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Proof of Evidence Rebuttal
Ornithology

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Contents

Rebuttal Proof of Murray Grant responding to the Updated Proof of Evidence of Dr A. McCluskie, RSPB

1. Introduction

1.1 My name is Dr Murray Grant and I am a Principal Ornithologist at Royal HaskoningDHV. I am acting as a witness on ornithology for Mentor Môn in relation to the Morlais Project (subsequently referred to as the Project).

1.2 This rebuttal Proof of Evidence is submitted in response to the proofs of evidence submitted for exchange.

1.3 I have read the various proofs of evidence submitted to the Inquiry and I am responding to issues raised on ornithology in the submitted Proof of Evidence, as updated on 23rd November 2020, of:

- Dr Aly McCluskie of RSPB

1.4 Insofar as I can usefully comment on the content and subject matter of the evidence of Dr McCluskie, I have. However, the absence of comments on any particular point should not be taken as agreement to it.

1.5 My responses to the evidence of Dr Aly McCluskie are set out below, according to each of the main sections of the RSPB Updated Proof of Evidence that the responses relate to. The paragraph numbers referred to are as for RSPB's Updated Proof of Evidence, with this updated proof subsequently referred to as RSPB's Proof of Evidence.

2. Species accounts

2.1 Section 3 of the RSPB Proof of Evidence (entitled '**The RSPB**') covers a range of topics, with the final subsection ('**Species Accounts'**) providing a general overview of the status and biology of the two species of seabird which are of key relevance to the Proof of Evidence (i.e. guillemot and razorbill).

2.2 These accounts describe how both species typically fly low over the sea surface and are regarded as having low manoeuvrability in flight (paragraphs 3.20 and 3.24 for guillemot and razorbill, respectively). The low flight manoeuvrability of these species is a result of the fact that they are adapted for swimming underwater, with their efficiency in this regard leading to an evolutionary trade-off with their flying ability[1].

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[1] Thaxter, CB, Wanless, S, Daunt, F, Harris, MP, Benvenuti, S, Watanuki, Y, Grémillet, D and Hamer, KC (2010) Influence of wing loading on the trade-off between pursuit-diving and flight in common guillemots and razorbills. *Journal of Experimental Biology*, **213**, 1018-1025.

2.3 However, it is important to note that this lack of flight manoeuvrability does not necessarily mean that these birds are likely to collide with above surface structures, as suggested in paragraphs 3.20 and 3.24 of RSPB's Proof of Evidence. I am aware of no direct evidence that these species frequently collide with such structures and, for example, collision with wind turbine bases is not considered as an impact pathway for these species in the numerous assessments undertaken for offshore wind farm developments in the UK. In a study ranking the sensitivity of different seabird species to adverse effects from tidal stream turbines and wave energy devices, Furness *et al.* (2012)[2] scored both guillemot and razorbill as 3 (on a scale of 1 to 5, where 1 equated to minimal risk of mortality and 5 to moderate risk of mortality) for the risk of colliding with wave energy devices, either in flight[3] or while swimming or diving. The authors state that, given the nature of wave energy devices, even a score of 5 on this factor would probably represent a relatively low risk compared to risks such as entanglement in netting.

2.4 Certain of the devices and designs which could be deployed in the Project would have structures that may protrude above the water surface, although this would not apply to the moving parts of any device. The assessment assumes that these structures could extend up to 6m above the surface. The potential for collisions with parts of the devices protruding above the water surface was screened out at scoping. The RSPB's responses to the Scoping Report did not raise this as an issue (see Table 11-7 of ES Chapter 11 – MDZ/A25.11, updated as MDZ/A31.11).

2.5 The species accounts presented in the RSPB's Proof of Evidence also identify that the study by Furness *et al.* (2012)[2], which ranks the sensitivity of different seabird species to adverse effects from tidal stream turbines and wave energy devices, classifies both guillemot and razorbill as being at high vulnerability to tidal turbines. This is correct but, importantly, this should not be taken as evidence of established impacts of tidal devices on these species. Rather, Furness *et al.* (2012) provide a classification based upon such factors as the extent to which the behaviours and habitat preferences of different species mean that they are likely to coincide with, and be exposed to, such devices, so that it identifies those species for which there is likely to be greatest need to investigate potential impacts.

3. Scope of proposed development (section 4 of the RSPB's Proof of Evidence)

3.1 Paragraphs 4.1 and 4.2 of RSPB's Proof of Evidence outline their ecological objections and key concerns with regard to offshore ornithology. In contrast to RSPB's position as set out in these paragraphs, the Applicant considers that:

- The ecological assessments carried out are robust, and employ appropriate methods, models and approaches to phasing considerations;

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[2] Furness, RW, Wade, HM, Robbins, AMC and Masden, EA (2012) Assessing the sensitivity of seabird populations to adverse effects from tidal stream turbines and wave energy devices. *ICES Journal of Marine Science*, **69**, 1466-1479.

[3] Noting that wave energy devices are usually floating structures so that there are above surface components with which birds in flight could (in theory) collide.

- The effects of the initial deployment on the guillemot and razorbill populations at the South Stack and Penlas colonies are predicted to be non-significant (where the initial deployment equates to the species-specific worst-case for which the predicted collision rate for bottlenose dolphins does not exceed the Potential Biological Removal, currently calculated as 0.7 animals per annum – see paragraph 3.3 of the Proof of Evidence for Ornithology (MDZ/P1) and Marine Ornithology Collision Risk Modelling Note (MDZ/F16, updated as MDZ/31.10));

- The initial deployment will not result in declines to any seabird populations;

- Subsequent deployment phases will depend on the findings from detailed monitoring to address uncertainties in the predicted collision risk, as determined in the Environmental Mitigation and Monitoring Plan (EMMP).

4. Offshore ornithology (section 5 of the RSPB's Proof of Evidence)

4.1 Within this section of the RSPB's Proof of Evidence a number of issues are raised concerning the avoidance rates which are applied to the outputs from the ERM and CRM. As detailed in paragraph 5.11 of the RSPB's Proof of Evidence, the avoidance rate is applied to account for the fact that a proportion of the animals predicted to pass through the rotor swept area of a turbine will not do so due to avoidance behaviour but, in addition, the avoidance rate also acts as a correction factor for model error.

Avoidance rates and addressing the high levels of uncertainty in predicted collision rates

4.2 Paragraphs 5.11 and 5.12 of the RSPB's Proof of Evidence make the case that the high level of uncertainty and potential error associated with the ERM and CRM mean that SNH's recommendation of applying and presenting a range of avoidance rates from 0% to 99% (MDZ/F19)[4] is required to properly express this uncertainty. This recommendation is followed in ES Chapter 11 (MDZ/A25.11, updated as MDZ/A31.11), where the ERM and CRM outputs are presented for the range of avoidance rates from 0% to 99.9%, allowing estimates for this full range to be examined and scrutinised. However, as described in the Proof of Evidence for Ornithology (MDZ/P1) the assessment of impacts on the key guillemot and razorbill populations focusses on a narrower range of avoidance rates from 95% to 99.9%, on the basis that these rates are considered to be most plausible. This range of avoidance rates encompasses a 50-fold difference in the estimated collisions (paragraph 6.19 of the Proof of Evidence for Ornithology (MDZ/P1)).

4.3 The approach of presenting the outputs for the full range of avoidance rates (0% - 99.9%) but focussing the assessment on a narrower range of the higher values is not inconsistent with the guidance, which states "*SNH recommends that all collision risk assessments using an avoidance factor should set out results using six avoidance rates: 0% (i.e. no avoidance), 50%, 90%, 95%, 98% and 99%*."[4] Thus, the guidance does not

[4] SNH (2016) Assessing collision risk between underwater turbines and marine wildlife. SNH Guidance Note (version 1) (MDZ/F19).

state that the assessment should necessarily focus on this wide range of avoidance rates, and SNH do not appear to expect such an approach to be taken. For example, the Habitats Regulations Appraisal (HRA) undertaken by SNH in relation to the proposed additional (sixth) turbine for the Shetland Tidal Array was based upon an assessment of collision risk to diving seabirds using the ERM with a 98% avoidance rate[5].

4.4 This higher range of avoidance rates is considered to be most plausible for the following two main reasons:

- First, both guillemot and razorbill are pursuit divers, using their wings for propulsion as they seek to catch fast moving fish which often use high burst speeds and high manoeuvrability to escape predators. Guillemot and razorbill are also visual foragers, relying on sight when locating, pursuing and catching their prey[6]. As such, these species are capable of making rapid response movements and have high levels of manoeuvrability when swimming underwater, suggesting that higher avoidance rates are likely to be appropriate.

- Second, as outlined in paragraph 6.20 of the Proof of Evidence for Ornithology (MDZ/P1), the avoidance rates applied to CRMs for onshore and offshore wind turbines in relation to birds in flight have increased for many bird species (and markedly so in some cases) from initial, precautionary, values of 95% or 98% as data availability and the understanding of interactions of birds with wind turbines has increased. Thus, for many species (or species groups) the avoidance rates that have now been calculated in relation to wind turbines are 99% or higher[7,8,9]. Although the RSPB Proof of Evidence points to one factor that could *potentially* cause the avoidance rates for use in assessments for tidal turbines to be lower than those used in assessments for wind turbines (at paragraph 5.14 and re-emphasised in paragraph 5.15), it is clearly the case that the range used in the Morlais assessment encompasses values considerably lower than the vast majority of the estimates derived for birds in flight in relation to wind turbines. Moreover, there are other factors that are likely to cause avoidance rates for tidal turbines to be higher than those for wind turbines, perhaps most notably the slower travel speeds of swimming than flying birds (see paragraph 6.20 of the Proof of Evidence for Ornithology (MDZ/P1)) and also that foraging activity by diving birds may be lower in those tidal states and

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[5] Letter from SNH to Marine Scotland, 2nd March 2018: SNH Advice – Marine Licence for the Deployment and Operation of the Shetland Tidal Array by Nova Innovation Ltd – 6 Turbines. Provided in **Appendix 1** of this rebuttal

[6] Hedd, A, Regular, PM, Montevecchi, WA, Buren, AD, Burke, CM and Fifield, DA (2008) Going deep: common murres dive into frigid water for aggregated, persistent and slow-moving capelin. *Marine Biology*, **156**, 741-751.

[7] Scottish Natural Heritage (2018) Avoidance rates for the onshore SNH Wind Farm Collision Risk Model. vs 2. (And references therein). Provided in **Appendix 2** of this rebuttal

[8] SNCBs (2014) Joint response from the Statutory Nature Conservation Bodies to the Marine Scotland Science avoidance rate review. Provided in **Appendix 3** of this rebuttal

[9] Bowgen, K. and Cook, A. (2018) Bird collision avoidance: Empirical evidence and impact assessments. JNCC Report, no. 614. Joint Nature Conservation Committee, Peterborough.

rates of flow associated with slow turbine rotation speeds or which are below cut-in speeds[10].

4.5 Paragraph 5.15 of the RSPB's Proof of Evidence states that the inherent uncertainty in avoidance rates, together with the complication of underwater turbines acting as ecological traps, mean that the avoidance rate that may be most reflective of biological reality, could actually be lower than those taken forward for population level assessment. This is re-emphasised in paragraph 5.31, which suggests that the available evidence does not support the use of higher avoidance rates.

4.6 However, as described above there are other (more compelling) reasons for considering the higher avoidance rates to be most plausible. Moreover, the case for underwater turbines acting as ecological traps (due to attraction of prey to their vicinity) should not be overstated. Whilst there is some evidence for prey attraction (as stated in paragraph 5.14 of the RSPB's Proof of Evidence), this is limited, and there is no evidence that this leads to the devices acting as ecological traps for diving seabirds. For example, long-term monitoring of diving bird abundance and activity at the Shetland Tidal Array site (encompassing pre- and post-deployment periods) reveals no evidence for any such effects. Rather, the monitoring data suggest low probabilities of turbine encounters for diving bird species (particularly during periods of operation), whilst the video footage from the operational turbines recorded no collisions or near misses between diving birds and turbines (with the only recorded occurrences of birds around the turbines being during periods when turbines were not operating)[10, 11]. These data derive from a development for which the predicted collision mortality for the most abundant diving bird species at the site (i.e. black guillemot) was 16.3 birds per breeding season, based upon the ERM with a 98% avoidance rate[5] (which is between the predicted breeding season collision mortality at 98% avoidance of guillemot (i.e. 25) and razorbill (i.e. 5) for the Project's proposed initial deployment).

4.7 The high levels of uncertainty associated with the outputs of the ERM and CRM is a major theme of the RSPB Proof of Evidence. This issue is highlighted in several parts of the document (in addition to the paragraphs specified above), with paragraph 5.30 also pointing out that these models are theoretical and unvalidated. Whilst this is not disputed, it is important to also emphasise that these are the models which are considered to provide the best available means of assessing collision risk to diving birds and which are recommended for this purpose by the relevant industry guidance provided by the Statutory Nature Conservation Bodies (MDZ/F19)[4].

4.8 Furthermore, the requirement for the assessment to adopt a highly precautionary approach as a consequence of the high levels of uncertainty associated with the ERM and CRM outputs (as advocated in paragraph 5.30 of the RSPB Proof of Evidence) is fully recognised. This is not limited to the application of a suitably precautionary range of avoidance rates. Rather, precaution is applied at several stages in the assessment

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10 Sparling, C.E., A.C. Seitz, E. Masden, and K. Smith (2020) Collision Risk for Animals around Turbines. In A.E. Copping and L.G. Hemery (Eds.), OES-Environmental 2020 State of the Science Report: Environmental Effects of Marine Renewable Energy Development Around the World. Report for Ocean Energy Systems (OES). (pp. 28-65). doi:10.2172/1632881

11 Smith, K. (2020) EnFait. Enabling Future Arrays in Tidal. D8.6 – Y3 Environmental Monitoring Report.

of the potential effects of the Project on seabird populations, including via the values used for certain of the ERM and CRM input parameters, as well as in the approach used for modelling the resultant population-level effects. Examples of other elements of precaution which have been incorporated into the assessment are set out in paragraphs 6.24 to 6.27 of the Proof of Evidence for Ornithology (MDZ/P1).

4.9 Paragraph 5.31 of the RSPB's Proof of Evidence states that the Applicant's discussion of avoidance rates fails to include consideration of "*uncertainty, variability and model error*". This is incorrect. Thus:

- Section 2.5.5 of ES Chapter 11, Appendix 11.3 (MDZ/27.5, updated as MDZ/31.12), entitled 'Selecting Appropriate Avoidance Rates', begins with the sentence "*There is considerable uncertainty regarding the avoidance rates for several reasons.*" This section then goes on to outline a range of the factors that lead to uncertainty in the avoidance rates, and much of this same text is also repeated in the main body of the chapter (ES Chapter 11 – MDZ/A25.11, updated as MDZ/A31.11). Section 2.2.2 of ES Chapter 11, Appendix 11.3 (MDZ/27.5, updated as MDZ/31.12) further alludes to uncertainty in the model outputs, stating "*The current body of evidence relating to diving bird behaviour, and likely avoidance, is limited. For this reason, a wide range of avoidance rates from 0 % to 99.9 % are considered.*"

- Paragraph 6 of ES Chapter 11, Appendix 11.3 (MDZ/27.5, updated as MDZ/31.12) states "*For both methods of collision risk modelling, the number of collisions is estimated by undertaking predictive modelling of the number of encounters between operational tidal devices and diving birds, and then adjusting this number by an avoidance rate. This is a catch-all term which describes a range of factors, which has a very large effect on the predicted number of collisions. Due to uncertainties surrounding avoidance rates of diving birds, a range of avoidance rates are presented as per SNH (2016) and discussions with the Ornithology TWG*." Although this is not an explicit statement concerning the incorporation of model error, it makes clear that the avoidance rate accounts for more than behavioural avoidance per se.

4.10 No mention is made of 'variability' in relation to the avoidance rates as applied to the ERM and CRM. The RSPB's Proof of Evidence is unclear as to exactly what is meant by variability in this context, but it is assumed to refer to statistical variability about model estimates. Both the ERM and CRM are what are termed deterministic models, meaning they do not account for statistical variability – i.e. they produce a single value for the estimate of collisions with no measure of the statistical variability that surrounds this estimate. Consequently, these models provide no means of determining the statistical confidence in the estimate. Given that these models are the industry standard and this is an understood and accepted limitation, in my view this point is largely academic.

4.11 Overall, in terms of discussing the avoidance rates, the assessment as presented in ES Chapter 11 (MDZ/A25.11, updated as MDZ/A31.11) and ES Chapter 11, Appendix 11.3 (MDZ/27.5, updated as MDZ/31.12) is clear in setting out the key issue surrounding the use of avoidance rates with the ERM and CRM, and it would be difficult to read these documents and not recognise that there are gaps in the knowledge base.

Applying a common avoidance rate to averaged ERM and CRM outputs

4.12 Paragraphs 5.11 and 5.13 of the RSPB's Proof of Evidence state that the avoidance rate used for a particular model (i.e. the ERM or CRM in this case) will be specific to that model and explain the reasons for this. Consequently, they point to the potential problems of taking the average ERM and CRM value and applying a common avoidance rate, as done in the collision assessment for the Project. This point is repeated in paragraph 5.27 of the RSPB's Proof of Evidence.

4.13 Whilst it is correct that the avoidance rate is model specific, this is of limited relevance with respect to the ERM and CRM as used in the Project assessment. In this case the averaging of the model outputs simply represents a pragmatic approach to dealing with the issue of generating outputs from two different models for multiple device scenarios and multiple species. As detailed in paragraphs 6.22 and 6.23 of the Proof of Evidence for Ornithology (MDZ/P1), adopting this pragmatic approach is reasonable (and valid) in this case because:

- Neither model is identified as being preferred by the relevant industry guidance (MDZ/F19)[4] and there is no basis for assuming that the estimates from one model are any more accurate than those from the other. Although the calculations and approach of the two models differ, in both cases the avoidance rate is applied to the model output as the end point in estimating collision mortality. Thus, it is equally likely that the ERM, CRM or averaged ERM/CRM outputs with a particular avoidance rate applied give the true collision estimate.

- The overall conclusions of the assessment are unaffected when the worst-case of the ERM and CRM is used instead of the averaged value, as detailed in paragraph 6.23 of the Proof of Evidence for Ornithology (MDZ/P1)[12].

Metrics used for interpreting the Population Viability Analysis (PVA) outputs

4.14 Paragraph 5.16 and 5.17 of the RSPB's Proof of Evidence describe the metric referred to as the Counterfactual of Population Size (CPS) and state that this is the most appropriate metric for interpreting outputs from Population Viability Analysis (PVA), with this having been determined in reviews commissioned by the Joint Nature Conservancy Council (JNCC) and Marine Scotland Science (MSS)[13,14]. This is not entirely correct because these reviews identify two metrics as being most appropriate for this purpose, with these being the CPS and the Counterfactual of Population Growth Rate (CPGR). Both of these metrics have been applied to guide the interpretation of the PVA outputs in the Project assessment of effects on seabird populations (see ES Chapter 11,

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12 Differences in the collision estimates for the averaged and worst-case of the ERM and CRM values can be determined from the files showing the workings for the calculations provided to RSPB on 26/10/10.

13 Cook, ASCP and Robinson, RA (2016) Testing sensitivity of metrics of seabird population response to offshore wind farm effects. JNCC report no. 553. Provided in **Appendix 4** of this rebuttal

14 Jitlal, M, Burthe, S, Freeman, S and Daunt, F (2017) Testing and validating metrics of change produced by population viability analysis (PVA). Scottish Marine and Freshwater Science, vol. 8, no. 23. Provided in **Appendix 5** of this rebuttal

Appendix 11.3 (MDZ/27.5, updated as MDZ/31.12) and Marine Ornithology Collision Risk Modelling Note (MDZ/F16, updated as MDZ/31.10)).

4.15 Paragraph 5.19 of RSPB's Proof of Evidence points out that neither of the reviews which assess the suitability of different metrics for interpreting PVA outputs[13,14] advocate the use of the "25 year population relative to current population", which is presented alongside the CPS and CPGR metrics in ES Chapter 11 (MDZ/A25.11, updated as MDZ/A31.11) and the Marine Ornithology Collision Risk Modelling Note (MDZ/F16, updated as MDZ/31.10). This is entirely correct and it is true that there is greater uncertainty associated with metrics which rely more on predictions of absolute future population sizes. However, whilst this additional metric is presented, it remains the case that the conclusions of the assessment are based upon an interpretation of the CPS and CPGR metrics. As a secondary point, it is considered that this additional metric does provide useful context because the South Stack and Penlas guillemot and razorbill populations have shown increasing population trends over recent years. Therefore, at a broad level, the predicted 25 year trend for these populations under baseline conditions is consistent with this and indicates (broadly) what the consequences of the different levels of predicted collision mortality could be *if* the wider environmental conditions sustaining these population increases were to be maintained into the future (although it is accepted that there is substantial uncertainty regarding this).

Independent verification of the ERM and CRM calculations

4.16 Paragraph 5.28 of RSPB's Proof of Evidence states that although the ERM and CRM modelling appeared (superficially at least) to have been undertaken appropriately, the spreadsheets detailing the calculations were not shared until 26th October 2020, so preventing confirmation of this until then. The point is made that the need for such independent verification was highlighted by the fact that immediately prior to sharing these spreadsheets with RSPB, the Applicant discovered an error in these workings.

4.17 Clearly, the oversight in not sharing these spreadsheets with RSPB until the 26th October is unfortunate, as was the error in the calculations. However, it is important to note that this error had (in the most part) only a small effect on the calculated collision estimates and the conclusions were unchanged following its correction. The Applicant fully concurs with RSPB on the point of making such calculations available for independent verification and in this regard it is important to note that these spreadsheets were shared with NRW in late 2019.

4.18 The statement in paragraph 5.42 of RSPB's Proof of Evidence that the source spreadsheets for the ERM and CRM calculations have not been provided is incorrect and assumed to be an oversight from the previous version of their Proof of Evidence.

Approach to the revised collision risk modelling and the predicted level of impacts to the guillemot and razorbill populations from the South Stack and Penlas colonies

4.19 Paragraphs 5.38 to 5.46 of RSPB's Proof of Evidence set out their position with regard to the ERM and CRM calculations undertaken to assess the impacts of the revised initial deployment on the guillemot and razorbill populations, as assessed in the Marine Ornithology Collision Risk Modelling Note (MDZ/F16, updated as MDZ/31.10). The

RSPB's Proof of Evidence suggests a number of problems with these calculations and, consequently, they do not base their conclusions on the revised deployment (instead continuing to base conclusions on the assessments for the 40MW and 240MW deployments, as presented in ES Chapter 11, Appendix 11.3 (MDZ/27.5, updated as MDZ/31.12)).

4.20 However, the 'problems' identified by the RSPB's Proof of Evidence are, in my view, spurious and their arguments in this regard are without credibility or logic. The reasons for this are outlined below:

- First, the RSPB's Proof of Evidence states that basing the revised collision estimates on a calculation of "collisions per MW" based upon the original ERM and CRM estimates (as presented in ES Chapter 11, Appendix 11.3 (MDZ/27.5, updated as MDZ/31.12)) appeared to be severely flawed, being underpinned by a wholly unproven and likely incorrect assumption of a linear relationship between MW output and mortality (paragraphs 5.38 and 5.39). However, the revised deployment only considers scenarios that comprise a single type of device, with each scenario comprising a different device (see Table 2.2 in the Marine Ornithology Collision Risk Modelling Note (MDZ/F16, updated as MDZ/31.10)). The calculation for both the ERM and the CRM involves estimating the collision risk associated with a single device and then multiplying that by the number of devices to be deployed. Therefore, if all devices for a particular scenario are the same, the relationship between MW output and the calculated collision estimate is clearly linear (e.g. to take a trivial example, if the collision estimate for a 10MW device is 1 bird, then there are 0.1 collisions per MW, and the total collision estimate for a deployment of 10 devices can be obtained either by multiplying the 1 bird per device by 10, or the 0.1 birds per MW by 100).
- Secondly, the RSPB's Proof of Evidence claims that the approach adopted by the Applicant is only acceptable if the extrapolation is on the basis of collisions per whole devices and not collision per MW (paragraph 5.40). The logic of this is unclear because the calculation is undertaken purely for the purposes of establishing the design envelope within which the predicted impacts must occur. It is of course the case that such an approach can lead to an estimate which involves a fraction of a device, as illustrated by the example in paragraph 5.45 of the RSPB's Proof of Evidence. And, it is also clearly impossible to install a fraction of a device. However, in terms of the assessment, this simply means that it is more precautionary in terms of the worst-case that is assessed because (using the example in paragraph 5.45), the predicted collision mortality for 5.89 devices is greater than that for five devices (which is what would be produced if extrapolating on the basis of whole devices). It also serves to provide greater flexibility in determining the final deployment because it uses the correct 'ceiling' as set by the worst-case collision estimate (provided of course that this worst-case level of predicted collisions is deemed acceptable by NRW in accordance with the requirements of the EMMP).

4.21 As to the statements in paragraph 5.45 of RSPB's Proof of Evidence that the approach of considering part devices is "*wholly unacceptable in an environmental assessment*" and “*An assessment should only be based on whole numbers of devices*”, I am not aware of any guidance or advice that exists to this effect. Rather, these statements appear to suggest that the purpose of establishing the design envelope is not understood. In this regard it is worth noting that the approach being taken to the Project Design Envelope (PDE) by this Project is supported in principle by NRW guidance (MDZ/15)[15].

4.22 Paragraph 5.43 of the RSPB's Proof of Evidence also states that the position taken by the RSPB at the meeting on 4th August 2020 is misrepresented in the Marine Ornithology Collision Risk Modelling Note (MDZ/F16, updated as MDZ/31.10). This was certainly not the intention of the Applicant and, presumably, results from a misunderstanding of the discussions had at that meeting.

4.23 As a consequence of the position adopted in the RSPB's Proof of Evidence, their conclusions on the assessment are based upon the predicted impacts for the 40MW and 240MW deployments. Paragraphs 5.33 to 5.37 of the RSPB's Proof of Evidence detail the levels of impact that are predicted for the guillemot and razorbill populations from the South Stack and Penlas colonies as a result of the 40MW and 240MW deployments that were assessed in ES Chapter 11 (MDZ/A25.11, updated as MDZ/A31.11). As described, these levels of predicted impact are high and are predicted to be moderate adverse or major adverse in ES Chapter 11 (MDZ/A25.11, updated as MDZ/A31.11).

4.24 The Applicant considers that these deployment scenarios have been superseded by the reduced, initial, deployment, and for the reasons detailed above considers that the assessment of this initial deployment is robust and entirely valid. As presented in the Marine Ornithology Collision Risk Modelling Note (MDZ/F16, updated as MDZ/31.10), the predicted impacts of this reduced deployment are concluded to be minor adverse (or non-significant in EIA terms) for both the guillemot and razorbill populations.

4.25 Through the adaptive management plan, the Project would aim to increase the level of deployment beyond this initial phase (ultimately to 240MW) but this would depend upon the monitoring established via the EMMP demonstrating that the level of risk to marine bird populations from the further deployments was acceptable. It is also considered that the extent of precaution incorporated into the assessment will mean that the predicted impacts from the 40MW and 240MW deployments represent substantial overestimates of the true impacts.

Other points

4.26 Paragraph 5.1 states that there is little information on surface and subsurface collisions with tidal turbines. As detailed in paragraph 2.3 above, there is no evidence to suggest that any above surface structures associated with the Project are likely to pose a collision risk to either guillemot or razorbill.

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[15] Sparling, C and Smith, K (2019) Defining project envelopes for marine energy projects: Review and tidal energy test facility and marine mammals case study. Natural Resources Wales, report no. 274.

5. Proposed adaptive management approach (section 8 of the RSPB's Proof of Evidence)

Technological issues

5.1 Paragraphs 8.3, 8.4 and 8.9 of the RSPB's Proof of Evidence outline some of the challenges involved in monitoring the impacts of tidal devices on marine bird populations (and specifically the populations of guillemot and razorbill at the South Stack and Penlas colonies). At several points in these paragraphs, it is stated that there are constraints related to the fact that the required technology does not exist, that the monitoring techniques are in their infancy or that the technology required for the proposed monitoring is not technologically capable of addressing the aims of the OEMMP.

5.2 It is undoubtedly the case that *some* parts of the monitoring proposed will be challenging but it is important to point out that of the three main components proposed for the seabird monitoring programme, these technological issues are essentially relevant to only one (i.e. the monitoring of collisions at devices using sonar and / or video). The work that is proposed to monitor annual numbers and breeding success at the colonies and to track the movements and diving behaviour of individual birds (the other two components of the monitoring programme) have been undertaken on the same species at numerous other sites and, to large degree, use established methods and techniques. Details are provided in paragraphs 6.10 to 6.14 of the Proof of Evidence for Ornithology ((MDZ/P1) and in the latest version of the OEMMP (MDZ/A16.8).

5.3 Data on annual numbers and breeding success at the colonies (and changes therein) and on the movements and diving behaviour of individual birds could provide important indications of effects from the deployment of the tidal devices and would allow refinement of the existing collision risk predictions (including important elements of avoidance, such as changes in the use of the waters around the array following deployment, and whether foraging effort varies with tidal state and flow rates which also determine when the turbines operate). It is also the case that active sonar (and / or video) may enable detection of collisions and / or near-field avoidance at the devices, although it seems likely that there would be limitations in the detection of the species involved (in which case it could be limited to providing a worst-case determination of collisions) (MDZ/F15.2)[16].

5.4 Thus, the RSPB's Proof of Evidence is too sweeping in appearing (in some instances at least) to suggest that these technological problems apply to the overall monitoring programme, as opposed to a single component of that programme. Furthermore, these technological problems, as they apply to the use of active sonar and / or video, are arguably overstated, given that they have met with some success at other tidal sites (including at the Shetland Tidal Array)[11,16].

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[16] ABPmer, (2020). Review of potential collision between tidal stream devices and marine animals, NRW Evidence Report No. 444, (ABPmer Report No. R.3322). A report produced by ABPmer for Cyfoeth Naturiol Cymru (Natural Resources Wales), June 2020 (MDZ/F15.2).

Colony counts

5.5 Paragraphs 8.20 and 8.21 of RSPB's Proof of Evidence describe the need for relatively detailed colony monitoring, including estimation of breeding productivity and the need for such work to be informed by power analysis. As detailed in paragraph 6.10 and Appendix 2 of the Proof of Evidence for Ornithology (MDZ/P1), it is proposed that such work would be undertaken.

Tagging studies

5.6 Paragraphs 8.25 to 8.32 of the RSPB's Proof of Evidence outline some of the challenges involved in establishing studies to monitor movements and diving behaviour of guillemots and razorbill via attachment of tags. These include the need to (safely) catch birds for tag attachment, issues of data download from the tags and the constraints in terms of the period during which such work is possible.

5.7 It is important to note that consideration has been given to issues of this type, with advice sought from RSPB on the potential for catching guillemots and razorbills at the South Stack and Penlas colonies and discussions held with other researchers involved in undertaking similar work on these species, including at nearby colonies in North Wales (as detailed in MDZ/A28.56). The advice received from RSPB indicates that it may be possible to catch approximately 15 individuals of each species during a single breeding season. Whilst establishing such studies to monitor movement and diving behaviour is undoubtedly challenging, such work has been undertaken on guillemots and razorbills at many locations in the UK and elsewhere.

5.8 Paragraph 8.29 of the RSPB's Proof of Evidence states that previous versions of the OEMMP indicated that tagging studies would be used as a method of monitoring collisions. The OEMMP no longer indicates this and it is recognised that tagging studies are not a suitable method for determining collisions.

5.9 Paragraph 8.32 of RSPB's Proof of Evidence indicates the need to consider possible effects of catching and tagging on those individual birds on which such studies are undertaken. These more detailed aspects of exactly how the tagging studies would be undertaken will be determined via the EMMP but it can be confirmed that consideration will be given to such potential effects (e.g. via comparisons of nest attendance, duration of foraging bouts and breeding success between tagged birds and a sample of untagged 'control' birds). Indeed, it is expected that RSPB would be a valuable source of advice in this respect, given their experience and long history of having undertaken tagging studies at breeding seabird colonies.

6. Summary and conclusions

6.1 My rebuttal Proof of Evidence details a number of points on which the RSPB's Proof of Evidence is considered to be incorrectly focussed or to be flawed or in error.

6.2 Of key relevance is that the main conclusions of the RSPB's Proof of Evidence remain focussed on the predicted impacts on the key guillemot and razorbill populations which result from the 40MW and 240MW deployments, as assessed in ES Chapter 11 (MDZ/A25.11, updated as MDZ/A31.11). However, in terms of the predicted impacts on these populations it is the smaller, revised, deployment which is relevant (as presented in Marine Ornithology Collision Risk Modelling Note (MDZ/F16, updated as MDZ/31.10). Predicted impacts from this revised deployment are considerably lower than for the original deployment scenarios, and are assessed as having non-significant effects on the guillemot and razorbill populations. RSPB consider that the assessment of the smaller, revised, deployment is not valid but, in my view, the reasons they provide to justify this position are entirely spurious and without credibility or logic. The Applicant considers that the assessment for the revised deployment is robust, and that there are no valid reasons for dismissing it in the way RSPB have done.

6.3 The RSPB's Proof of Evidence suggests that the assessment should focus on ERM and CRM outputs using a range of avoidance rates from 0% to 99% but such an approach gives no consideration to the known biology of the species in question or to other relevant sources of evidence in determining what is most plausible in this respect. It is considered that such considerations suggest that higher avoidance rates are most plausible and on this basis, avoidance rates of 95% to 99.9% are considered to encompass a sufficiently precautionary range of values. This approach is not inconsistent with the guidance from the Statutory Nature Conservation Bodies (SNCBs), and collision assessments for other tidal developments by SNCBs may also rely on such higher avoidance rates.

6.4 In considering the conclusions of the assessment, the RSPB's Proof of Evidence also fails to take full overview and account of the range of precautionary approaches and assumptions that have been used to derive the predictions on the population-level impacts, instead focussing almost exclusively on the avoidance rates. Prominence is also given to issues that from a practical perspective are of little relevance, whilst there is a failure to recognise that the assessment has given due consideration and attention to the uncertainty associated with the use of avoidance rates in relation to the ERM and CRM.

6.5 Some aspects of the background information presented in the RSPB's Proof of Evidence on the basic biology and relative vulnerability of the key species to tidal devices fail to provide sufficient context and could be taken to imply greater effects, or a greater potential for effects, than considered by the assessment. Thus, it is important to be clear as to the implications that can made from this background information.

6.6 In terms of the monitoring proposed as part of the adaptive management strategy to enable deployments to increase beyond the initial phase, the RSPB's Proof of Evidence is considered to be potentially misleading in some important respects. Thus, the extent of the technological limitations and challenges that will be faced in establishing an effective programme are over-emphasised and fail to acknowledge the potential for valuable monitoring that will be able to inform and refine the predicted collision risk to diving seabirds. Whilst emphasising the challenges faced by certain elements of this monitoring programme, the RSPB's Proof of Evidence also fails to acknowledge the extent to which much of the monitoring follows approaches that have (and are being) used elsewhere and which would rely upon established methods.

Appendix 1 - Letter from SNH to Marine Scotland, 2nd March 2018: SNH Advice – Marine Licence for the Deployment and Operation of the Shetland Tidal Array by Nova Innovation Ltd – 6 Turbines

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

Your ref: 06642

Our ref: CNS/REN/TP/Shetland – Bluemull Sound – Nova Innovation – 6 Tidal Turbine Array/CLC149573

Date: 2 March 2018

Sophie Humphries
Marine Scotland
Marine Laboratory
PO Box 101
375 Victoria Road
Aberdeen
AB11 9DB

By email only:
MS.MarineRenewables@gov.scot

Dear Sophie,

SNH ADVICE – MARINE LICENCE FOR THE DEPLOYMENT AND OPERATION OF THE SHETLAND TIDAL ARRAY BY NOVA INNOVATION LTD - 6 TURBINES

Thank you for consulting us on 22 February 2018 for the marine licence for the Shetland tidal array by Nova Innovation Ltd at Bluemull Sound. This current proposal includes the deployment of an additional single tidal turbine increasing the total number of turbines from 5 to 6 within the array.

Background

We provided screening advice for a Shetland Islands Council marine works licence and a marine licence with respect to this proposal on 19 December 2017. We subsequently met with Nova Innovation, MS LOT and Shetland Islands Council on 26 January 2018 to discuss this proposal and ongoing monitoring for the Shetland Tidal Array.

Following previous advice (letters of 24 June 2013, 27 August 2015, 26 January 2016) with regard to the licences and discussions at the recent meeting, and taking account of the documents provided in support of the 6 turbine work licence application, we have updated our advice to take account of this proposed additional turbine, notably with respect to collision risk.

We provided updated collision risk assessments for the 6 turbine array to inform this marine licence and related Shetland Islands Council (SIC) works licence application (9 February 2018). We include an assessment for the Bluemull and Colgrave Sounds proposed SPA (pSPA) for breeding red-throated diver qualifying interests. This site has been proposed as a pSPA since the consent of the 5 turbine array and therefore is a relevant consideration in our assessment for the 6 turbine proposal. We note and welcome that the advice we provided on 9 February 2018 has been incorporated into the latest *Shetland Tidal Array Extension – Environmental Assessment Report* submitted by Nova for this marine licence.

Scottish Natural Heritage, Battleby, Redgorton, Perthshire PH1 3EW
Tel: 01876 580236 E-mail: tracey.begg@snh.gov.uk www.nature.scot

Advice

We consider that the deployment and operation of this array of 6 tidal turbines and associated infrastructure can be implemented without serious adverse effects on natural heritage interests. However, the proposal requires consideration of natural heritage issues of international and national importance. Appendices A, B and C include our detailed advice.

The proposed array is likely to have a significant effect on qualifying interests of:

- Yell Sound Coast Special Area of Conservation (SAC) (harbour seals; see Appendix A)
- Hermaness, Saxa Vord and Valla Field Special Protection Area (SPA) and
- Bluemull and Colgrave Sounds proposed SPA (pSPA) (see Appendix B).

We have concluded that the project **will not have an adverse effect on site integrity for these Natura sites**.

In addition, we advise that a European Protected Species (EPS) will be required with respect to relevant cetaceans for the project construction phase (see Appendix C – Advice on natural heritage interests). We have considered other relevant marine species (see Appendix C) and have concluded that significant adverse effects can be avoided.

Environmental Monitoring and Mitigation Plan (EMMP)

We refer you to our most recent detailed advice relating to the EMMP (letter of 15 August 2017) for the previous works and marine licence applications. This advice remains relevant for the current 6 turbine array application. We recommend continued liaison between Marine Scotland and SIC in the formation of licence conditions, notably with respect to details of monitoring requirements within the EMMP.

Further information and advice

We hope this advice is helpful. If further information or advice is required please contact me in the first instance: tracey.begg@snh.gov.uk or 01876 580236.

Yours faithfully

Dr Tracey Begg
Policy & advice officer - Marine energy and seaweed harvesting

Cc marine.planning@shetland.gov.uk

APPENDIX A

NOVA INNOVATION TIDAL ARRAY, BLUEMULL SOUND, SHETLAND

HABITATS REGULATIONS APPRAISAL – SPECIAL AREA OF CONSERVATION (SAC)

1. Where a plan or project could affect a Natura site, the Habitats Regulations require the competent authority – the authority with the power to undertake or grant consent, permission or other authorisation for the plan or project in question – to consider the provisions of regulation 48. This means that the competent authority has a duty to:

- determine whether the proposal is directly connected with or necessary to site management for conservation; and, if not,
- determine whether the proposal is likely to have a significant effect on the site either individually or in combination with other plans or projects; and, if so, then,
- make an appropriate assessment of the implications (of the proposal) for the site in view of that site's conservation objectives.

2. This process is now commonly referred to as **Habitats Regulations Appraisal** (HRA). HRA applies to any plan or project which has the potential to affect the qualifying interests of a Natura site, even when those interests may be at some distance from that site.

3. The competent authority, with advice from SNH, decides whether an appropriate assessment is necessary and carries it out if so. It is the applicant who is usually required to provide the information to inform the assessment. Appropriate assessment focuses exclusively on the qualifying interests of the Natura site affected and their conservation objectives. A plan or project can only be consented if it can be ascertained that it will not adversely affect the integrity of a Natura site (subject to regulation 49 considerations).

4. SACs relevant under this HRA can be determined by (a) species observed at the site during site survey, (b) the distance between SACs and the proposed development site, and (c) the foraging range of species designated as qualifying interests. Consequently, we recommend that the only SAC relevant for consideration under HRA is Yell Sound Coast SAC.

Yell Sound Coast SAC

5. Yell Sound Coast SAC is designated for harbour seals and otters. The proposal is approximately 28km from the nearest part of the SAC.

Step 1:	Is the proposal directly connected with or necessary for the conservation management of the SAC?

6. The proposal is not directly connected with or necessary for the conservation management of the Yell Sound Coast SAC.

Step 2:	Is the proposal likely to have a significant effect on the qualifying features of the SAC either alone or in combination with other plans or projects?

7. The conservation objectives of the site are:

(i) to avoid deterioration of the habitats of the qualifying species or (ii) significant disturbance to the qualifying species, thus ensuring that the integrity of the site is maintained and the site makes an appropriate contribution to achieving favourable conservation status for each of the qualifying features;

and to ensure for the qualifying species that the following are maintained in the long term:

(iii) Population of the species as a viable component of the site,
(iv) Distribution of the species within site,
(v) Distribution and extent of habitat supporting the species,
(vi) Structure, function and supporting processes of habitats supporting the species,
repeat of (ii) No significant disturbance of the species

8. Otters designated as qualifying interests of the Yell Sound Coast SAC are unlikely to have connectivity with the proposal due to the distances and depths involved. Consequently, we advise that there is no likely significant effect upon otters and no further consideration of this species is required within HRA.

