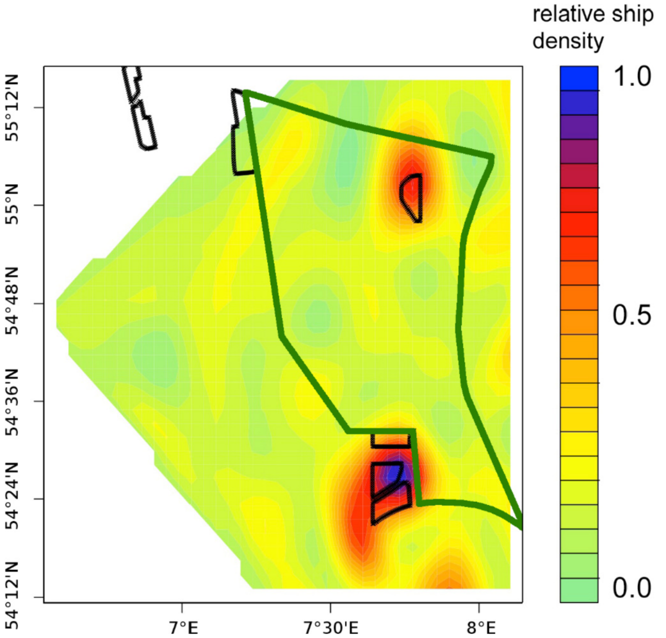

Fig. 5. Spatial density plot of ship distribution in the 'after' period based on AIS data.

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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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.