9. This distance separating the proposed development site and Yell Sound Coast SAC is well within the foraging range of harbour seals and we therefore advise that there is a **likely significant effect upon harbour seals as a qualifying feature of the Yell Sound Coast SAC**. As a consequence, Marine Scotland and SIC, as the competent authorities, are required to carry out appropriate assessments (AA) in view of the site's conservation objectives for this qualifying interest. Impacts upon harbour seals are of particular concern due to declining populations, including a condition status of 'unfavourable declining' for the Yell Sound Coast SAC. **We provide an appraisal of proposal below, in relation to seals as a qualifying feature of the SAC.**

Step 3: Can it be ascertained that the proposal will not adversely affect the integrity of the SPAs either alone or in combination with other plans or projects?

10. Potential sources of impact upon harbour seals are discussed in turn:

<u>Potential disturbance and displacement of seals:</u>

11. The use of gravity-bases, as opposed to rock-drilling, greatly reduces the potential for disturbance by limiting the sources of anthropogenic noise and allowing more rapid deployment of devices. The relatively small size of devices and the vessels therefore required for deployment and maintenance works also limit the potential for disturbance. In addition, the construction programme involves the deployment of devices spaced over an extended period of time, further limiting the potential for any sustained source of disturbance. Overall, we advise that potential disturbance of harbour seals is not of a scale or severity that would lead to an adverse effect on site integrity.

<u>Potential collision with operational tidal turbines:</u>

12. Table 1 below contains the collision risk estimates from the updated ERM model with a 98% avoidance rate applied for the harbour seal qualifying interest from Yell Sound Coast SAC for which LSE was previously identified.

Table 1: Collision risk estimates for the harbour seal qualifying interest of Yell Sound Coast SAC

Species	Updated ERM model with updated turbine parameters – BREEDING SEASON (June to August)	Updated ERM model with updated turbine parameters – ALL YEAR	Updated ERM model with updated turbine parameters – Seals-at-sea density (availability accounted for)
Harbour seal	0.17	3.96	4.00

13. The rate of collision predicted from the updated modelling during the breeding season is very similar to what was previously calculated (0-1 seal per year) and the current PBR for harbour seals for the Shetland Seal Management Unit [1] is 20 individuals. Furthermore, we are mindful of the underwater camera monitoring undertaken for the deployed turbines in Bluemull Sound for which no collision or near misses were detected during operational periods to date, and the ongoing commitment by Nova Innovation through their EMMP for further collision risk monitoring together with the emergency shutdown protocol in the event of any collision.

14. Through consideration of the above points, our advice is that there will be **no adverse effect on the integrity of the Yell Sound Coast SAC** according to its conservation objectives.

15. Cumulative / in-combination assessment: We advise that, based on our appraisal of this proposal and our knowledge of other developments/activities in Shetland, any potential cumulative and in combination effects will not adversely affect the integrity of this SAC.

[1] http://www.gov.scot/Topics/marine/Licensing/SealLicensing

APPENDIX B

NOVA INNOVATION TIDAL ARRAY, BLUEMULL SOUND, SHETLAND

HABITATS REGULATIONS APPRAISAL (HRA) – SPECIAL PROTECTION AREA (SPA)

See Appendix A for information on the HRA process and role of the competent authority.

SPAs relevant under this HRA can be determined by (a) species observed at the site during site survey, (b) the distance between SPAs and the proposed development site, (c) the foraging range and diving ability of birds designated as qualifying species and (d) the scale of the proposal. Relevant SPAs for consideration under HRA are Hermaness, Saxa Vord and Valla Field SPA and Bluemull and Colgrave Sounds proposed SPA (pSPA).

1. Hermaness, Saxa Vord and Valla Field SPA

1. Hermaness, Saxa Vord and Valla Field SPA is designated for a suite of breeding-bird interests. The proposal is approximately 3km from the nearest part of the SPA.

Step 1:	Is the proposal directly connected with or necessary for the conservation management of the SPA?

2. The proposal is not directly connected with or necessary for the conservation management of the Hermaness, Saxa Vord and Valla Field SPA.

Step 2:	Is the proposal likely to have a significant effect on the qualifying features of the SPA either alone or in combination with other plans or projects?

3. The conservation objectives for Hermaness, Saxa Vord and Valla Field SPA are:

> (i) to avoid deterioration of the habitats of the qualifying species or (ii) significant disturbance to the qualifying species, thus ensuring that the integrity of the site is maintained;
> and to ensure for the qualifying species that the following are maintained in the long term:
>
> (iii) Population of the species as a viable component of the site,
> (iv) Distribution of the species within site,
> (v) Distribution and extent of habitat supporting the species,
> (vi) Structure, function and supporting processes of habitats supporting the species,
> repeat of (ii) No significant disturbance of the species

4. Qualifying species for Hermaness, Saxa Vord and Valla Field SPA are as follows (*indicates assemblage qualifier only):
    a. Fulmar (*Fulmarus glacialis)**
    b. Gannet (*Morus bassana)*
    c. Great skua (*Catharacta skua)*
    d. Guillemot (*Uria aalge)**
    e. Kittiwake (*Rissa tridactyla*)*
    f. Puffin (*Fratercula arctica)*
    g. Red-throated diver (*Gavia stellata)*
    h. Shag (*Phalacrocorax aristotelis)**

i. Seabird assemblage

5. The conservation objectives for which consideration is required are (ii) and (iii) as listed above. The other objectives require no further consideration due to the distance between the SPA and the proposed development site and/or the small scale of the proposal.

6. Conservation objective (ii) is concerned with ensuring that there is no significant disturbance of species designated as qualifying interests of the site, such as through vessel activity or other cause of bird displacement. Although the proposed development would be within the foraging range of all of the above listed breeding populations for Hermaness, Saxa Vord and Valla Field SPA, we advise that there is no likely significant effect in this regard, due to the small scale of the development, the expected limited duration of installation procedures and the distance from nesting sites.

7. In this case, conservation objective (v) is relevant to the risk of collision between birds and operational turbines. Consequently it is only relevant to birds with diving capabilities that may place them at risk of interaction with the device. **We advise that there is a likely significant effect for gannets, puffins, red-throated divers, guillemots and shags from Hermaness, Saxa Vord and Valla Field SPA.**

2. <u>Bluemull and Colgrave Sounds pSPA</u>

9. Bluemull and Colgrave Sounds pSPA is designated for breeding red throated diver qualifying interests. The proposal is within the pSPA.

Step 1:	Is the proposal directly connected with or necessary for the conservation management of the pSPA?

10. The proposal is not directly connected with or necessary for the conservation management of Bluemull and Colgrave Sounds pSPA.

Step 2:	Is the proposal likely to have a significant effect on the qualifying features of the pSPA either alone or in combination with other plans or projects?

11. The draft conservation objectives for Bluemull and Colgrave Sounds pSPA are:

To avoid deterioration of the habitats of the qualifying species or significant disturbance to the qualifying species, subject to natural change, thus ensuring that the integrity of the site is maintained in the long-term and it continues to make an appropriate contribution to achieving the aims of the Birds Directive for each of the qualifying species.

This contribution will be achieved through delivering the following objectives for each of the site's qualifying features:

a) Avoid significant mortality, injury and disturbance of the qualifying features, so that the distribution of the species and ability to use the site are maintained in the long-term;
b) To maintain the habitats and food resources of the qualifying features in favourable condition.

12. Conservation objective a) is concerned with ensuring that there is no significant mortality, injury and disturbance of species designated as qualifying interests of the site, such as through collisions, vessel activity or other cause of bird displacement. In this case, there is risk of collision between red-throated divers and operational turbines that may place them at risk of interaction with the device.

13. For conservation objective b) although the proposed development would be within the foraging range of the breeding population for Bluemull and Colgrave Sounds pSPA, we advise that there is no likely significant effect in this regard, due to the small scale of the development relative to available habitats and food resources within the pSPA.

14. We advise that due to the risk of collisions with operational turbines **there is a likely significant effect for breeding red-throated from Bluemull and Colgrave Sounds pSPA.**

15. As a consequence, Marine Scotland and SIC, as the competent authorities, are required to carry out appropriate assessments (AA) in view of the site's conservation objectives for breeding red-throated diver. **We provide an appraisal of potential collision impacts below.**

Step 3: Can it be ascertained that the proposal will not adversely affect the integrity of the SPAs either alone or in combination with other plans or projects?

Collision risk assessment

16. Collision risk impacts for the following European sites and their qualifying interests for which a Likely Significant Effect (LSE) was previously identified are outlined below (Table 1).

Table 1: Bird interests and sites for which LSE is identified with respect to collision risk

European site	Qualifying interest(s)
Hermaness, Saxa Vord and Valla Field SPA	Atlantic puffin
	Red-throated diver
	Northern gannet
	Common guillemot
	European shag
Bluemull and Colgrave Sounds pSPA	Red-throated diver

17. We consider that our original advice still remains relevant (letter dated 24 June 2013), except where it has been updated with respect to the project-specific Environmental Mitigation and Monitoring Plan (EMMP). We refer you to our most recent detailed advice relating to the EMMP (letter of 15 August 2017).

1. Hermaness, Saxa Vord and Valla Field SPA
Bluemull and Colgrave Sounds pSPA

18. Table 2 below contains the collision risk estimates from the updated ERM model with a 98% avoidance rate applied for the SPA / pSPA breeding bird species for which LSE is identified. We have manually extracted the monthly densities for gannet and shag in order to be able to calculate the breeding season more accurately. We have not undertaken this for puffin, red-throated diver or common guillemot, and so the breeding season for these three species is taken as March to October, reflecting the way in which the survey data was presented to us.

Table 2: Collision risk estimates for SPA qualifying interests

Species	**Updated ERM model with updated 6 turbine parameter – BREEDING SEASON**	**Updated ERM model with updated 6 turbine parameter – ALL YEAR**
Atlantic puffin	1.45	1.36
Red-throated diver	0.13	0.15
Northern gannet	0.00	0.00
Common guillemot	0.37	0.36
European shag	4.87	11.25

19. For all of the above mentioned species, the collision risk estimates (using a 98% avoidance rate) are of a magnitude similar to previous predictions. **We are therefore content that these collision rates will not lead to an adverse effect on site integrity for Hermaness, Saxa Vord and Valla Field SPA and Bluemull and Colgrave Sounds pSPA.**

20. Cumulative / in-combination assessment: We advise that based on our appraisal of this proposal and our knowledge of other developments/activities in Shetland, any potential cumulative and in-combination effects will not adversely affect the integrity of these SPAs/pSPAs.

APPENDIX C

NOVA INNOVATION TIDAL ARRAY, BLUEMULL SOUND, SHETLAND

ADVICE ON NATURAL HERITAGE INTERESTS

Below we provide advice on the following natural heritage interests:

- Protected species
- Marine Protected Area qualifying interests

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1. <u>Protected species</u>

European Protected Species (EPS)

European Protected Species (EPS) are species listed in Annex IV of the Habitats Directive and are afforded protection under The Conservation (Natural Habitats, &c.) Regulations 1994 (as amended) and the Conservation of Offshore Marine Habitats and Species Regulations 2017. Marine Scotland provides guidance on the protection of Marine European Protected Species from injury and disturbance for Scottish Inshore Waters[2].

<u>EPS – Cetaceans</u>

On 19 January 2018 we were consulted by Marine Scotland regarding an extension to the EPS licence (expiry date 23 January 2018) for the array. We advised that it would be appropriate to issue a short extension to the licence in advance of gaining a better understanding about the results of the monitoring, the potential for activities causing disturbance and also for any further deployment activity associated with the 6 turbine array.

Further information available from the documents submitted for the licence allows us to provide further advice with respect to EPS.

- Disturbance during construction

The construction programme involves the phased deployment of infrastructure and turbines: cable installation Q3, 2019; turbine 4 installation, Q3, 2019; turbines 5 and 6, Q2, 2020; reconfiguration of array relocating turbines 4,5 and 6, Q1, 2021.

There is the potential for disturbance from installation works such as cable and turbine installation and associated vessel movements. However, installation works will be temporary and noise levels are likely to be relatively low and unlikely to cause significant disturbance.

The use of gravity-bases, as opposed to rock-drilling, greatly reduces the potential for disturbance, by limiting the sources of anthropogenic noise and allowing more rapid deployment of devices. The relatively small size of turbines and the vessels therefore required for deployment works also limit the potential for disturbance.

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[2] http://www.gov.scot/Resource/0044/00446679.pdf

Overall, we advise that installation works including cable and turbine installation / relocation and associated vessel activities could potentially cause disturbance to cetaceans and **we advise that an EPS licence for all relevant cetacean species with respect to disturbance during construction is required**.

We conclude that for the reasons outlined above, it is unlikely that there will be any significant disturbance, and project **will not be detrimental to the maintenance of the populations of relevant cetacean species at a favourable conservation status in their natural range.**

We recommend good practice should be applied during all marine and coastal works by following the guidelines associated with the Scottish Marine Wildlife Watching Code (SMWCC)[3].

- Collision with operational turbines

There is a potential for injury and mortality due to collision risk with the operational turbines. Due to the current lack of monitoring data from operational tidal arrays, the behaviour of marine mammals around tidal turbines is uncertain and the collision risk estimated.

Table 3 below contains the collision risk estimates from the updated ERM model with a 98% avoidance rate applied for the other marine animal species found in Bluemull Sound. Due to the way the survey information has been supplied we have used an 'all year' density figure for all three species presented in the table, apart from grey seal where we have manually extracted the monthly densities to be able to calculate the breeding season more accurately.

Table 3: Collision risk estimates for marine mammals recorded in the Bluemull Sound

Species	**Updated ERM model with updated turbine parameter – BREEDING SEASON**	**Updated ERM model with updated turbine parameter – ALL YEAR**	**Updated ERM model with updated turbine parameters – SCANSII (Area J) (Availability accounted for)**
Grey seal	2.85	7.15	N/A
Harbour porpoise	N/A	2.20	1.74
Minke whale	N/A	0.16	1.06

We have considered the updated collision risk estimates against the population estimates for the relevant management units for harbour porpoise (1-2 per year from a population of 228, 000) and minke whale (0-1 per year from a population of 229,000).

The level of predicted collisions for cetaceans is low and **will not be detrimental to the maintenance of the populations of relevant cetacean species at a favourable conservation status in their natural range.**

[3] https://www.nature.scot/professional-advice/land-and-sea-management/managing-coasts-and-seas/scottish-marine-wildlife-watching-code

We advise that an EPS licence for the operational phase is <u>not</u> required, unless through monitoring the modelled predictions and reality indicated a need to consider this further.

The EMMP should include sufficient detail with respect to ongoing monitoring as agreed with Marine Scotland and SIC. This monitoring should focus on gathering data on the behaviour of marine mammals in close proximity to the tidal turbines. Monitoring results will allow review to inform our EPS advice for cetaceans with respect to future licensing requirements.

<u>EPS - Otters</u>

Otters are EPS commonly seen in various parts of Bluemull Sound. We provided advice in relation to otters for earlier licence applications (letter of 4 June 2013). This advice remains relevant for this application and as a result, we advise that there is no requirement for further EPS licensing considerations in relation to otters.

Basking sharks

Basking sharks receive protection through the Wildlife and Countryside Act 1981 (as amended, including the Wildlife and Natural Environment (Scotland) Act 2010), with licensing requirements similar to EPS.

Although there are no established population estimates for basking sharks, they are a very wide-ranging species. There has been only one basking shark observation for this development since monitoring began in 2010, from the land based or underwater monitoring. Consequently, the applicant will not require a basking shark licence to address potential disturbance during installation or operational collision risk. **We consider that the Shetland Tidal Array will not have a negative impact on the conservation status of basking sharks.**

Seals

Seals as a qualifying feature of SACs are addressed in Appendix A. However, there is potential for impact upon harbour seal and grey seal interests not connected with Natura sites. Seals are protected under the Marine (Scotland) Act 2010. Impacts upon harbour seals are of particular concern due to their declining status across UK waters. Potential impact types are discussed in turn below:

- Potential disturbance and displacement of seals:

For reasons described in relation to the HRA for harbour seals in Appendix A, for both harbour and grey seals not connected to Natura sites we advise that potential disturbance would not be of a scale or severity of particular concern. We advise on the need for monitoring to improve our knowledge and understanding as to whether any patterns in the distribution and behaviour of seals in Bluemull Sound varies concurrently with the presence and or operation of the turbines. Also, good-practice should be applied during all marine and coastal works by following the guidelines associated with the SMWCC.

- Potential collision with operational tidal turbines:

The outcome of collision risk modelling for harbour seals is detailed in Appendix A. As grey seals also frequently occur in Bluemull Sound, collision risk estimates for this species have been generated (Table 3). The collision risk estimate (between 3-7 animals per year) is within the PBR limits (239) for the Shetland Seal Management Unit. **We are therefore content that these rates of collision do not necessitate mitigation for wider seal interests, as previously advised.**

Black guillemots

Black guillemots are the most frequently occurring non-SPA bird species recorded at the development site. The species is also a feature of the nearby Fetlar to Haroldswick nature conservation Marine Protected Area (NC MPA).

- Potential disturbance and displacement:

Due to the small scale of the development compared to the availability of suitable foraging and loafing habitat for black guillemots, disturbance away from the proposed development site is unlikely to be important at the population level.

- Potential collision with operational tidal turbines:

Table 4 below contains the collision risk estimates from the updated ERM model with a 98% avoidance rate applied for black guillemot.

Table 4: Collision risk estimates for black guillemot

Species	**Updated ERM model with updated turbine parameter – BREEDING SEASON**	**Updated ERM model with updated turbine parameter – ALL YEAR**
Black guillemot	16.27	27.72

Using Seabird 2000 and other recent counts, we consider that Shetland has a regional population of 15,329 black guillemots. If the higher number of predicted collisions is used (28), then this would equate to a small percentage - 0.2% of the population each year. **We consider that the Shetland Tidal Array will not have a negative impact on the conservation status of black guillemot.**

Appendix 2 - Scottish Natural Heritage (2018) Avoidance rates for the onshore SNH Wind Farm Collision Risk Model.

Scottish Natural Heritage

Avoidance Rates for the onshore SNH Wind Farm Collision Risk Model

September 2018 v2

1. Purpose

The potential for birds to collide with turbines is one of the main impacts to consider in the development of a wind farm. The SNH Collision Risk Model (CRM) (also known as the Band model (Band *et al.,* 2007; SNH, 2000)) provides a method based on vantage point data to estimate the number of birds likely to collide with turbines at a proposed wind farm. This allows pre-construction assessment of collision impacts on local and national populations. As birds may avoid a wind farm (for example some may be displaced from the area, while others may avoid turbines or take other evasive action to prevent a collision), the CRM accounts for this by applying an **avoidance rate**. This guidance provides recommended avoidance rates for a number of key species for use in the SNH Collision Risk Model.

This document is intended for anyone involved with collision risk modelling for birds at onshore wind farms (e.g. developers and their ecological consultants, SNH staff and those within consenting authorities). It is updated when robust new information becomes available. This document replaces previous versions (SNH, 2010, 2016) and includes an update for red kite.

2. Recommended avoidance rates

Table 1 presents the current recommended avoidance rates for key species, with links to supporting evidence for these. For species not listed in Table 1, we recommend a default value of 98%.

Table 1: Recommended avoidance rates for use with the onshore SNH Collision Risk Model for key bird species commonly identified in wind farm environmental statements for which supporting evidence is available. The recommended default avoidance rate for species not listed is 98%.

Species	Recommended avoidance rate	Rationale/supporting evidence
Red-throated diver	99.5%	Furness (2015)
Black-throated diver	99.5%	Breeding birds show similar behaviour to red-throated diver; Furness (2015)
Swans (all species)	99.5%	Increased from previous rate of 98% based on evidence presented in Whitfield & Urquhart (2015), but slightly more precautionary than report recommendation given this was based on a short run of data from one study
Geese (all species)	99.8%	SNH (2013)
Red kite	99%	Urquhart & Whitfield (2016)
Hen harrier	99%	Whitfield & Madders (2006a)
Golden eagle	99%	Whitfield (2009)
White-tailed eagle	95%	Sufficient evidence from flight behaviour and collision monitoring studies in Norway for vulnerability to collisions; see May *et al.* (2011)
Kestrel	95%	Sufficient evidence from flight behaviour (including hovering) and collision monitoring studies for vulnerability to collisions; see Whitfield & Madders (2006b)
Great skua	99.5%	Furness (2015)
Arctic skua	99.5%	Similar behaviour of breeding birds to great skua; Furness (2015)

3. Contact:

If you have any comments or queries about this guidance, please contact Dr Jessica Shaw at the SNH office at Stilligarry, South Uist, HS8 5RS, or email: jessica.shaw@nature.scot

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Appendix 3 - Joint response from the Statutory Nature Conservation Bodies to the Marine Scotland Science avoidance rate review (2014)

Joint Response from the Statutory Nature Conservation Bodies to the Marine Scotland Science Avoidance Rate Review

25th November 2014

1. Summary of recommendations

This joint response from the Statutory Nature Conservation Bodies (SNCBs)[1] is intended to provide recommendations on how the Offshore Wind Farm (OWF) industry could appropriately apply findings from the Marine Scotland Science Avoidance Rate Review[2] (hereafter 'the report') to the impact assessment process. This section provides a summary of our recommendations on best practise impact assessment using Collision Risk Modelling (CRM) in light of the report. The rationale for these recommendations is outlined within the main body of the paper.

Basic Band model (Options 1 and 2) recommendations

Whenever the Basic Band model (Options 1 or 2) are used for collision mortality estimation:

- Collision mortality estimates should be presented using the mean total avoidance rate (as detailed in Table 1 below) as well as a range of avoidance rates that reflects the variability and uncertainty linked to it (i.e. ±2SD).

Basic Band model (Option 2) recommendations

Whenever the Basic Band model (Option 2) is used for collision mortality estimation:

- Collision mortality estimates should be presented using the mean total avoidance rate (as detailed in Table 1 below) as well as a range of avoidance rates that reflects the variability and uncertainty linked to it (i.e. ±2SD).

Furthermore, the following information should also be provided:

- Presentation and comparison of both site-specific and generic flight height data (including median and confidence limits).
- A range of collision mortality estimates using the lower and upper confidence limits of the generic modelled flight distribution.

Extended Band model (Option 3) recommendations

It is **not** **appropriate** to use the **Extended Band model** in predicting collisions for **northern gannet** or **black-legged kittiwake**, at the current time.

Whenever the Extended Band model (Option 3) is used for **large gull** collision mortality estimation:

- Collision mortality estimates should be presented using the mean total avoidance rate (as detailed in Table 2 below) as well as a range of avoidance rates that reflects the variability and uncertainty linked to it (i.e. ±2SD).

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[1] To be read as comprising the Joint Nature Conservation Committee (JNCC), Natural England (NE), Natural Resource Wales (NRW), Northern Ireland Environment Agency (NIEA), Scottish Natural Heritage (SNH).

[2] Cook, A.S.C.P., Humphries, E.M., Masden, E.A., and Burton, N.H.K. 2014. The avoidance rates of collision between birds and offshore turbines. BTO research Report No 656 to Marine Scotland Science.

Furthermore, the following information should also be provided:

- Presentation and comparison of both site-specific and generic flight height data (including median and upper and lower confidence limits).
- Presentation of both Basic Band model outputs (Options 1 and 2) with the measures of confidence outlined in **Section 3.4**, in addition to Extended Band model outputs.
- A range of Extended Band model collision mortality estimates using lower and upper confidence limits of the generic flight distribution.

2. Introduction

The SNCBs welcome this important piece of work and congratulate Marine Scotland Science (MSS) for taking the initiative to commission this report and the British Trust for Ornithology (BTO) for conducting such a thorough review.

We note that a key finding of the report is the absence of studies of collision mortality and avoidance rates at offshore wind farms. The report concludes that the bulk of avoidance rate studies are from onshore or coastal wind farms. Having reviewed this body of work the report concludes that for many species (or groups of species) there are insufficient empirical data to derive meaningful avoidance rates at micro-, meso- or macro-scales. To a large degree, this inability to quantify these separate components of overall avoidance rates was due to lack of spatial resolution in empirical data and/or technical capacity to separate these components of overall avoidance.

The lack of empirical data from offshore wind farms contributing to the report's conclusions must be considered in the future applicability of recommended avoidance rates in an offshore context. Nevertheless, with many offshore projects at critical junctures in the decision-making process, we support some of the report's findings for use in offshore wind farm collision risk modelling, until such time as more empirical data are available.

This joint SNCB position represents a considerable shift in advice on avoidance rates for use with collision risk modelling in light of the report. This reflects the obligation on SNCBs to amend their advice as the best available evidence continues to evolve. However, it must be recognised that further empirical data on bird avoidance, flight heights and activity at offshore wind farms will continue to accrue and may alter our understanding of the likelihood of seabird collisions in the future. Therefore, the SNCBs position on avoidance rates may, as the current response bears testimony, be subject to change as more empirical data become available, e.g. ORJIP study (refer to section 6).

The following advice is applicable only to collision risk modelling for the five priority species and other gull species covered by the report. For other seabirds (e.g. skuas) and waterbirds (e.g. divers, seaducks, etc.) the report does not conduct an analysis or provide recommended avoidance rates for any version of the Band model. In light of this, the SNCBs continue to recommend the basic Band model, in conjunction with a default 98% avoidance rate, for predicting collisions of species other than those detailed here, until such time as further species-specific work has been undertaken.

3. General Statements of Agreement

3.1 Avoidance rates for use with the Basic Band model

The SNCBs support the recommended avoidance rates (AR) presented in the report in relation to four of the five priority species (the exception being black-legged kittiwake) as we consider these rates to be the best available evidence regarding the average avoidance rates for use with these species (Table 1 below). However, it should be noted that in several instances these are not derived from species-specific information and as such represent avoidance rates for species groupings (e.g. 'large gulls') rather than for an individual species.

The SNCBs also recommend that the estimated variance in empirically derived estimates of within windfarm avoidance rates, as presented within the report, be acknowledged and explored in any application of these total avoidance rates in future collision risk modelling.

Collision mortality estimates should be presented using the mean total avoidance rate as well as a range of avoidance rates that reflects the variability and uncertainty linked to it (i.e. ±2SD).

Table 1. Basic Band avoidance rates derived from MSS avoidance rate report Table 7.2. This table represents new avoidance rates (± 2SD) supported by the SNCBs for use in impact assessment collision risk modelling.

Species (rate used)	Basic Band model avoidance rate (2SD)
Northern gannet (all gull avoidance rate)	**0.989** (± 0.002)
Black-legged kittiwake (~~small~~ all gull avoidance rate)	~~0.992~~ **0.989** (± 0.002)*
Lesser black-backed gull (large gull avoidance rate)	**0.995** (± 0.001)
Herring gull (species-specific avoidance rate)	**0.995** (± 0.001)
Great black-backed gull (large gull avoidance rate)	**0.995** (± 0.001)

** Note: 'strike-through' data as presented in Table 7.2 of the report; data in 'bold' as recommended by SNCBs (see section 4.1 below for further explanation).*

3.2 Northern Gannet avoidance rates for Basic Band model

We note that the northern gannet avoidance rate represents, in reality, an 'all gull' avoidance rate, due to the absence of species-specific within windfarm avoidance data. We agree it is inappropriate to combine a within wind farm avoidance rate for this species based on the rates established for gulls with the gannet-specific macro-avoidance rate of 0.64, as this would result in a non-evidence based total avoidance rate higher than for any of the other groups considered. However, we agree that, without a within windfarm avoidance component for gannets, and acknowledging their more marked tendency to exhibit macro-avoidance behaviour; it is reasonable to ascribe to gannets the lowest of the total avoidance rates determined for any of the other groups (i.e. the 'all gull' category). In the absence of gannet-specific data for all elements of avoidance, this is also appropriately precautionary.

3.3 Use of avoidance rates to 3 decimal places

The SNCBs advise that, following recommendations in the report, practitioners of collision risk modelling now use avoidance rates to three decimal places as outlined above rather than rounding figures to two as typically done previously (e.g. 0.98). The report presents within windfarm avoidance rates to 4 decimal places (Table 7.1) but given the inherent uncertainty in the data the final recommended total avoidance rates are presented to only 3 decimal places (Table 7.2). The SNCBs agree with the recommendation in the report to use avoidance rates to three decimal places, until such time as improvements are made to the characterisation of uncertainty within the models, avoidance rates and flight height distributions used.

3.4 Recommended avoidance rates for use with Band model Option 2

We acknowledge that Options 1 and 2 of the Band model are mathematically identical (the Basic Band model) and consequently that it is appropriate to use the same predictive avoidance rate for both options. But the estimates of avoidance rates within section 5.4 of the report derived using Option 2 were in every case lower than using Option 1.

The SNCBs accept that this reflects the mismatch between the observed site-specific values of the proportion of birds recorded flying at predicted collision risk height (PCH) and the equivalent values derived using generic modelled flight height distribution data, and hence that the lower avoidance rates derived under Option 2 are anomalous.

We accept the recommendation that the higher avoidance rates derived using Option 1 should be used with the Basic Band model. For any future application of these recommended Basic Band model avoidance rates in combination with generic modelled flight height distribution data (i.e. use of Option 2), we advise the following is included:

- Presentation and comparison of both site-specific and generic flight height data (including median and confidence limits).
- A range of collision mortality estimates using the lower and upper confidence limits of the generic modelled flight distribution.
- A range of collision mortality estimates reflecting the empirically derived range of uncertainty around the mean avoidance rate (as detailed in Table 1 above).

This is to ensure due consideration is given to the uncertainty surrounding the generic flight height distribution and its applicability to the wind farm in question and the uncertainty around the avoidance rate itself.

4. Areas of Disagreement or Uncertainty

4.1 Kittiwake avoidance rates for Basic Band model

The SNCBs consider that the principles applied to northern gannet avoidance rate recommendations in the face of lack of species-specific data (i.e. application of the lowest “all gull” alternative rate derived by the review) should also be applied to black-legged kittiwake avoidance rates. The report includes kittiwake within the ‘small gull’ category, the data for which are predominantly derived from common gulls and black-headed gulls. Indeed, no species-specific data for kittiwakes are represented within the ‘small gull’ category at all.

While the report provides a theoretical argument towards the inclusion of kittiwakes within the ‘small gull’ category, there are equally arguments that could be put forward in support of their treatment as part of the ‘large gull’ category (i.e. typical flight speeds and generally more marine behaviour). Consequently, we feel these somewhat subjective arguments should be discounted in favour of a more consistent and precautionary approach with regards the treatment of other species lacking species-specific within windfarm avoidance rate data (namely gannets).

Therefore, we recommend that, until such time as it is possible to calculate a species-specific avoidance rate for kittiwakes, they are classed under the more generic (and precautionary) ‘all gull’ category.

4.2 Applicability of Extended Band model avoidance rates

The SNCBs highlight that the report makes no recommendation regarding avoidances rates for use with the Extended Band model for northern gannets and black-legged kittiwakes due to a lack of species-specific data.

This means it is not appropriate to use the Extended Band model in predicting collision figures for these species at the current time.

For the other three priority species covered by the report (see Table 2 below), we note that while we accept the work undertaken to derive avoidance rates for use with Option 3; we remain concerned over the use of the Extended Band model. In particular, we have concerns regarding its sensitivity to flight height distribution data, and the uncertainty this component introduces to variation in estimates of collision.

Table 2. Extended Band avoidance rates taken from MSS avoidance rate report Table 7.2.

Species (rate used)	Extended Band model avoidance rate (2SD)
Northern gannet	Not available
Black-legged kittiwake	Not available
Lesser black-backed gull (large gull avoidance rate)	**0.989** (± 0.002)
Herring gull (species-specific avoidance rate)	**0.990** (± 0.002)
Great black-backed gull (large gull avoidance rate)	**0.989** (± 0.002)

We advise those wishing to present Extended Band model predictions for those species/groupings where sufficient data on appropriate avoidance rates has been compiled within the report (i.e. those noted in Table 2 above), that the following information must also be provided:

- Presentation and comparison of both site-specific and generic flight height data (including median and upper and lower confidence limits).
- Presentation of both Basic Band model outputs (Options 1 and 2) with the measures of confidence outlined in **Section 3.4**, in addition to Extended Band model outputs.
- A range of collision mortality estimates reflecting the empirically derived range of uncertainty around the mean avoidance rate applicable to the output of the extended Band model (as detailed in Table 2 above).
- A range of Extended Band model collision mortality estimates using lower and upper confidence limits of the generic flight distribution.

Presentation of uncertainty around both flight heights and avoidance rates and incorporation into the analysis in this way, will provide clarity over the range of possible collision mortality outcomes and which collision risk model outputs are most appropriate for the assessment of the wind farm(s) in question.

5. Further Detailed Explanation of SNCB Positioning

5.1 Constraints on the wider applicability of Extended Band model avoidance rates

The report highlights, in many instances, significant differences between the observed proportion of birds at PCH (within the studies used to derive the avoidance rate estimates) and the proportion predicted to be at collision risk height derived from generic modelled distributions of flight heights. The latter estimates are almost invariably lower than the former.

In the case of 'small gulls' this discrepancy is so great that the report concludes it would be inappropriate to use avoidance rates derived for the Extended Band model for this group. Similar discrepancies, although less marked, also occur in the case of 'all gulls', 'large gulls' and 'herring gull'. Therefore, while accepting that the greater discrepancy in the case of 'small gulls' is such that the resultant extended model avoidance rate for that group (0.9027) and for 'all gulls' (which includes small gulls) (0.9672) are so unreliable as to be of no practical use, we can accept the use of the Extended Band model ARs derived for herring gulls and the other two larger species of gulls (Table 2 above), provided this is accompanied by acknowledgement of uncertainty around the underlying flight height data, and provided that equivalent Basic Band model AR outputs are presented for consideration alongside those from the extended Band model.

5.2 Need for on-going exploration of other aspects of uncertainty within the collision risk modelling framework

The SNCBs acknowledge that the Extended Band model is a more refined mathematical model than the Basic Band model in that it allows consideration of the fine-scale variation in the distribution of flight heights of birds flying within the rotor swept height band, and the variation as a function of height within that risk band in the probability of: i) passing within the perimeter of the rotating disc and ii) being hit during that passage. This Extended Band model is therefore a more advanced tool with which to derive estimates of the non-avoidance collision mortality.

However, the use of Option 3 in collision risk modelling is dependent upon; i) the availability of appropriate non-avoidance rates to apply to its non-avoidance estimate of collision mortality and ii) the degree of uncertainty around and confidence in the general applicability of the modelled flight height distribution on which it is based.

The report presents two pieces of evidence that highlight the significance of having robust estimates of the proportion of birds at PCH. These are:

- Deriving an Extended Band model AR for 'small gulls' was thwarted by the consistent mismatch between generic modelled flight height distributions and the observed proportion of birds flying at PCH in the empirical studies from which ARs were being derived. This may be because the empirical studies used within the report to derive ARs were all onshore, while the suite of studies used to model generic flight height distributions included more offshore data. In any event, this mismatch indicates extreme caution is needed when applying the generic flight height distribution required of the Extended Band model.

- The exploration of the sensitivity of the non-avoidance rate to variation in several key parameters indicates that the non-avoidance rate predicted by the Extended Band model can be highly sensitive to variation in the simulated flight height distribution. Although this appears not to be a consistent issue, it occurs sufficiently often to support the assertion above that extreme caution is needed in application of the generic flight height distributions to different sites.

Finally, there remains the issue of whether the derivation of collision mortality estimates using the Extended Band model is or is not more sensitive to errors in the attribution of birds to differing flight height bands in the field. Irrespective of the relative sensitivity of the Basic Band model and the Extended Band model in this respect, it is clear that errors in height estimation is another factor which needs to be considered in applying any estimate of flight height in collision risk modelling.

5.3 Issues limiting applicability of the correction factor g

The SNCBs note that the report shows in Annex 1 how the avoidance rates for use with the Basic and Extended Band models are related. The Basic model gives an estimate of no avoidance collision mortality, if information is available on turbine and bird parameters and the number of bird flights at risk height. The Extended model refines this estimate to take account of the distribution of flight heights, if detailed information on the latter is also available. The ratio between the Extended Band model estimate of collision rate and that from the Basic Band model, **if the same height distribution data are used in the latter to calculate the proportion of flights at collision risk height**, is termed the *g* factor.

Annex 1 of the report shows that if both models are applied to a reference windfarm, working back from an observed collision rate such as to derive the avoidance rate appropriate for each model, the non-avoidance rate for use with the Extended model **must be** *1/g* times the non-avoidance rate for the Basic model. This non-avoidance rate may then be used in estimating collision mortality at any new windfarm, using the Extended model, **if the flight height distribution at the new windfarm site is known**. **Thus to make use of the Extended model requires knowledge of the flight height distribution at both the reference site and at the new windfarm site**.

g factors have been estimated in Appendix 7 of the report, by comparison of the non-avoidance collision mortality estimates from the Basic and Extended Band models, both being based on assumed generic flight height distributions. It is clear, though, that there is a substantial mismatch between the observed values of the proportion of birds at collision risk height (PCH) and the proportion at risk height calculated from the generic modelled flight height distributions. The SNCBs consider it likely that estimates based on the latter are in many cases unreliable. The current review indicates that there is very little site-based information on the flight height distribution at the 'reference' windfarms reviewed, such as to enable *g* factors to be derived at each of these reference sites on the basis of site-specific data.

Until detailed flight height distributions are derived on a site-specific basis for a reference windfarm (or the applicability of a generic flight height distribution confirmed), the **SNCBs advise that the *g* factors presented in Appendix 7 should not be used to derive a windfarm avoidance rate for use with the Extended model at any new offshore windfarm**. In particular it would be wholly wrong to use avoidance rates appropriate for the Basic Band model, but based on observed values of the proportion of flights at risk, in conjunction with the *g* factors in Appendix 7 of the report which are calculated based on the generic flight distributions.

Where the report recommends use of avoidance rates for use with the Extended Band model, these are based on the assumed generic flight height distributions and hence may also be inaccurate. However, for these reference windfarms, the generic flight height distributions almost always predict a substantially smaller proportion of bird flights at risk height than have been observed in the site data. A correspondingly greater proportion of birds must be deemed not to take avoiding action in order to match the observed rate of collision at each reference windfarm. Hence the avoidance rates so calculated are precautionary, that is to say the true avoidance rates are most likely to be greater. **For those species for which they are quoted, the SNCBs accept the use of these avoidance rates with the Extended Band model (Table 2 above), subject to the qualifications set out in the report and presentation of the additional information as set out in section 4.2 above.**

The SNCBs acknowledge that as more detailed flight height information is acquired, it may prove possible to derive more reliable estimates of the non-avoidance rates for use with the Extended Band model, and the associated *g* factors. Nonetheless we advise that even then, any future application of the Extended Band model in collision risk mortality estimation should take account of the degree of uncertainty in all aspects of the underlying flight height data used, and present a range of possible outputs which reflect the degree of uncertainty around the assumed flight height distribution.

6. Next Steps

As outlined, this joint SNCB position reflects the obligation on SNCBs to amend their advice as the best available evidence continues to evolve. Consequently, this SNCB position statement will be subject to review as more empirical data become available (e.g. ORJIP study). Further to this, we advise that:

1. A review of this position statement will be undertaken by the SNCBs once ongoing work to quantify error and uncertainty in flight height distributions and collision risk modelling reports are completed. A NERC funded project, undertaken by Dr Liz Masden[3] , is expected to address some of these outstanding questions by spring 2015.

2. A strategic data collection programme should be drawn-up and agreed between all interested parties to supplement data collected under ORJIP. This should be aimed at gathering additional species-specific avoidance behaviour data (particularly for gannets and kittiwakes) to allow derivation of more refined avoidance rates than those recommended in the MSS report. Implementation of the programme should be overseen by regulatory bodies in recognition of their key role in the consenting process and formulation of licence conditions.

[3] Environmental Research Institute, University of Highlands and Islands, Thurso.

Appendix 4 - Cook, ASCP and Robinson, RA (2016) Testing sensitivity of metrics of seabird population response to offshore wind farm effects.

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Testing sensitivity of metrics of seabird population response to offshore wind farm effects

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The opinions expressed and statements made within this report are those of the BTO and do not necessarily represent those of the SNCBs and regulators. Please consult with the relevant SNCB and regulator as to the use of metrics in any impact assessment work.

Summary

Population models are often used to help understand the population level consequences of the impacts of offshore wind farms on seabirds. Metrics can be derived from these models in order to quantify the population level consequences of the impacts associated with the wind farm.

Based on a previous review of assessments of offshore wind farms, we identified 11 metrics which have been used to assess the population level consequences of impacts from offshore wind farms on seabirds. However, seabird demography is often not quantified accurately and may be subject to significant levels of uncertainty. This leads to concerns that these metrics may be sensitive to assumptions about population trend, demographic parameters and density dependence. As a consequence, it is important to understand the extent to which conclusions based on these metrics may be influenced by the assumptions underpinning them. With this in mind, we tested the sensitivity of each metric to assumptions about population trend, life history strategy, mis-specification of demographic parameters, the incorporation of density dependence and whether the metric was derived from a stochastic or deterministic population model.

Our analysis revealed that all metrics were sensitive to at least some of the assumptions underpinning them, but that some were more sensitive than others. We describe these sensitivities, indicating how to use each metric most effectively.

Contents

1 Introduction

Offshore wind farms potentially have a number of negative effects on seabird populations. These include displacement from preferred foraging areas, the risk of collision with turbines and the wind farm acting as a barrier to migrating or commuting birds (Garthe & Huppop 2004; Drewitt & Langston 2006; Everaert & Stienen 2007; Petersen & Fox 2007; Masden *et al* 2009; Krijgsveld *et al* 2011; Vanermen *et al* 2013; Furness *et al* 2013). As part of the consenting process, it is necessary to understand what impact these effects are likely to have at a population level. In the UK, the potential for a proposed offshore wind farm development to affect seabird populations has previously been assessed using demographic models, for example Population Viability Analysis (PVA) or Potential Biological Removal (PBR) (Wade 1998; Maclean *et al* 2007; Dillingham & Fletcher 2008; WWT 2012; Bennet 2013).

These demographic models can be used either to compare the trajectory of the population over time with and without the development, or quantify the impact of the development within a risk-based framework (for example, the probability that the population declines). To date, a range of different metrics have been used to undertake such assessments including the ratio of the predicted population size with and without the development, the ratio of the growth rates for each population, the increase in the probability of recording a growth rate less than 1 (*i.e.* a population decline) and the increase in the probability of a population decreasing by a fixed amount (*e.g.* a 50% decline over 10 years) (e.g. Green 2014; Trinder 2014). However, there is likely to be significant uncertainty and variability amongst the input parameters used for the demographic models, leading to concerns that metrics may not always allow a clear understanding of the population level consequences associated with offshore wind farms (Green 2014; Cook & Robinson 2015). The importance of capturing uncertainty when estimating the effects associated with offshore wind farms has recently been highlighted (Masden *et al* 2014; Green 2014). Without capturing this uncertainty, it is difficult to know how likely any of the scenarios put forward are, potentially posing problems for the consenting process.

The quantity and quality of data available to inform demographic models for different species is highly variable (Maclean *et al* 2007; Horswill & Robinson 2015). For some species time-specific data will be available from particular colonies of interest for at least some parameters whereas, for other species even basic estimates may be lacking. In the absence of detailed demographic data, especially at a site-specific level, for the species under consideration, it is important to understand whether these metrics are unduly sensitive to mis-specification of demographic parameters in the models used to derive them. The impact of this sensitivity may depend on the specific circumstances under consideration. For example if strong evidence is available to support estimates of adult survival rates, but the evidence underpinning the immature survival rates is less robust, metrics which may be more sensitive to the estimate of adult survival but less sensitive to estimates of immature survival would be favoured. Based on the guidance given by the steering group we set out criteria against which sensitivity to different attributes could be assessed and which metrics should be favoured based on this assessment. While this report has looked at the sensitivity of metrics, it has not explicitly considered how that sensitivity is influenced by the value chosen for the rule derived for that metric.

2 Approach

2.1 Metrics used to quantify population level consequences of effects from offshore wind farms on seabirds

As part of the Strategic Ornithological Support Service (SOSS) work programme, WWT (2012) produced guidance on the use of Population Viability Analysis (PVA) to assess the population level consequences of effects from offshore wind farms on seabirds. Using a stochastic population model, the effect of the additional mortality arising as a result of birds colliding with turbines on northern gannet *Morus bassanus* populations is assessed by considering the probability of collision mortality causing populations to decline at colonies across the North Sea. Subsequently, a broad range of metrics, have been derived from population models with which to assess population level effects (Centrica Energy 2009; Trinder 2014; Green 2014; Cook & Robinson 2015).

As part of Cook and Robinson (2015), we reviewed Habitats Regulations Assessments (HRA) undertaken for offshore wind farms currently within the planning process. We identified eleven metrics which can be derived from population models and that have been used to assess the population level effects of impacts from offshore wind farms on seabirds. These can be split into two broad categories, probabilistic approaches (e.g. the probability of the population declining) or ratio approaches (e.g. the ratio of the population size in the presence and absence of the wind farm). However, many of these approaches have been criticised as, potentially, being sensitive to both uncertainties in the demographic parameters used in the underlying population models and to uncertainties in the magnitude of the impact predicted (Green 2014). Cook and Robinson (2015) assessed the validity of these criticisms and highlighted some areas where further analyses were required in order to draw firm conclusions about the usefulness of these approaches. The purpose of this report is to use population models, to quantify how sensitive conclusions drawn from each are 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.

Table 1. Description of metrics used to assess population responses to impacts of offshore wind farms and included in the sensitivity analysis and scale over which the metrics operate.

	Metric	Scale	Description
1	**Population growth rate** **(GR)** **Section 3.2.3**	A value of 1 indicates a stable population, a value <1 indicates a declining population and a value >1 indicates an increasing population.	By considering the growth rate of the population in the presence of an offshore wind farm, it will be possible to consider whether the population will remain stable, increase or decrease through the life time of the project. A value of 1 indicates a stable population, <1 a declining population and >1 an increasing population. Growth rate is calculated as a mean rate over the study period: $\left(\frac{End\ Population\ Size}{Start\ Population\ Size}\right)^{1/Nyears}$
2	**Ratio of median impacted to unimpacted growth rate** **(RI:U)** **Section 3.2.4**	From 0 – 1, with 1 indicating the impacted population growth rate is the same as the unimpacted growth rate (i.e. no population-level consequence) and values close to 0 indicating a large difference between the impacted and unimpacted population growth rates (i.e. a strong population-level consequence).	Considering only the growth rate of a population 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. By comparing the growth rate of the population in the presence of a wind farm to that expected in the absence of a wind farm it may be possible to demonstrate what impact the development is having on a population.

	Metric	Scale	Description
3	**Ratio of impacted to unimpacted population size** **(RI:U25)** **Section 3.2.5**	From 0 – 1, with 1 indicating the impacted population size is the same as the unimpacted growth rate (i.e. no population-level consequence) and values close to 0 indicating a large difference between the impacted and unimpacted population sizes (i.e. a strong population-level consequence).	Population models can be used to estimate the size of a population through time both with and without the impact of an offshore wind farm. Comparing the ratio of the size of these two populations offers a relatively easy to interpret statistic with which to assess the population level impact of an offshore wind farm. 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. 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.
4	**Probability that growth rate <1, 0.95, 0.8** **(P(GR<1))** **Section 3.2.6**	From 0 – 1 with 0 indicating that none of the simulations from a stochastic model result in a growth rate <1 and 1 indicating 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 (i.e. has a growth rate <1). The probability of a population declining is typically assessed over the lifetime of the project. However, it would also be possible to examine the probability of the population declining at any point during the lifetime of the project. Alternatively, 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.
5	**Change in probability that growth rate <1, 0.95, 0.8** **(dP(GR<1))** **Section 3.2.7**	From 0 – 1, with 0 indicating that there is no likely change in the probability of the growth rate being <1 between impacted and unimpacted populations (i.e. no population-level consequence) and values approaching 1 indicating there is an almost certain change in the probability of the growth rate being <1 between the impacted and unimpacted populations (i.e. a population-level consequence).	Where simulations show that a 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, simply quantifying the probability of a population decline in the presence of an offshore wind farm may not be meaningful. To assess the population level impact of a development it is therefore necessary 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 could be done either at a single fixed point in time, or at intervals throughout the life time of the project.

	Metric	Scale	Description
6	**Probability that population is below initial size at any point in time** **($P(p<p_0)$)** **Section 3.2.8**	From 0 -1 with 0 indicating that none of the simulations from a stochastic model result in a population below its initial size at any point in time and 1 indicating that all of the simulations from a stochastic model result in a population below its initial size at any point in time.	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.
7	**Probability of a 10, 25 or 50% population decline** **(P(ld>0.25))** **Section 3.2.9**	From 0 – 1, with 0 indicating that none of the simulations from a stochastic model show the impacted population declining by a given magnitude (i.e. no population-level consequence) and 1 indicating that all simulations show the impacted population declining by at least the given magnitude.	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.
8	**Change in probability of a 10, 25 or 50% decline** **(dP(ld>0.25))** **Section 3.2.10**	From 0 – 1, with 0 indicating that there is no change in the probability of the population decreasing by a given magnitude between the impacted and unimpacted populations (i.e. no population-level consequence) and values approaching 1 indicating there is a large change in the probability of the population decreasing by a given magnitude between the impacted and unimpacted populations (i.e. a population-level consequence).	At many colonies throughout the UK, seabird populations are already declining (JNCC 2013). As a consequence, the presence of a wind farm may not 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 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 10% increase in the probability of a 5% decline.

	Metric	Scale	Description
9	**Probability of a population being a given magnitude below the median size predicted in the absence of an impact** **P(I<25)** **Section 3.2.11**	From 0 – 1, with 0 indicating that none of the simulations from a stochastic model show the impacted population being a given magnitude below the unimpacted population (i.e. no population-level consequence) and 1 indicating that all simulations show the impacted population a given magnitude below the unimpacted population.	A metric to assess the population level impact of a development could be derived by estimating a median size for a population in the absence of 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.
10	**Probability that impacted population growth rate is 2.5, 5 or 10% less than unimpacted growth rate** **(P(IGR<2.5))** **Section 3.2.12**	From 0 – 1, with 0 indicating that none of the simulations from a stochastic model show the impacted population growth rate being a given magnitude below the unimpacted population (i.e. no population-level consequence) and 1 indicating that all simulations show the impacted population growth rate a given magnitude below the unimpacted population.	With growth rates simulated from stochastic models, it may be desirable to estimate a mean or median value for the unimpacted population and calculate the proportion of simulations in which the growth rate of the impacted population is lower, or a given percentage lower, than this value. This approach has the advantage of allowing a probabilistic forecast of the impact of the offshore wind farm on a population, e.g. there is a 50% chance that the wind farm will reduce the population growth rate by 10%.
11	**Overlap of Impacted and Unimpacted Populations** **(OI:U)** **Section 3.2.13**	From 0 – 1, with 0 indicating 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 unimpacted population and 1 indicating 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 unimpacted population.	Using stochastic models, the population size at a fixed point in time (i.e. at the end of a project lifetime) may be expressed as a distribution. In these circumstances, it may be desirable to compare the distributions of the impacted and unimpacted populations. Where there is greater overlap between the two populations, impacts may be deemed less significant.

2.2 Sensitivity Analysis

2.2.1 Baseline Scenario

Initially, each metric listed above was derived for a baseline scenario, against which sensitivity to mis-specification of demographic parameters, model structure and the magnitude of any impacts could be considered.

Table 2. Parameters used in population models for r-selected and K-selected seabird species with stable, increasing or decreasing population trends (± 1SD).

	R-selected			**K-selected**
	Increasing	**Stable**	**Decreasing**	**Stable**
Age at first breeding	4			9
Initial Population Size	10,000 Breeding adults			
Sex Ratio	0.5			
Adult Survival	0.89 (± 0.085)	0.89 (± 0.085)	0.866 (± 0.024)	0.953 (± 0.030)
Immature Survival	0.850 (± 0.200)	0.741 (± 0.200)	0.741 (± 0.206)	0.845 (± 0.150)
First year Survival	0.441 (± 0.200)	0.441 (± 0.200)	0.358 (± 0.219)	0.845 (± 0.150)
Productivity	1.590 (± 0.175)	1.030 (± 0.175)	0.920 (± 0.175)	0.540 (± 0.089)

Rather than consider any particular species, we focused on a generic “r-selected” species and a generic “K-selected” species, with suitable demographic rates informed by a recent review of seabird demography (Horswill & Robinson 2015). Whilst we acknowledge that, in general, seabirds are K-selected species, there is variation in life history across the group as a whole. For example some, such as terns, may be considered to be more r-selected (lower survival rates, higher productivity rates), whilst others, for example fulmar or gannet, may be considered more K-selected (higher survival rates, lower productivity rates). For simplicity throughout this report we refer to these two groups as r-selected and K-selected respectively. We based demographic parameters for the r-selected species on those presented for terns and parameters for the K-selected species on those presented for northern gannet and northern fulmar *Fulmarus glacialis*, these species reflecting opposing ends of the life history spectrum occupied by seabirds.

This baseline scenario considered a stochastic population model for a stable population of an r-selected seabird species with the demographic parameters given in Table 2 and informed by a recent review of demography (Horswill & Robinson 2015). Following the guidance given in Horswill and Robinson (2015), for the stochastic models we sampled demographic parameters from distributions based on the mean values given in Table 2 and bounded by one standard deviation. Each metric is derived using a “matched runs” approach, as specified in WWT (2012) and Green (2014), whereby stochasticity is applied to the population, but the survival and productivity rates used for the impacted and unimpacted populations at each time step are the same, prior to any impact from an offshore wind farm being applied.

Table 3. Sources of uncertainty considered as part of sensitivity analysis of metrics used to quantify population level impacts of offshore wind farms on birds.

Population Trajectory	R/K-selected species	Type of model	Parameter affected	Demographic parameter varied	Low (10%)/moderate (20%)/high (40%) impact
Stable	R-selected	Deterministic	Survival	Adult survival	Low
					Moderate
					High
				Immature Survival	Low
					Moderate
					High
				Chick Survival	Low
					Moderate
					High
				Productivity	Low
					Moderate
					High
			Productivity	Adult survival	Low
					Moderate
					High
				Immature Survival	Low
					Moderate
					High
				Chick Survival	Low
					Moderate
					High
				Productivity	Low
					Moderate
					High
		Stochastic	Survival	Adult survival	Low
					Moderate
					High
				Immature Survival	Low
					Moderate
					High
				Chick Survival	Low
					Moderate
					High
				Productivity	Low
					Moderate
					High
			Productivity	Adult survival	Low
					Moderate
					High
				Immature Survival	Low
					Moderate
					High
				Chick Survival	Low
					Moderate
					High
				Productivity	Low
					Moderate
					High
		Density dependent impact on	Survival	MaxF	Low
					Moderate
					High

Population Trajectory	R/K-selected species	Type of model	Parameter affected	Demographic parameter varied	Low (10%)/moderate (20%)/high (40%) impact
		survival		Shape parameter (b)	Low
					High
		Density dependent impact on Productivity	Survival	MaxF	Low
					Moderate
					High
				Shape parameter (b)	Low
					Moderate
					High
Increasing	R-selected	Stochastic	Survival	NA	High
					Moderate
					Low
			Productivity	NA	High
					Moderate
					Low
Decreasing	R-selected	Stochastic	Survival	NA	High
					Moderate
					Low
			Productivity	NA	High
					Moderate
					Low
Stable	K-selected	Stochastic	Survival	NA	High
					Moderate
					Low
			Productivity	NA	High
					Moderate
					Low

Following discussion with the project steering group, we derived metrics from stochastic population models run over a 25 year time period for 1,000 iterations. It was not feasible to test every conceivable combination of impact, impact magnitude, demographic parameter, population trend, life history strategy and model structure. We therefore selected 77 scenarios (Table 3) from which general inferences could be drawn about the relationship between each metric and parameters to which they may be sensitive.

2.2.2 Deterministic or stochastic models

In a deterministic population model, each of the demographic parameters is assumed to have a single value, which is constant over time, whilst in a stochastic model each parameter is drawn from a distribution, with a different value prevailing in each simulated year. It has been argued that where there is significant uncertainty surrounding demographic parameters and the magnitude of impacts predicted from offshore wind farms, using deterministic models may be a more "honest" approach than using stochastic models (WWT 2012). This is because the confidence intervals presented surrounding the outputs from a stochastic model may imply a level of precision that the underlying data do not justify.

Initial simulations by Green (2014) suggest that there is little difference between metrics derived from deterministic models and equivalent metrics derived from stochastic models. However, stochastic models are inherently more conservative than deterministic models as environmental stochasticity causes the long-run growth rate to be below the mean growth rate (Lande *et al* 2003). Given the preference of some authors for deterministic models as a result of the uncertainty associated with demographic parameters (WWT 2012; Green 2014) it is important to understand the extent to which they may be less conservative than stochastic models.

Deterministic models do not produce the distribution of potential results necessary to estimate the probabilistic metrics. Therefore, comparisons between deterministic and stochastic models are restricted to the ratio based metrics derived from a stable population of an r-selected species. For each model we consider a 10% or 20% increase in mortality or a 10% or 20% decrease in productivity.

2.2.3 Sensitivity to impact

Whilst collisions would be expected to affect seabird survival rates, displacement may affect either survival or productivity rates. For example, displacement from a preferred feeding area may mean birds are unable to meet their energy requirements, resulting in a reduction in the survival rate (e.g. Searle *et al* 2014). Alternatively, if birds are unable to meet their energy requirements, they may abandon breeding attempts in the year concerned, resulting in an overall reduction in productivity rates. As K-selected species, seabird populations are thought to be more resilient to impacts on productivity than impacts on, especially adult, survival. For this reason we modelled impacts on survival and productivity separately. Following discussion with the project steering group, we considered impacts arising as a result of the presence of an offshore wind farm of up to a 40% increase in the mortality rate or up to a 40% reduction in the productivity rate, in line with the magnitude of impacts predicted for some offshore wind farms. Note that, in contrast to previous work (e.g. Searle *et al* 2014) these figures relate to a percentage of the mortality rate, rather than survival rates. Also note that it considers relative, rather than absolute increases in mortality or decreases in productivity, i.e. if baseline mortality were 10%, a 40% increase in mortality would give a total mortality rate of 14 % (i.e. 10% + 40% of 10%), not 50% (i.e. 10% + 40%).

A reduction in the productivity rate of 40% would lead to a change from 1.030 chicks fledging per nest in the absence of the wind farm, to 0.618 chicks/nest fledging in the presence of the wind farm. Assuming a stable population of an r-selected species with a starting size of 10,000 breeding adults, the baseline mortality rate (i.e. mortality in the absence of an offshore wind farm) would be 0.11, or 1,100 adult birds per annum. A 40% increase in the mortality rate would lead to an additional 440 deaths per annum and a total annual mortality of 1540 adult birds per annum. Under this scenario the annual mortality rate would increase from 0.11 to 0.15 and the survival rate would reduce from 0.89 to 0.85 (see below for calculations).

Population size:	10,000 breeding adults
Annual mortality:	(1-0.89) * 10,000 = 1,100 adults/year
40% increase in mortality:	1,100 * 1.40 = 1,540 adults/year
New mortality rate:	1,540/10,000 = 0.15
New survival rate:	1 – 0.15 = 0.85

Increased mortality rates were calculated on an annual basis, as using a single value for additional mortality (i.e. adding the additional mortality predicted for year 1 to each subsequent time step) would mean that mortality would not vary in proportion with the population size at each time step. As a consequence, additional mortality will vary from year to year across each simulation. In reality, population-level impacts will be greater than the 440 birds per annum given in the above example as this figure only relates to breeding adults and not juvenile, sub-adult or non-breeding adult birds. However, this approach is consistent with the way in which Habitats Regulations Assessments (HRA) are currently applied. HRA considers the impact of a development on a designated population which, at present, typically relates to the number of breeding adults at a protected site.

These initial analyses were used to investigate how each metric varied in response to up to a 40% increase in mortality and up to a 40% reduction in productivity.

2.2.4 Sensitivity to population trend

A key criticism made of metrics used to assess the population level consequence of impacts from offshore wind farms is that they are sensitive to the population trends of the species concerned (Green 2014). This may mean that decisions about whether or not the impacts associated with a wind farm may be deemed acceptable may be based on the population trend prior to construction (i.e. whether it was increasing, decreasing or stable) rather than the magnitude of the population-level consequences associated with the wind farm.

In order to evaluate how each metric varied in response to current population status we adjusted the population models using biologically plausible values (Table 2) determined from the review of seabird demography by Horswill and Robinson (2015) in order to give increasing and declining trends. We compare the metrics derived from populations with increasing and declining trends with those derived from a stable population using stochastic models and assuming a 10% or 20% increase in mortality or decrease in productivity. Ideally metrics would have an identical value for a given magnitude of impact regardless of whether a population was increasing, stable or declining, prior to the assumed effect of the offshore wind farm.

2.2.5 Sensitivity to life history strategy

As described above, whilst seabirds are primarily K-selected species they cover a spectrum from species such as terns, with lower survival rates and higher productivity rates, to species like northern fulmar and northern gannet, with higher survival rates and lower productivity rates. Therefore, it is important to understand how the metrics may reflect differences in life history strategies.

To test this we compare the metrics derived using a stable r-selected species to those derived using a stable K-selected species with the demographic characteristics given in Table 2 and using stochastic models.

2.2.6 Sensitivity to mis-specification of demographic parameters

There is often significant uncertainty surrounding the demographic parameters used in the population models from which each metric is derived (Robinson & Horswill 2015). As a consequence, concerns have been raised that the metric may be highly sensitive to mis-specification of these parameters (Green 2014), and that small changes in the demographic rates considered may result in relatively large changes in the metric derived. Ideally, changes in the metric should primarily be driven by changes in the magnitude of the impact as opposed to mis-specification of the parameters used to derive the metric.
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 for both stochastic and deterministic models (where appropriate), assuming a stable population of an r-selected species and considering a 10% or 20% increase in mortality or decrease 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). Ideally, mis-specification of any of the demographic parameters should result in minimal change to the metric concerned and we assess this in relation to a subjective threshold of 1% of the parameter value.

2.2.7 Sensitivity to density dependence

Where seabird population sizes are reduced as a result of the effects from offshore wind farms, density dependent responses may partially compensate for any losses through increases in survival or productivity. In these circumstances, the inclusion of density dependence in any population models may lead to metrics which are less precautionary than those derived from population models which do not incorporate density dependence. Furthermore, whilst there is strong evidence for density dependent population responses in some seabirds (Horswill & Robinson 2015), there is considerable uncertainty about the form and strength of these relationships, as they will generally depend on processes acting at a site specific level. As a consequence, it is important to determine the extent to which the use of density independent models when deriving the metrics is a precautionary approach and, if density dependent models are to be used, the sensitivity of the metrics to assumptions about the form of that density dependence. Ideally, if density dependent models are to be used, metrics should be relatively insensitive to assumptions about its form.

In order to assess the impact of incorporating density dependence on the metrics we updated our population models for a stable population of an r-selected seabird species to incorporate density dependent regulation of productivity or survival. Using this approach, we estimated density dependent responses of productivity and survival to population size within a stochastic model. It was assumed that density dependent impacts on survival would apply only to adults at their breeding colonies where they are constrained as central place foragers (Thaxter *et al* 2012) and therefore less able to make use of alternative foraging habitats.

A variety of functions can be used to represent density dependent responses (Horswill & Robinson 2015), however, the Weibull function (Eq. 1) has been found to be a realistic representation across a variety of species (Cury *et al* 2011).

Equation 1. $D = maxD * \exp(-a * N^b)$

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. We consider the sensitivity of each metric to both *maxD* and *b*. The scale parameter, *a*, is estimated with reference to *b,* consequently it is inappropriate to consider sensitivity of the metrics to *a* in isolation. The relationship between the population size and productivity or survival for different values of the shape parameter and *maxD* are shown in Figures 1 and 2.

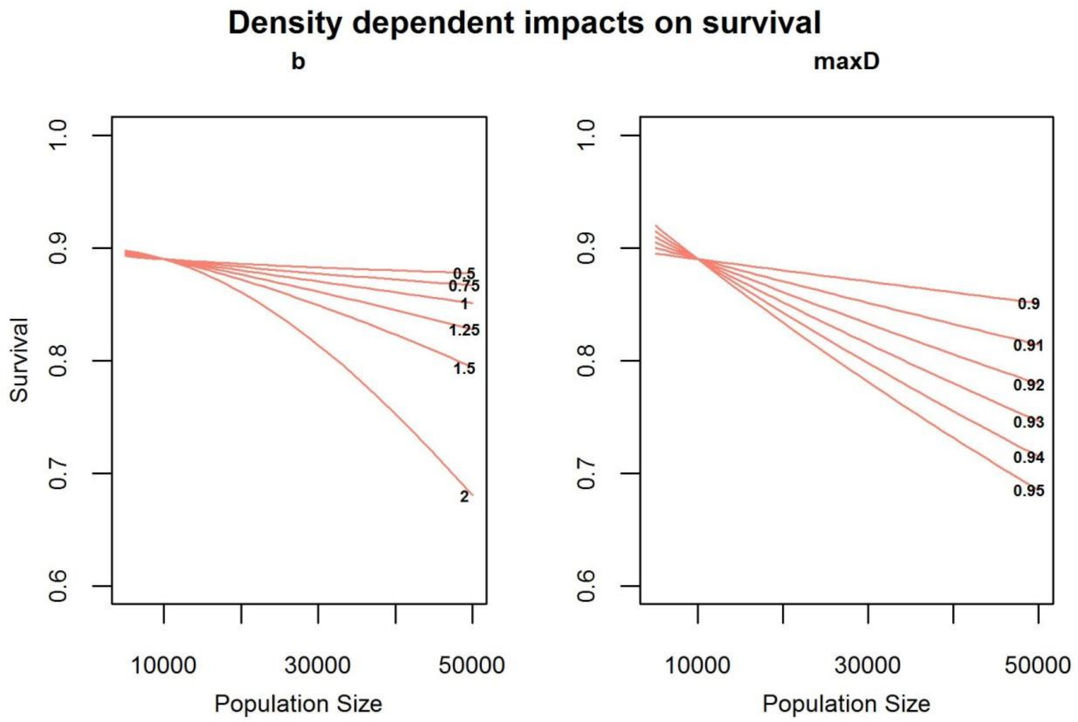

Figure 1. Density dependent relationship between survival and population size (number of breeding adults) for different values of the shape parameter, b (0.5, 0.75, 1, 1.25, 1.5, 2), assuming a maximum survival rate of 0.9 and the maximum survival rate (0.90, 0.91, 0.92, 0.93, 0.94, 0.95), assuming a value of 1 for the shape parameter.

Figure 1 shows how the adult survival rate changes for different values of *b* and *maxD*. This plot assumes a stable population with a carrying capacity of 10,000 breeding adults. Where the population drops below the carrying capacity, there are increases in the survival rate to compensate. Where the population rises above carrying capacity, the survival rate declines through mechanisms such as increased competition. The strength of the relationship between the survival rate and the population size is determined by both the shape parameter, *b*, and the maximum value permitted for survival, *maxD*. As the value assumed for the shape parameter *b* increases, the survival rate responds more quickly to changes in population size. Similarly, as the maximum value allowed for survival increases, so too does the strength of the relationship between population size and survival rate. Where the survival rate assumed in the population models is closer to the maximum allowable survival rate, there is less potential for it to increase and therefore less variability in relation to population size. Figure 2 shows how the population changes through time assuming density dependent regulation of survival with different values assumed for *b* and *maxD*. Over time, the population rises towards an asymptote, which reflects the carrying capacity of the population concerned. The shape parameter, *b*, determines the approximate size of the population at carrying capacity (and is therefore effectively the strength of the density dependent relationship) whilst the maximum allowable survival rate, *maxD*, influences variation around this population size.

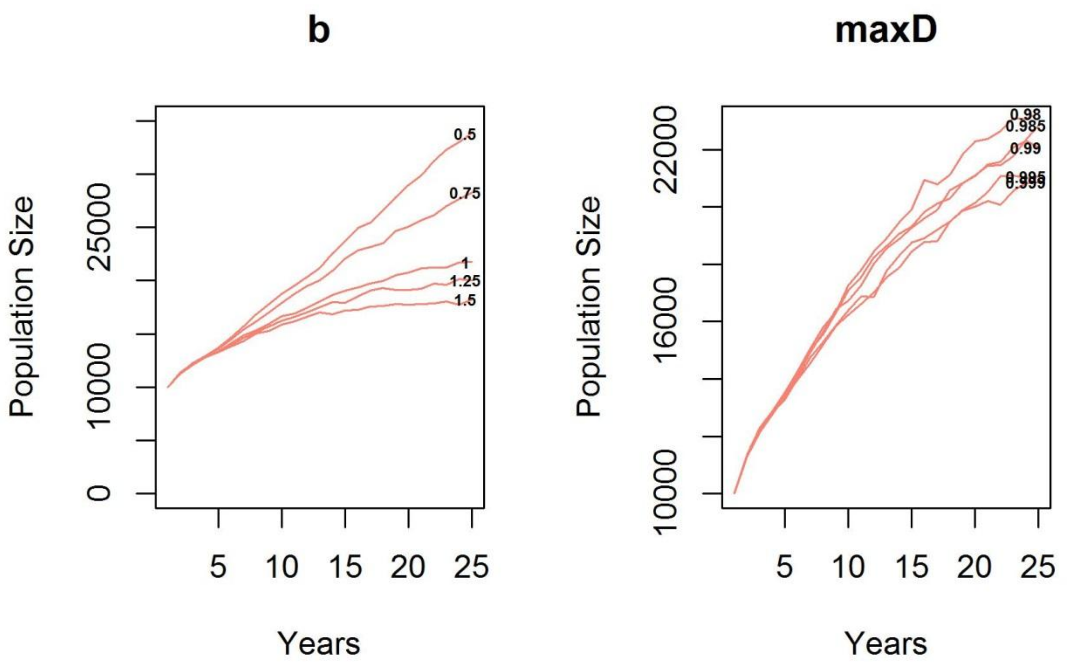

Figure 2. Influence of density dependent regulation of survival on population size through time. Models assume an increasing population based on the demographic parameters in Table 2. Left hand graph shows population trajectory for different values of the shape parameter *b* (0.5, 0.75, 1, 1.25 and 1.5) assuming a maximum survival rate of 0.98. Right hand graph shows population trajectory for different maximum survival rates (0.98, 0.985, 0.99, 0.995, 0.999) assuming a shape parameter of 1.

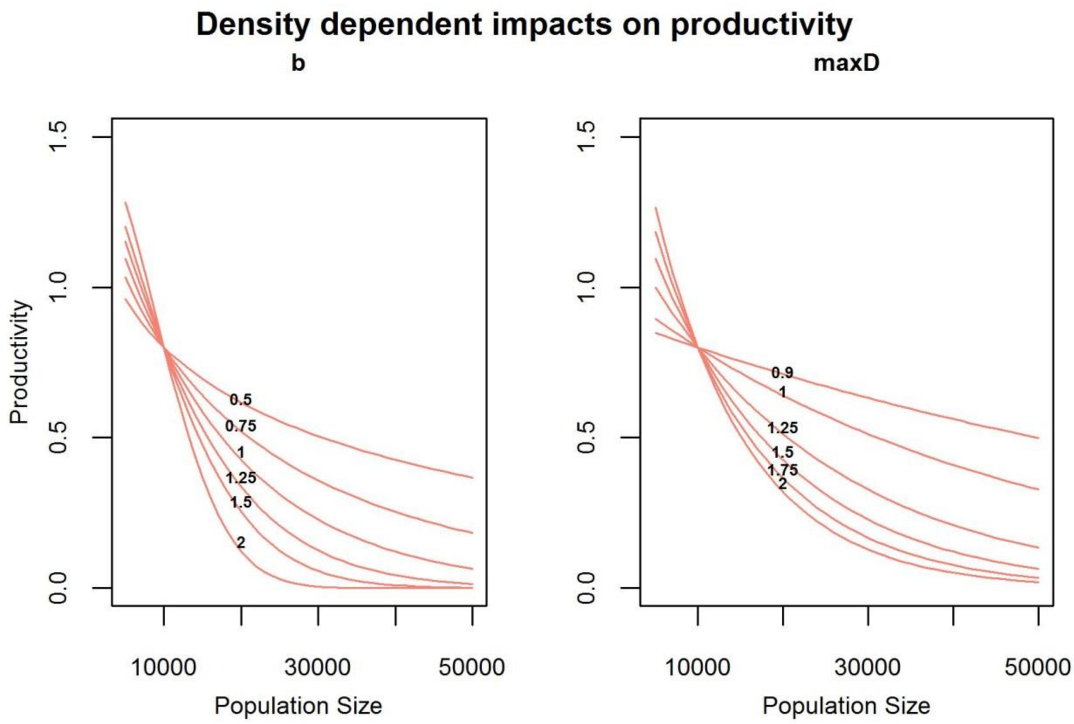

Figure 3. Density dependent relationship between productivity and population size for different values of the shape parameter, b (0.5, 0.75, 1, 1.25, 1.5, 2), assuming a maximum productivity rate of 1.5, and maximum productivity (0.9, 1 1.2, 1.3, 1.5, 1.75, 2), assuming a value of 1 for the shape parameter.

Figure 3 shows how the productivity rate varies in response to changes in population size. As with survival, where the population drops below its carrying capacity, there is an increase in productivity to compensate. Similarly, where the population size exceeds the carrying capacity, there is a drop in the productivity rate as a result of factors such as increased competition for food, nest sites or nest predation from con-specifics. However, it is noticeable that the density dependent relationship between productivity and population size is much more pronounced than that for survival (Fig. 1). This pattern is likely to reflect the fact that productivity rates are more variable than survival rates (Horswill and Robinson 2015). For higher values of *b* and *maxD*, as the population size rises above the carrying capacity, the productivity rate quickly declines to 0, suggesting that, in these cases, density dependence means large populations may fail to produce any chicks in a given year. Figure 4 shows how the population changes through time assuming density dependent regulation of productivity with different values assumed for *b* and *maxD*. Over time, the population rises towards an asymptote, which reflects the carrying capacity of the population concerned. The shape parameter, *b*, determines the approximate size of the population at carrying capacity whilst the maximum allowable productivity rate, *maxD*, influences variation around this population size.

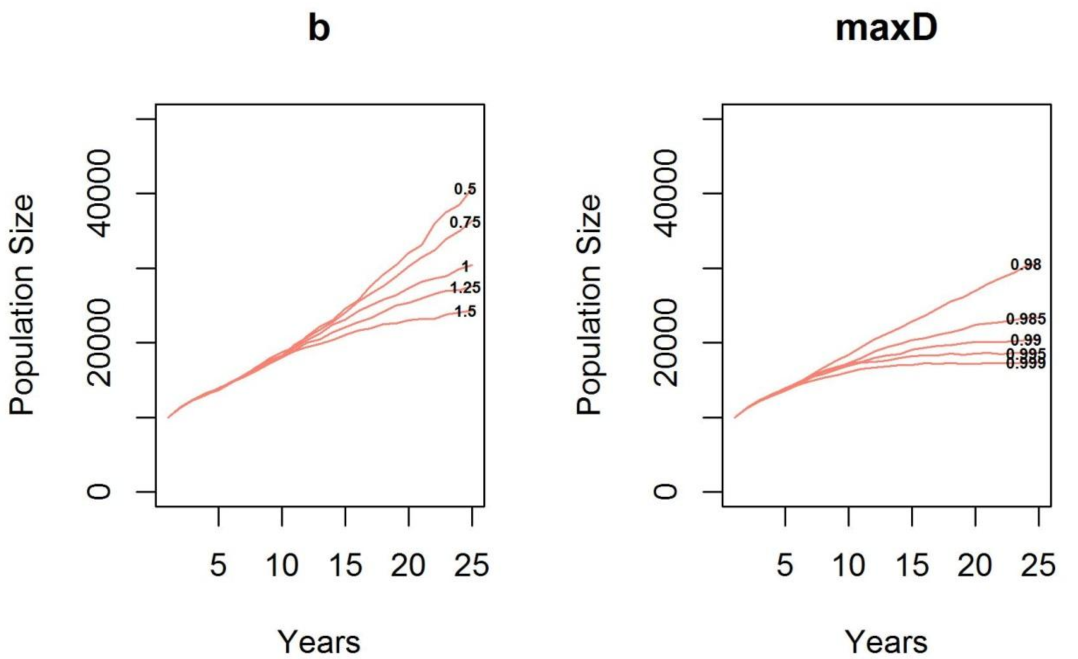

Figure 4. Influence of density dependent regulation of productivity on population size through time. Models assume an increasing population based on the demographic parameters in Table 2. Left hand graph shows population trajectory for different values of the shape parameter *b* (0.5, 0.75, 1, 1.25 and 1.5) assuming a maximum survival rate of 0.98. Right hand graph shows population trajectory for different maximum survival rates (0.98, 0.985, 0.99, 0.995, 0.999) assuming a shape parameter of 1.

To test the influence of density dependence on the metrics we consider, we compare metrics derived from density dependent stochastic models of an r-selected seabird with a stable population size to metrics derived from density independent stochastic models of the same population. We consider density dependent regulation of survival and productivity separately. We also consider sensitivity to assumptions about the shape parameter and maximum value selected for each demographic parameter.

2.3 Population Models

In order to test sensitivity to each of the factors listed above, we built seven different population models based on the demographic parameters set out in Table 2:

- a deterministic population model for a seabird with the characteristics of an r-selected species with a stable population
- a stochastic population model for a seabird with the characteristics of an r-selected species with an increasing population
- a stochastic population model for a seabird with the characteristics of an r-selected species with an stable population
- a stochastic population model for a seabird with the characteristics of an r-selected species with a declining population
- a stochastic population model for a seabird with the characteristics of a K-selected species with a stable population
- a stochastic population model with a density dependent response to productivity for a seabird with the characteristics of an r-selected species
- a stochastic population model with a density dependent response to survival for a seabird with the characteristics of an r-selected species

The purpose of these analyses was to examine general patterns and trends in each of the metrics as opposed to considering sensitivity to every conceivable combination of parameters. We use a stochastic population model for a stable r-selected population as a basis for comparing the sensitivity of each metric to demographic parameters. Useful metrics would be expected to respond in a similar fashion across different models, for example metrics derived from a declining population of a K-selected species would be expected to show a similar pattern to metrics from a declining population of an r-selected species. By restricting the analyses to a limited subset of models and extrapolating across them, we aim to simplify the interpretation of the results, aiding clarity by drawing broad conclusions about expected patterns in the metrics as opposed to a detailed discussion covering every possible combination of sources of sensitivity.

As described above, following discussions with the project steering group about selecting realistic levels for simulated effects arising as a result of impacts from offshore wind farms, we considered up to a 40% increase in mortality and up to a 40% reduction in productivity. It is worth noting that, in absolute terms, these figures reflect a greater effect on productivity than on survival. For example, assuming a stable population of an r-selected species (Table 2), a 40% decrease in productivity would result in a change from 1.030 chicks/nest to 0.618 chicks/nest. In contrast, a 40% increase in adult mortality would result in a decrease in the adult survival rate from 0.890 (1,100 deaths per year assuming a population of 10,000 adults) to 0.846 (1,540 deaths per year assuming a population of 10,000 adults). However, the purpose of these metrics is to determine what impact these changes have at a population level.

2.3.1 Deterministic population model for r-selected species

Population trends were simulated over a typical 25 year life time of an offshore wind farm using Leslie Matrix Models. Based on the review of Horswill and Robinson (2015), for the r-selected species we considered a model with four age-classes with annual survival transition probabilities between the ages of 0-1, 1-2, 2-3 and for an adult age class (Matrix *a*). Reproduction was confined to the adult age class:

$$a = \begin{Bmatrix} 0 & 0 & 0 & 0.515 \\ 0.441 & 0 & 0 & 0 \\ 0 & 0.741 & 0 & 0 \\ 0 & 0 & 0.741 & 0.890 \end{Bmatrix}$$

Matrix *a* assumes a stable population of an r-selected species with four age-classes. The top row reflects the annual productivity rate for each age class and is derived from half the productivity rate in Table 2, to give per individual productivity as opposed to a per pair productivity. The values on the diagonal give the survival probability for each of the immature age classes, and the final value the adult survival rate, i.e. the rate at which adults continue to be (living) adults.

This model can be thought of as a post-breeding census of the population concerned, with the first age class giving the number of birds that fledged per breeding individual that calendar year. For the unimpacted population, each bird has the probability of surviving until the next calendar year given in the diagonal of matrix *a*, where each adult will raise the number of chicks given in row 1, column 4 to fledging. For the impacted population, at each 1-year time step, impacts of an additional mortality of between 0 and 40% across all age classes or a reduction in productivity of between 0 and 40% were applied.

2.3.2 Stochastic population model for r-selected species

The format of the stochastic population models is similar to that given above in matrix *a* for the deterministic model. The key difference is that an element of random variation is introduced into the model by sampling each demographic parameter from a distribution for each iteration (year) of the model, rather than assuming a fixed value across all iterations. The model is then run many times (in our case, 1000) in order to give an indicative trend for the population concerned and an estimate of the uncertainty surrounding that trend.

Two types of stochasticity can be incorporated within the models, demographic stochasticity and/or environmental stochasticity. Demographic stochasticity can be considered to be variation between individuals (e.g. variation in individual quality), which affects likelihood of transition between states (age classes). Environmental stochasticity can be considered to be variation arising as a result of changes in the environment (e.g. between year differences in weather conditions) affecting all individuals within a group. Environmental stochasticity is considered to be more important than demographic stochasticity, particularly in the case of large populations (Lande 1993; Fox & Kendall 2002); for this reason, in our analyses we only consider environmental stochasticity.

Environmental stochasticity was incorporated into the models by considering one standard deviation around the mean values given for each parameter (Table 2). 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. A matched runs approach was used to compare the impacted and unimpacted populations. For each iteration, a productivity rate and a survival rate for each age class was simulated. These productivity and survival rates were then used to estimate the size of the unimpacted population. To estimate the size of the impacted population, as described above for the deterministic model, impacts of a 0-40% increase in mortality or reduction in productivity were then applied to these simulated rates.