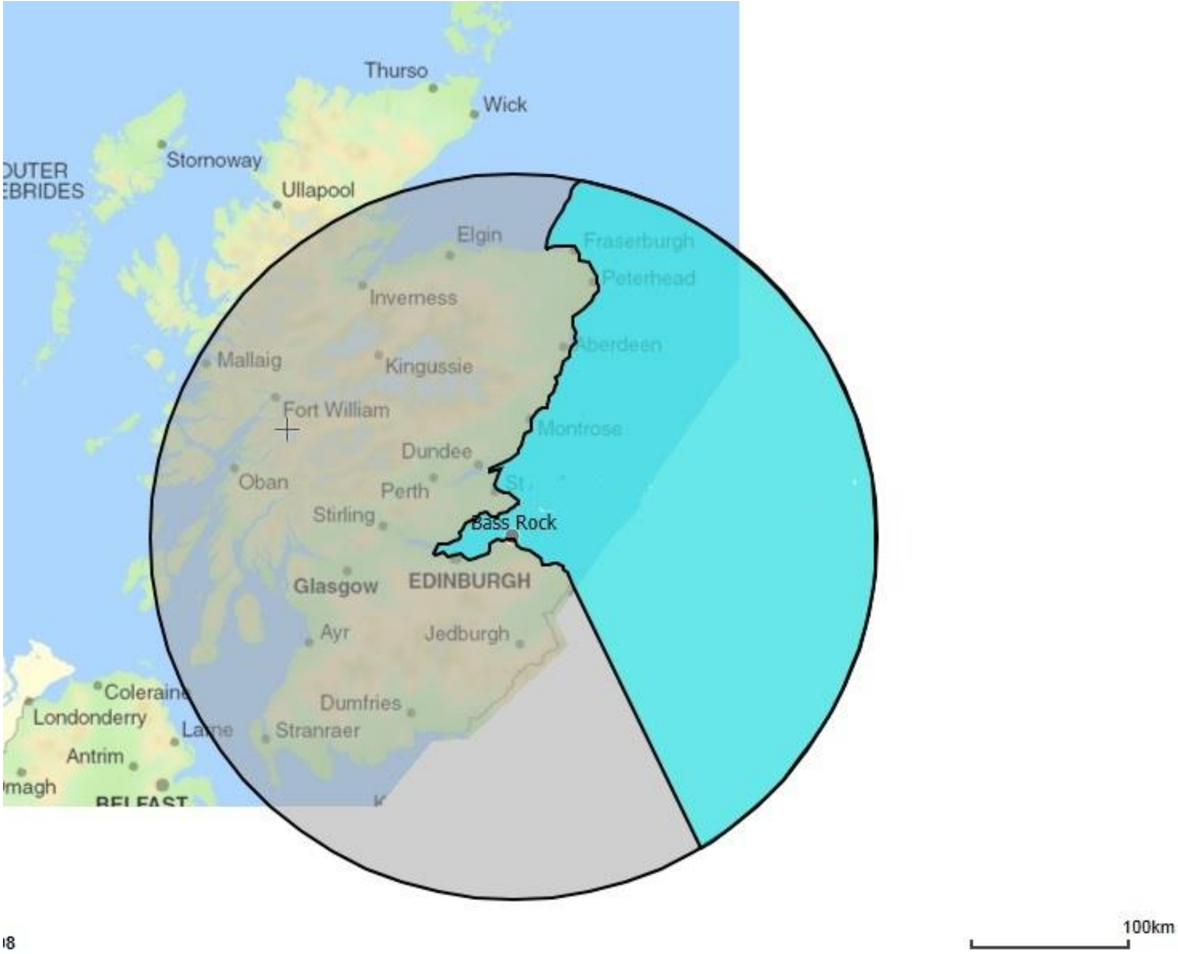


Figure 1. Illustration of effect of sea-route foraging range versus ‘straight-line’ foraging range. The blue area is the true area of sea accessible to a species with a foraging range of 230km from Bas Rock, whereas the grey circle shows the result of applying the straight-line distance, which includes sea areas on the west coast which would not be accessible.

Weighting by sea area

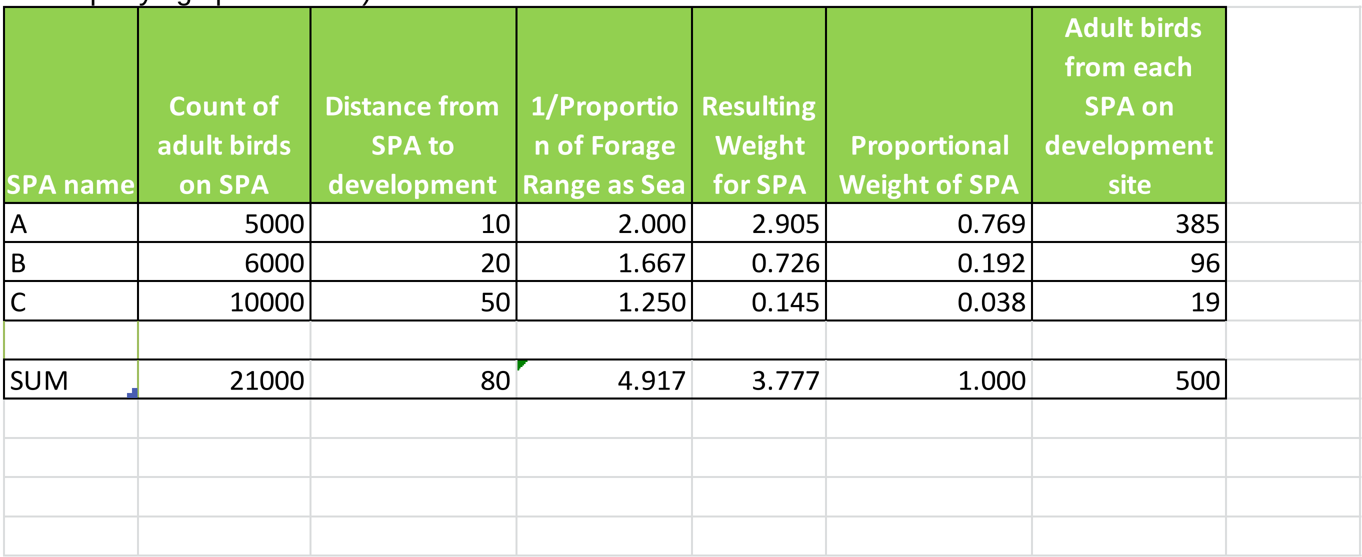
We suggest that a correction is also made for the available sea area. The logic behind this is that for a colony on an island, seabird foragers may have all 360^o of marine habitat to forage over, whereas for a mainland colony, birds may only have about 180^o of marine habitat. The available sea area 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.

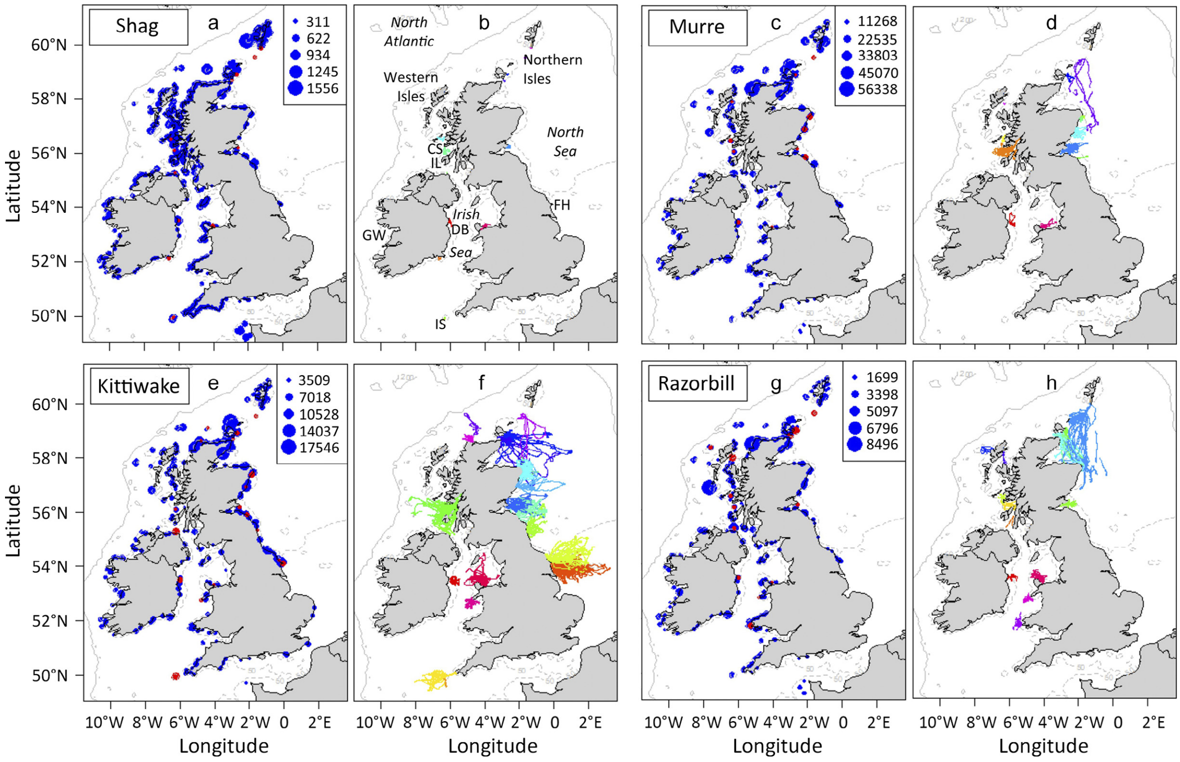

FIG. 1. Breeding distribution and individual movement data used to estimate the distribution at sea of seabirds foraging from UK colonies. Panels a, c, e, and g show numbers of apparently occupied nests recorded during the Seabird 2000 census (Mitchell et al. 2004; red indicates study colonies). Panels 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

$$l_{\mathrm{IPP}}(\beta; y, y_0, \mathrm{w}) \approx \sum_{i=1}^{m} w_i(s_i \log\lambda_i - \lambda_i) \tag{2}$$

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

$$\begin{aligned} \lambda_{k,i} &\sim \mathrm{Poisson}(\mu_{k,i}) \Rightarrow E(\lambda_{k,i}) \sim \mu_{k,i} \\ \log(\mu_{k,i}) &= \mathrm{offset}(n_k) + \beta_0 + \sum_{j=1}^{m} x_{i,j}\beta_j + u_k \end{aligned} \tag{3}$$

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

$$\overline{\mathrm{BA}} = \frac{\sum\limits_{\mathrm{All}k} n_k \mathrm{BA}_k}{\sum\limits_{\mathrm{All}k} n_k} \tag{5}$$

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

$$A_{k,i} = \sum_{\mathrm{All}x} a_x \delta_{k,x},\ \delta_{k,x} = \begin{cases} 1 & \text{if } d_{k,x} \leq d_{k,i} \\ 0 & \text{otherwise} \end{cases} \tag{6}$$

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

$$\theta = \sum_{\mathrm{All}k} \frac{\mathrm{N}_k}{d_{h,k} + 1} \qquad (7)$$

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

$$\theta' = \sum_{\mathrm{All}k} \frac{\sqrt{\mathrm{N}_k}}{d_{k,i} + 1} \qquad (8)$$

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

$$\rho_{h,i} = \frac{\lambda_{h,i}\mathrm{N}_h}{\sum_{k \neq h} \lambda_{k,i}\mathrm{N}_k} \qquad (9)$$

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

[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

$$\mathrm{UD}_{P,i} = \sum_{\mathrm{All}x} \mathrm{UD}_{s,x}\mathrm{N}_s \tag{10}$$

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
$\log(A) \times \sqrt{N}$	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