2.3.3 Stochastic population model for increasing or decreasing r-selected species

In order to extend these models to consider how the metrics responded to populations that were increasing or decreasing, we adjusted the demographic parameters given in matrix *a* above using biologically plausible values identified in Horswill and Robinson (2015). Through trial and error, we aimed to identify a combination of demographic parameters that would result in a moderate increase of roughly 5% per year and a decrease of roughly 5% per year; which would result in the population doubling, or halving, in approximately 14 years. This was achieved using matrix *b* for an increasing population and matrix *c* for a declining population.

$$b = \begin{Bmatrix} 0 & 0 & 0 & 0.795 \\ 0.441 & 0 & 0 & 0 \\ 0 & 0.850 & 0 & 0 \\ 0 & 0 & 0.850 & 0.890 \end{Bmatrix}$$

$$c = \begin{Bmatrix} 0 & 0 & 0 & 0.460 \\ 0.358 & 0 & 0 & 0 \\ 0 & 0.741 & 0 & 0 \\ 0 & 0 & 0.741 & 0.866 \end{Bmatrix}$$

We incorporated stochasticity as described above, and considered impacts of a 0-40% increase in mortality or decrease in productivity.

2.3.4 Stochastic population model for an r-selected species incorporating density dependent impacts on survival or productivity

In order to extend these models to incorporate the density dependent regulation of survival or productivity, we used the Weibull function (Equation 1), as described above, to estimate productivity and survival rates given the estimated size of the impacted and unimpacted populations at each time step. As described above, in the case of the unimpacted populations these values were used to estimate population size. In the case of the impacted populations, an additional mortality of between 0 and 40% or a reduction in productivity of between 0 and 40% was then applied before estimating population size. For simplicity, it was assumed that density dependent regulation of survival would only occur in adult birds as other age classes may be less constrained in their foraging areas and are therefore, less likely to be affected by competition.

2.3.5 Stochastic Population Model for K-selected species

In order to consider how the metrics may change in response to a more K-selected species we used Horswill and Robinson (2015) to identify a range of biologically plausible values for a species at the opposite end of the seabird life history spectrum from those considered above. Based on this review, we consider a nine age class model with higher survival rates and lower productivity rates than described above (matrix *d*).

$$d = \begin{Bmatrix} 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0.270 \\ 0.845 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 \\ 0 & 0.845 & 0 & 0 & 0 & 0 & 0 & 0 & 0 \\ 0 & 0 & 0.845 & 0 & 0 & 0 & 0 & 0 & 0 \\ 0 & 0 & 0 & 0.845 & 0 & 0 & 0 & 0 & 0 \\ 0 & 0 & 0 & 0 & 0.845 & 0 & 0 & 0 & 0 \\ 0 & 0 & 0 & 0 & 0 & 0.845 & 0 & 0 & 0 \\ 0 & 0 & 0 & 0 & 0 & 0 & 0.845 & 0 & 0 \\ 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0.845 & 0.953 \end{Bmatrix}$$

2.3.6 Additional Analyses

For promising metrics, we undertook some further analyses to allow us to more carefully consider their suitability. Concern has been raised that the value estimated for a metric may be determined by uncertainty surrounding the demographic parameters used in the population models (Green 2014). Non-biological factors, such as sampling variance, may make a significant contribution to the total variance associated with demographic parameters (Gould & 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). We recalculate these metrics based on distributions of adult survival rates using a fixed mean estimate (0.89, see Table 2) and a range of values for the standard deviation informed by the review of Horswill and Robinson (2015). The resulting distributions are shown in Figure 5. Ideally, the metrics 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.

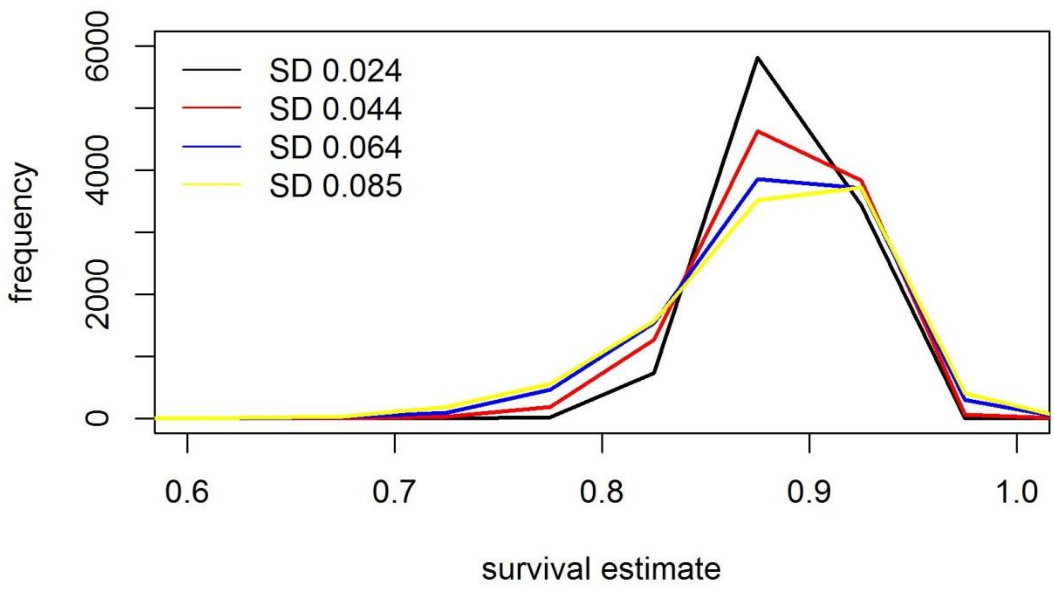

Figure 5. Frequency distributions estimated for adult survival rates based on a mean of 0.89 and standard deviations of 0.024, 0.044, 0.064 and 0.085.

Metrics may be calculated at different points in time (e.g. five years post-construction, 10 years post-construction etc.) meaning it is possible to have a snapshot of what population level impacts are likely to be at any point in the lifetime of the wind farm. However, it is unclear whether sensitivity of the metrics to demographic parameters may vary in time. For example, over a longer time period, variation in demographic parameters may average out meaning metrics would be expected to be less sensitive to mis-specification of demographic parameters at the end of a project lifetime than at earlier stages of the project. We test this for periods of five and 10 years post-construction in relation to adult survival, and compare these values to those obtained in the earlier analyses for 25 years post-construction.

Finally, we more closely examine how incorporating density dependence into population models influences the final metrics. Firstly, we consider how sensitive density dependent models are to mis-specification of adult survival, following the approach take in section 2.2.6. If density dependent models are to be used, then ideally they should be no more sensitive to mis-specification of adult survival than density independent models.

Density dependence may also influence a metric’s sensitivity to population trend, for example where a population is increasing, density dependent processes may cause growth rates of impacted and unimpacted populations to be equalised. We therefore consider the sensitivity of selected metrics to population trend, when derived from a density dependent model. Lastly, we consider how incorporating density dependence influences the metrics through time.

2.4 Assessing the value of metrics

In order to determine how robust different metrics are to uncertainty, it is important identify key criteria against which each metric should be assessed. Following a discussion with the project steering group, it was agreed the criteria for assessing the metrics should be:

- Have a consistent response to the magnitude of the estimated impact (i.e. the relationship between the magnitude of the impact and the metric should be linear)
- Have a clear relationship with the magnitude of the impact (i.e. there should be a noticeable change in the metric in response to impacts of increasing magnitude)
- Insensitive to mis-specification of the input demographic parameters (see section 2.2.6)
- Insensitive to population trend (see section 2.2.4)
- Insensitive to the incorporation of density dependence (see section 2.2.7)
- Insensitive to uncertainty in the form of density dependence (see section 2.2.7)
- Insensitive to whether it is derived from a deterministic or stochastic model (see section 2.2.2)

Ideally, metrics should have a clear and consistent relationship with the magnitude of the estimated impact in order to help interpret the population-level consequences of any impact. Such a relationship will help decision makers understand what the extent of the risk of over or underestimating the severity of any population level effect is. For example, in Figure 6, we consider two possible relationships between a metric and the magnitude of the impact under consideration. If there is a consistent linear trend between the metric and the magnitude of the impact under consideration, it is easier to predict what the effect at a population level will be of an increase or decrease in the magnitude of the predicted impact. In contrast, if there is a curved relationship between the metric and the magnitude of the impact, extrapolating the metric value for an increase or decrease in the impact will be less straightforward as the rate of change will differ between high and low impacts.

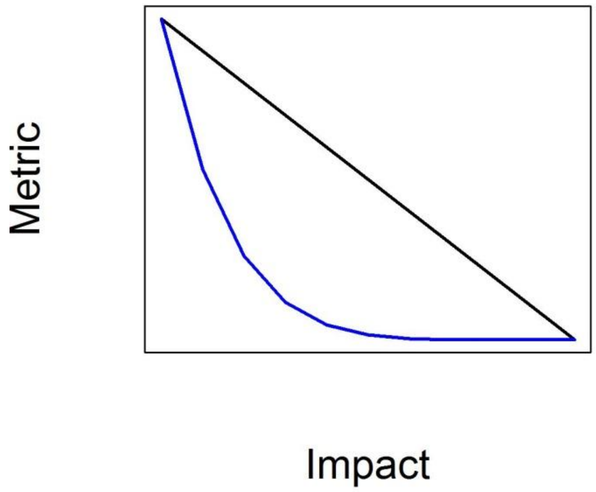

Figure 6. Possible linear or non-linear relationships between metrics used to assess the population-level consequences of impacts from offshore wind farms and the magnitude of the impacts concerned.

Given the uncertainties in the data used to derive these metrics, it is important the relationship between the metric and the magnitude of the impact is clear. For example, in the case of the straight line in Figure 6, this should have a steep gradient. If the gradient is less steep, then the limited range of values over which the metric operates may mean that significant effects at a population level are masked by relatively minor changes in the metric value. Additionally, it may be difficult to determine the extent to which any change results from an increase in the magnitude of the impact as opposed to uncertainty in the demographic parameters used in the population models. Linked to this, it is important that the metrics are insensitive to mis-specification of the demographic parameters used in population models. If metrics are sensitive to mis-specification of demographic parameters, it may lead to inappropriate conclusions, or not, as a result of incomplete knowledge of the demography of the species concerned. Metrics may also be sensitive to the degree of uncertainty surrounding estimates of demographic parameters. Whilst it is important to understand this sensitivity, we felt that it was less important than the sensitivity to the mis-specification of the demographic parameters themselves. For this reason, we restricted analysis of the sensitivity to the level of uncertainty surrounding demographic parameters to metrics which had performed strongly in relation to other criteria.

Seabird populations are known to be affected by immigration (the recruitment of breeding birds from elsewhere) and emigration (the loss of breeding birds to other populations). Metrics used to assess the population consequences of impacts from offshore wind farms may be sensitive to these changes. However, immigration and emigration will be reflected at a population level by changes in the number of breeding adults present. These changes would be similar to an increase (immigration) or decrease (emigration) in the adult survival rate. For this reason, we felt it would be more appropriate to focus on changes in survival as part of our sensitivity analysis, rather than considering immigration or emigration directly.

The purpose of these metrics is to understand what the population level impact of the wind farm is, after all other factors have been accounted for. For this reason, it is important that metrics are insensitive to population trend so that the metric reflects the impact of the wind farm, as opposed to the status of the population concerned. This means that, for impacts of a similar magnitude, metrics for a population that was increasing prior to the construction of the wind farm should have the same value as those for a population that was decreasing prior to the construction of the wind farm. However, such an approach does not preclude taking population trends into account when assessing the impact of a development at a population level. For example, it would be possible to specify that a more severe population level effect was permissible if that population were increasing, rather than stable or declining.

For similar reasons it is desirable that metrics should be insensitive to the incorporation of density dependence and the form of any density dependence assumed. The latter is particularly important given uncertainty over the forms of density dependence that may be present in seabird populations (Horswill & Robinson 2015). For example, if density dependence were depensatory (reduction in survival/productivity in response to reductions in population density) this would mean that impacted populations would be expected to decline more quickly than unimpacted populations.

Finally, ideally metrics should be unaffected by whether they are derived from a deterministic or stochastic population model to minimise the uncertainty which may arise if one approach suggests an impact may be acceptable and the other approach suggests that it is not. As described above, the metrics should focus on the population effects of offshore wind farms, once all other factors have been accounted for. This means that ideally, where the same demographic parameters are used, and impacts are of the same magnitude, the metrics derived should be the same, regardless of whether a stochastic or a deterministic model is used. However, it is important to note that probabilistic metrics cannot be derived from deterministic models.

2.5 R Code

All models were developed in the R statistical package (R-project 2015), and the code used is available as an electronic appendix to this report. The code used for the population models was written in consultation with the BTO Ecological Statistician, Dr Alison Johnston.

3 Results

Each of the metrics described in Table 1 was derived from a population model. The population models used to derive these metrics are presented below in section 3.1. The metrics themselves and their sensitivity to different parameters, population trends and model structures are then presented in section 3.2, with the key findings and recommendations presented in sections 3.2.1 and 3.2.2. The first metric text (Section 3.2.3) contains guidance on how to interpret the results presented in assessment of each metric.

3.1 Population Models

3.1.1 Deterministic population model for a seabird with the characteristics of an r-selected species with a stable population

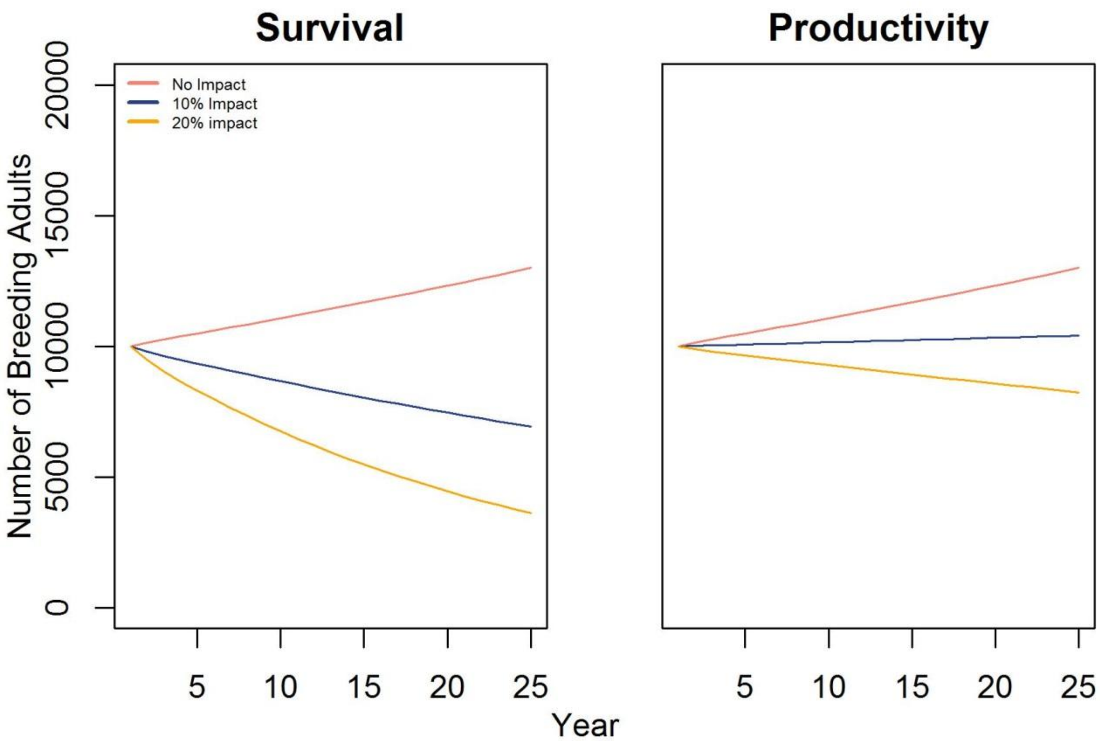

Figure 7. Deterministic population model for an r-selected seabird species with a stable population over 25 years. Plots show the baseline (i.e. no impact scenario) as well as the impact of a 10% or 20% increase in mortality (left hand graph) and a 10% or 20% decrease in productivity (right hand graph). Note that deterministic models are naturally less precautionary than stochastic models (Lande *et al* 2003).

Using a deterministic model, the growth rate of the unimpacted population was 1.010. Following a 10% increase in mortality, the growth rate fell to 0.985 and following a 20% increase in mortality the growth rate fell to 0.959. For impacts on productivity, a 10% reduction led to the growth rate falling to 1.001 and a 20% reduction led to the growth rate falling to 0.992 (Fig. 7, Table 4). Whilst a 10% reduction in productivity results in a growth rate that is closer to 1 than the baseline scenario, the baseline demographic parameters were selected as they resulted in both the deterministic and stochastic models having a growth rate close to 1 (i.e. essentially stable). Stochastic models naturally result in more conservative estimates of population size as the impact of environmental stochasticity generally causes the long-term growth rate to be less than the mean growth rate (Lande *et al* 2003). Consequently, parameters resulting in a slight increase in population size with a deterministic model would result in a slight decrease for a stochastic model. In order to compare metrics derived from deterministic and stochastic models, with all other factors being equal, we therefore selected demographic parameters that would result in a growth rate as close to 1 (i.e. a stable population) as possible.

Table 4. Population growth rates obtained from different models and different levels of offshore wind farm impact (95% CIs for Stochastic models).

	No Impact	Impact on Survival			Impact on Productivity		
		10%	20%	40%	10%	20%	40%
Deterministic, Stable, r-selected	1.010	0.985	0.959	0.908	1.001	0.992	0.971
Stochastic, Stable, r-selected	0.996 (0.961 – 1.028)	0.969 (0.933 – 1.004)	0.942 (0.907 - 0.976)	0.889 (0.843 – 0.928)	0.988 (0.958 – 1.016)	0.980 (0.951 – 1.008)	0.962 (0.935 – 0.987)
Stochastic, Increasing, r-selected	1.073 (1.037 – 1.109)	1.044 (1.006 – 1.080)	1.012 (0.966 – 1.053)	0.944 (0.881 – 1.001)	1.061 (1.027 – 1.095)	1.049 (1.017 - 1.082)	1.022 (0.993 – 1.051)
Stochastic, Declining, r-selected	0.952 (0.923 – 0.982)	0.922 (0.888 – 0.957)	0.891 (0.851 – 0.927)	0.829 (0.778 – 0.875)	0.945 (0.913 – 0.974)	0.938 (0.907 – 0.966)	0.922 (0.894 – 0.949)
Stochastic, Stable, K-selected	0.998 (0.978 – 1.017)	0.987 (0.966 – 1.007)	0.978 (0.957 – 0.998)	0.958 (0.935 – 0.979)	0.995 (0.977 – 1.014)	0.992 (0.975 – 1.011)	0.987 (0.971 – 1.003)
Stochastic, Stable, r-selected, Density dependent survival	1.002 (0.961 – 1.041)	0.986 (0.944 – 1.029)	0.972 (0.925 – 1.018)	0.939 (0.987 – 0.874)	0.997 (0.960 – 1.039)	0.993 (0.958 – 1.033)	0.984 (0.945 – 1.023)
Stochastic, Stable, r-selected, Density dependent productivity	0.997 (0.975 – 1.020)	0.983 (0.958 – 1.007)	0.965 (0.933 – 0.994)	0.916 (0.857 – 0.961)	0.994 (0.972 – 1.016)	0.990 (0.969 – 1.009)	0.982 (0.962 – 0.999)

3.1.2 Stochastic population model for a seabird with the characteristics of an r-selected species with a stable population

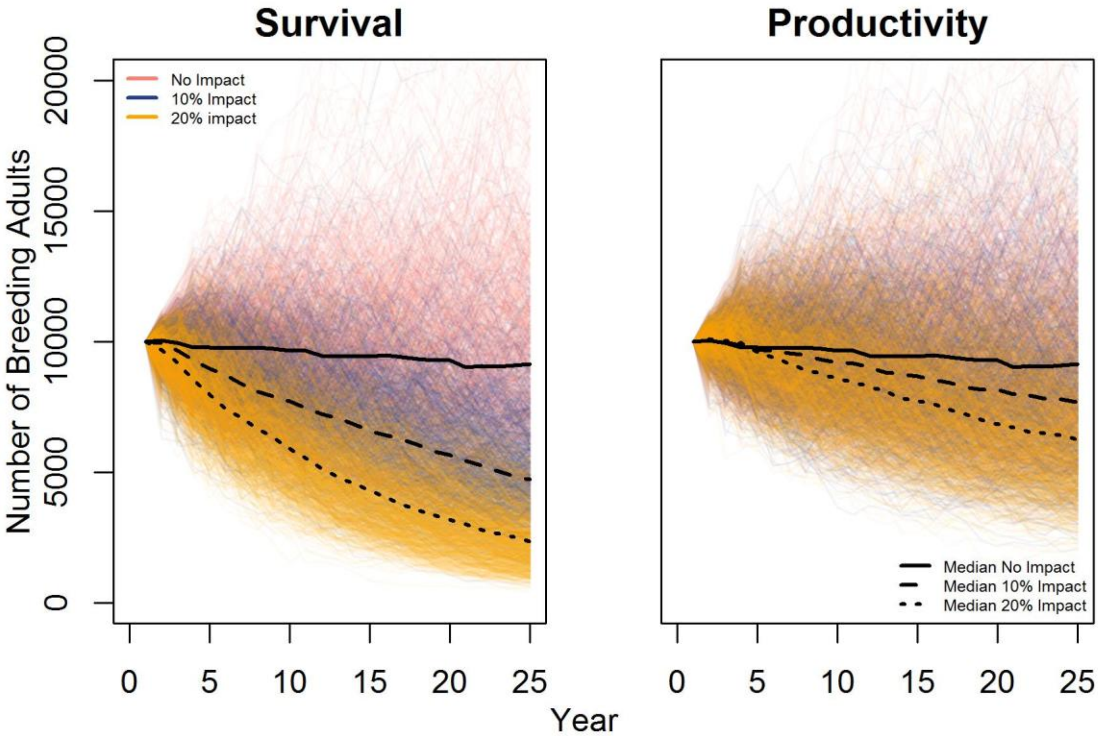

Figure 8. Stochastic population model (1000 bootstraps) for an r-selected seabird species with a stable population over 25 years. Plots show the baseline (i.e. no impact scenario) as well as the impact of a 10% or 20% increase in mortality (left hand graph) and a 10% or 20% decrease in productivity (right hand graph).

Stochastic population models were run over 1000 bootstraps. Each coloured line in Figure 8, above, represents population changes in a single model bootstrap run over 25 years. The red lines indicate populations exposed to no impact from offshore wind farms, the blue lines reflect a 10% impact from offshore wind farms and the orange lines reflect a 20% impact. Where effects have a strong impact at a population level, the red, blue and orange lines would be clearly distinguishable, as is the case for the plot on the left. Where the effects have a less severe impact at a population level, the lines would be less clearly distinguishable, as is the case for the plot on the right. In addition to each bootstrap, the black lines in Figure 6 show the median population trajectories under each scenario (solid lines = no impact, dashed line = 10% impact, dotted line = 20% impact). These roughly correspond to the median population growth rates shown in Table 4. The variance around these population trajectories can be inferred with reference to the individual bootstraps for each scenario; darker areas indicate more frequent population trajectories, paler areas those represented by fewer bootstrap runs. They show for example, that although impacts on productivity can result in reduced population sizes, with reasonable environmental variation the differences may be hard to distinguish. Even when there are relatively large impacts on survival the impacts may not be consistently manifested for several years, highlighting the need for consistent post-consent monitoring.

Using a stochastic model, the population growth rate of the stable population was 0.996. Following a 10% increase in mortality, this rate fell to 0.969 and 0.942 in response to a 20% increase in mortality. For a 10% reduction in productivity, the population growth rate fell to 0.988 and 0.980 in response to a 20% reduction in productivity. As explained above (section 3.1.1), using demographic parameters equivalent to those used in the deterministic model results in a relatively minor rate of decline, in comparison to the relatively minor rate of increase observed in the deterministic model. Based on the criteria described above, any metrics used to quantify the population level effects of impacts from an offshore wind farm would be insensitive to such differences.

3.1.3 Stochastic population model for a seabird with the characteristics of an r-selected species with an increasing population

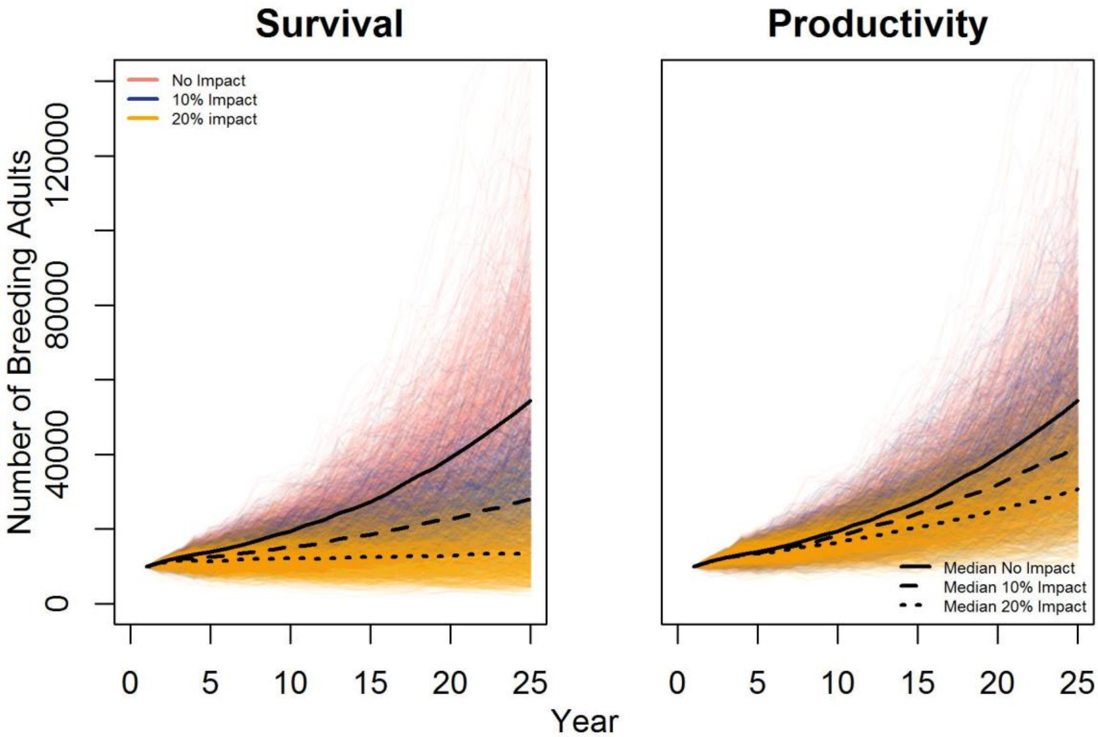

Figure 9. Stochastic population model (1000 bootstraps) for an r-selected seabird species with an increasing population over 25 years. Plots show the baseline (i.e. no impact scenario) as well as the impact of a 10% or 20% increase in mortality (left hand graph) and a 10% or 20% decrease in productivity (right hand graph).

Using a stochastic model, the population growth rate of the increasing population was 1.073 (Fig. 9). Following a 10% increase in mortality, this rate fell to 1.044 and 1.012 in response to a 20% increase in mortality. For a 10% reduction in productivity, the population growth rate fell to 1.061 and 1.049 in response to a 20% reduction in productivity.

3.1.4 Stochastic population model for a seabird with the characteristics of an r-selected species with a declining population

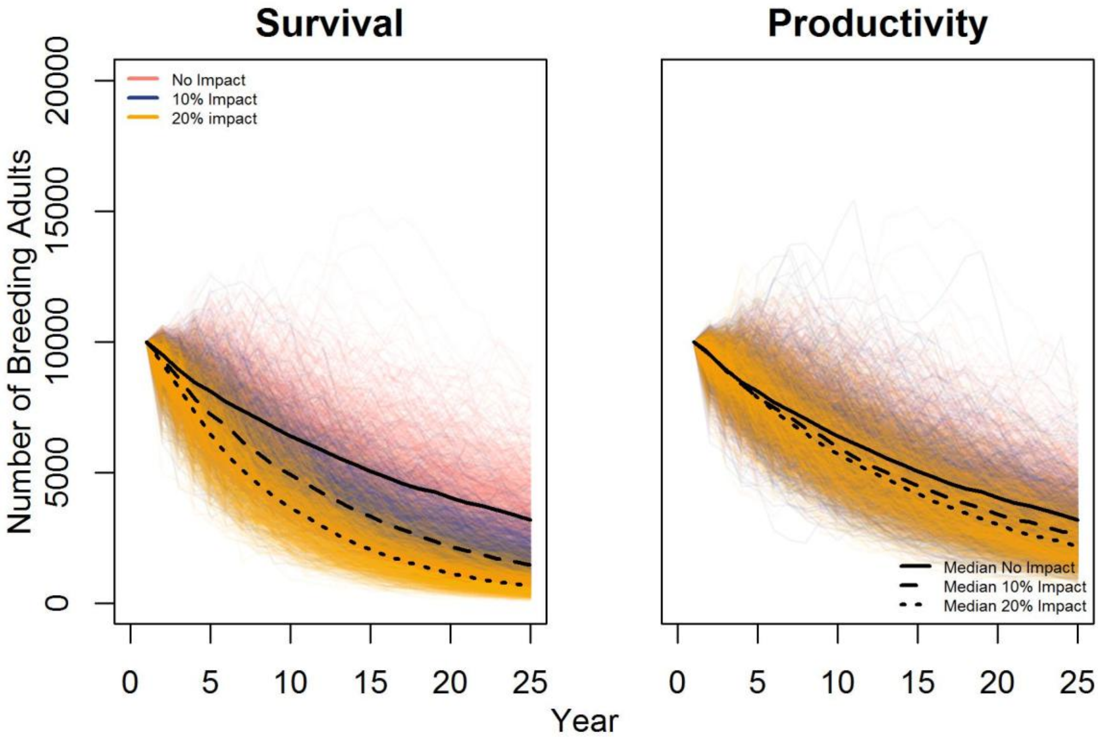

Figure 10. Stochastic population model (1000 bootstraps) for an r-selected seabird species with a declining population over 25 years. Plots show the baseline (i.e. no impact scenario) as well as the impact of a 10% or 20% increase in mortality (left hand graph) and a 10% or 20% decrease in productivity (right hand graph).

Using a stochastic model, the population growth rate of the decreasing population was 0.952 (Fig. 10). Following a 10% increase in mortality, this rate fell to 0.922 and 0.891 in response to a 20% increase in mortality. For a 10% reduction in productivity, the population growth rate fell to 0.945 and 0.938 in response to a 20% reduction in productivity.

3.1.5 Stochastic population model for a seabird with the characteristics of a K-selected species with a stable population

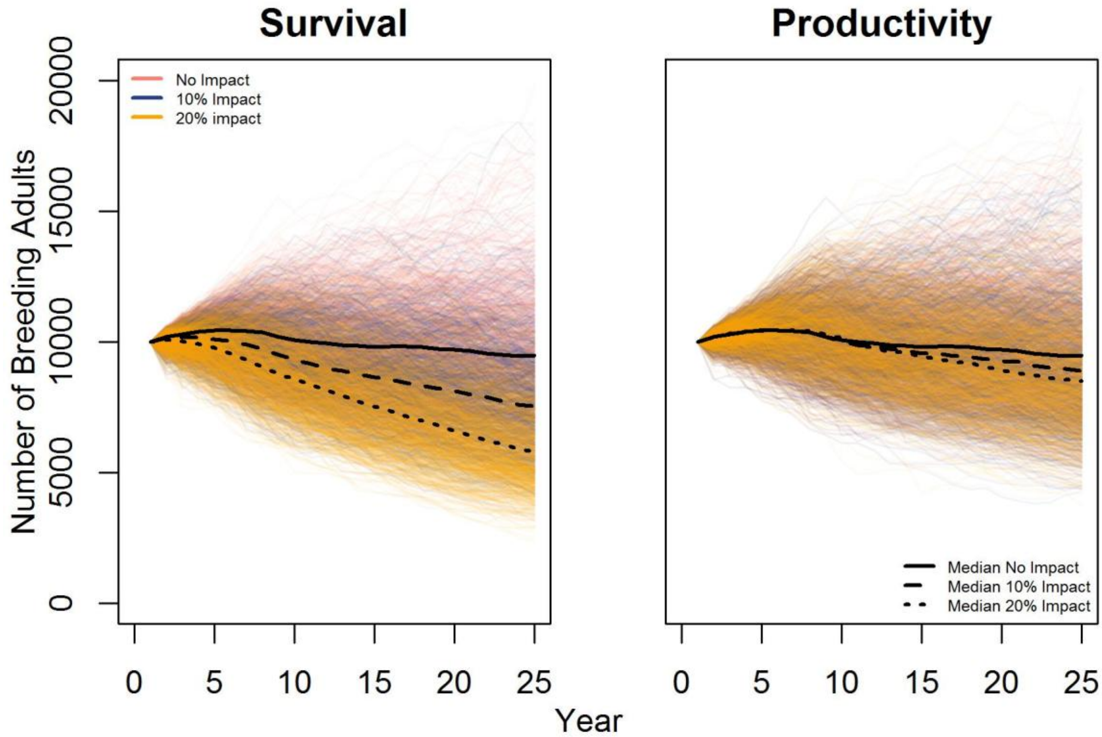

Figure 11. Stochastic population model (1000 bootstraps) for a K-selected seabird species with a stable population over 25 years. Plots show the baseline (i.e. no impact scenario) as well as the impact of a 10% or 20% increase in mortality (left hand graph) and a 10% or 20% decrease in productivity (right hand graph).

Using a stochastic model, the population growth rate of the stable K-selected population was 0.998 (Fig. 11). Following a 10% increase in mortality, this rate fell to 0.987 and 0.978 in response to a 20% increase in mortality. For a 10% reduction in productivity, the population growth rate fell to 0.995 and 0.992 in response to a 20% reduction in productivity. The greater overlap in the bootstrap population trajectories, indicate that impacts on productivity may be harder to detect than impacts on survival.

In comparison to a stable population of an r-selected species, populations of the K-selected species appear to be less affected by changes in survival and productivity. This means that the growth rate of the impacted K-selected species is likely to decline less than the growth rate of the impacted r-selected species. It is also worth noting that the longer generation times of the K-selected species mean that where productivity, rather than survival, is impacted, these changes will not become evident for a longer period than they would for an r-selected species.

3.1.6 Stochastic population model with a density dependent response to productivity for a seabird with the characteristics of an r-selected species

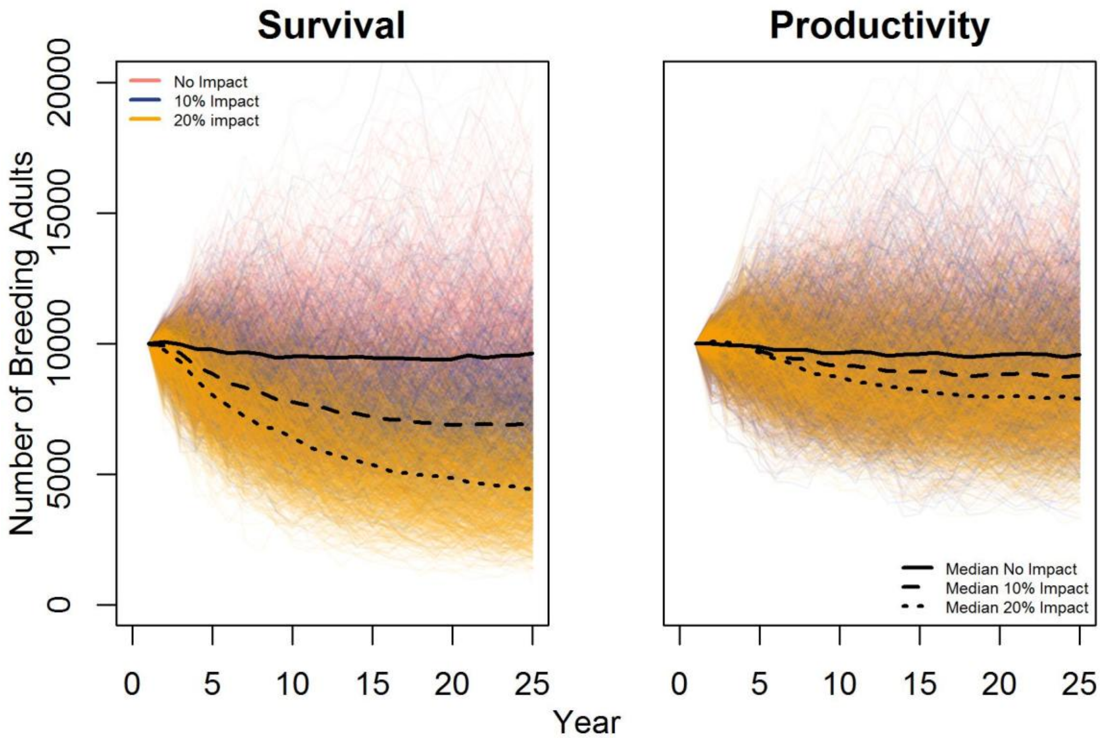

Figure 12. Stochastic population model (1000 bootstraps) for an r-selected seabird species with a stable population over 25 years assuming density dependent productivity. Plots show the baseline (i.e. no impact scenario) as well as the impact of a 10% or 20% increase in mortality (left hand graph) and a 10% or 20% decrease in productivity (right hand graph).

Assuming density dependent regulation of the population growth rate, changes in the population size are compensated for by changes in the productivity rate. As a consequence, when the population size decreases, for example as a result of the impacts arising from offshore wind farms, the productivity rate increases so that the population growth rate is maintained at a higher level than for a density independent model.

Using a stochastic model, the population growth rate over the 25 years of the study period for a stable population with density dependent regulation of productivity was 0.997 (Fig. 12), assuming a maximum productivity rate (maxD) of 1.5 chicks/nest and a shape parameter (*b*) of 1. Following a 10% increase in mortality, this rate fell to 0.983 and 0.965 in response to a 20% increase in mortality. For a 10% reduction in productivity, the population growth rate fell to 0.994 and 0.990 in response to a 20% reduction in productivity. For the reasons set out above, these growth rates are higher than were recorded for the density independent model of a stable r-selected seabird population (see section 3.1.2).

3.1.7 Stochastic population model with a density dependent response to survival for a seabird with the characteristics of an r-selected species

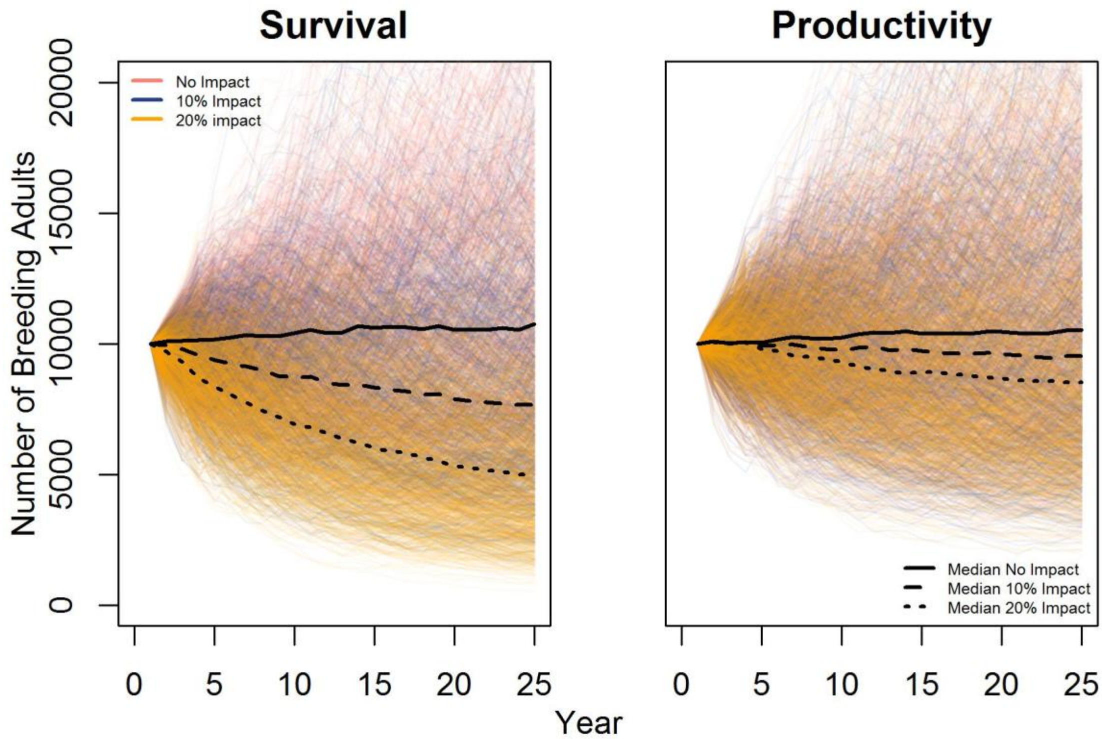

Figure 13. Stochastic population model (1000 bootstraps) for an r-selected seabird species with a stable population over 25 years assuming density dependent survival. Plots show the baseline (i.e. no impact scenario) as well as the impact of a 10% or 20% increase in mortality (left hand graph) and a 10% or 20% decrease in productivity (right hand graph).

Assuming density dependent regulation of the survival rate, changes in the population size are compensated for by changes in the survival rate. As a consequence, when the population size decreases, for example as a result of the impacts arising from offshore wind farms, the survival rate increases so that the population growth rate is maintained at a higher level than for a density independent model.

Using a stochastic model, the population growth rate of a stable population with density dependent regulation of survival was 1.002 (Fig. 13), assuming a maximum survival rate (maxD) of 0.98 and a shape parameter (*b*) of 1. Following a 10% increase in mortality, this rate fell to 0.986 and 0.972 in response to a 20% increase in mortality. For a 10% reduction in productivity, the population growth rate fell to 0.997 and 0.993 in response to a 20% reduction in productivity. For the reasons set out above, these growth rates are higher than were recorded for the density independent model of a stable r-selected seabird population (see section 3.1.2).