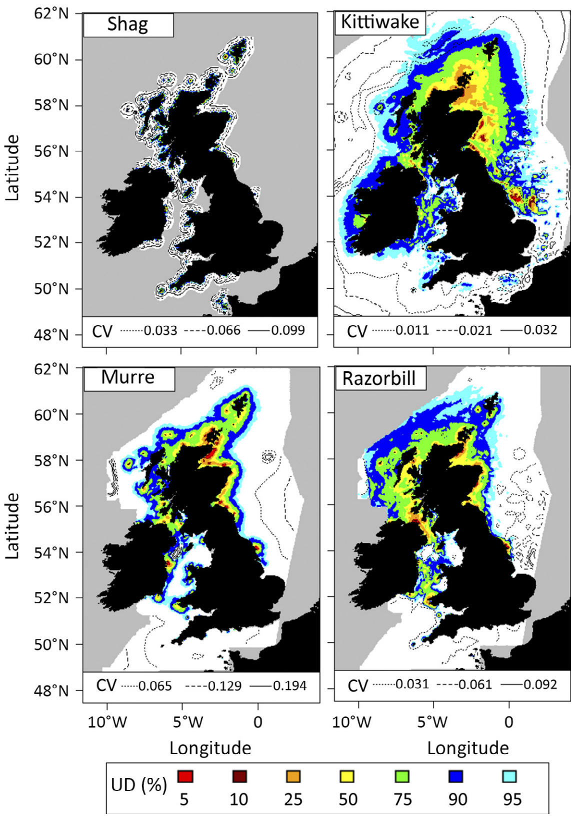

FIG. 2. Percentage at-sea utilization distribution (UD) of seabirds breeding within Britain and Ireland during late incubation/early chick-rearing estimated as functions of colony distance, coast geometry, intra-specific competition, and habitat (models V–VIII). Warmer colors indicate higher usage. Isopleths 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).

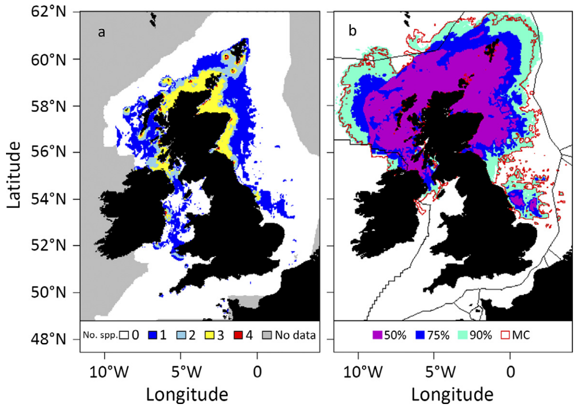

FIG. 3. Predicted multi-species hotspots. (a) Overlap between estimated core areas used by the four study species during late incubation/early chick-rearing. Colors indicate number of overlapping species' core areas (75% of the species' utilization distribution, UD; see Appendix S2: Fig. S7 for equivalent 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

[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.

Submission for Deadline 4
for
The Royal Society for the Protection of Birds

15 January 2019

Planning Act 2008 (as amended)

In the matter of:

Application by Ørsted Hornsea Project Three (UK) Ltd for an Order Granting Development Consent for the

Hornsea Project Three Offshore Wind Farm

Planning Inspectorate Ref: EN010080
Registration Identification Ref: 20010702

2. Ecology – Offshore		
Ref:	**Question to**	**Questions**
		Ornithology
Q2.2.2	NE, Royal Society for the Protection of Birds (RSPB)	**Notwithstanding the use of two out of four cameras, do you agree that the digital aerial survey data forms an adequate ornithological baseline for the months where data were collected over two separate years?** The RSPB agrees that the digital aerial survey data gathered forms an adequate baseline for the months where the data was collected.
Q2.2.5	RSPB	**In the draft SoCG [REP2-012] submitted at Deadline 2 you have highlighted that the baseline survey remains inadequate despite the results of baseline sensitivity testing [REP1-141].** **Is there anything else that can be done to improve the robustness of the baseline or is your position final on this matter bearing in mind any subsequent evidence that has been submitted?** The only improvement to the robustness of the baseline survey that we can see as being possible at this stage is for the Applicant present the data from four of the cameras used for the aerial survey or to carry out further survey.
Q2.2.6	RSPB, NE	**The Data Hierarchy Report [APP-110] indicates that more limited variation in bird density was observed between December and March.** **Is it reasonable to assume that missing data for this period would have less impact on the confidence in the modelling than missing data from other months?**

		As the principal ornithological issue relates to the effect of the project on the breeding bird assemblage at the Flamborough and Filey Coast Special Protection Area (SPA), why does it matter if there is missing data between December and March? **In practical terms, how is the additional information you are seeking likely to alter the conclusions of the Environmental Statement (ES) and Habitat Regulations Assessment (HRA)?** It is reasonable to assume to assume that missing data from this period would have less impact on the confidence in the modelling than missing data from other months. The RSPB would also reiterate that 2 years is an absolute minimum survey period in order to characterise the variability in bird density, and is already a compromise. It is impossible to know the extent on the influence on the conclusion of the ES and HRA without the missing 4 months data.
Q2.2.14	RSPB, NE	**The Applicant has advised that the nocturnal activity factors (NAF) historically used for collision risk modelling are not taken directly from Garthe and Hüppop (2004) but are instead based on an incorrect representation of the scores by Band (2012). The Applicant goes on to state that Band (2012) recommends that empirical data should be used when available, as has been the case for gannet and kittiwake.** **Please comment on these views and the empirical robustness of the studies that were used to justify the use of different NAF by the Applicant, as set out in [REP1-188].** **Please provide copies of any publications you wish to rely upon in evidence that have not already been provided.** The Applicant's argument that CRM nocturnal activity factors are not taken directly from Garthe and Hüppop (2004) is incorrect; they are taken directly from the paper and converted to

		percentages. Furness *et al.* (2018) point put that this method converts categorical data to linear data, and while this is likely to be correct (we only have information from other indices presented by Garthe and Hüppop (2004) as to whether they are categorical), Band is clear that this approach is a necessary simplification in the absence of survey data so it is wrong to say that they are an "incorrect representation". However we are in agreement with the Applicant that where empirical evidence is available it should be used. As such we request that empirical evidence of the timings of surveys is presented alongside any evidence of temporal variability in foraging activity, such as that described in Furness *et al*. (2018).
Q2.2.15	RSPB	**The Applicant has stated [REP1-122] that Marine Scotland has previously noted that your position appears to conflate nocturnal activity with colony attendance, foraging activity and timing of at-sea surveys and lacks an adequate empirical basis.** **How do you respond?** **Please provide copies of any publications you wish to rely upon in evidence that have not already been provided.** The Marine Scotland note needs further clarification as to its meaning but the note appears unjustified. Our argument is that the aerial surveys relied upon by the applicant may underestimate the number of birds at risk of collision as the surveys took place outwith the peaks of foraging activity recorded by tagging. The empirical evidence for this variability in the timing of foraging is reported in Furness *et al.* (2018). We are unable to assess this further, however, as the applicant has not provided details of the time of day of survey. The issue of how this is conflated with colony attendance is unclear; evidence presented in Furness *et al.* (2018) figure 3, is for diving only, a behaviour associated with foraging not colony attendance. It should also be noted that the Marine Scotland comment predates the publication of Furness *et al.* (2018).

Q2.2.16	RSPB	**The Applicant has stated in [REP1-122] that peaks in abundance that may occur at first light should not be accounted for by increasing the NAF which is used in the CRM to calculate the collision risk at night. The Applicant notes that the nocturnal activity rate used represents the activity expected as a proportion of daylight activity and, as such, the application of a nocturnal activity factor does not require consideration of peaks in activity that may occur at first light as the amount of nocturnal activity is the same regardless of the activity that occurs in daylight hours.** **How do you respond?** **Please provide copies of any publications you wish to rely upon in evidence that have not already been provided.** We do not argue for an increase in NAF. NAF, like avoidance rate, is a simple correction factor and if it is to be modified full consideration of all aspects relating to temporal activity cycles, including survey timing, should be made.
Q2.2.18	RSPB, NE	**Cook et al (2018) recommends avoidance rates for kittiwake and lesser black-backed gull that are different to those proposed by JNCC et al (2014) and the RSPB.** **Please comment on the results of the additional modelling, its empirical basis and the implications for the ES and HRA as set out by the Applicant in Appendix 10 at Deadline 1[REP1-188].** The Cook *et al*. (2018) Avoidance Rates are the same, and derived from the same data, as the Cook *et al*. (2014) Avoidance Rates, that the JNCC *et al*. (2014) advice is based on. The JNCC advised rates are identical to Cook *et al.* (2014 and 2018) except with regard to the rate recommended for kittiwake using the Basic model (Options 1 and 2). During the review that underpins both 2014 and 2018 papers there was inadequate data to assign a species-specific