For both the density dependent scenarios, but especially where survival is density-dependent, in the face of environmental stochasticity, determining the consequences of the impacts becomes more difficult as there is less separation between bootstraps from each scenario (Fig's 12 and 13).

3.2 Metrics for Population Level Impacts

3.2.1 Assessing the metrics

This section summarises the key findings from the sensitivity analysis of the eleven metrics listed in section 2.1.4, a more detailed summary of the results from the sensitivity analysis of each of these metrics is provided below in sections 3.2.3 – 3.2.12.

The metrics 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), impacts on survival had a more significant impact at a population level. In reality, the life history strategies of seabirds mean that indirect impacts on survival (through displacement or barrier effects) are likely to be of a lower magnitude than effects on productivity as seabirds are likely to abandon breeding attempts in sub-optimal conditions and thus minimise impacts on survival (e.g. Monaghan *et al* 1989; Aebischer & Wanless 1992; Klomp & Furness 1992). Such situations may occur if birds are displaced from favoured foraging areas in response to the presence of an offshore wind farm. As a consequence, where impacts are on survival, metrics may reflect the lower end of the range of impacts considered here, whereas for impacts on productivity metrics may be towards the upper end of those considered.

Of the metrics we considered in this sensitivity analysis, none showed both a clear and a consistent response to impacts of increasing magnitudes (Table 5). A clear response makes it more straightforward to distinguish between population level consequences arising from impacts of differing magnitudes. For example, if metrics vary only over a limited range then relatively small changes in their value may mask more severe changes at a population level. Additionally, if metrics do not show a clear response to impacts arising as a result of offshore wind farms, it may be difficult to discern any effects from what would be expected in response to natural variation in the population concerned. However, this may be partly addressed through the use of matched runs, as advocated by Green (2014) and WWT Consulting (2012), in our analyses.

A consistent response facilitates understanding the relationship between the metric and the impact at a population level. It also allows us to more easily understand what the consequences of under or over-estimating the magnitude of any impact at a population level would be. For example, if there is a linear relationship between the metric and the magnitude of the impact, it is possible to quickly determine what the implications are of under or over-estimating the impact. This is less straightforward for a curved relationship as the implications of under of over-estimating the impact will depend on the magnitude of the impact predicted (see Figure 6), making the conclusion more vulnerable to mis-specification of the model parameters.

Of the metrics considered, only the population growth rate and the ratio of the impacted to unimpacted population growth rate showed a consistent linear relationship with impacts of increasing magnitude (Table 5). However, the confidence intervals associated with these metrics, in combination with the range over which they operate, meant that impacts arising as a result of the presence of an offshore wind farm could not be clearly detected unless they were particularly severe. In contrast, the ratio of the impacted to unimpacted population size after 25 years showed a clear response to the range of impacts considered. However, this response was asymptotic. Where an asymptote is reached, it can be difficult to distinguish between population level impacts of differing magnitudes. These metrics were similar regardless of whether they were derived from deterministic or stochastic models. None of the probabilistic metrics gave responses which were clear or consistent (Table 5).

Ideally the metrics should be able to separate the population level consequences of impacts from offshore wind farms from the underlying trend of the population concerned. For this reason, it is desirable that metrics should give the same value for a similar magnitude of impact, regardless of whether the population is stable, increasing or declining. Of the metrics considered, only probability of the population growth rate of the impacted population being 2.5% less than the population growth rate of the unimpacted population was insensitive to population trend (Table 5). However, this metric had no clear and consistent relationship with the magnitude of the impact concerned, making it more difficult to draw conclusions about the population consequence of any impact, and was particularly sensitive to mis-specification of the adult survival rate (Table 5).

Of the remaining metrics, most showed clear differences between populations which were stable and those that were declining and/or increasing. For example, the probability of a population growth rate being less than 1 changes from 0.968, for a 10% increase in mortality in a stable population, to 0.029 for a 10% increase in mortality in an increasing population (Section 3.2.6 Table 22), meaning it was not possible to separate the impact of the offshore wind farm from the underlying trend of the population concerned with this metric. However, these differences were less pronounced for the ratio of the impacted to unimpacted population growth rate and the ratio of the impacted to unimpacted population size after 25 years (Table 5). For the ratio of the impacted to unimpacted growth rate the values of this ratio for stable and increasing populations were the same (0.973) for a 10% increase in mortality, but decreased to 0.966 for a declining population (Section 3.2.4 Table 10). Whilst such a change may appear small, the resultant change in the growth rate over the lifetime of the project is likely to have a significant impact at a population level. For the ratio of the impacted to unimpacted population size after 25 years, for a 10% increase in mortality values for an increasing and stable population are similar (0.509 and 0.515 respectively), but decrease substantially (to 0.460) for a declining population (Section 3.2.5 Table 16).

Of the demographic parameters considered, all metrics were most sensitive to mis-specification of adult survival. As is often the case in long-lived species, variation in adult survival has the greatest impact on population growth rates (Lande *et al* 2003). However, the uncertainty surrounding estimates of adult survival can be lower than the uncertainty surrounding estimates of other demographic parameters, such as recruitment rates, as adult survival is generally easier to measure (Horswill & Robinson 2015). This greater uncertainty means other parameters are more likely to be mis-specified than adult survival and, as a result, it is desirable that metrics should be considerably less sensitive to them than to adult survival. This was true for every metric with the exception of the change in the probability of the population growth rate being <1 (Table 5). Even when considered over a realistic range (defined by the observed mean and standard deviation of each parameter, so accounting for the fact that adult survival rates are known more precisely) in percentage terms, mis-specification of adult survival still has a more significant effect on the final value than other demographic parameters for all metrics, with the exception of the probability of the change in the population growth rate being <1 (Table 5). Consequently, every effort should be made to obtain accurate and representative estimates of adult survival for the species concerned.

In general, the sensitivity of each metric to mis-specification of the parameters used in demographic models increases with the magnitude of the impact considered. This means that where impacts are likely to have a greater magnitude, there is likely to be greater uncertainty in the conclusions drawn about the population level consequences associated with the offshore wind farms. As the parameters used in the demographic models are mean values, mis-specification means that these impacts are as likely to under-estimated as over-estimated. Therefore, the values presented throughout this report reflect the impact of mis-specification in either direction.

Of the metrics considered, only population growth rate and the ratio of the impacted to unimpacted population growth rate were relatively insensitive to mis-specification of adult survival (i.e. a 1% mis-specification of adult survival resulted in <1 % change in the metric) (Table 5), with a 1% mis-specification of adult survival resulting in a 0.75% and 0.12% change in the two metrics respectively, assuming a 10% increase in mortality (Tables 4 and 5). Of the remaining metrics, the ratio of the impacted to unimpacted population size after 25 years and the probability that the population drops below its initial size at any point in time are less sensitive to mis-specification of adult survival than the others (Table 5). The probability of the growth rate being <1, the change in the probability of the growth rate being <1, the probability of a 25% decline and the change in probability of a 25% decline were also sensitive to mis-specification of the immature survival rate. Of those, the last three were also sensitive to mis-specification of chick survival and productivity as well (Table 5).

As might be expected, most metrics were sensitive to the incorporation of density dependence in the population models. The exceptions to this were the probability that the population was below its initial size at any point in time and the probability that the growth rate of the impacted population was 2.5% less than the unimpacted population (Table 5). Of the remaining metrics, sensitivity to density dependence reflected the range of values over which they operated. This meant that population growth rate, the ratio of the impacted to unimpacted population growth rate, the change in probability of the growth rate being <1 and the change in probability of a 25% decline were less sensitive to the incorporation of density dependence than the ratio of the impacted to unimpacted population size after 25 years, the probability of the growth rate being <1 and the probability of a 25% population decline. However, the metrics appeared to be relatively insensitive to the form of density dependence assumed, with similar population level consequences predicted regardless of the values assumed for the shape parameter, *b*, and the maximum level set for each demographic parameter. These results suggest that where there is good evidence about the presence and direction of any density dependent relationship in seabirds, it can be incorporated into population models and metrics generated may not be unduly sensitive to mis-specification of either the shape parameter or the maximum value for the demographic rate under consideration. However, it is important to note that density dependent relationships appear to operate on a highly site-specific basis and cannot be assumed to be present at all sites (Horswill & Robinson 2015).

Table 5. Assessment of metrics used to determine the population level effects of impacts from offshore wind farms on seabird populations. Colour indicates how well each metric matches each criterion – Light gray indicates close match, Dark gray indicates moderate match, black indicates poor match. Note that probabilistic metrics cannot be calculated from deterministic models, so the comparison between stochastic and deterministic models is not applicable for these metrics. Of the factors assessed, a clear and consistent relationship between the magnitude of the impact and the metric was considered to be key (highlighted by black outline around columns).

Metric (Acronym)	**Consistent** – see Figure 6, light gray indicates linear relationship, dark gray = curved relationship, black = stepped relationship	**Clear** – light gray, clear difference between metrics for impacts of increasing magnitude, dark gray, metric varies over a very narrow range, black the metric quickly reaches an asymptote when compared to impacts of increasing magnitude	**Insensitive to population trend** – light gray metrics identical regardless of population trend, dark gray <10 percentage change in metric in relation to population trend, black >10 percentage change in metric in relation to population trend	**Insensitive to Adult Survival** – light gray sensitivity to misspecification <1% for 10%l impacts on survival/productivity, dark gray sensitivity <5% for 10% impacts, black sensitivity >5% for 10 % impacts	**Insensitive to Immature Survival** – light gray sensitivity to misspecification <1% for 10%l impacts on survival/productivity, dark gray sensitivity <5% for 10% impacts, black sensitivity >5% for 10 % impacts	**Insensitive to Chick Survival** – light gray sensitivity to misspecification <1% for 10%l impacts on survival/productivity, dark gray sensitivity <5% for 10% impacts, black sensitivity >5% for 10 % impacts	**Insensitive to Productivity** – light gray sensitivity to misspecification <1% for 10%l impacts on survival/productivity, dark gray sensitivity <5% for 10% impacts, black sensitivity >5% for 10 % impacts	**Insensitive to incorporation of density dependence** – light gray median values the same for density dependent and independent, Dark gray <10 percentage change in metric between density dependent/independent, Black >10% change	**Insensitive to form of density dependence** – Light gray straight line regardless of values for *b* or maxD, dark gray wavy line, black clear +ve or –ve relationship	**Insensitive to Stochastic / deterministic model** – light gray median values the same for each model, dark gray <10% change between models, black >10% change between models
Population Growth Rate (GR)										
Ratio of Impacted to unimpacted population growth rate (RI:U)										
Ratio of impacted to unimpacted population size after 25 years (RI:U25)										
Probability of growth rate being <1 P(GR<1)										
Change in probability of growth rate being <1 dP(GR<1)										
Probability that population is below initial size at any point in time P(p<p_0)										
Probability of a 25% population decline P(ld<0.25)										
Change in Probability of a 25% population decline P(ld<0.25)										
Probability impacted population 50% below unimpacted population(P(I<25))										
Probability that impacted population growth rate is 2.5% less than unimpacted population growth rate P(lgr>2.5)										
Overlap of Impacted and Unimpacted Populations (OI:U)										

3.2.2 Recommendations

Our analyses suggest that metrics derived from deterministic models consistently predict lower population level consequences associated with the impacts of offshore wind farms than those derived from stochastic models. Given the differences between population level consequences predicted using deterministic and stochastic models and the fact that stochastic models are likely to be more realistic (Lande *et al* 2003), we suggest that metrics should be derived using stochastic models.

Incorporating compensatory density dependence into the population models can be considered to make the resulting metrics less precautionary. Horswill & Robinson (2015) found that density dependent effects were highly site-specific. For this reason, we suggest that compensatory density dependence should only be incorporated into models where there is sufficient evidence for it operating on the population concerned. Given our modelling suggests that predicted impacts may be reasonably robust to mis-specification of the relationship, the form of that relationship may not, necessarily, be important. Where density dependent processes are incorporated in population models, metrics should be estimated for the end of the project life cycle, so that the compensatory mechanisms of density dependence are accounted for. However, in small populations, depensatory density dependence may operate (Horswill & Robinson 2015). It is important that, where applicable, depensatory processes are incorporated into the population models in order to ensure any metrics are suitably precautionary.

We found that none of the metrics fulfil all of the criteria set out in section 2.4. Whilst we suggest that some metrics are better than others, we recognise that the selection of a metric with which to examine the population level consequences of an offshore wind farm may depend on the specific circumstances of the site concerned. With this in mind, in Table 6, we list each metric we have examined and outline the strengths and weaknesses of each approach, as well as suggesting how each should be used.

Notwithstanding the issues highlighted in Table 6, based on our analyses, we recommend that the ratio of impacted to unimpacted population growth rate (RI:U) and the ratio of impacted to unimpacted population size after 25 years (RI:U25) are likely to be the most generally useful metrics. However, RI:U varies over a limited range, meaning it may be difficult to distinguish between the between the population level consequences arising from impacts of different magnitudes, and RI:U25 has a non-linear relationship with impacts of increasing magnitude, making it harder to understand the consequences of incorrectly predicting the magnitude of any impact. For this reason, when assessing the population level consequences of impacts from offshore wind farms, we suggest referencing both metrics: the ratio of growth rates to quantify the consequence of impacts at a population level and the ratio of population sizes to present these impacts in an easily understandable context, for example:

"The impacts associated with *Offshore Wind Farm X* are predicted to result in the annual population growth rate at *Breeding Colony Y* declining from 0.994 to 0.967, a ratio of impacted to unimpacted population growth rate of 0.973. This means that after the 25 year life time of *Offshore Wind Farm X*, the population size of *Breeding Colony Y* is expected to be 51.5% of what it would have been in the absence of *Offshore Wind Farm X*."

These metrics should be derived using stochastic population models. Density dependence should only be incorporated into these models where there is suitable evidence of it acting on the population concerned at a site-specific level.

Table 6. Strengths, weaknesses and guidance on the usage of metrics presented in this report.

METRIC	STRENGTHS	WEAKNESSES	HOW TO USE/INTERPRET
Population Growth Rate (GR)	Easy to interpret. Relatively insensitive to mis-specification of demographic parameters.	No comparison with an unimpacted population means that, on its own, the metric cannot be used to assess the population level effects associated with impacts from offshore wind farms. Variability around estimates of population growth rate mean that it can be difficult to distinguish between the impact of an offshore wind farm and variation in the baseline population growth rate.	On its own, the population growth rate is not a meaningful metric with which to assess the population level effects of impacts arising from offshore wind farms. If selected, the population growth rate of the impacted population should be compared to the population growth rate of the unimpacted population. Care must be taken when comparing the growth rates of impacted and unimpacted populations. The overlapping confidence intervals may make it difficult to distinguish between the two, but a lack of a significant difference between impacted and unimpacted population growth rates does not necessarily reflect no population-level consequence over the 25 year life time of an offshore wind farm.

METRIC	STRENGTHS	WEAKNESSES	HOW TO USE/INTERPRET
Ratio of Impacted to unimpacted population growth rate (RI:U)	Consistent relationship between metric and the magnitude of any impact, making it easier to assess what the likely implication of incorrectly predicting the magnitude of any impact will be at the population level. Insensitive to mis-specification of demographic parameters and relatively insensitive to estimates of uncertainty surrounding these parameters. Insensitive to population trend, meaning that metric reflects only the impact of the offshore wind farm and not the status of the population concerned.	The metric varies over a limited numeric range, which combined with the overlapping confidence intervals of the two population GR's, may make it harder to infer population level effects from impacts of different magnitudes. The limited range of the metric makes it harder to assess what the effect of the offshore wind farm means in a population context.	RI:U can be used to assess the population level effect of impacts arising from offshore wind farms regardless of population status or trend. The metric should be presented as a median value with 95% confidence intervals. Thresholds for determining whether or not an impact is deemed acceptable will be subjective, but could be set with reference to the status or trend of the population concerned.
Ratio of impacted to unimpacted population size after 25 years (RI:U25)	Metric is easy to interpret in a population context. Clear relationship between metric and the magnitude of any impact making it easier to assess what the population level effects of any impact will be. Relatively insensitive to the estimate of uncertainty surrounding demographic parameters.	Sensitive to whether population is declining rather than stable/increasing. More sensitive to mis-specification of demographic parameters than GR or RI:U.	On its own, RI:U25 can be used to assess the population level effects of impacts arising from offshore wind farms for stable or increasing populations. However, the metric may also offer a useful context for the RI:U metric, regardless of population trend. The metric should be presented as a median value with 95% confidence intervals. Thresholds for determining whether or not an impact is deemed acceptable will be subjective, but could be set with reference to the status or trend of the population concerned.

METRIC	STRENGTHS	WEAKNESSES	HOW TO USE/INTERPRET
Probability of growth rate being <1 P(GR<1)	Produces a metric which is intuitive and easy to understand.	No comparison with an unimpacted population means that, on its own, the metric cannot be used to assess the population level effects associated with impacts from offshore wind farms. Sensitive to mis-specification of adult survival rate. Sensitive to whether population is increasing or stable/declining. Where population is stable/declining the metric only varies over a limited range, making it more difficult to discern population level effects associated with different impacts.	Needs comparison with an unimpacted population to understand the population level effect associated with any wind farm. Only suitable for use in situations where population was increasing prior to the construction of offshore wind farms. If the metric is to be used, robust data describing adult survival rates, at a site specific level, are required. Thresholds for determining whether or not an impact is deemed acceptable will be subjective, but could be set with reference to the status or trend of the population concerned.
Change in probability of growth rate being <1 dP(GR<1)	Easy to understand metric that quantifies the change in probability of a population declining as a result of an offshore wind farm.	Sensitive to population trend. Sensitive to mis-specification of demographic parameters.	Not suitable for use in populations where the populations were declining prior to the construction of an offshore wind farm, where the P(GR<1) is already close to 1. If metric is to be used, robust, site-specific data describing demographic parameters are required. Thresholds for determining whether or not an impact is deemed acceptable will be subjective, but could be set with reference to the status or trend of the population concerned.

METRIC	STRENGTHS	WEAKNESSES	HOW TO USE/INTERPRET
Probability that population is below initial size at any point in time $P(p<p_0)$	Takes into account the fact that populations may recover from initial declines over the lifetime of an offshore wind farm.	No comparison with an unimpacted population means that, on its own, the metric cannot be used to assess the population level effects associated with impacts from offshore wind farms. Sensitive to whether population is increasing or stable/declining prior to offshore wind farm construction. Sensitive to mis-specification of adult survival rate.	Needs comparison with an unimpacted population to understand the population level effect associated with any wind farm. Only suitable for use in situations where population was increasing prior to the construction of offshore wind farms. If the metric is to be used, robust data describing adult survival rates, at a site specific level, is required. Thresholds for determining whether or not an impact is deemed acceptable will be subjective, but could be set with reference to the status or trend of the population concerned.
Probability of a 25% population decline $P(ld<0.25)$	Relatively easy to understand metric which could, potentially, be related to established conservation assessments (e.g. Birds of Conservation Concern Red/Amber lists – Eaton *et al* 2009).	No comparison with an unimpacted population means that, on its own, the metric cannot be used to assess the population level effects associated with impacts from offshore wind farms. Sensitive to whether population is increasing or stable/declining prior to offshore wind farm construction. Sensitive to mis-specification of demographic parameters.	Needs comparison with an unimpacted population to understand the population level effect associated with any wind farm. Only suitable for use in situations where population was increasing prior to the construction of offshore wind farms. If the metric is to be used, robust data describing adult survival rates, at a site specific level, is required. Thresholds for determining whether or not an impact is deemed acceptable will be subjective, but could be set with reference to the status or trend of the population concerned.

METRIC	STRENGTHS	WEAKNESSES	HOW TO USE/INTERPRET
Change in probability of a 25% population decline P(Id<0.25)	Easy to understand metric that quantifies the change in probability of a population declining by 25% as a result of an offshore wind farm.	Sensitive to whether population is stable or increasing/declining prior to offshore wind farm construction. Sensitive to mis-specification of demographic parameters.	Not suitable for use in populations where the populations were declining prior to the construction of an offshore wind farm, where the P(GR<1) is already close to 1. If metric is to be used, robust, site-specific data describing demographic parameters are required.
Probability impacted population 25% below unimpacted population (P(I<25))	Straightforward, easy to understand comparison of impacted and unimpacted populations. Can be easily related to criteria used to assess conservation status of species (i.e. Birds of Conservation Concern – Eaton *et al* 2009).	Some sensitivity to underlying population trend. Sensitive to mis-specification of demographic parameters.	A difference of 25% between the impacted and unimpacted populations is a subjective threshold. Careful consideration should be given as to whether this is an appropriate threshold for the population concerned, drawing on information about the importance and status of the population concerned. If metric is to be used, robust, site-specific data describing demographic parameters are required. Sensitivity to inclusion and form of density dependence, mean density dependent models should only be used where there is strong, clear evidence for it in the population.

METRIC	STRENGTHS	WEAKNESSES	HOW TO USE/INTERPRET
Probability that impacted population growth rate is 2.5% less than unimpacted population growth rate P(lgr>2.5)	Metric which assesses the impacted population growth rate relative to the unimpacted population growth rate.	May be difficult to understand in a population context. In practice it may be statistically difficult to detect a 2.5% drop in the population growth between the impacted and unimpacted populations. It is possible to consider a greater change, but more severe impacts would be required to detect such a change. Sensitive to whether population is stable/increasing or declining prior to offshore wind farm construction. Sensitive to mis-specification of demographic parameters.	Not suitable for use in populations where the populations were declining prior to the construction of an offshore wind farm. If metric is to be used, robust, site-specific data describing demographic parameters are required. Sensitivity to inclusion and form of density dependence, mean density dependent models should only be used where there is strong, clear evidence for it in the population.

METRIC	STRENGTHS	WEAKNESSES	HOW TO USE/INTERPRET
Overlap of Impacted and Unimpacted Populations (OI:U)	Straightforward comparison which enables understanding of how similar the outputs from models of the impacted and unimpacted populations are.	Sensitive to population trend. Sensitive to mis-specification of demographic parameters. Sensitive to estimates of uncertainty surrounding the demographic parameters. Value may depend on the number of simulations used in the demographic models used to derive metric.	Sensitivity to population trend means that the metric should only be used where there is a clear understanding of the status of the population concerned. If metric is to be used, robust, site-specific data describing demographic parameters, and the uncertainty surrounding these parameters, are required. Sensitivity to inclusion and form of density dependence, mean density dependent models should only be used where there is clear evidence for it in the population. Careful analysis is needed to ensure sufficient simulations are used in demographic models.

Clarifying the direction of sensitivity could be an important consideration for those wishing to decide how appropriate it is to use the metric. In this report, the results for the population growth rate metric provide a detailed account of how to interpret the presentation of the sensitivity analyses. Subsequent metrics are presented in a more concise manner.

3.2.3 Population Growth Rate (GR)

By considering the growth rate of the population in the presence of an offshore wind farm, it will be possible to consider whether the population will remain stable, increase or decrease through the lifetime of the project. A value of 1 indicates a stable population, <1 a declining population and >1 an increasing population.

Growth rate is calculated as a mean rate over the study period:

$$\left(\frac{End\ Population\ Size}{Start\ Population\ Size}\right)^{1/Nyears}$$

Mean annual growth rate over the study period is used in preference to the year-specific figures because, as a result of year to year variation in demographic parameters influencing population size, this approach gives a value that is more representative of any impacts at a population level. This definition is used for all subsequent metric which rely on a population growth rate. With this metric, a value of 1 indicates a stable population, a value <1 indicates a declining population and a value >1 indicates an increasing population.

Initial Results

Outputs from the sensitivity analyses of each metric are presented in a series of graphs and tables. Graphs are purposefully presented on the same scale for each metric in order to aide comparison between different methods. The first plots in each section (Fig. 14) show how the metric changes in response to impacts ranging from a 0 – 40% increase in mortality and a 0 – 40% reduction in productivity.

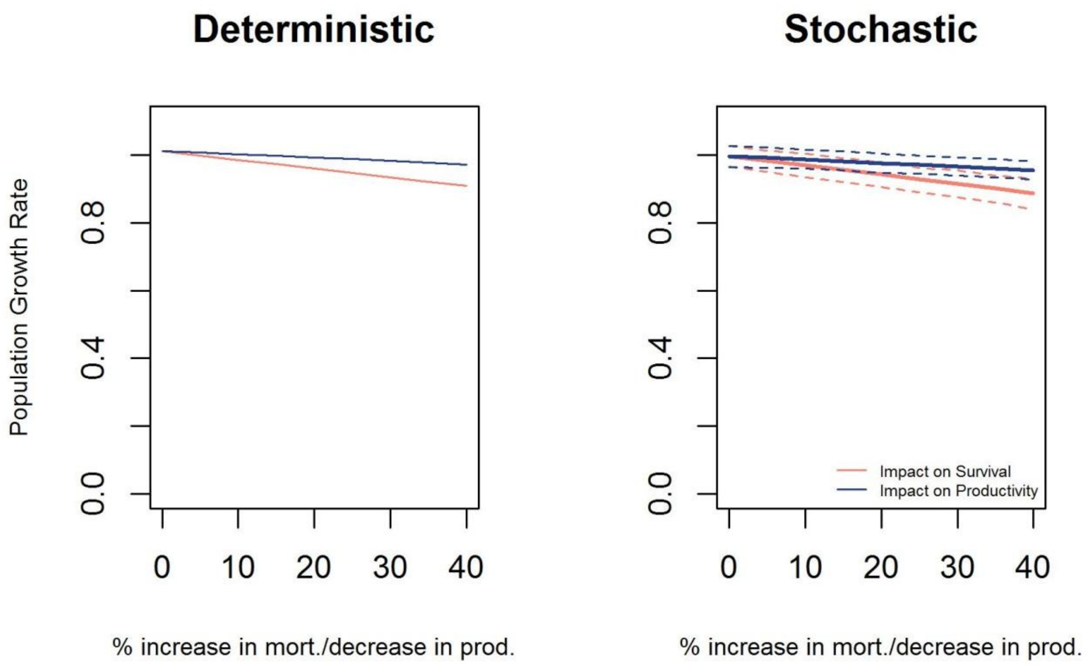

Figure 14. Impact of up to a 40% increase in mortality or up to a 40% decrease in productivity on the growth rate of a population of seabirds with an r-selected life history strategy and a stable population trajectory derived from deterministic and stochastic models (95% CIs for the latter given by broken lines).

Where appropriate, these figures are presented for the metric derived from both deterministic and stochastic models. If metrics are based on mean or median values, plots illustrating those derived from stochastic models are bounded by the 95% confidence intervals (shown by broken lines). In these plots, red illustrates the metric derived when an increase in mortality is considered, whilst blue illustrates the metric derived when a reduction in productivity is considered. Ideally, these plots should show a linear response to impacts of increasing magnitude, with a strong gradient. A non-linear response may make it harder to infer what the consequences of inaccurately estimating the impact arising from an offshore wind farm would be as this would depend on the magnitude of the impact concerned.

As might be expected, increases in mortality or decreases in productivity cause the population growth rate (GR) to decline over the life time of an offshore wind farm (Fig. 14). This decline is more severe when the impact is on survival than when the impact is on productivity, consistent with what would be expected amongst a group of species with high survival and relatively low productivity. Consistent with the finding that stochastic models are likely to result in more severe declines than deterministic models (Lande *et al* 2003), predicted GR over 25 years declined from 1.010 to 0.903 using a deterministic model and assuming up to a 40% increase in mortality, and from 0.994 (95% CIs 0.96 – 1.03) to 0.877 (95% CIs 0.82 – 0.92) using a stochastic model. Assuming up to a 40% decrease in productivity, GR would decline to 0.971 using a deterministic model and 0.954 (95% CIs 0.92 – 0.98) using a stochastic model. It should be noted that the one standard deviation around the population growth rate assuming no offshore wind farm impact overlaps with one standard deviation around a population with up to a 20% increase in mortality, suggesting that using population growth rate to estimate the population level impact of a development may only be possible where these impacts are large (>20% increase in mortality).

Sensitivity to life history strategy, population trend and density dependence

The following table, Table 7, illustrates how the metric would change (assuming a 10 or 20% increase in mortality or reduction in productivity) if populations were stable, increasing, decreasing, regulated by density dependence or based on a stable population of a K-selected species rather than an r-selected species. Ideally, values of the metric should be identical for similar levels of impact, regardless of whether the population concerned is increasing, decreasing or stable. The metric should also be similar when density dependent regulation of productivity or survival is introduced into the population models. It is expected that the metric would differ between stable populations of r or K-selected species, reflecting the different life history strategies of these species.

As might be expected, if a population were increasing prior to any wind farm impact, GR is higher for an impacted population than would be the case if the population were stable prior to the wind farm impact. Similarly, were the population already declining prior to the wind farm impact, the impacted GR is lower than would be the case if the population were stable prior to the wind farm impact (Table 7). As expected, if density dependent processes operate on the population this may mitigate the any impacts arising from the wind farm (Table 7). It appears to make relatively little difference whether density dependence is assumed to operate on productivity or survival. These results confirm that where there is uncertainty over density dependent processes in a population, assuming no density dependence is present is likely to be the most precautionary assumption, unless there is a depensatory density dependent relationship.

Changes in GR appear to be more pronounced for species which have a more r-selected life history than for those which have a more K-selected strategy (Table 7). This may be linked to the generation times of the respective species meaning that there is a greater chance of changes in populations of r-selected species (which have shorter generation times) becoming apparent over the 25 year lifetime of an offshore wind farm than is the case for K-selected species (which have longer generation times) in response to a given magnitude of impct.

Table 7. Population growth rates (95% CIs) resulting from a 10%, 20% or 40% increase in mortality or a 10% decrease in productivity estimated from a stochastic model and assuming an increasing, stable or decreasing population of an r-selected seabird species, a stable population of a K-selected seabird species and a stable population of an r-selected species with density dependent regulation of survival or productivity.

	Unimpacted population	Survival			Productivity		
		10%	20%	40%	10%	20%	40%
Increasing	1.073 (1.037 – 1.109)	1.044 (1.006 – 1.080)	1.012 (0.966 – 1.053)	0.944 (0.881 – 1.001)	1.061 (1.027 – 1.095)	1.049 (1.017 - 1.082)	1.022 (0.993 – 1.051)
Stable	0.996 (0.961 – 1.028)	0.969 (0.933 – 1.004)	0.942 (0.907 - 0.976)	0.889 (0.843 – 0.928)	0.988 (0.958 – 1.016)	0.980 (0.951 – 1.008)	0.962 (0.935 – 0.987)
Decreasing	0.952 (0.923 – 0.982)	0.922 (0.888 – 0.957)	0.891 (0.851 – 0.927)	0.829 (0.778 – 0.875)	0.945 (0.913 – 0.974)	0.938 (0.907 – 0.966)	0.922 (0.894 – 0.949)
K selected	0.998 (0.978 – 1.017)	0.987 (0.966 – 1.007)	0.978 (0.957 – 0.998)	0.958 (0.935 – 0.979)	0.995 (0.977 – 1.014)	0.992 (0.975 – 1.011)	0.987 (0.971 – 1.003)
Density Dependent Survival	1.002 (0.961 – 1.041)	0.986 (0.944 – 1.029)	0.972 (0.925 – 1.018)	0.939 (0.987 – 0.874)	0.997 (0.960 – 1.039)	0.993 (0.958 – 1.033)	0.984 (0.945 – 1.023)
Density Dependent Productivity	0.997 (0.975 – 1.020)	0.983 (0.958 – 1.007)	0.965 (0.933 – 0.994)	0.916 (0.857 – 0.961)	0.994 (0.972 – 1.016)	0.990 (0.969 – 1.009)	0.982 (0.962 – 0.999)

Sensitivity to mis-specification of demographic parameters

The next figure (Fig. 15) and tables (Tables 8 and 9) illustrate how sensitive each metric is to the mis-specification of each demographic parameter in the models. For this, we consider a 1 percentage change in each parameter and what change this makes, in percentage terms, to the value of the metric. Figure 13, illustrates this graphically. It is important to note that the y-axis of this barchart is **not** linear. To aid interpretation, we use a horizontal, red line to illustrate the point at which a 1% mis-specification of the demographic parameter concerned would correspond to a 1 percentage change in the metric (this is purely to help interpretation, and should not be taken to imply this represents any sort of threshold, acceptable or otherwise). Tables 8 and 9 tabulate the values in this figure. Ideally, each of these values should be close to 0, indicating that mis-specification of the input parameters has little impact on the value of the derived metric. Where appropriate, Tables 8 and 9 and Figure 15 are shown for both the deterministic and stochastic models.

Of the demographic parameters considered, GR appears to be most sensitive to the mis-specification of the adult survival rate (Fig. 13 and Tables 8 and 9). However, this sensitivity is relatively minor, with a mis-specification of 1% in the adult survival rate resulting in less than a 1 percentage change in the population growth rate (Fig. 13 & Tables 8 and 9). However, the sensitivity of GR to the adult survival rate appears to increase as the magnitude of the impact increases (Tables 8 and 9). These results are consistent, regardless of whether a deterministic or stochastic model is used.

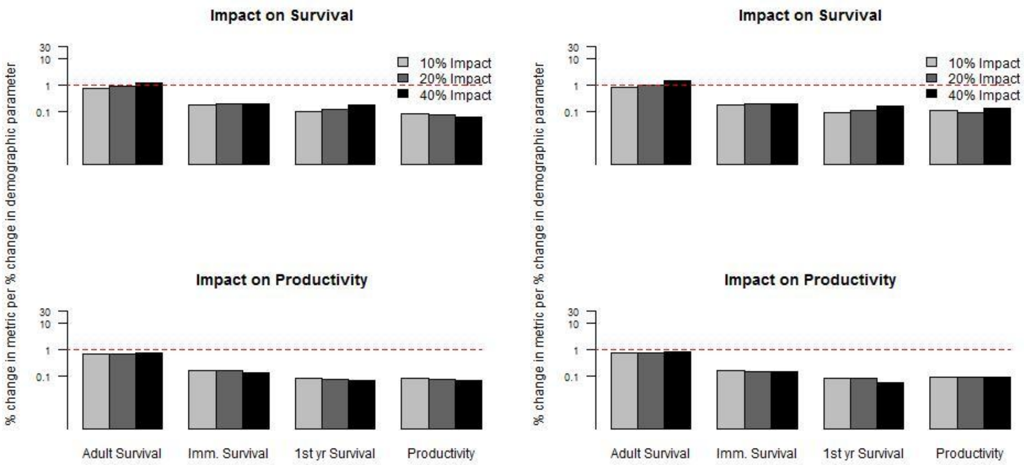

Figure 15. Percentage change in population growth rate per percentage change in adult survival, immature survival, first year survival and productivity for offshore wind farm impacts on survival and productivity from deterministic (left) and stochastic (right) models. Note that the Y-axis is on a non-linear scale. Data tabulated in Tables 8 and 9.

Table 8. Influence of a 1% mis-specification of each demographic parameter on the growth rate of an r-selected seabird species estimated from deterministic and stochastic models (assessed as % change in metric), assuming a 10%, 20% or 40% increase in mortality. Illustrated graphically in Figure 15.

	Deterministic			Stochastic		
	10%	20%	40%	10%	20%	40%
Adult Survival	0.75	0.89	1.25	0.84	0.97	1.32
Immature Survival	0.18	0.19	0.20	0.18	0.19	0.16
Chick Survival	0.12	0.12	0.17	0.10	0.10	0.13
Productivity	0.08	0.08	0.06	0.08	0.08	0.10

Table 9. Influence of a 1% mis-specification of each demographic parameter on the growth rate (as assessed by % change in metric) of an r-selected seabird species estimated from deterministic and stochastic models, assuming a 10%, 20% or 40% reduction in productivity. Illustrated graphically in Figure 15.

	Deterministic			Stochastic		
	10%	20%	40%	10%	20%	40%
Adult Survival	0.65	0.68	0.75	0.74	0.75	0.80
Immature Survival	0.17	0.17	0.14	0.16	0.16	0.14
Chick Survival	0.08	0.08	0.07	0.09	0.08	0.06
Productivity	0.08	0.08	0.07	0.09	0.09	0.06

Sensitivity to form of density dependence

The final figure, Figure 16, illustrates the impact of mis-specifying the parameters used to estimate the density dependent response of survival or productivity on the value of the metric derived. Here, ideally, the lines on each plot should be flat, indicating that an error in estimating these parameters has little or no impact on the value of the metric derived.

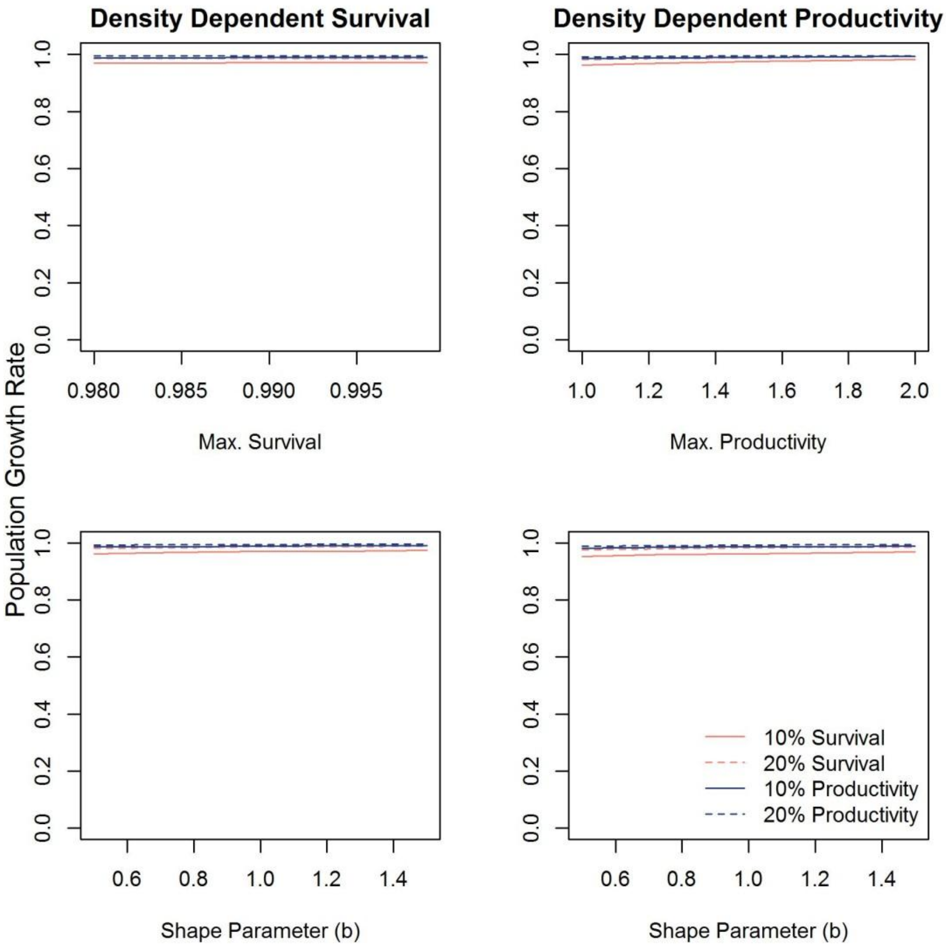

Figure 16. Impact of mis-specifying the shape parameter and maximum survival or productivity rate in a stable population of an r-selected seabird when using a stochastic model with density dependent regulation of survival or productivity.

If density dependence is introduced into the models, the impacted GR does not appear to be particularly sensitive to either mis-specification of the shape parameter or mis-specification of the maximum survival or productivity rate.

Metric overview

GR shows a consistent decline as the magnitude of any impact increases. This decline is more severe where survival is impacted than where productivity is impacted. Stochastic and deterministic models produce similar results although stochastic models may be more precautionary as they result in stronger declines. Whilst growth rates derived from stochastic models are lower than those derived from deterministic models, they may better reflect biological reality by accounting for the changeable nature of the environment in which these species live. However, when using stochastic models it is only possible to clearly distinguish between the growth rates of an impacted and unimpacted population when the predicted impacts of an offshore wind farm are relatively severe. This means that more moderate changes in the population growth rate, which may still have a significant population level impact over the lifetime of a project, may not be identified. GR, as a metric, is relatively insensitive to mis-specification of the input demographic parameters using either a deterministic or stochastic model and regardless of whether density dependence is incorporated.

3.2.4 Ratio of the Median Impacted to Unimpacted Population Growth Rate (RI:U)

Considering only the growth rate of a population 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. By comparing the growth rate of the population in the presence of a wind farm to that expected in the absence of a wind farm it may be possible to demonstrate what impact the development is having on a population.

This metric is on a scale from 0 – 1, with 1 indicating the impacted population growth rate is the same as the unimpacted growth rate (i.e. no population-level consequence) and values close to 0 indicating a large difference between the impacted and unimpacted population growth rates (i.e. a strong population-level consequence). Changes in the metric reflect increases or decreases in the impacted population growth rate.