		rate for kittiwake, so Cook *et al.* assigned it to the small gull species group. JNCC *et al.* argued that because of flight speed and species morphology, that kittiwake should be assigned to the large gull species group. The RSPB agreed with the JNCC *et al.* approach. The only area of further disagreement with the RSPB is in terms of gannet avoidance rate during the breeding season. As highlighted by Cook *et al.* (2018), the majority of the data used to calculate avoidance for gannet was collected outwith the breeding season, and they further highlight that "*the extent to which this information is relevant to birds when they are tied to their colonies is unclear*". It is likely that birds will show less avoidance during the breeding season, as they are under the constraints of Central Place Foraging, so for this reason the RSPB prefer that a lower avoidance rate is used at this time. In Appendix 10 at Deadline 1 [REP1-188] the Applicant presents additional modelling with changes made to three input variables: • Flight Speed • Nocturnal Activity Factor • Avoidance rate The RSPB agree that having empirical evidence to underpin these variables is an important process in refining our understanding of potential impacts. In terms of flight speed, the RSPB argued for the use of empirical data through the inquiry into Hornsea Project Two (see for example para 50 of our submission to deadline 6[1]). As such we welcome the inclusion of the flight speeds measured during the ORJIP BCA project. However, while these are valuable, we would highlight that they are from a single site and outwith the breeding season. It is unclear whether these data are transferable to other sites and to the breeding season. We have dealt with the Nocturnal Activity Factors in other answers, but in addition, partial analysis of data from thermal imaging cameras was carried out in the Skov *et al*. (2018) ORJIP BCA report, and is included in Appendix 10. However this analysis was incomplete and did not fully account for the distinction between the definition of daylight as used in the Band model and

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[1] https://infrastructure.planninginspectorate.gov.uk/wp-content/ipc/uploads/projects/EN010053/EN010053-001224-RSPB.pdf

		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 NAFs based on Skov *et al.* 2018. The empirically derived avoidance rates presented in Skov *et al.* (2018) are functionally different from the Avoidance Rates used in the Band (2012) model, as the latter incorporate error and variability in relation to both the data used and the model itself (Cook *et al*. 2014), which means that Band model Avoidances Rates will be lower than empirically derived avoidance rates. Debate is ongoing as to how to apply the EARs into the Band model and so it is not clear how, if at all, predicted mortalities would be different if the Skov *et al.* rates were considered. It is also important to highlight that there are difficulties in the manner in which Skov *et al.* (2018) calculated the Empirical Avoidance Rates, particularly for macro-avoidance. As there were no pre-construction data available for this calculation, the study estimated macro-avoidance by comparing the density of bird tracks within the wind farm to the density of bird tracks in a 3 km buffer around the wind farm. However this calculation assumed that there is no attraction by birds to the wind farm area. Other research has suggested that birds may be attracted to wind farm sites e.g. Vanermen et al (2015). Birds may also be attracted to the presence of birds funnelling or otherwise aggregating outside the wind farm. Furthermore it appears that fishing vessels were frequently recorded in the wind farm buffer which would increase the attraction to birds. Previous studies (Krijgsveld et al. 2011) noted gulls being attracted to fishing vessels on the edge of a wind farm and observers noted a similar effect as part of the ORJIP BCA study. In such circumstances, birds will be responding to the fishing vessels rather than the turbines and this will strongly bias the results. As such little confidence can be placed in this calculation. Therefore while the presentation of results based on these revisions if of interest, they should not be relied upon.

<table>
<tr>
<td>Q2.2.19</td>
<td>RSPB</td>
<td>
<p>The Applicant has stated [REP1-122] that no colony specific data from Flamborough and Filey Coast SPA were made available and that it is, in any event, irrelevant to the seasons in the array area.</p>
<p>Bearing in mind the typical foraging distances of breeding birds from this colony, why are the colony specific seasons relevant to what happens 150km away in the array area?</p>
<p>How many breeding individuals have been tracked and shown to be entering the array area each year?</p>
<p>Please provide copies of any publications you wish to rely upon in evidence that have not already been provided.</p>
<hr>
<p>The RSPB consider that the kittiwake and gannet recorded at the array area during the breeding season are most likely to have originated from the Flamborough and Filey Coast SPA. For HRA the impacts specific to this SPA must be considered for a likely significant effect, and so the colony represented by this SPA is the most relevant to define breeding seasons.</p>
<p>The RSPB tracked kittiwake from sites at Bempton and Filey within the SPA from 2010 to 2015. The details of the numbers of breeding individual entering the array area are given below</p>
<table>
<tr><th>Site</th><th>Year</th><th>No. birds tracked</th><th>No. birds entering Hornsea3</th><th>Proportion</th></tr>
<tr><td>Bempton</td><td>2010</td><td>25</td><td>0</td><td>0</td></tr>
<tr><td></td><td>2011</td><td>17</td><td>0</td><td>0</td></tr>
<tr><td></td><td>2012</td><td>9</td><td>2</td><td>0.22</td></tr>
<tr><td></td><td>2013</td><td>21</td><td>0</td><td>0</td></tr>
<tr><td></td><td>2014</td><td>17</td><td>1</td><td>0.06</td></tr>
<tr><td></td><td>2015</td><td>15</td><td>3</td><td>0.20</td></tr>
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<table>
<tr><td>Filey</td><td>2013</td><td>20</td><td>0</td><td>0</td></tr>
<tr><td></td><td>2014</td><td>16</td><td>0</td><td>0</td></tr>
<tr><td></td><td>2015</td><td>14</td><td>0</td><td>0</td></tr>
</table>
The RSPB will provide a breakdown of more recent tagging and overlap with the array area for Deadline 5.
</td></tr>
<tr><td>Q2.2.21</td><td>RSPB</td><td>How many kittiwakes were tracked in Cleasby et al (2018) [REP1-144] to give the hotspot results in section 3.1?
What proportion originated from the Flamborough and Filey Coast SPA?
642 kittiwake were tracked as part of the work that informed Cleasby <i>et al</i>. (2018). 154 were from the Flamborough and Filey Coast SPA, 24%.</td></tr>
<tr><td>Q2.2.22</td><td>RSPB</td><td>In [REP2-025] you note that kittiwake productivity has been in decline at the Flamborough and Filey Coast SPA since 2009 as set out in Aitken et al (2018).
Please provide a copy of this publication if you wish to rely upon it as evidence.
This is now provided.</td></tr>
<tr><td>Q2.2.23</td><td>RSPB</td><td>During ISH2 there was some discussion concerning the evidence underpinning the differences of opinion over how breeding seasons were defined. NE suggested that it had relied on an internal RSPB report.
Please confirm the details with NE and submit the report as evidence at Deadline 4.</td></tr>
</table>

		This document is the Guillemot and Kittiwake Phenology 2016-17 document that the RSPB submitted at Deadline 3.[2]
Q2.2.26	RSPB	**You did not answer question Q1.2.69 about how predicted displacement mortality should be evaluated against background displacement mortality. The Applicant is of the view that you approved this approach in the Evidence Plan Meeting.** **Have you departed from your original views?** **If so, what has changed?** **Please provide copies of any publications you wish to rely upon in evidence that have not already been provided.** In our answer to Q1.2.69 we stated that 1% is an arbitrary value which has no biological meaning and therefore cannot be used as a measure of significance of negative effect. This position has been clear throughout the Evidence Plan process so we have not departed from our original views. The RSPB support Natural England's position with regard to definitions of seasons, summation of seasonal mortalities, and use of differing mortality rates for different seasons.
Q2.2.27	RSPB	**You have stated that support vessels for servicing the turbines during the operational phase may cause displacement of divers and that a distance of 4km should be considered as the minimum distance within which impacts during this phase should have been considered. You cite a paper by Mendel et al (unpublished) in support of this view.**

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[2] https://infrastructure.planninginspectorate.gov.uk/wp-content/ipc/uploads/projects/EN010080/EN010080-001350-Royal%20Society%20for%20the%20Protection%20of%20Birds%20%20-%20Guillemot%20Razorbill%20and%20Kittiwake%20Phenology%202016-17.pdf

		As there are no loons off the north Norfolk coast you appear to be making a generalisation between this species and the red throated diver. What evidence do you have to suggest that their ethology and mortality risk are the same in all respects? **Given that this is unpublished work that cannot be submitted to the examination library how can the ExA give it any weight?** **Please provide copies of any publications you wish to rely upon in evidence that have not already been provided.** The RSPB apologise for not providing this paper. It is now provided. The authors of the paper use the American word "Loon" to refer to birds from the genus Gavia, which includes red-throated divers, *Gavia stellata*, and in the paper the authors highlight that 90% of the "loons" in the study area are red-throated divers (or "red-throated loons" as the paper says). The use of Latin names prevents any misunderstanding of these meanings. The results of this paper suggest a significant reduction in diver densities within a radius of up to 5 km from the vicinity of ships. As such this paper is highly relevant to consideration of the impacts of support vessels on the diver population off the north Norfolk coast.
Q2.2.28	RSPB	**You state that the correct manner in which to deal with uncertainties is through a properly quantified precautionary approach and not the qualitative approach taken by the Applicant [REP1-111]. The Applicant has set out the detail of the assessment from 5.9.2.24 in the ES [APP-065].** **How should the quantitative approach you advocate be carried out?** **Why is the assessment set out in the ES [APP-065] not adequate?**

Please provide copies of any publications you wish to rely upon in evidence that have not already been provided.