Initial Results

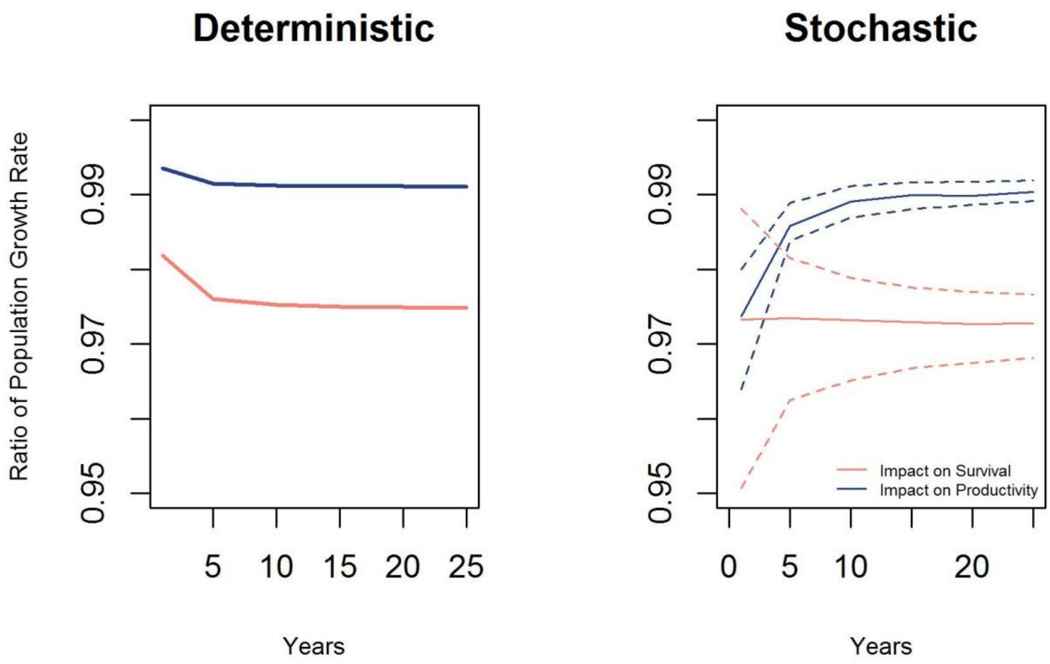

Figure 17. Change in value of the ratio of the impacted to unimpacted population growth rate through time calculated using a deterministic (left) and stochastic (right) model and assuming a 10% increase in mortality or a 10% reduction in productivity. Broken lines indicate 95% confidence intervals. Note, first point is shown for year 1, not year 0.

The ratio of the median impacted to unimpacted population growth rate (RI:U) can be calculated for any point over the lifetime of a development. As time increases, the ratio of the impacted to unimpacted population growth rate stabilises, suggesting a new stable age-structure is reached (Fig. 17). Impacts on survival result in a larger change than those on productivity. Using the stochastic model, the uncertainty surrounding the ratio appears to decrease through time as a result of regression to the mean. Given this, it makes sense to use the ratio estimated at the end of the lifetime of the project, in this case after 25 years, when predicted impacts will be at their greatest. For this reason, subsequent discussion of the metric is based on RI:U after 25 years. Apparent differences in the value of the metric derived from different models relate to the fact the stochastic value comes from multiple simulations whereas the deterministic value comes from a single calculation. Furthermore, as described above, values from stochastic models are slightly lower than those from deterministic models as a result of accounting for variation in demographic rates.

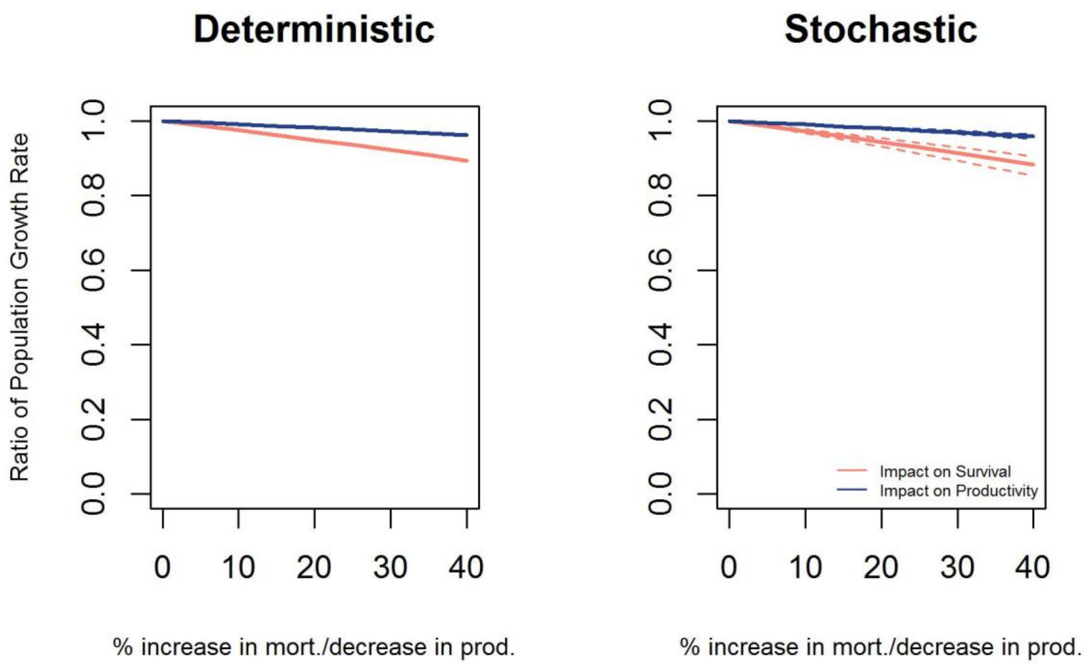

Figure 18. Impact of up to a 40% increase in mortality or up to a 40% decrease in productivity on the ratio of the median growth rate of an impacted and unimpacted population of seabirds with an r-selected life history strategy and a stable population trajectory derived from deterministic and stochastic models (95% CIs given by broken lines).

RI:U decreases as the magnitude of any impact increases (Fig. 18). Ratios are similar, regardless of whether a stochastic or deterministic model is used. Assuming a 40% increase in mortality results in a ratio of 0.894 using the deterministic model and 0.882 using the stochastic model. However, given that the growth rates of impacted and unimpacted populations estimated from the stochastic models overlap for impacts of up to 20% on survival or up to 40% on productivity (see above, section 3.2.1 Table 7), these values must be interpreted with caution.

Sensitivity to life history strategy, population trend and density dependence

In contrast to GR, RI:U appears to be relatively insensitive to the underlying population trend, with extremely similar values obtained for increasing and stable populations, and only a small change where the population is decreasing (Table 10). As expected, if density dependent processes operate on the population this may mitigate the impact of any impacts arising from the wind farm (Table 10). It appears to make relatively little difference whether density dependence is assumed to operate on productivity or survival. These results confirm that where there is uncertainty over density dependent processes in a population, assuming no density dependence is present is likely to be the most precautionary assumption.

Table 10. Ratio of impacted to unimpacted population growth rates (95% CIs) resulting from a 10%, 20% or 40% increase in mortality or a 10%, 20% or 40% decrease in productivity estimated from a stochastic model and assuming an increasing, stable or decreasing population of an r-selected seabird species, a stable population of a K-selected seabird species and a stable population of an r-selected species with density dependent regulation of survival or productivity. Values close to 1 indicate no impact from offshore wind farm, values close to 0 indicate strong impact from offshore wind farm.

	Survival			Productivity		
	10%	20%	40%	10%	20%	40%
Increasing	0.973 (0.966 – 0.978)	0.945 (0.929 – 0.956)	0.882 (0.840 – 0.908)	0.988 (0.987 – 0.989)	0.975 (0.973 – 0.977)	0.946 (0.941 – 0.951)
Stable	0.973 (0.965 – 0.976)	0.943 (0.931 – 0.953)	0.881 (0.851 – 0.903)	0.990 (0.989 – 0.991)	0.980 (0.977 – 0.982)	0.958 (0.952 – 0.963)
Decreasing	0.966 (0.971 – 0.958)	0.930 (0.912 – 0.942)	0.853 (0.801 – 0.881)	0.991 (0.989 – 0.991)	0.981 (0.978 – 0.983)	0.959 (0.952 – 0.965)
K selected	0.989 (0.987 – 0.990)	0.978 (0.974 – 0.981)	0.958 (0.949 – 0.964)	0.995 (0.994 – 0.996)	0.990 (0.989 – 0.992)	0.980 (0.976 – 0.984)
Density Dependent Survival	0.985 (0.983 – 0.991)	0.969 (0.961 – 0.980)	0.933 (0.896 – 0.948)	0.994 (0.994 – 0.995)	0.988 (0.987 – 0.990)	0.972 (0.969 – 0.976)
Density Dependent Productivity	0.989 (0.978 – 0.994)	0.971 (0.949 – 0.986)	0.934 (0.874 – 0.965)	0.996 (0.993 – 0.997)	0.991 (0.987 – 0.994)	0.984 (0.977 – 0.988)

As highlighted previously (section 3.2.1), the growth rates of K-selected seabird species appear to be more resilient to impacts from offshore wind farms as the RI:U is higher than that for r-selected species (Table 10).

Sensitivity to mis-specification of demographic parameters

Of the demographic parameters considered, RI:U appears to be most sensitive to the mis-specification of the adult survival rate (Fig. 19 & Tables 11 and 12). However, this sensitivity is relatively minor, with a mis-specification of 1% in the adult survival rate resulting in less than a 1 percentage change in the population growth rate (Fig. 19 & Tables 11 and 12). However, the sensitivity of the ratio of population growth rate to the adult survival rate appears to increase as the magnitude of the impact increases (Tables 11 and 12). These results are consistent, regardless of whether a deterministic or stochastic model is used.

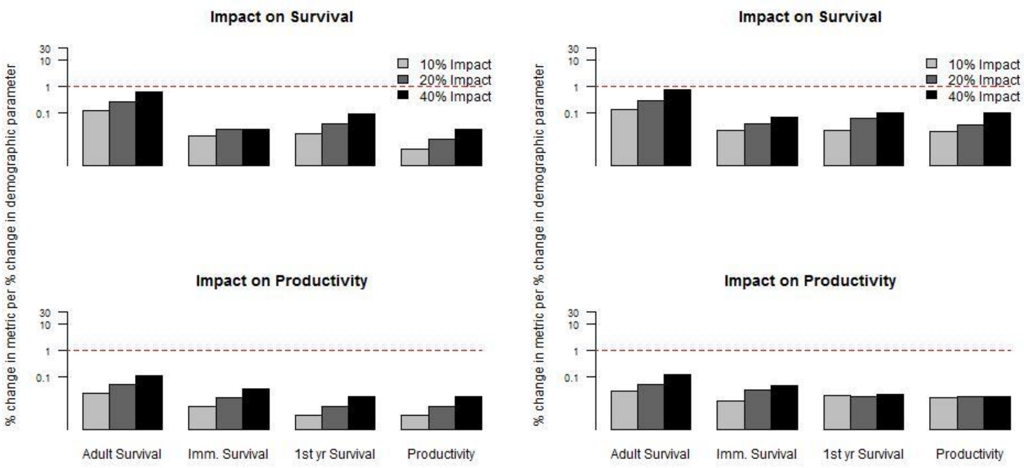

Figure 19. Percentage change in the ratio of impacted to unimpacted population growth rate per percentage change in adult survival, immature survival, first year survival and productivity for offshore wind farm impacts on survival and productivity from deterministic (left) and stochastic (right) models. Note that the Y-axis is on a non-linear scale. Data tabulated in Tables 10 and 11.

Table 11. influence of a 1% mis-specification of each demographic parameter on the ratio of the growth rate of an impacted to unimpacted population (as assessed by % change in metric) of an r-selected seabird species estimated from a deterministic and stochastic model, assuming a 10%, 20% or 40% increase in mortality. Illustrated graphically in Figure 19.

	Deterministic			Stochastic		
	10%	20%	40%	10%	20%	40%
Adult Survival	0.12	0.26	0.62	0.14	0.30	0.74
Immature Survival	0.01	0.01	0.02	0.02	0.04	0.07
Chick Survival	0.02	0.04	0.09	0.02	0.06	0.10
Productivity	<0.01	0.01	0.02	0.02	0.04	0.11

Table 12. Influence of a 1% mis-specification of each demographic parameter on the ratio of the growth rate of an impacted to unimpacted population (as assessed by % change in metric) of an r-selected seabird species estimated from a deterministic and stochastic model, assuming a 10%, 20% or 40% reduction in productivity. Illustrated graphically in Figure 19.

	Deterministic			Stochastic		
	10%	20%	40%	10%	20%	40%
Adult Survival	0.02	0.05	0.11	0.03	0.05	0.11
Immature Survival	<0.01	0.01	0.04	0.01	0.03	0.04
Chick Survival	<0.01	0.01	0.02	0.02	0.02	0.02
Productivity	<0.01	0.01	0.02	0.02	0.02	0.03

Sensitivity to form of density dependence

If density dependence is introduced into the models, RI:U does not appear to be particularly sensitive to either mis-specification of the shape parameter or mis-specification of the maximum survival or productivity rate (Fig. 20).

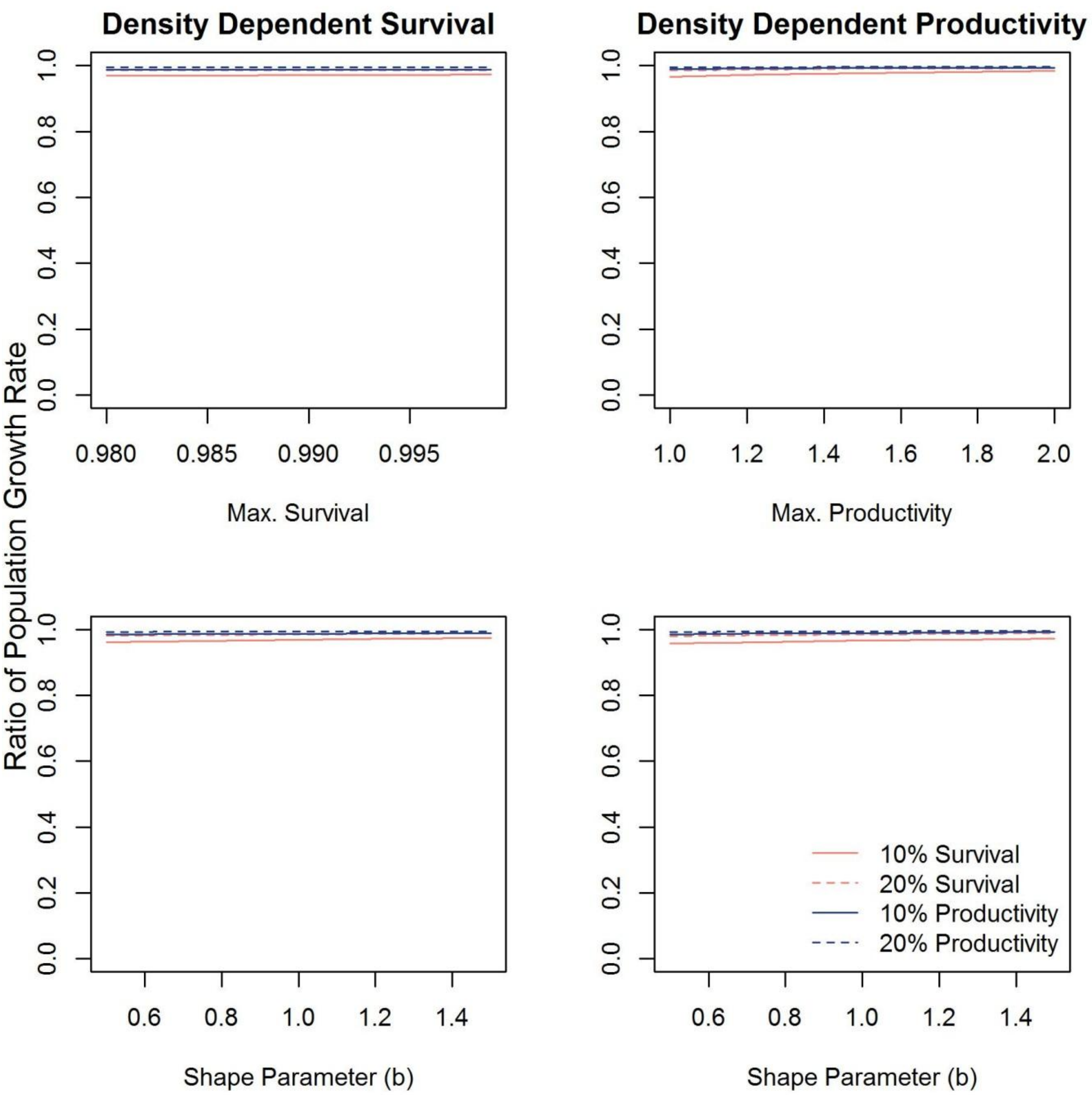

Figure 20. Impact of mis-specifying the shape parameter and maximum survival or productivity rate in a stable population of an r-selected seabird when using a stochastic model with density dependent regulation of survival or productivity.

Additional Analysis

Since initial results suggested that this metric may be of use in estimating the population level effects of impacts arising from offshore wind farms, additional analyses were undertaken.

RI:U can be estimated at any point over the lifetime of a project. However, it is unclear whether sensitivity to mis-specification of demographic parameters is constant through time, i.e. if we estimate RI:U after 5 or 10 years, is it more important that we use accurate survival estimates than if we estimate it after 25 years. In order to understand this, we investigate how mis-specification of adult survival rates may affect the metric at different points in time (5 and 10 years post-construction). These results suggest that the sensitivity of the metric to mis-specification of demographic parameters is likely to be broadly similar regardless of when in a project’s lifetime it is calculated (Fig. 21 & Table 13). These results imply that the metric may be interpreted with the same degree of confidence regardless of whether it is estimated 5, 10 or 15 years post-construction.

Table 13. Impact of a 1% mis-specification of adult survival rate on the ratio of the growth rate of an impacted to unimpacted population after 5 or 10 years for an r-selected seabird species estimated from a stochastic model, assuming a 10%, 20% or 40% increase in mortality or a 10%, 20% or 40% reduction in productivity. Illustrated graphically in Figure 19.

	Impact on Survival			Impact on Productivity		
	10%	20%	40%	10%	20%	40%
After 5 years	0.12	0.27	0.59	0.04	0.06	0.13
After 10 Years	0.13	0.28	0.67	0.03	0.06	0.12

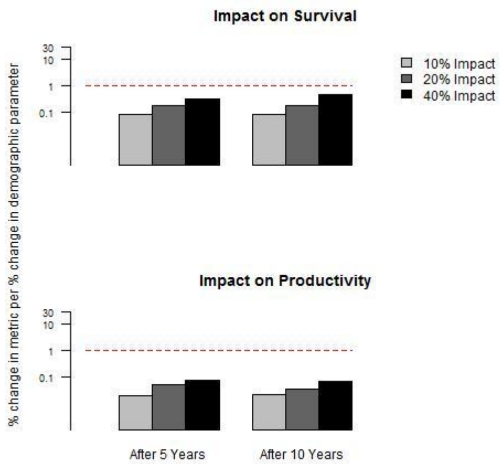

Figure 21. Percentage change in the ratio of impacted to unimpacted population growth after 5 and 10 years rate per percentage change in adult survival for offshore wind farm impacts on survival and productivity from a stochastic model. Note that the Y-axis is on a non-linear scale. Data tabulated in Table 13.

Additional analyses revealed that RI:U showed some sensitivity to the extent of uncertainty surrounding the demographic parameters used in population models (Fig. 22). However, this sensitivity did not appear to be greater than the sensitivity of the metric to mis-specification of demographic parameters. These analyses show that in populations where there is greater uncertainty surrounding the demographic parameters, RI:U is lower, implying a stronger population-level consequence, than is the case where there is less uncertainty surrounding the demographic parameters. This is because the wider confidence intervals surrounding the demographic parameters enable more extreme values to be selected. In the case of adult survival rates, 0.89 in this analysis, there is more scope for lower values to be selected than higher values (as the maximum possible rate is 1). Consequently, when averaged over multiple simulations the population with the wider confidence limits surrounding adult survival will show a greater impact than the population with narrower confidence limits.

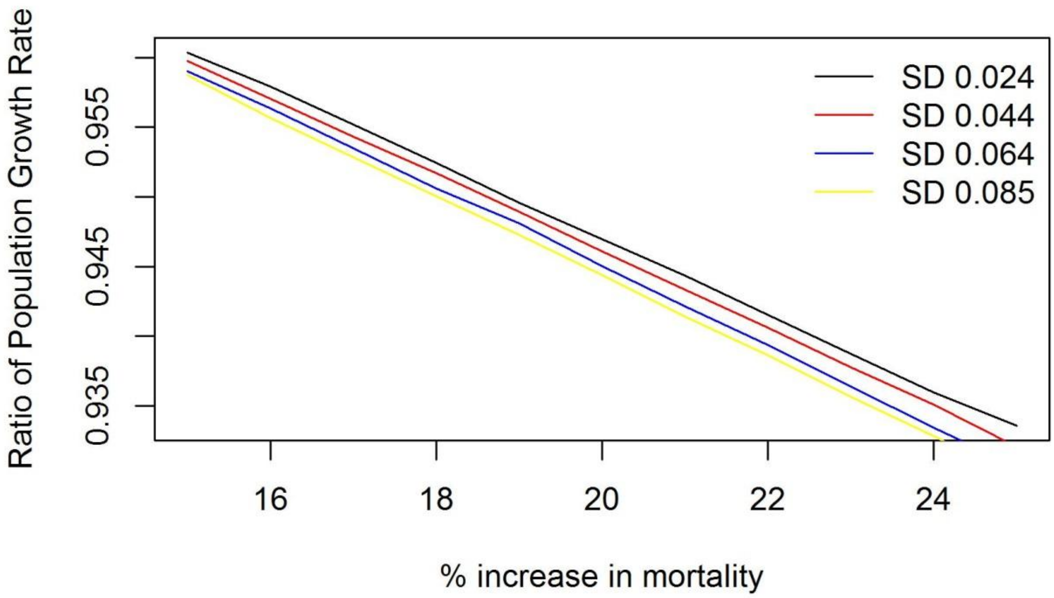

Figure 22. Impact of 15% to 25% increase in mortality on the ratio of impacted to unimpacted population growth rate for a population of seabirds with an r-selected life history strategy and a stable population trajectory, assuming an adult survival rate of 0.89 and standard deviations of 0.024, 0.044, 0.064 and 0.085.

Sensitivity to population trend may vary depending on whether or not density dependence is incorporated in the population models used to estimate RI:U. We therefore estimated this metric using a model assuming a density dependent impact on productivity and increasing, stable and declining populations. As with density independent models, when derived from density dependent models, RI:U is insensitive to whether the population is stable or increasing, but does appear to be sensitive to whether the population is declining (Table 14), i.e. including a density dependent response does not change how the metric behaves.

Table 14. Sensitivity of the ratio of impacted to unimpacted population growth rate after 25 years to population trend, derived from stochastic models of an r-selected seabird assuming stable, increasing or declining populations with density dependent regulation of productivity.

	Impact on survival			Impact on productivity		
	10%	20%	40%	10%	20%	40%
Increasing	0.989 (0.982 – 0.993)	0.976 (0.958 – 0.985)	0.932 (0.866 – 0.962)	0.994 (0.992 – 0.995)	0.988 (0.984 – 0.991)	0.977 (0.967 – 0.982)
Stable	0.989 (0.978 – 0.994)	0.971 (0.949 – 0.986)	0.934 (0.874 – 0.965)	0.996 (0.993 – 0.997)	0.991 (0.987 – 0.994)	0.984 (0.977 – 0.988)
Declining	0.977 (0.966 – 0.986)	0.950 (0.922 – 0.968)	0.865 (0.446 – 0.916)	0.996 (0.995 – 0.997)	0.993 (0.990 – 0.995)	0.986 (0.980 – 0.990)

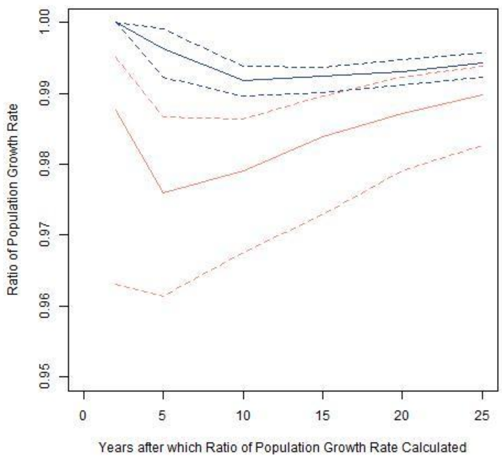

Figure 23. Change in value of the ratio of the impacted to unimpacted population growth rate through time calculated using a stochastic model with density dependent regulation of productivity and assuming a 10% increase in mortality (red lines) or a 10% reduction in productivity (blue lines). Broken lines indicate 95% confidence intervals.

In contrast to density independent models, when using a density dependent model (Fig. 23), after an initial decrease in RI:U over the first five years of a project lifetime, the metric increases again over the remaining time period. This suggests that, over time, density dependent mechanisms are compensating for the impacts associated with the offshore wind farm and that, as a result, the growth rates of the impacted and unimpacted populations become more similar (i.e. the metric takes values closer to one).

RI:U can be estimated from a density dependent model at any point over the lifetime of a project. However, it is unclear how incorporating density dependence into the models will affect sensitivity to mis-specification of demographic parameters. To test this, we focus on adult survival rate, which previous analyses showed was the parameter the metric was most sensitive to, and consider the sensitivity of the metric to mis-specification after 5, 10 and 25 years using a stochastic model with density dependent regulation of productivity. These results suggest that when density dependence is incorporated into population models, the metric may be less sensitive to mis-specification of demographic parameters than is the case for density independent models. These results are consistent regardless of whether the metric is estimated 5,10 or 25 years post-construction (Table 15, Figure 24).

Table 15. Influence of a 1% mis-specification of adult survival rate on the ratio of the growth rate (as assessed using by % change in metric) of an impacted to unimpacted population after 5, 10 or 25 years for an r-selected seabird species estimated from a stochastic model with density dependent regulation of productivity, assuming a 10%, 20% or 40% increase in mortality or a 10%, 20% or 40% reduction in productivity.

	Impact on Survival			Impact on Productivity		
	10%	20%	40%	10%	20%	40%
After 5 years	0.07	0.17	0.44	0.02	0.03	0.08
After 10 Years	0.09	0.22	0.54	0.02	0.03	0.05
After 25 Years	0.08	0.22	0.67	0.02	0.03	0.04

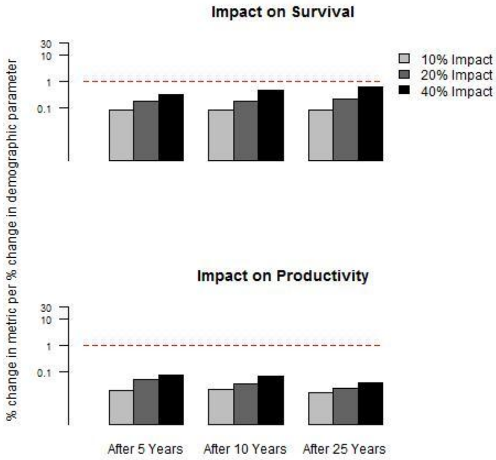

Figure 24. Percentage change in the ratio of impacted to unimpacted population growth after 5, 10 and 25 years rate per 1 percentage change in adult survival for offshore wind farm impacts on survival and productivity from a stochastic model with density dependent regulation of productivity. Note that the Y-axis is on a non-linear scale.

Metric overview

RI:U shows a consistent decline as the magnitude of any impact increases. This decline is more severe where survival is impacted than where productivity is impacted. Stochastic and deterministic models produce similar results, although stochastic models may be more precautionary as they predict stronger declines. However, as highlighted previously this metric must be interpreted with caution given the overlapping confidence intervals recorded in the population growth rates of unimpacted populations and populations exposed to moderate impacts as a result of an offshore wind farm (see Figure 14, section 3.2.3). This may make interpreting the outputs of this metric difficult, particularly when combined with the fact that it only varies over a fairly limited range. Despite this, it is important to note that the metric is relatively insensitive to mis-specification of demographic parameters. Where there is greater uncertainty surrounding the appropriate values for these parameters, this is reflected by the metric predicting a more severe population-level consequence. The metric can be estimated using a density dependent model, and this approach is no more sensitive to assumptions about the underlying population trend or to mis-specification of demographic parameters than is the case for density independent models. However, incorporating density dependence into the models means that RI:U does not remain constant through time, as is the case for density independent models. Consequently, if this approach is taken, it is only appropriate to estimate RI:U at the end of the project lifetime (in the above example, after 25 years).

3.2.5 Ratio of Impacted to Unimpacted Population Size (RI:U25)

Population models can be used to estimate the size of a population through time both with and without the impact of an offshore wind farm. Comparing the ratio of the size of these two populations offers a relatively easy to interpret statistic with which to assess the population level impact of an offshore wind farm. The ratio could be derived either from a simple deterministic model or taken from the mean or median values simulated using a more complex stochastic model with matched runs for impacted and unimpacted populations. 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.

The metric is on a scale from 0 – 1, with 1 indicating the impacted population size is the same as the unimpacted growth rate (i.e. no population-level consequence) and values close to 0 indicating a large difference between the impacted and unimpacted population sizes (i.e. a strong population-level consequence).

Initial Results

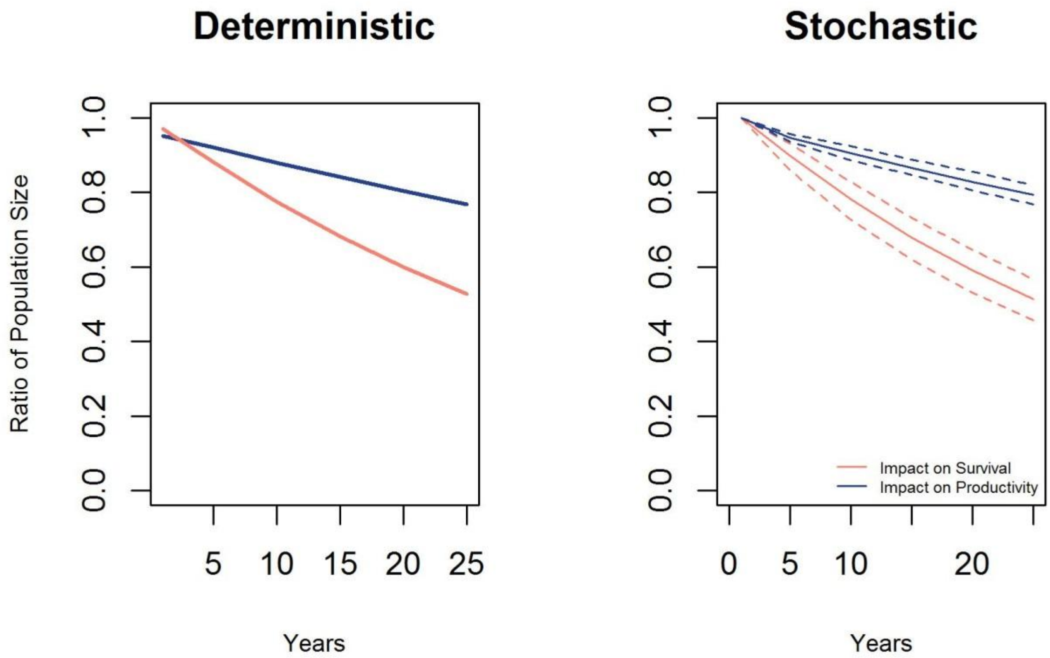

Figure 25. Change in value of the ratio of the impacted to unimpacted population size through time calculated using a deterministic (left) and stochastic (right) model and assuming a 10% increase in mortality or a 10% reduction in productivity. Broken lines show 95% Confidence Intervals.

The ratio of the impacted to unimpacted population size can be calculated for any point over the lifetime of a development. As time increases, the ratio of the impacted to unimpacted population decreases (Fig. 25). As with previous metrics, impacts on survival result in a more significant impact than those on productivity. Using the stochastic model, after the first 5 years the uncertainty surrounding the ratio appears relatively constant. Given this, it makes sense to use the ratio estimated at the end of the lifetime of the project, in this case after 25 years, when predicted impacts will be at their greatest. For this reason, subsequent discussion of the metric is based on the ratio of the impacted to unimpacted population size after 25 years (RI:U25).

RI:U25 decreases with increasing magnitude of the impact resulting from the offshore wind farm (Fig. 26). The metric is similar for both stochastic and deterministic models (Fig. 26). Using a deterministic model, RI:U25, assuming a 40% increase in mortality, is 0.07, in comparison to a value of 0.06 (± 0.02) from a stochastic model. For a 40% reduction in productivity the equivalent figures are 0.32 based on a deterministic model and 0.37 (0.03) based on a stochastic model. However, it is important to note that the relationship between the metric and the magnitude of the impact is not linear and approaches an asymptote where predicted offshore wind farm impacts are most severe (Fig. 26). This is an inevitable feature of the metric because as the population declines towards extinction the value of the metric cannot go below 0. As a consequence, whist distinguishing between low-moderate impacts is likely to be relatively straightforward, this distinction may be more difficult when predicted impacts are severe.

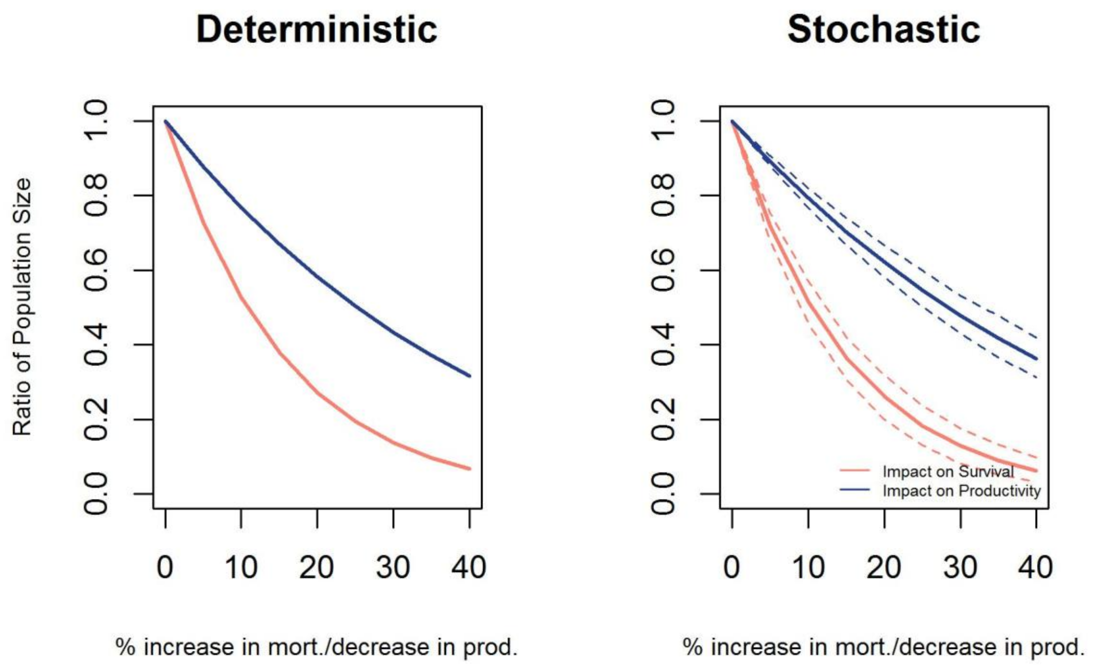

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Figure 26. Impact of up to a 40% increase in mortality or up to a 40% decrease in productivity on the ratio of the population size after 25 years for an impacted and unimpacted population of seabirds with an r-selected life history strategy and a stable population trajectory derived from deterministic and stochastic models (95% CIs given by broken lines).

Sensitivity to life history strategy, population trend and density dependence

There appears to be relatively little difference between RI:U25 estimated for populations which are stable or increasing (Table 16). However, there is a more noticeable difference between the ratios estimated from stable and decreasing populations, suggesting that despite impacts of a similar magnitude, they may have a more significant effect on a declining population. Considering a species which may have a more K-selected life history strategy results in significantly higher values for the ratio of the impacted to unimpacted population size (Table 16), implying that these species may be better able to withstand these impacts on their populations. As expected, if density dependent processes operate on the population this may mitigate the impact of any impacts arising from the wind farm (Table 16). It appears to make relatively little difference whether density dependence is assumed to operate on productivity or survival. These results confirm that where there is uncertainty over density dependent processes in a population, assuming no density dependence is present is likely to be the most precautionary assumption.

Table 16. Ratio of impacted to unimpacted population size after 25 years (95% CIs) resulting from a 10% or 20% increase in mortality or a 10% decrease in productivity estimated from a stochastic model and assuming an increasing, stable or decreasing population of an r-selected seabird species, a stable population of a K-selected seabird species and a stable population of an r-selected species with density dependent regulation of survival or productivity.

	Survival			Productivity		
	10%	20%	40%	10%	20%	40%
Increasing	0.509 (0.437 – 0.565)	0.248 (0.171 – 0.319)	0.050 (0.017 – 0.089)	0.738 (0.716 – 0.761)	0.532 (0.499 – 0.564)	0.255 (0.222 – 0.292)
Stable	0.515 (0.452 –	0.263 (0.202 -	0.061 (0.032 –	0.795 (0.768 –	0.624 (0.579 –	0.365 (0.313 –

	0.567)	0.319)	0.095)	0.821)	0.667)	0.427)
Decreasing	0.460 (0.397 – 0.517)	0.204 (0.144 – 0.261)	0.036 (0.011 – 0.061)	0.809 (0.778 – 0.833)	0.647 (0.600 – 0.692)	0.395 (0.334 – 0.460)
K selected	0.773 (0.740 – 0.801)	0.599 (0.549 – 0.643)	0.363 (0.307 – 0.418	0.897 (0.879 – 0.915)	0.798 (0.767 – 0.836)	0.620 (0.568 – 0.687)
Density Dependent Survival	0.742 (0.671 – 0.802)	0.528 (0.409 – 0.623)	0.195 (0.091 – 0.282)	0.883 (0.867 – 0.898)	0.773 (0.741 – 0.801)	0.522 (0.471 – 0.575)
Density Dependent Productivity	0.755 (0.594 – 0.883)	0.503 (0.299 – 0.717)	0.213 (0.049 – 0.430)	0.904 (0.862 – 0.932)	0.814 (0.729 – 0.866)	0.685 (0.587 – 0.765)

Sensitivity to mis-specification of demographic parameters

In comparison to the metrics based on population growth rate, this metric is more sensitive to mis-specification of adult survival, but not to mis-specification of other demographic parameters. There was strong evidence that this sensitivity varied with the magnitude of the predicted impact. With a 10% increase in mortality, a 1% mis-specification of adult survival led to a 2.45 percentage change in the ratio based on a deterministic model, or a 2.81% change based on a stochastic model (Fig. 27 & Tables 17 and 18). Were mortality to increase by 20%, the sensitivity to this mis-specification increases to 5.30% and 6.10% respectively. These patterns likely reflect the non-linear relationship between the metric and the magnitude of the predicted impact.

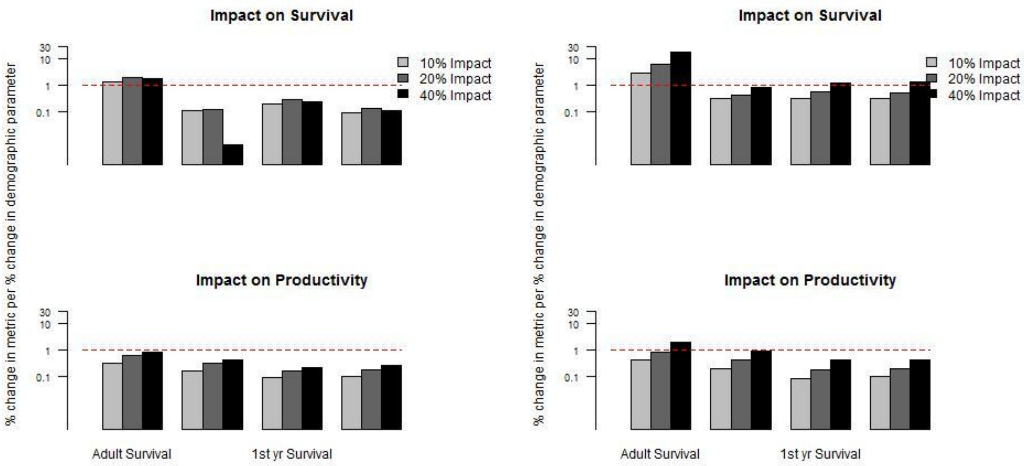

Figure 27. Percentage change in the ratio of impacted to unimpacted population size after 25 years per percentage change in adult survival, immature survival, first year survival and productivity for offshore wind farm impacts on survival and productivity from deterministic (left) and stochastic (right) models. Note that the Y-axis is on a non-linear scale. Data tabulated in Tables 17 and 18.

Table 17. Influence of a 1% mis-specification of each demographic parameter on the ratio of the population's size after 25 years (as assessed by % change in metric) of an impacted to unimpacted population of an r-selected seabird species estimated from a deterministic and stochastic model, assuming a 10%, 20% or 40% increase in mortality. Illustrated graphically in Figure 27.

	Deterministic			Stochastic		
	10%	20%	40%	10%	20%	40%
Adult Survival	2.45	5.30	13.73	2.81	6.07	16.34
Immature Survival	0.16	0.41	0.04	0.28	0.50	0.86
Chick Survival	0.37	0.78	1.76	0.29	0.59	1.51
Productivity	0.17	0.36	0.82	0.22	0.54	1.25

Table 18. Influence of a 1% mis-specification of each demographic parameter on the ratio of the population's size after 25 years (as assessed by % change in metric) of an impacted to unimpacted population of an r-selected seabird species estimated from a deterministic and stochastic model, assuming a 10%, 20% or 40% reduction in productivity. Illustrated graphically in Figure 27.