Uncertainty is inherent in scientific assessment and need not be detrimental but should always be described either quantitatively where possible or qualitatively, to provide a measure of confidence in the data which underpin decisions. In the assessment, where an estimate for a parameter is cited, a measure of uncertainty should be given with this estimate. The uncertainty associated with underlying modelling should also have been assessed. For example, the Skov *et al.,* 2018 report has been cited widely by the Applicant, but they have omitted to mention that analysis of this work suggested that

- Underpinning the Band (2012) collision risk model is the calculation of pColl, the probability of collision of birds passing through the rotor swept area of a turbine. The ORJIP BCA study was the first time that it has been possible to validate this calculation with empirical data. This showed that the calculation of probability of collision by the Band model may underestimate collision by a factor of four (Bowgen & Cook, 2018).
- Avian flight heights recorded during the ORJIP study were considerably higher than those previously estimated, and included in the assessment (Bowgen & Cook, 2018).
- Calculation of Avoidance Rates for the Skov *et al.,* (2018) report neglected to consider the influence of fishing vessels on bird distributions.

To facilitate a full assessment with consideration of uncertainty details of confounding factors such as these should be presented clearly.

There are well-established quantitative methods for calculating and expressing uncertainty, such as confidence limits which may be estimated directly or by techniques such as bootstrapping. These metrics present a measure of confidence in the data which is unambiguous and therefore should be presented where possible. Sensitivity analyses should also be carried out to identify the potential effect of any uncertainty and to show how potential variation in key results should information or data in the study be incorrect. There is precedent for such analysis in wind farm

		assessment, for example, sensitivity analyses were used to assess the effects of data uncertainty in a population viability analysis of the north Norfolk Sandwich tern population related to wind farm developments (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).
Q2.2.29	RSPB	**The Applicant has provided additional population viability assessment modelling outputs in [REP1-135].** **Has the model re-run addressed your concerns?** **If not, in your view, what is needed to be able to evaluate potential impacts?** The RSPB welcome the inclusion of this report and in particular the inclusion in the outputs of the RSPB preferred metric, the Counterfactual of Population Size (Green *et al.* (2016) and subsequently endorsed by Cook & Robinson (2017) and Jitlal *et al.* (2017)). We also welcome the use of the matched runs approach as recommended by Cook & Robinson (2017) and Jitlal *et al.* (2017). However we note that the demographic rates used are based on those carried out in 2012 as part of the Hornsea Project One Examination and this approach assumes that there have been no changes in population sizes or productivity in the intervening years. This is of particular concern for kittiwake whose productivity has been in decline at the pSPA since 2009 (Aitken *et al.* 2018). As such we can only have a limited amount of confidence in the conclusions for this species. We recommend that the models are re-run using the most up-to-date population data available.
Q2.2.31	RSPB	**You stated that you did not agree with the "evidence based displacement rates" for the array area but have not suggested any alternative values, as requested in question Q1.2.70.** **Is there any empirical evidence which suggests that the use of different values would be more robust?**

		Please provide copies of any publications you wish to rely upon in evidence that have not already been provided. The RSPB agree with the approach advocated by Natural England as per SNCB advice that a range of displacement rates be applied in a matrix approach. The reason for this is that the evidence base for displacement rates is scant and frequently equivocal.
Q2.2.32	RSPB	**You stated that the apportioning of impacts on kittiwake to the Flamborough and Filey Coast SPA was scientifically unjustified [REP1-111]. The Applicant has requested that you provide any information to the contrary to support a different apportioning rate.** **Is there any empirical evidence to the contrary to suggest the use of different values would be more robust?** **Please provide copies of any publications you wish to rely upon in evidence that have not already been provided.** 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 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. 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.
Q2.2.41	RSPB	**Please provide the following publications that you have relied upon in evidence:**

		Cook and Robinson (2017) [REP2-025] **Jithal et al (2017) [REP2-025]** **Green et al (2016) [REP2-025]** **Masden & Cook (2016) [REP2-025]** **Ferrer et al (2012) [REP2-025]** **de Lucas et al (2008) [REP2-025]** **Horswill and Robinson (2015) [REP1-111]** The documents requested above are listed in bold italic below, along with other documents which the RSPB has supplied: Appendix A - Aitken et al (2017) Appendix B – Bowgen & Cook (2018) ***Appendix C – Cook & Robinson (2017)*** ***Appendix D – Ferrer et al (2012)*** Appendix E – Garthe et al (2004) ***Appendix F – Green et al (2016)*** ***Appendix G – Horswill & Robison (2015)*** ***Appendix H – Jitlal et al (2017)*** ***Appendix I – Lucas et al (2008)*** Appendix J – Mackenzie & Perrow (2009) ***Appendix K – Masden & Cook (2018)*** Appendix L – Mendel at al (2019) Appendix M – SNH (2018) Appendix N – Wakefield et al (2017)

End of questions