	Deterministic			Stochastic		
	10%	20%	40%	10%	20%	40%
Adult Survival	0.43	0.89	2.01	0.42	0.88	1.98
Immature Survival	0.12	0.29	1.03	0.17	0.36	0.89
Chick Survival	0.11	0.24	0.55	0.08	0.18	0.46
Productivity	0.13	0.27	0.61	0.11	0.21	0.53

Sensitivity to form of density dependence

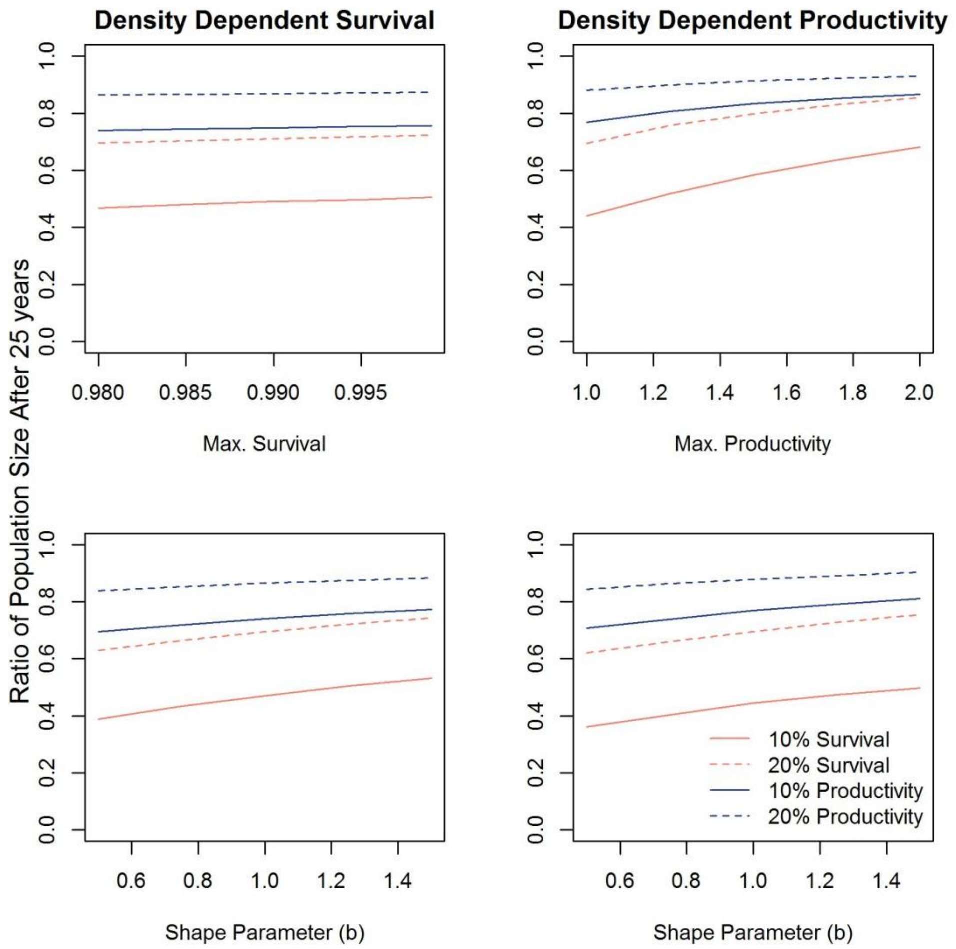

Figure 28. Impact of mis-specifying the shape parameter and maximum survival or productivity rate in a stable population of an r-selected seabird when using a stochastic model with density dependent regulation of survival or productivity.

RI:U25 is more sensitive to the shape parameter and the maximum productivity rate when estimated from a density dependent model than is the case for either GR or RI:U (Fig's 16, 20, and 28). Where survival is regulated, the maximum survival rate has little effect on RI:U25. However, as the shape parameter increases, the metric also increases. Where it is productivity that is regulated by density dependence, increases in both the maximum productivity rate and the shape parameter result in increases in RI:U25.

Additional Analysis

Since initial results suggested that this metric may be of use in estimating the population level effects of impacts arising from offshore wind farms, additional analyses were undertaken.

RI:U25 can be estimated at any point over the lifetime of a project. However, it is unclear whether sensitivity to mis-specification of demographic parameters is constant through time, i.e. if we estimate RI:U25 after 5 or 10 years, is it more important that we use accurate survival estimates than if we estimate it after 25 years. In order to understand this, we investigate how mis-specification of adult survival rates may affect the metric at different points in time (5 and 10 years post-construction). In contrast to the ratio of population growth rates (section 3.2.4), it appears that the ratio of impacted to unimpacted population size becomes more sensitive to mis-specification of demographic parameters over time (Fig. 29 & Table 19). This is because the population size in any given year is a product of the population size in the previous year and a matrix of the demographic parameters. As a consequence, any mis-specification of the demographic parameters becomes magnified through time, leading to a noticeable change in the metric value.

Table 19. Influence of a 1% mis-specification of adult survival rate on the ratio of the growth rate (as assessed by % change in metric) of an impacted to unimpacted population after 5 or 10 years for an r-selected seabird species estimated from a deterministic and stochastic model, assuming a 10%, 20% or 40% increase in mortality or a 10%, 20% or 40% reduction in productivity. Illustrated graphically in Figure 28.

	Impact on Survival			Impact on Productivity		
	10%	20%	40%	10%	20%	40%
After 5 years	0.42	0.87	1.94	0.05	0.10	0.23
After 10 Years	1.00	2.11	4.85	0.14	0.30	0.68

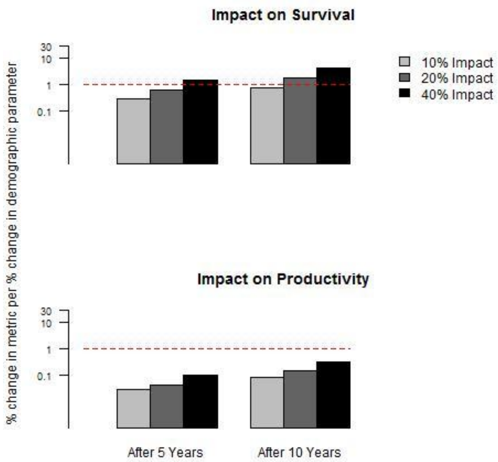

Figure 29. Percentage change in the ratio of impacted to unimpacted population size after 5 and 10 years rate per percentage change in adult survival for offshore wind farm impacts on survival and

productivity from a stochastic model. Note that the Y-axis is on a non-linear scale. Data tabulated in Table 18.

Additional analyses revealed that RI:U25 showed some sensitivity to the extent of uncertainty surrounding the demographic parameters used in population models (Fig. 30). However, this sensitivity did not appear to be greater than the sensitivity of the metric to mis-specification of demographic parameters. These analyses show that in populations where there is greater uncertainty surrounding the demographic parameters, RI:U is lower, implying a stronger population-level consequence, than is the case where there is less uncertainty surrounding the demographic parameters.

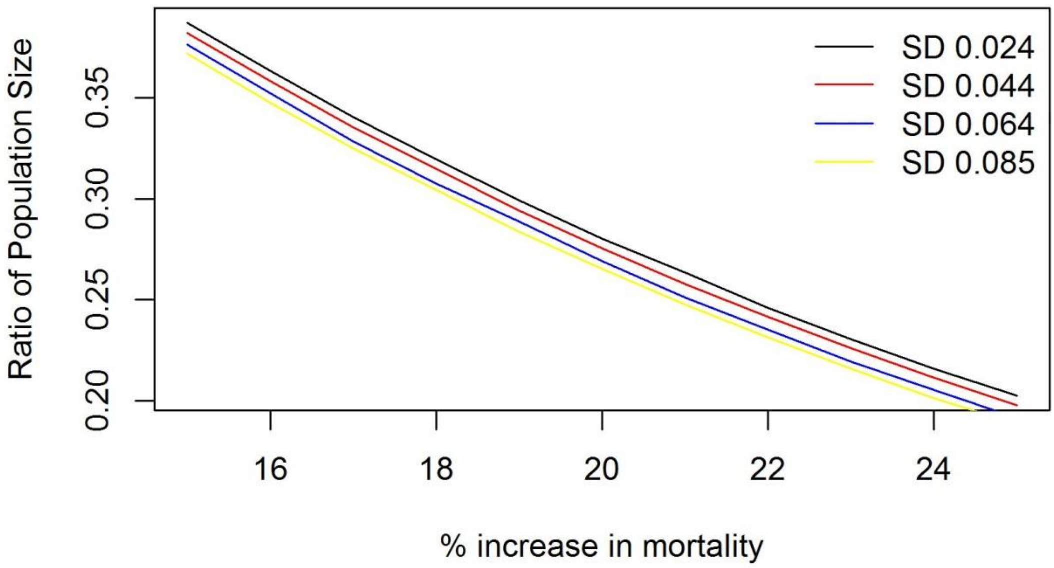

Figure 30. Impact of 15% to 25% increase in mortality on the ratio of impacted to unimpacted population size for a population of seabirds with an r-selected life history strategy and a stable population trajectory, assuming an adult survival rate of 0.89 and standard deviations of 0.024, 0.044, 0.064 and 0.085.

Sensitivity to population trend may vary depending on whether or not density dependence is incorporated in the population models used to estimate RI:U25. We therefore estimated this metric using a model assuming a density dependent impact on productivity and increasing, stable and declining populations. As with density independent models, when derived from density dependent models, where the impact is on survival, RI:U25 is insensitive to whether the population is stable or increasing, but does appear to be sensitive to whether the population is declining. Where the impact is on productivity, the metric is sensitive to whether the population is stable, declining or increasing (Table 20).

Table 20. Sensitivity of the ratio of impacted to unimpacted population size after 25 years to population trend, derived from stochastic models of an r-selected seabird assuming stable, increasing or declining populations with density dependent regulation of productivity.

	Impact on survival			Impact on productivity		
	10%	20%	40%	10%	20%	40%
Increasing	0.768 (0.652 – 0.860)	0.550 (0.361 – 0.703)	0.193 (0.037 – 0.388)	0.865 (0.821 – 0.901)	0.748 (0.673 – 0.808)	0.560 (0.441 – 0.660)
Stable	0.755	0.503	0.213	0.904	0.814	0.685

	(0.594 – 0.883)	(0.299 – 0.717)	(0.049 – 0.430)	(0.862 – 0.932)	(0.729 – 0.866)	(0.587 – 0.765)
Declining	0.600 (0.465 – 0.724)	0.314 (0.167 – 0.471)	0.049 (0 – 0.138)	0.924 (0.896 – 0.945)	0.852 (0.803 – 0.892)	0.722 (0.623 – 0.800)

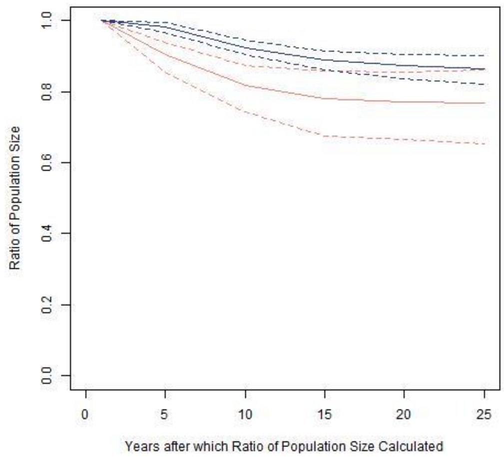

Figure 31. Change in value of the ratio of the impacted to unimpacted population size through time calculated using a stochastic model with density dependent regulation of productivity and assuming a 10% increase in mortality or a 10% reduction in productivity.

In contrast to density independent models, when using a density dependent model (Fig. 31), after an initial decrease in the ratio of impacted to unimpacted population size over the first 10 years of a project lifetime, the metric stabilises over the remaining time period. This suggests that, over time, density dependent mechanisms are compensating for the impacts associated with the offshore wind farm and that, as a result, an asymptote is reached in the ratio of impacted to unimpacted population size.

The ratio of impacted to unimpacted population growth size can be estimated from a density dependent model at any point over the lifetime of a project. However, it is unclear how incorporating density dependence into the models will affect sensitivity to mis-specification of demographic parameters. To test this, we focus on adult survival rate, which previous analyses showed was the parameter the metric was most sensitive to, and consider the sensitivity of the metric to mis-specification after 5, 10 and 25 years using a stochastic model with density dependent regulation of productivity. These results suggest that density dependent models may not be any more sensitive to mis-specification of demographic parameters than density independent models (Table 21, Figure 32).

Table 21. Influence of a 1% mis-specification of adult survival rate on the ratio of the population size (as assessed by % change in metric) of an impacted to unimpacted population after 5, 10 or 25 years for an r-selected seabird species estimated from a stochastic model with density dependent regulation of productivity, assuming a 10%, 20% or 40% increase in mortality or a 10%, 20% or 40% reduction in productivity. Illustrated graphically in Figure 32.

	Impact on Survival			Impact on Productivity		
	10%	20%	40%	10%	20%	40%
After 5 years	0.28	0.62	1.46	0.03	0.06	0.15
After 10 Years	0.73	1.69	4.12	0.07	0.14	0.31
After 25 Years	1.91	4.54	14.36	0.25	0.47	0.84

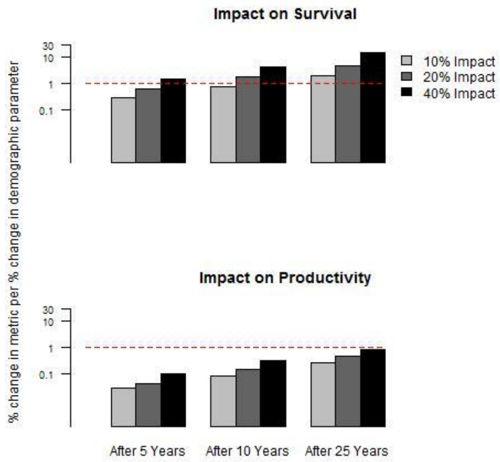

Figure 32. Percentage change in the ratio of impacted to unimpacted population size after 5, 10 and 25 years rate per percentage change in adult survival for offshore wind farm impacts on survival and productivity from a stochastic model with density dependent regulation of productivity. Note that the Y-axis is on a non-linear scale. Data tabulated in Table 21.

Metric overview

RI:U25 shows a clear and consistent relationship with increasing impacts resulting from offshore wind farms. However, the non-linear relationship between the magnitude of any impact and the metric may make interpreting the metrics more complicated when the impacted population size after 25 years is small. This metric appears to be more sensitive to mis-specification of demographic parameters, notably adult survival rate, than either GR, or RI:U. It also appears to be more sensitive to assumptions about density dependence than either of the preceding metrics. As with the preceding metrics, similar conclusions are reached about sensitivity to population trends and mis-specification of demographic parameters regardless of whether stochastic or deterministic models are used.

3.2.6 Probability of population growth rate being <1, 0.95 or 0.8 (P(GR<1))

Calculated from a stochastic model based on the proportion of simulations where the population growth rate is less than 1 (i.e. declining) or less than 0.95 or 0.8 (indicating more severe declines. The probabilities are typically assessed over the lifetime of the project. However, it would also be possible to examine these probabilities at any point during the lifetime of the project.

The metric is on a scale from 0 – 1 with 0 indicating that none of the simulations from a stochastic model result in a growth rate <1, 0.95 or 0.8 and 1 indicating that all of the simulations from a stochastic model result in a growth rate <1, 0.95 or 0.8.

Initial Results

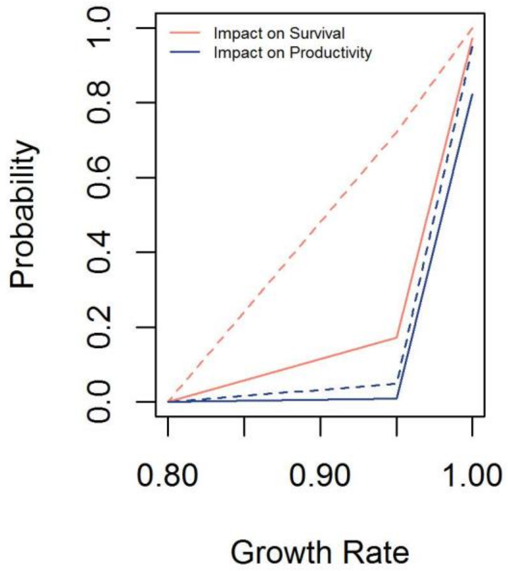

Figure 33. Probability of growth rate being less than 0.8, 0.95 or 1 assuming a stable population of an r-selected seabird species and a 10% (solid lines) or 20% (broken lines) increase in mortality or reduction in productivity.

Probabilistic metrics are estimated from the population growth rates estimated in simulations from a stochastic model. For this reason they cannot be derived from a deterministic model. The probability of the population growth rate of an impacted population being less than 0.8, 0.95 or 1 was considered (Fig. 33). For a stable population, in the absence of any impact, the probability of the population growth rate being <1 should be close to 0.5. Once an impact from an offshore wind farm was applied to the population, this probability would be expected to rise towards 1, depending on the magnitude of the impact. Initial trials suggested that, for a stable population, an increase of 20% in mortality would result in a noticeable increase in the probability of the population growth rate being less than 0.95 or less than 1. However, the clearest response for either a 10% or 20% increase in mortality or reduction in productivity is for the probability of the growth rate being less than 1. For this reason, subsequent discussion focuses the metric of the probability of the growth rate being less than 1 (P(GR<1)). It is likely that any issues raised by this metric will be equally applicable to a metric of the probability of the growth rate being less than 0.95 or less than 0.8.

If the median growth rate of the population under consideration were precisely 1, the probability of the growth being <1 would be approximately 0.5 as half of the simulations from the stochastic model would have a growth rate <1 and half would have a growth rate >1. However, as the median growth rate of this population is slightly less than 1 (see section 3.2.3), there is greater probability of the unimpacted population having a growth rate <1, in this case 0.598 (Fig. 34). As the magnitude of the impact on survival or productivity increases, P(GR<1) approaches one (Fig. 34). Less severe impacts are required on survival before the probability of the growth rate being less than 1 reaches 1, than is the case for impacts on productivity. A 15% increase in mortality results in a probability of the growth rate being less than 1 of 0.998. In contrast, a 30% reduction in productivity is required before a 0.996 probability of the growth rate being less than one is estimated.

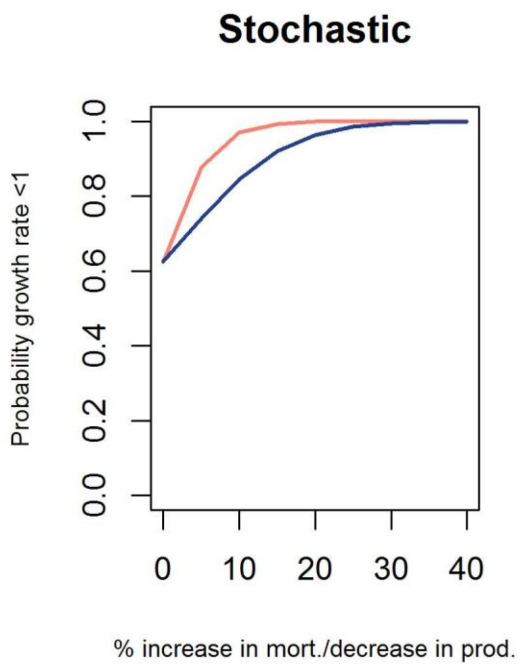

Figure 34. Impact of up to a 40% increase in mortality (red line) or up to a 40% decrease in productivity on the probability (blue line) of the growth rate of a population of seabirds with an r-selected life history strategy and a stable population trajectory.

Sensitivity to life history strategy, population trend and density dependence

There are clear differences between P(GR<1) for increasing and stable populations (Table 22). Differences are also evident between stable and decreasing populations although these narrow as the magnitude of the predicted impact increases. Species which have a more K-selected life history strategy appear to be more resilient to impacts than those with an r-selected strategy and the probability of a stable population of these species having a growth rate of less than 1 is lower for an equivalent impact level than is the case for an r-selected species (Table 22).

Table 22. Probability of population growth rate being less than one resulting from a 10% or 20% increase in mortality or a 10% decrease in productivity estimated from a stochastic model and assuming an increasing, stable or decreasing population of an r-selected seabird species, a stable population of a K-selected seabird species and a stable population of an r-selected species with density dependent regulation of survival or productivity.

	Survival			Productivity		
	10%	20%	40%	10%	20%	40%
Increasing	0.009	0.296	0.970	<0.001	0.001	0.064
Stable	0.963	0.998	1.000	0.792	0.910	0.999
Decreasing	1.000	1.000	1.000	1.000	1.000	1.000
K selected	0.901	0.982	1.000	0.677	0.777	0.931
Density Dependent Survival	0.730	0.876	0.990	0.537	0.653	0.809
Density Dependent Productivity	0.901	0.989	1.000	0.725	0.835	0.975

As might be expected, density dependent processes appear to mitigate the impacts arising from offshore wind farms on seabirds. Density dependent regulation of survival appears to more effectively regulate against any impacts than density dependent regulation of productivity (Table 22).

Sensitivity to mis-specification of demographic parameters

P(GR<1) is more sensitive to mis-specification of input parameters than the metrics discussed previously (Fig. 35, Tables 23 and 24, and see sections 3.2.1 - 3.2.3 above). As previously, the metric is most sensitive to mis-specification of adult survival. However, in contrast to these preceding metrics, it may also be fairly sensitive to mis-specification of other demographic parameters, particularly when productivity is affected by a development. The sensitivity of the metric to mis-specification of demographic parameters declines with the magnitude of the predicted impact, and for a 40% increase in mortality the sensitivity to mis-specification of demographic parameters other than adult survival is 0. This is because, for the stable, r-selected population considered, an increase in mortality of 40% will, almost always, result in a growth rate less than 1 regardless of the value assumed for immature or first year survival or productivity. It is worth noting that, in contrast to previous metrics (see sections 3.2.3 – 3.2.5), there is less scope for this metric to vary as it cannot exceed 1, and for no impact, the probability of a population growth rate of <1 is 0.6, meaning the metric must be bound by these two values.

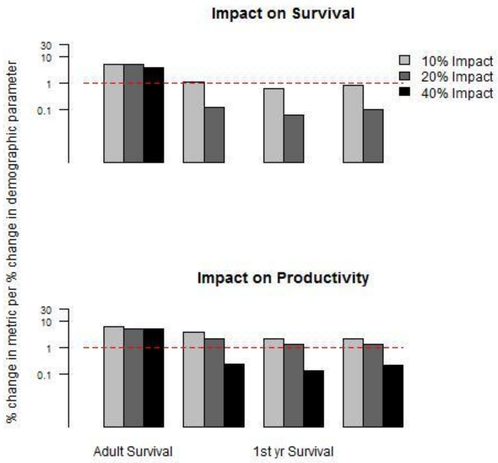

Figure 35. Percentage change in the probability of the population growth rate being less than 1 per percentage change in adult survival, immature survival, first year survival and productivity for offshore wind farm impacts on survival and productivity from deterministic (left) and stochastic (right) models. Note that the Y-axis is on a non-linear scale. Data tabulated in Tables 23 and 24.

Table 23. Influence of a 1% mis-specification of each demographic parameter on the probability of the population growth rate being less than 1 (as assessed by % change in metric) for an r-selected seabird species estimated from a stochastic model assuming a 10%, 20% or 40% increase in mortality. Illustrated graphically in Figure 35.

	Stochastic		
	10%	20%	40%
Adult Survival	5.19	5.00	3.77
Immature Survival	1.21	0.19	<0.01
Chick Survival	0.76	0.13	<0.01
Productivity	0.90	0.15	<0.01

Table 24. Influence of a 1% mis-specification of each demographic parameter on probability of the population growth rate being less than 1 (as assessed by % change in metric) for an r-selected seabird species estimated from a stochastic model assuming a 10%, 20% or 40% reduction in productivity. Illustrated graphically in Figure 35.

	Stochastic		
	10%	20%	40%
Adult Survival	6.26	5.45	5.01
Immature Survival	3.82	2.20	0.22
Chick Survival	2.18	1.29	0.10
Productivity	2.42	1.17	0.14

Sensitivity to form of density dependence

If density dependence is introduced into the models, P(GR<1) appears to show some sensitivity to mis-specification of the shape parameter and mis-specification of the maximum survival or productivity rate (Fig. 36), with some variation in the metric when alternative values are assumed for each parameter.

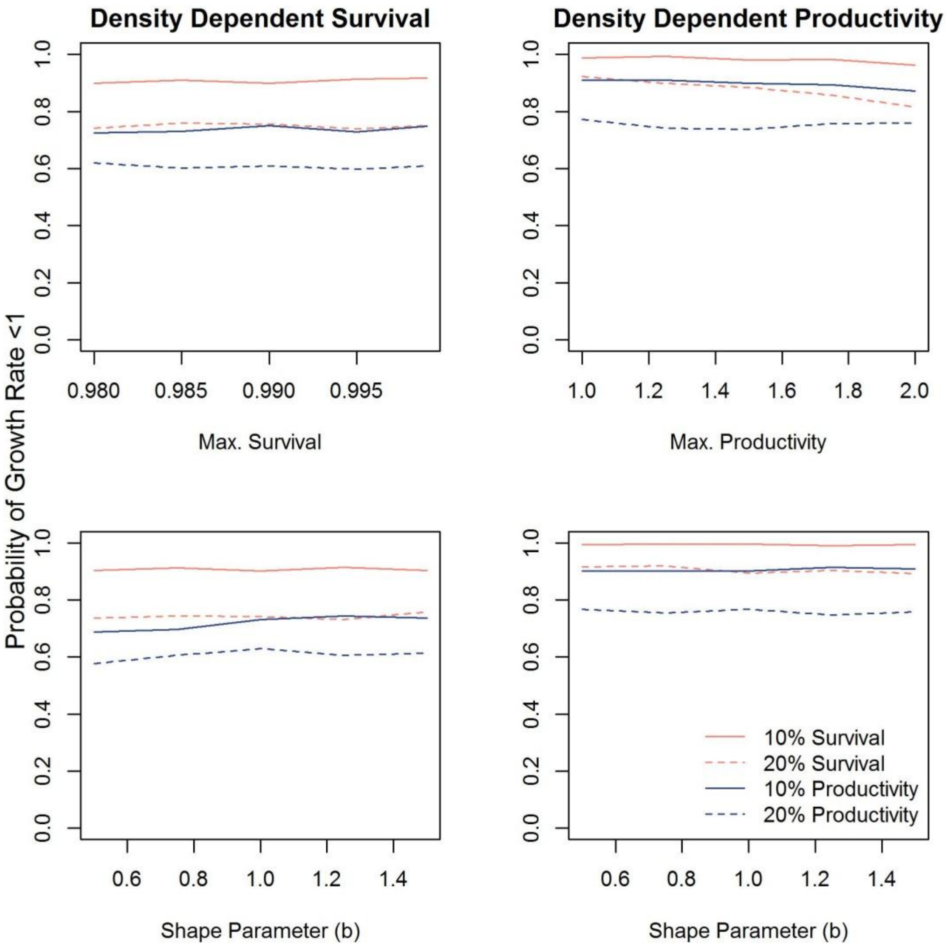

Figure 36. Impact of mis-specifying the shape parameter and maximum survival or productivity rate in a stable population of an r-selected seabird when using a stochastic model with density dependent regulation of survival or productivity.

Metric overview

P(GR<1) reaches an asymptote (Fig. 34), making it harder to understand differences in the population level effects of a development when impacts are moderate to severe, particularly in the case of impacts on survival. The metric appears to be more sensitive to mis-specification of demographic parameters than those discussed previously. As with previous metrics it is most sensitive to mis-specification of adult survival. However, there are also indications that it may be sensitive to mis-specification of other demographic parameters. Whilst incorporating density dependent regulation of survival and/or productivity into the models reduces the value estimated for this metric, it appears to have some sensitivity to assumptions about the form of this density dependence (Fig. 36).

3.2.7 Change in the probability of the population growth rate being <1, 0.95 or 0.8 (dP(GR<1))

Where simulations show that a 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, simply quantifying the probability of a population decline in the presence of an offshore wind farm may not be meaningful. To assess the population level impact of a development it is therefore necessary 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 could be done either at a single fixed point in time, or at intervals throughout the life time of the project.

The metric is on a scale from 0 – 1, with 0 indicating that there is no change in the probability of the growth rate being <1, 0.95 or 0.8 between impacted and unimpacted populations (i.e. no population-level consequence) and values approaching 1 indicating there is a change in the probability of the growth rate being <1, 0.95 or 0.8 between the impacted and unimpacted populations (i.e. a population-level consequence).

Initial Results

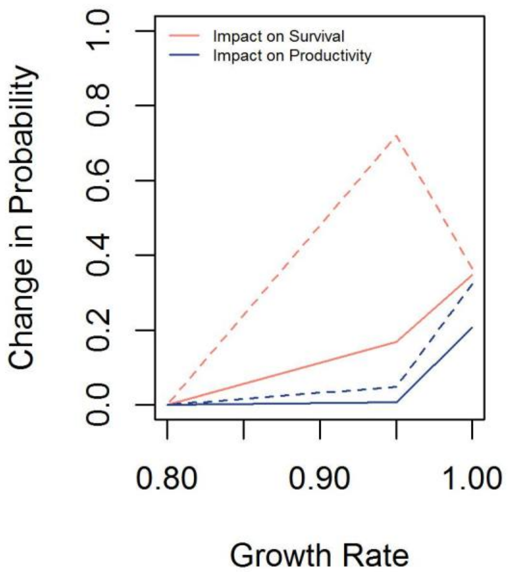

Figure 37. Change in probability of growth rate being less than 0.8, 0.95 or 1 assuming a stable population of an r-selected seabird species and a 10% (solid lines) or 20% (broken lines) increase in mortality or reduction in productivity.

Initial trials suggested that, for a stable population, the change in probability of the growth rate being 0.8, 0.95 or 1 peaked at different points depending on the magnitude of the impact considered (Fig. 37). The probability of the population growth rate being <1 for a stable population in the absence of impacts from an offshore wind farm should be close to 0.5. As a consequence, once impacts from an offshore wind farm are applied to a population, the maximum value possible for the change in probability of the population growth rate being <1 is 0.5 (i.e. the impact associated with an offshore wind farm causes the probability of the growth rate being <1 to increase from 0.5 to 1.0). Under the scenarios considered in Figure 37 a 10% impact on mortality or a 20% impact on survival or mortality causes the probability of the population growth rate being <1 to increase by between 0.35 and 0.4 (i.e. from a probability of 0.5 for a stable population in the absence of any offshore wind farm impact to 0.85-0.90 for the same population in the presence of an offshore wind farm impact). Similarly, for a stable population, the probability of a growth rate of <0.95 or <0.80 should be close to 0. Therefore, there is greater scope for a change in the probability of the growth rate being lower than either of these two values than is the case for a growth rate of <1. Figure 37 demonstrates that a severe impact on survival can result in a large change in the probability of the growth rate being <0.95. However, even with a 20% increase in mortality, the probability of the population growth rate being <0.8 was still close to 0. For this reason, subsequent discussion focuses on the probability of the population growth rate being less than 1 (dP(GR<1)). It is likely that this discussion would be equally applicable to metrics of the change in probability of the population growth rate being less than 0.95 or 0.8. Based on the stable population of an r-selected seabird considered in this report, the minimum value for P(GR<1) is 0.598 (see section 3.2.6, above). As a consequence, the maximum value for dP(GR<1), assuming a stable population, is 0.402 as 1 is the upper limit on P(GR<1). As the magnitude of the impact on survival or productivity increases, the change in the probability of the population growth rate being less than 1 reaches an asymptote at approximately 0.40 (Fig. 38). Less severe impacts are required on survival before the probability of the growth rate being less than 1 reaches an asymptote, than is the case for impacts on productivity. A 15% increase in mortality results in the change in the probability of the growth rate being less than 1 of 0.389. In contrast, a 30% reduction in productivity is required before this value is reached.

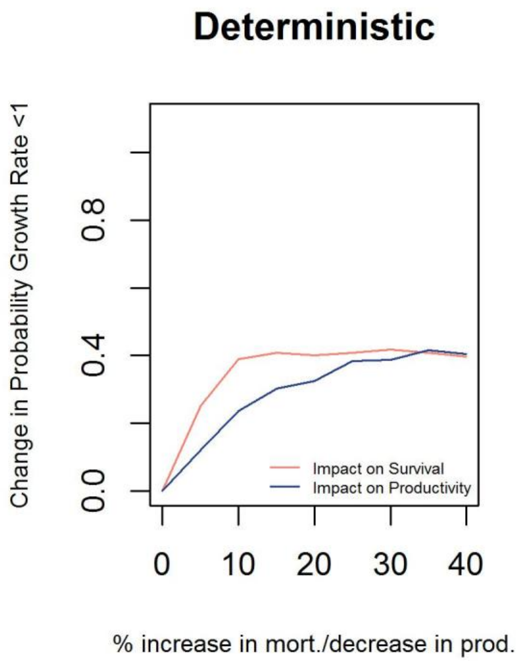

Figure 38. Impact of up to a 40% increase in mortality or up to a 40% decrease in productivity on the change in the probability of the growth rate being less than 1 for a population of seabirds with an r-selected life history strategy and a stable population trajectory.

Sensitivity to life history strategy, population trend and density dependence

dP(GR<1) appears to be sensitive to the underlying trajectory of the population concerned. Where a population is stable, the metric may indicate the impact of any development relatively clearly. However, where the population is increasing or decreasing, the metric is less capable of detecting any impact when these impacts are less severe. In the case of an increasing population, dP(GR<1) is 0.98 for a 40% impact on survival, indicating that the growth rate from the simulations for the unimpacted population was almost always >1, but almost always <1 for the impacted population.

Species which have a more K-selected life history strategy appear to be more resilient to impacts than those with an r-selected strategy and the probability of a stable population of these species having a growth rate of less than 1 is lower for an equivalent impact level than is the case for an r-selected species (Table 25).

Table 25. Change in probability of population growth rate being less than 1 after 25 years resulting from a 10% or 20% increase in mortality or a 10% decrease in productivity estimated from a stochastic model and assuming an increasing, stable or decreasing population of an r-selected seabird species, a stable population of a K-selected seabird species and a stable population of an r-selected species with density dependent regulation of survival or productivity.

	Survival			Productivity		
	10%	20%	40%	10%	20%	40%
Increasing	0.017	0.296	0.980	<0.001	0.005	0.074
Stable	0.383	0.379	0.400	0.185	0.328	0.400
Decreasing	0.001	0.002	0.002	<0.001	0.001	0.001
K selected	0.263	0.325	0.333	0.146	0.250	0.340
Density Dependent Survival	0.241	0.456	0.516	0.077	0.170	0.333
Density Dependent Productivity	0. 298	0.405	0.403	0.162	0.236	0.391

As might be expected, density dependent processes appear to largely mitigate the impacts arising from offshore wind farms on seabirds. However, in contrast to previous metrics, it appears that when impacts on survival are more severe, incorporating density dependence into the model may indicate a more severe impact at a population level. This is likely to be because the growth rate of the unimpacted populations is slightly higher when density dependence is incorporated (see Table 25), meaning a lower proportion of simulations will have a growth rate <1 than would be the case for density independent populations. Consequently, there is more scope for dP(GR<1) to increase when the density dependence is incorporated into the models.

Sensitivity to mis-specification of demographic parameters

As with previous metrics dP(GR<1) is most sensitive to adult survival (Fig. 39 and Tables 26 and 27). However, the change in probability of the population growth rate being less than one is also extremely sensitive to mis-specification of each of the other demographic parameters.

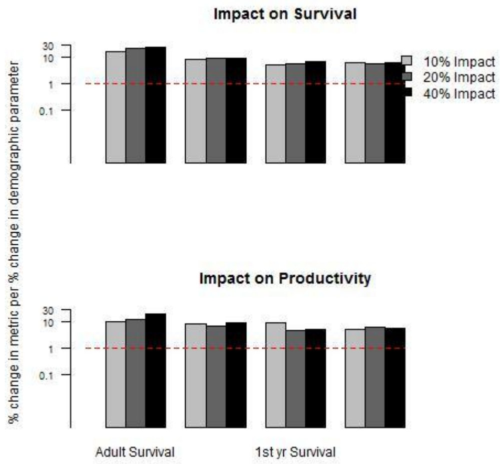

Figure 39. Percentage change in the change in probability of the population growth rate being less than 1 per percentage change in adult survival, immature survival, first year survival and productivity for offshore wind farm impacts on survival and productivity stochastic models. Data tabulated in Tables 26 and 27.

Table 26. Influence of a 1% mis-specification of each demographic parameter on the change in the probability of the population growth rate being less than 1 (as assessed by % change in metric) for an r-selected seabird species estimated from a stochastic model assuming a 10%, 20% or 40% increase in mortality. Illustrated graphically in Figure 39.

	Stochastic		
	10%	20%	40%
Adult Survival	16.88	23.03	23.75
Immature Survival	7.54	9.06	10.01
Chick Survival	5.18	5.99	6.03
Productivity	6.02	6.40	6.68

Table 27. Influence of a 1% mis-specification of each demographic parameter on the change in the probability of the population growth rate being less than 1 (as assessed by % change in metric) for an r-selected seabird species estimated from a stochastic model assuming a 10%, 20% or 40% reduction in productivity. Illustrated graphically in Figure 39.

	Stochastic		
	10%	20%	40%
Adult Survival	13.37	14.79	20.28
Immature Survival	10.40	8.63	8.81
Chick Survival	5.83	7.61	5.86
Productivity	6.22	6.36	7.19

Sensitivity to form of density dependence

If density dependence is introduced into the models, dP(GR<1) appears to have some sensitivity to both mis-specification of the shape parameter and mis-specification of the maximum survival or productivity rate (Fig. 40), with some variation in the metric when alternative values are assumed for each parameter.

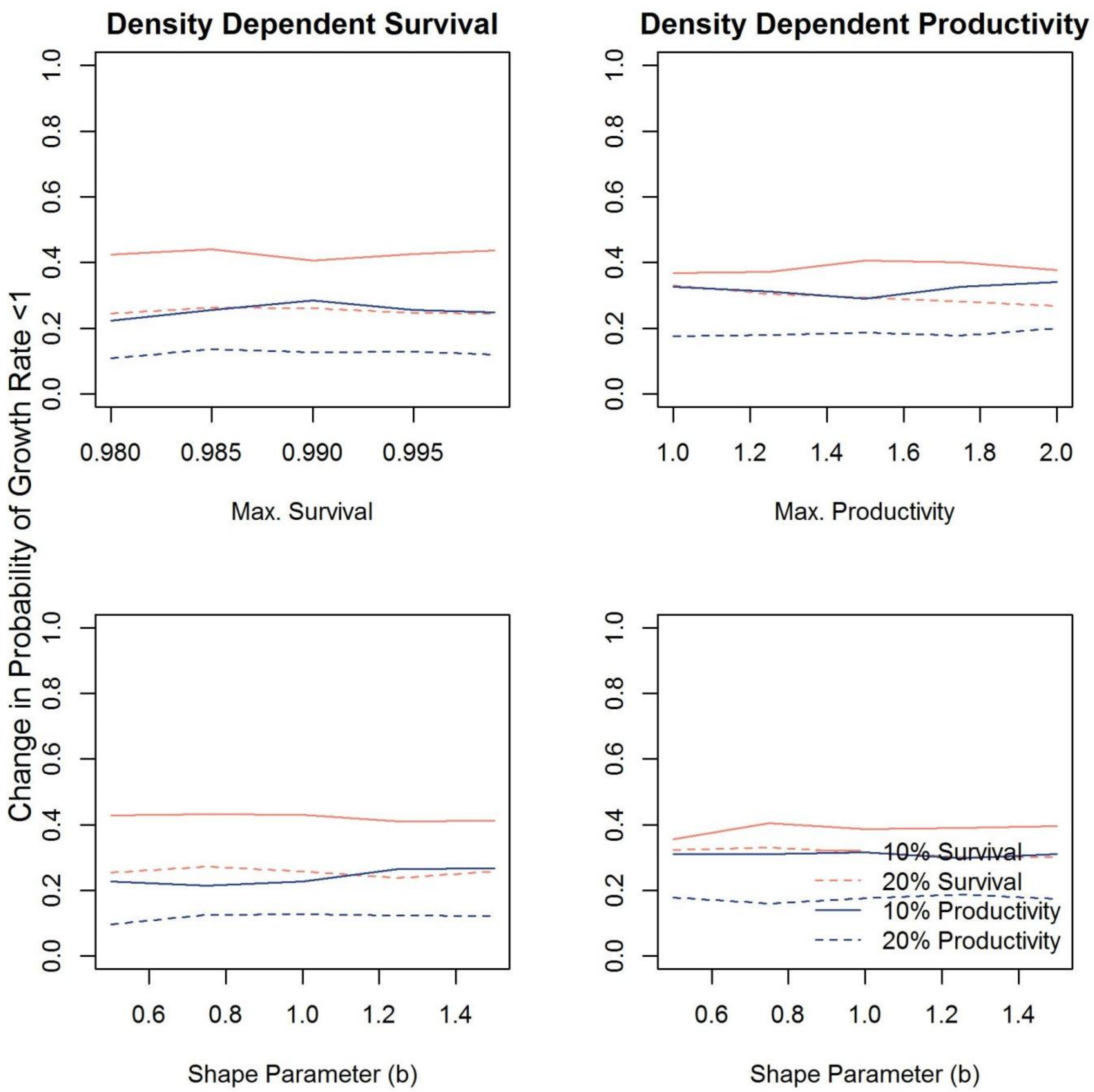

Figure 40. Impact of mis-specifying the shape parameter and maximum survival or productivity rate in a stable population of an r-selected seabird when using a stochastic model with density dependent regulation of survival or productivity.

Metric overview

dP(GR<1) quickly reaches an asymptote (Fig. 38), making it harder to understand differences in the population level effects of a development when impacts are moderate to severe, particularly in the case of impacts on survival. The metric appears to be more sensitive to mis-specification of demographic parameters than those discussed previously. As with previous metrics it is most sensitive to mis-specification of adult survival. However, there are also indications that it is also sensitive to mis-specification of other demographic parameters. Incorporating density dependent regulation of survival and/or productivity into the models reduces the value estimated for this metric, it also appears to have some sensitivity to assumptions about the form of this density dependence (Fig. 40).

3.2.8 Probability of the population declining below its initial size at any point in time ($P(p<p_0)$)

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.

The metric is on a scale from 0 -1 with 0 indicating that none of the simulations from a stochastic model result in a population below its initial size at any point in time and 1 indicating that all of the simulations from a stochastic model result in a population below its initial size at any point in time.

Initial Results

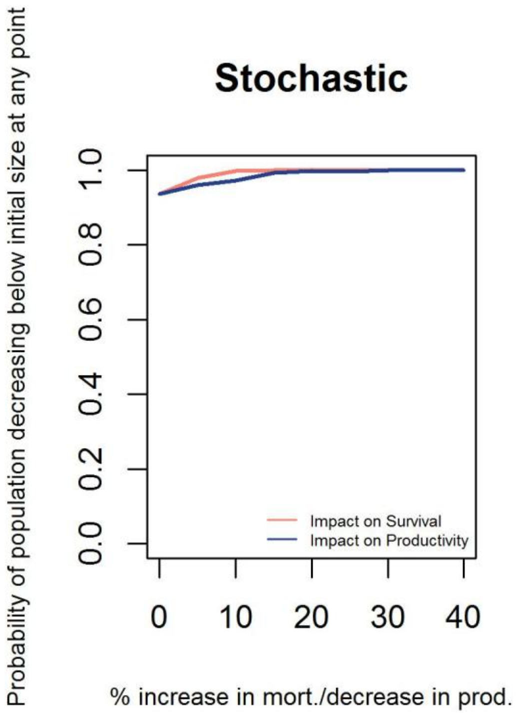

Figure 41. Impact of up to a 40% increase in mortality or up to a 40% decrease in productivity on the probability of the population dropping below its initial size at any point in time for a population of seabirds with an r-selected life history strategy and a stable population trajectory.

In contrast to the previous metrics, this considers the probability of a population deceasing below its initial size at any point in time ($P(p<p_0)$), as opposed to being below its initial size at the end of the project lifetime (i.e. has an average population growth rate <1). Therefore, this metric allows for the possibility that the impact from a development may initially cause a population to decline, but that it may then recover over the lifetime of the project. Assuming a stable population, this metric is at, or close to, 1 regardless of the magnitude of impact a population experiences (Fig. 41).

Sensitivity to life history strategy, population trend and density dependence

There are clear differences between $P(p<p_0)$ for increasing and stable populations (Table 28). However, differences are less clear for stable or decreasing populations. Species which have a more K-selected life history strategy appear to be more resilient to impacts than those with an r-selected strategy and the probability of a stable population of these species

having $P(p<p_0)$ is lower for an equivalent impact level than is the case for an r-selected species (Table 28).

As might be expected, density dependent processes mitigate against the impact of any impacts from development. Density dependent regulation of survival appears to more effectively mitigate the impacts of any development than density dependent regulation of productivity (Table 28).

Table 28. Probability of a population decreasing below its initial size at any point in time resulting from a 10% or 20% increase in mortality or a 10% decrease in productivity estimated from a stochastic model and assuming an increasing, stable or decreasing population of an r-selected seabird species, a stable population of a K-selected seabird species and a stable population of an r-selected species with density dependent regulation of survival or productivity.

	Survival			Productivity		
	10%	20%	40%	10%	20%	40%
Increasing	0.617	0.865	1.000	0.517	0.710	1.000
Stable	0.997	1.000	1.000	0.980	0.997	1.000
Decreasing	1.000	1.000	1.000	1.000	1.000	1.000
K selected	0.989	0.999	1.000	0.980	0.998	1.000
Density Dependent Survival	0.967	0.990	0.998	0.914	0.974	0.995
Density Dependent Productivity	0.994	1.000	1.000	0.989	0.996	1.000

Sensitivity to mis-specification of demographic parameters

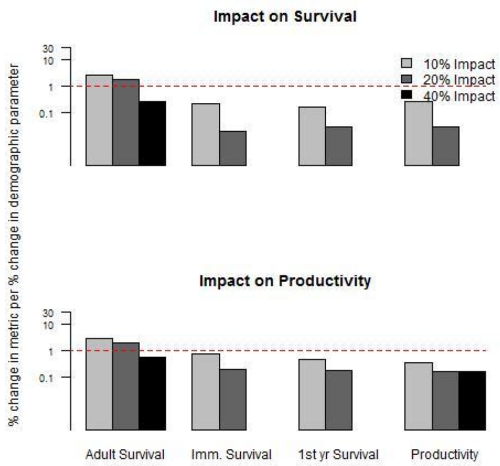

Figure 42. Percentage change in the change in probability of a population decreasing below its initial size at any point per percentage change in adult survival, immature survival, first year survival and productivity for offshore wind farm impacts on survival and productivity from stochastic models. Data tabulated in Tables 29 and 30.

As with previous metrics, P(p<p0) is most sensitive to mis-specification of adult survival. However, it is relatively insensitive to mis-specification of other parameters (Fig. 42 and Tables 29 and 30). Sensitivity to mis-specification of the demographic parameters varied according to the magnitude of the impact predicted, and the metric was less sensitive to mis-specification when more severe impacts were estimated.

Table 29. Influence of a 1% mis-specification of each demographic parameter on the probability of a population decreasing below its initial size at any point (as assessed by % change in metric) in time for an r-selected seabird species estimated from a stochastic model assuming a 10%, 20% or 40% increase in mortality. Data illustrated graphically in Figure 42.

	Stochastic		
	10%	20%	40%
Adult Survival	2.73	1.70	0.26
Immature Survival	0.22	0.02	<0.01
Chick Survival	0.16	0.03	<0.01
Productivity	0.25	0.03	<0.01

Table 30. Influence of a 1% mis-specification of each demographic parameter on the probability of a population decreasing below its initial size at any point in time (as assessed by % change in metric) for an r-selected seabird species estimated from a stochastic model assuming a 10%, 20% or 40% reduction in productivity. Data illustrated graphically in Figure 42.

	Stochastic		
	10%	20%	40%
Adult Survival	2.80	1.98	0.58
Immature Survival	0.75	0.20	<0.01
Chick Survival	0.16	0.03	<0.01
Productivity	0.33	0.16	<0.01

Sensitivity to form of density dependence

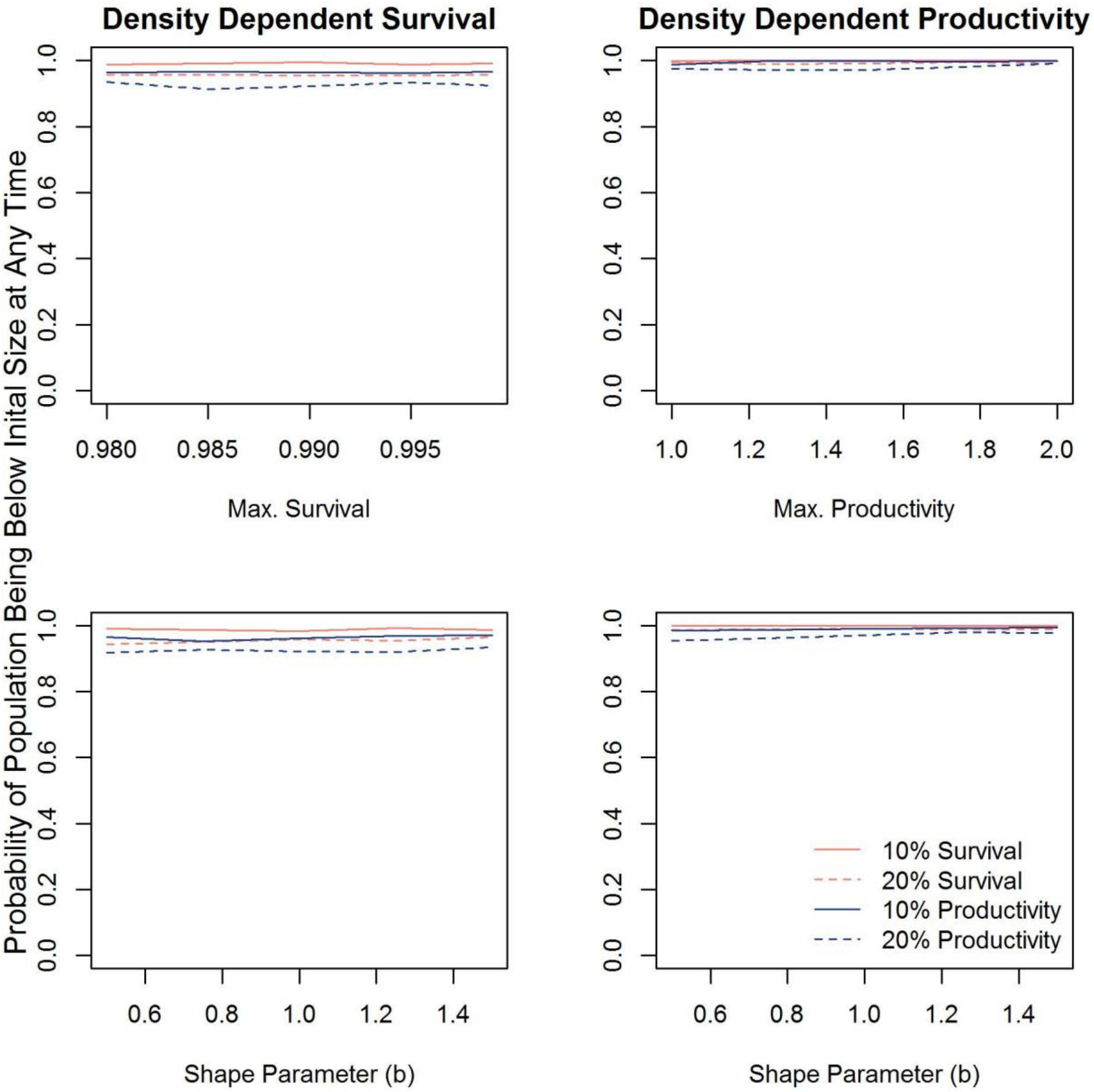

Figure 43. Impact of mis-specifying the shape parameter and maximum survival or productivity rate in a stable population of an r-selected seabird when using a stochastic model with density dependent regulation of survival or productivity.

If density dependence is introduced into the models, $P(p<p_0)$ does not appear to be particularly sensitive to either mis-specification of the shape parameter or mis-specification of the maximum survival or productivity rate (Fig. 43).

Metric overview

The metric is sensitive to the underlying trend of the population concerned, with clear differences when calculated for increasing or stable/decreasing populations. Where the pre-impact population at a site is stable, $P(p<p_0)$ rapidly plateaus at 1, regardless of whether impacts are on productivity or survival. However, the metric is relatively insensitive to mis-specification of demographic parameters and the assumptions made about density dependence.

3.2.9 Probability of a 10, 25 or 50% population decline (P(Id>0.25))

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 which in which a decline of a given magnitude was recorded.

The metric is on a scale from 0 – 1, with 0 indicating that none of the simulations from a stochastic model show the impacted population declining by a given magnitude (i.e. no population-level consequence) and 1 indicating that all simulations show the impacted population declining by a given magnitude.

Initial Results

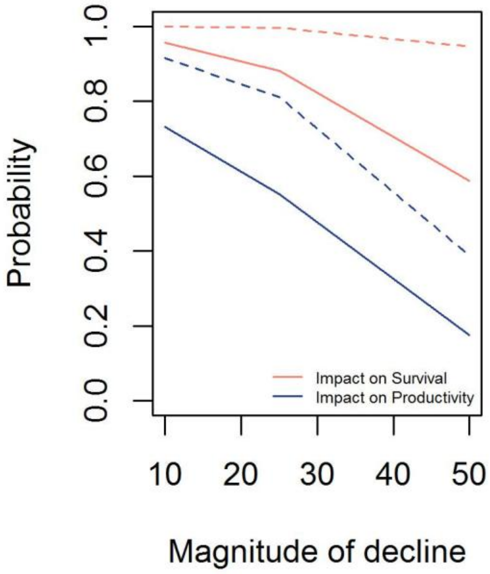

Figure 44. Probability of a decline of 10, 25 or 50% assuming a stable population of an r-selected seabird species and a 10% (solid lines) or 20% (broken lines) increase in mortality or reduction in productivity.

This metric considered the probability of a population declining by 10, 25 or 50% as a result of the impacts associated with an offshore wind farm after the 25 year life time of a wind farm (Fig. 44). Assuming a 20% increase in mortality, the metric is close, or equal, to 1 regardless of whether a 10, 25 or 50% decline is under consideration. For a 10% increase in mortality or a 10 or 20% decrease in productivity, the probability of detecting changes of these magnitudes declines sharply. Further discussion of this metric focusses on 25% decline, although these comments are likely to be equally applicable to a 10 or 50% decline.

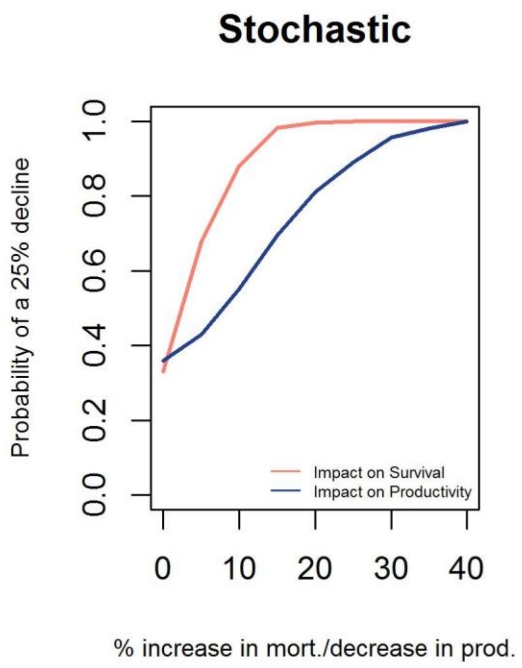

Figure 45. Impact of up to a 40% increase in mortality or up to a 40% decrease in productivity on the probability of a decline of 25% for a population of seabirds with an r-selected life history strategy and a stable population trajectory.

The probability of a population decline of 25% (P(Id>0.25)) reaches an asymptote at 1 for a 20% increase in mortality. Therefore, this metric may be unable to distinguish between the population-level consequences of medium (20% mortality) and large (40% mortality) magnitude impacts. If the impact is on productivity, the probability of a population decline of 25% rises more gradually before also reaching 1 after a 40% decrease in productivity (Fig. 45).

Sensitivity to life history strategy, population trend and density dependence

There are clear differences between P(Id<0.25) for populations which are increasing and for those that are stable (Table 31). There are also clear differences between stable and decreasing populations where there is a 10% increase in mortality or a 10 or 20% impact on productivity. However, where there are more severe increases in mortality, these differences are less clear. Species which have a more K-selected life history strategy appear to be more resilient to the impacts of offshore wind farms.

As might be expected, density dependent regulation of the population can mitigate the impacts arising as a result of offshore wind farms. It appears that density dependent regulation of productivity more effectively mitigates population level effects than density dependent regulation of productivity (Table 31).

Table 31. Probability of a population decline of 25% resulting from a 10% or 20% increase in mortality or a 10% decrease in productivity estimated from a stochastic model and assuming an increasing, stable or decreasing population of an r-selected seabird species, a stable population of a K-selected seabird species and a stable population of an r-selected species with density dependent regulation of survival or productivity.

	Survival			Productivity		
	10%	20%	40%	10%	20%	40%
Increasing	<0.001	0.170	0.944	<0.001	<0.001	0.059
Stable	0.882	0.994	1.000	0.594	0.798	1.000
Decreasing	1.000	1.000	1.000	1.000	1.000	1.000
K selected	0.596	0.884	0.997	0.354	0.589	0.916
Density Dependent Survival	0.512	0.804	0.978	0.372	0.524	0.765
Density Dependent Productivity	0.450	0.841	0.997	0.221	0.327	0.732

Sensitivity to mis-specification of demographic parameters

As with previous metrics, P(Id<0.25) is most sensitive to mis-specification of the adult survival rate. However, it is also sensitive to mis-specification of other demographic parameters, particularly when impacts are predicted to affect productivity (Fig. 46 and Tables 32 and 33).

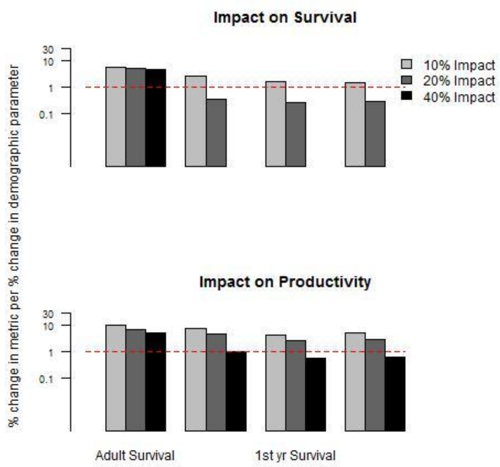

Figure 46. Percentage change in the probability of the population declining by 25% per percentage change in adult survival, immature survival, first year survival and productivity for offshore wind farm impacts on survival and productivity from stochastic models. Data tabulated in Tables 32 and 33.

Table 32. Influence of a 1% mis-specification of each demographic parameter on the probability of a population decreasing by 25% (as assessed by % change in metric) for an r-selected seabird species estimated from a stochastic model assuming a 10%, 20% or 40% increase in mortality. Illustrated graphically in Figure 46.

	Stochastic		
	10%	20%	40%
Adult Survival	5.86	5.02	4.47
Immature Survival	2.76	0.43	<0.01
Chick Survival	1.51	0.19	<0.01
Productivity	1.64	0.19	<0.01

Table 33. Influence of a 1% mis-specification of each demographic parameter on the probability of a population decreasing by 25% (as assessed by % change in metric) for an r-selected seabird species estimated from a stochastic model assuming a 10%, 20% or 40% reduction in productivity. Illustrated graphically in Figure 46.

	Stochastic		
	10%	20%	40%
Adult Survival	9.41	6.54	5.02
Immature Survival	7.07	3.60	0.37
Chick Survival	4.58	1.97	0.29
Productivity	4.30	2.23	0.31

Sensitivity to form of density dependence

Where a population has density dependent regulation of survival, P(Id>0.25) is relatively insensitive to assumptions about the maximum survival rate and the shape parameter (Fig. 47). However, where productivity is regulated by density dependence, P(Id>0.25) is sensitive to both of these parameters. This is because density dependence triggers an increase in the productivity rate of the impacted population, reducing the proportion of simulations in which it undergoes a 25% decline.

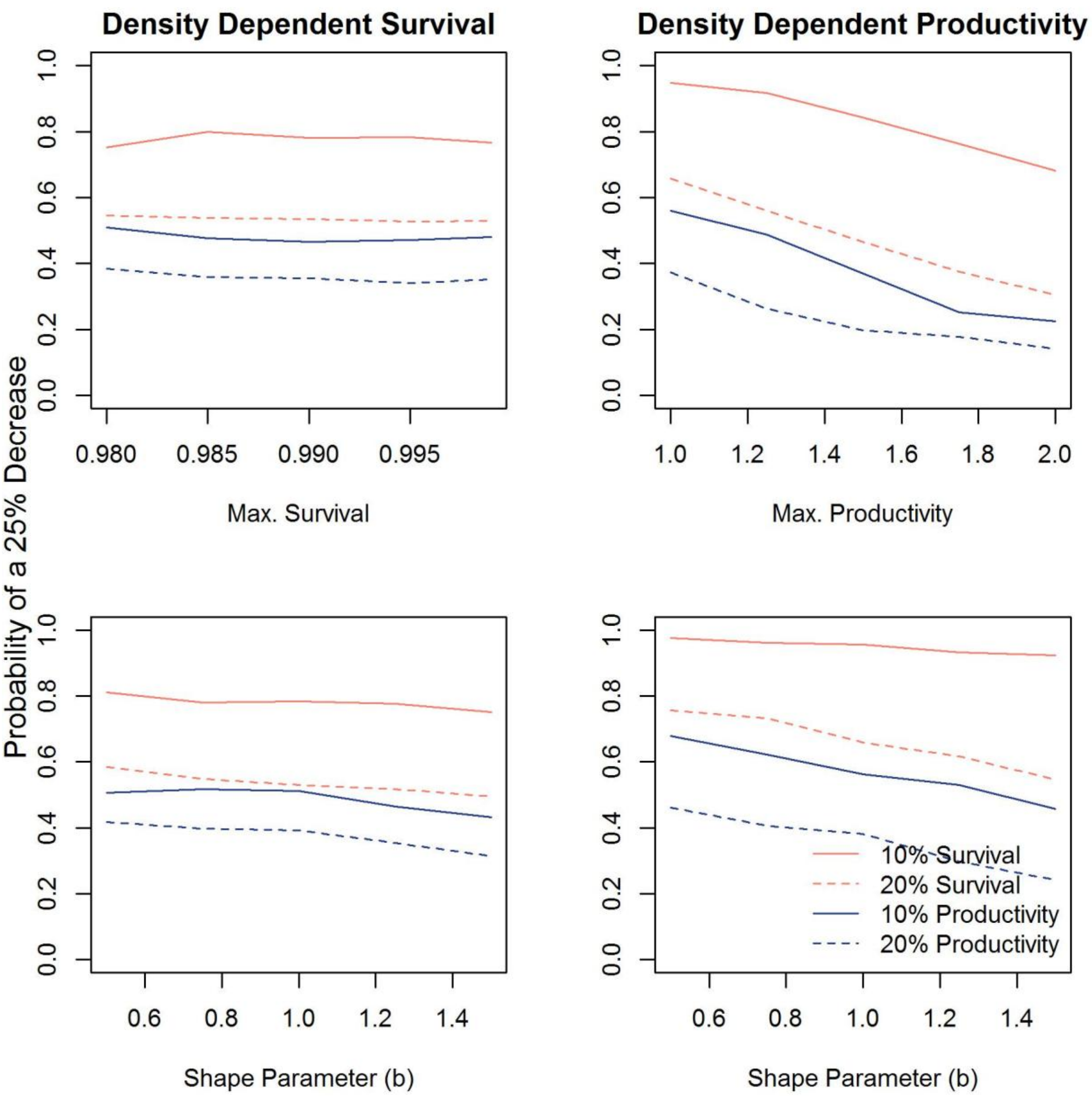

Figure 47. Impact of mis-specifying the shape parameter and maximum survival or productivity rate in a stable population of an r-selected seabird when using a stochastic model with density dependent regulation of survival or productivity.

Metric overview

P(Id>0.25) reaches an asymptote, making it difficult to distinguish the population level effects of more severe impacts, particularly in relation to increases in mortality. The metric appears to be sensitive to assumptions about the underlying trend of the population concerned and to the demographic parameters used in the population models, particularly when impacts are on productivity. The metric is also sensitive to assumptions about the density dependent regulation of productivity.

3.2.10 Change in probability of a 10, 25 or 50% decline (dP(Id.0.1), dP(Id>0.25), dP(Id>0.5))

At many colonies throughout the UK seabird populations are already declining (JNCC 2013). As a consequence, the presence of a wind farm may not 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 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 10% increase in the probability of a 5% decline.

From 0 – 1, with 0 indicating that there is no change in the probability of the population decreasing by a given magnitude between the impacted and unimpacted populations (i.e. no population-level consequence) and values approaching 1 indicating there is a large change in the probability of the population decreasing by a given magnitude between the impacted and unimpacted populations (i.e. a population-level consequence).

Initial Results

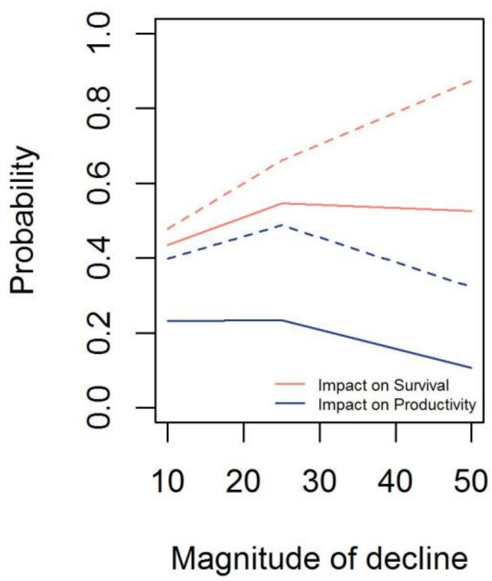

Figure 48. Change in probability of a decline of 10, 25 or 50% assuming a stable population of an r-selected seabird species and a 10% (solid lines) or 20% (broken lines) increase in mortality or reduction in productivity.

When considering the probability of a population decline of 10, 25 or 50% (dP(Id>0.1), dP(Id>0.25), dP(Id>0.5)), the relationship between the probability and the magnitude varies according to the level of impact predicted (i.e. 10% or 20%) and whether survival or productivity is affected (Fig. 48). For a 20% increase in mortality, the change in probability increases between dP(Id>0.1) and dP(Id>0.5). In contrast, for a 10% reduction in productivity, the change in probability decreases over this range. For a 10% increase in mortality, or a 20% decrease in productivity, the change in probability appears to peak at dP(Id>0.25). The reason for this peak is that whilst a 10 % increase in mortality, or up to a 20% decrease in productivity, are sufficient to increase the proportion of simulations showing a 25% decline for an impacted population relative to the unimpacted population, these impacts are not severe enough to cause a similar increase in the proportion of simulations showing a 50% decline. Consequently, further discussion of this metric is focussed on the change in probability of a 25% decline (dP(Id>0.25).

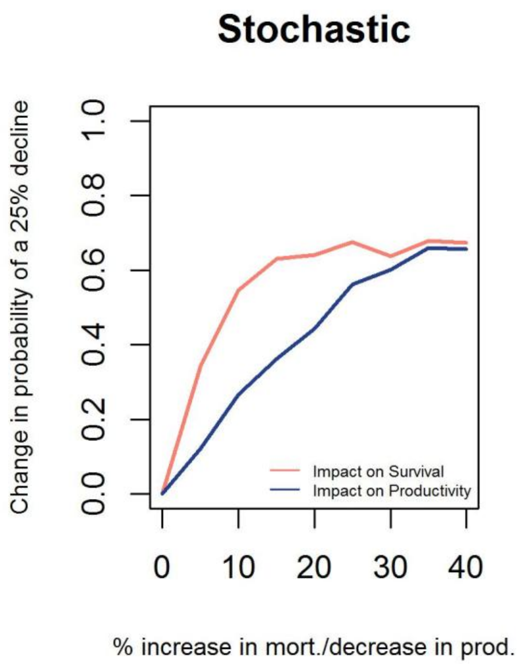

Figure 49. Impact of up to a 40% increase in mortality or up to a 40% decrease in productivity on the change in probability of a decline of 25% for a population of seabirds with an r-selected life history strategy and a stable population trajectory.

dP(ld>0.25) reaches an asymptote at around 0.65 for a 20% increase in mortality. If the impact is on productivity, the probability of a population decline of 25% rises more gradually before also reaching 0.65 after a 40% decrease in productivity (Fig. 49). It is worth noting that dP(ld>0.25) cannot exceed 0.65 as P(ld>0.25) in an unimpacted population is approximately 0.35 (Fig. 44) and cannot exceed 1.

Sensitivity to life history strategy, population trend and density dependence

dP(ld>0.25) appears to be sensitive to the underlying trajectory of the population concerned as there is limited scope for the metric to change in stable and declining populations. Where a population is increasing the metric may indicate the impact of any development relatively clearly. However, where the population is increasing or decreasing, the metric is less capable of detecting any impact, particularly when these impacts are less severe. Species which have a more K-selected life history strategy appear to be more resilient to impacts than those with an r-selected strategy and the probability of a stable population of these species having a growth rate of less than 1 is lower for an equivalent impact level than is the case for an r-selected species (Table 34).

As might be expected, density dependent processes appear to largely mitigate the impacts arising from offshore wind farms on seabirds. However, in contrast to previous metrics, it appears that when impacts on survival are more severe, incorporating density dependence into the model may indicate a more severe impact at a population level (Table 34).

Table 34. Change in probability of a population decline of 25% resulting from a 10% or 20% increase in mortality or a 10% decrease in productivity estimated from a stochastic model and assuming an increasing, stable or decreasing population of an r-selected seabird species, a stable population of a K-selected seabird species and a stable population of an r-selected species with density dependent regulation of survival or productivity.

	Survival			Productivity		
	10%	20%	40%	10%	20%	40%
Increasing	0.007	0.187	0.944	<0.001	<0.001	0.059
Stable	0.552	0.667	0.677	0.255	0.471	0.695
Decreasing	0.016	0.012	0.014	0.007	0.015	0.014
K selected	0.380	0.618	0.762	0.156	0.325	0.680
Density Dependent Survival	0.276	0.504	0.730	0.112	0.233	0.524
Density Dependent Productivity	0.333	0.722	0.899	0.102	0.249	0.643

Sensitivity to mis-specification of demographic parameters

As with previous metrics, dP(Id>0.25) is most sensitive to mis-specification of the adult survival rate. However, it is also sensitive to mis-specification of other demographic parameters (Fig. 50 and Tables 35 and 36)

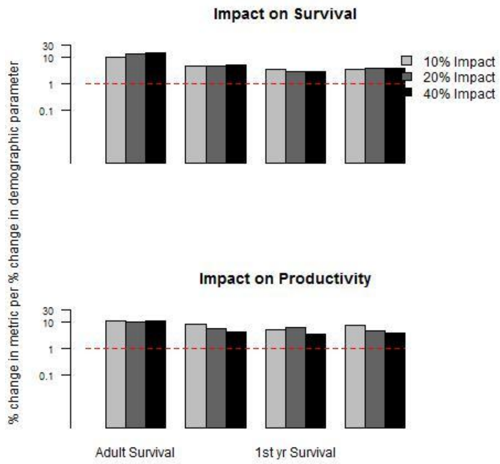

Figure 50. Percentage change in the change in probability of the population declining by 25% per percentage change in adult survival, immature survival, first year survival and productivity for offshore wind farm impacts on survival and productivity from stochastic models. Data tabulated in Tables 35 and 36.

Table 35. Influence of a 1% mis-specification of each demographic parameter on the change in probability of a population decreasing by 25% (as assessed by % change in metric) for an r-selected seabird species estimated from a stochastic model assuming a 10%, 20% or 40% increase in mortality. Data illustrated graphically in Figure 50.

	Stochastic		
	10%	20%	40%
Adult Survival	10.49	13.14	13.44
Immature Survival	4.90	4.80	4.71
Chick Survival	3.21	2.55	2.72
Productivity	4.04	3.03	2.97

Table 36. Influence of a 1% mis-specification of each demographic parameter on the change in probability of a population decreasing by 25% (as assessed by % change in metric) for an r-selected seabird species estimated from a stochastic model assuming a 10%, 20% or 40% reduction in productivity. Data illustrated graphically in Figure 50.

	Stochastic		
	10%	20%	40%
Adult Survival	10.54	9.97	12.33
Immature Survival	5.57	5.93	4.67
Chick Survival	5.12	2.53	2.99
Productivity	7.06	3.27	2.89

Sensitivity to form of density dependence

Where a population has density dependent regulation of survival, dP(ld>0.25) is relatively insensitive to assumptions about the maximum survival rate and the shape parameter. However, where productivity is regulated by density dependence, the metric is sensitive to both of these parameters (Fig. 51).

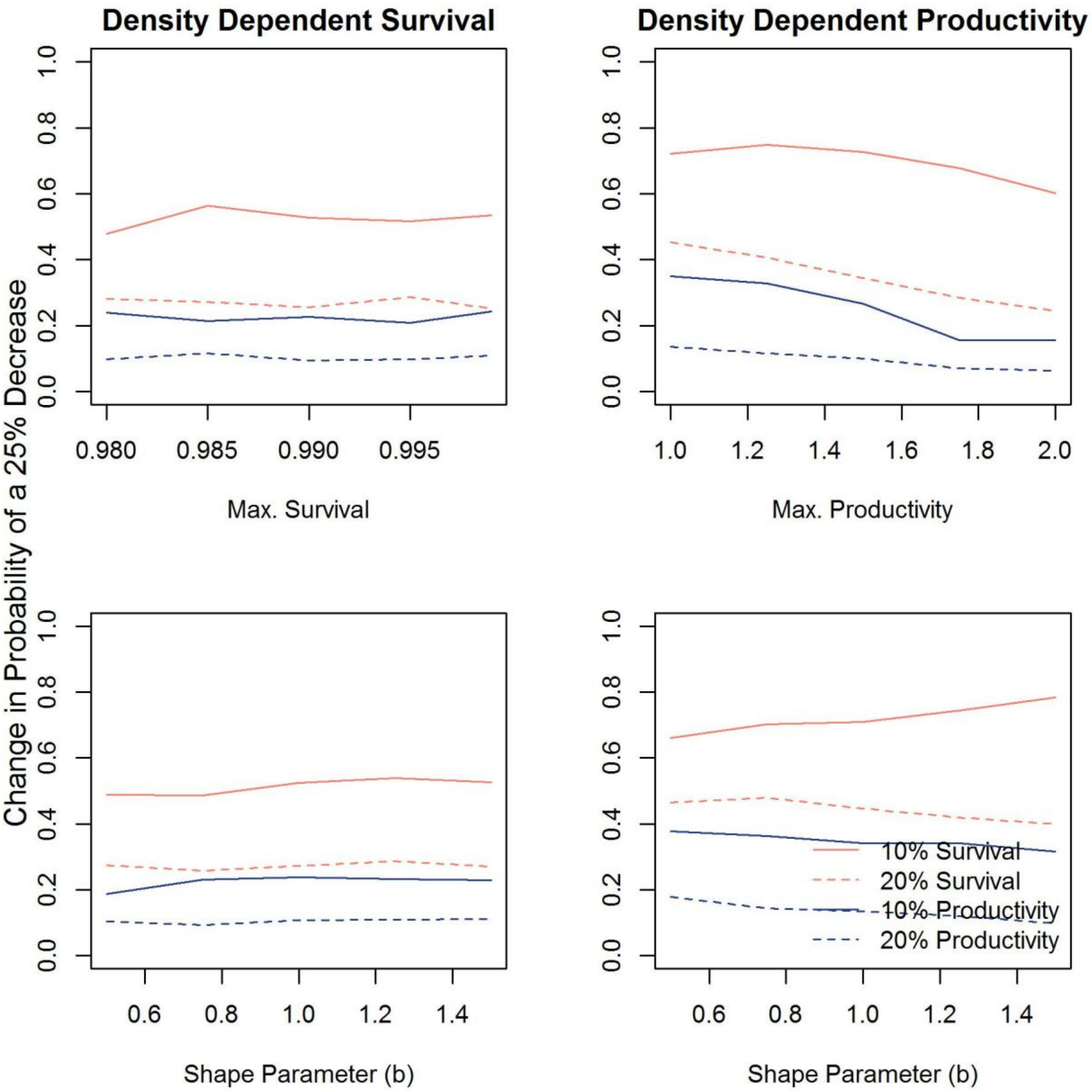

Figure 51. Impact of mis-specifying the shape parameter and maximum survival or productivity rate in a stable population of an r-selected seabird when using a stochastic model with density dependent regulation of survival or productivity.

Metric overview

dP(Id>0.25) reaches an asymptote, making it difficult to distinguish the population level effects of more severe impacts, particularly in relation to increases in mortality. The metric appears to be sensitive to assumptions about the underlying trend of the population concerned and to the demographic parameters used in the population models, particularly when impacts are on productivity. The metric is also sensitive to assumptions about the density dependent regulation of productivity.

3.2.11 Probability of an impacted population being a given magnitude below the median size predicted in the absence of an impact (P(I<25))

A metric to assess the population level impact of a development could be derived by estimating a median size for a population in the absence of 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.

The metric is on a scale from 0 – 1, with 0 indicating that none of the simulations from a stochastic model show the impacted population being a given magnitude below the unimpacted population (i.e. no population-level consequence) and 1 indicating that all simulations show the impacted population a given magnitude below the unimpacted population.

Initial Results

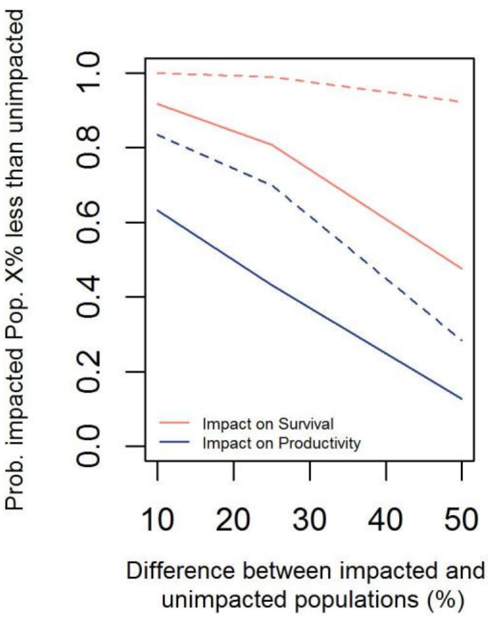

Figure 52. Probability of the impacted population being 10, 25 or 50% less than the unimpacted population after 25 years, assuming a stable population of an r-selected seabird species and a 10% (solid lines) or 20% (broken lines) increase in mortality or reduction in productivity.

When considering whether a population impacted by an offshore wind farm is 10, 25 or 50% less than it would be in the absence of the offshore wind farm, a range of probabilities are obtained for each value depending on the magnitude of the impact considered (Fig. 52).

However, for a 20% increase in mortality, the probability of a 10% change is similar to the probability of a 25% change. For this reason further discussion here focuses on the probability of a 25% decline (P(I<25)).

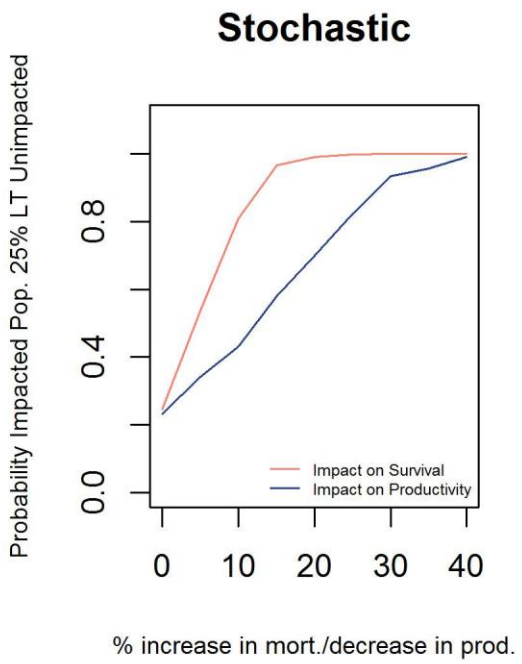

Figure 53. Impact of up to a 40% increase in mortality or up to a 40% decrease in productivity on the probability that the impacted population is 25% less than the unimpacted population after 25 years for a population of seabirds with an r-selected life history strategy and a stable population trajectory.

The probability of an impacted population being 25% less than an unimpacted population (P(I<25)) approaches an asymptote at 1 for a 20% increase in mortality. If the impact is on productivity, P(IGR<2.5) appears to have a more linear relationship with the magnitude of the impact under consideration, reaching 1 for a 40% reduction in productivity (Fig. 53).

Sensitivity to life history strategy, population trend and density dependence

P(I<25) is sensitive to whether the population is declining, rather than increasing or stable (Table 37). There is also evidence to suggest that P(I<25) is more sensitive to population trend if productivity, rather than survival, is affected. Using this metric, K-selected species are far less likely to reveal any population-level impact than r-selected species. As might be expected, incorporating density dependence into the model reduces the magnitude of the population level effect.

Table 37. Probability of the impacted population being 25% less than the unimpacted population resulting from a 10%, 20% or 40% increase in mortality or a 10% decrease in productivity estimated from a stochastic model and assuming an increasing, stable or decreasing population of an r-selected seabird species, a stable population of a K-selected seabird species and a stable population of an r-selected species with density dependent regulation of survival or productivity.

	Survival			Productivity		
	10%	20%	40%	10%	20%	40%
Increasing	0.833	0.985	1.000	0.468	0.730	0.992
Stable	0.829	0.988	1.000	0.391	0.607	0.947
Decreasing	0.853	0.996	1.000	0.383	0.562	0.915
K selected	0.433	0.782	0.991	0.180	0.231	0.458
Density Dependent Survival	0.533	0.746	0.971	0.365	0.441	0.622
Density Dependent Productivity	0.561	0.900	1.000	0.216	0.320	0.622

Sensitivity to mis-specification of demographic parameters

P(I<25) is sensitive to the demographic parameters incorporated in the models (Fig. 54 and Tables 38 and 39). Sensitivity appears to decrease as the magnitude of the impact increases. Sensitivity appears to be greatest where impacts are on productivity, rather than survival.

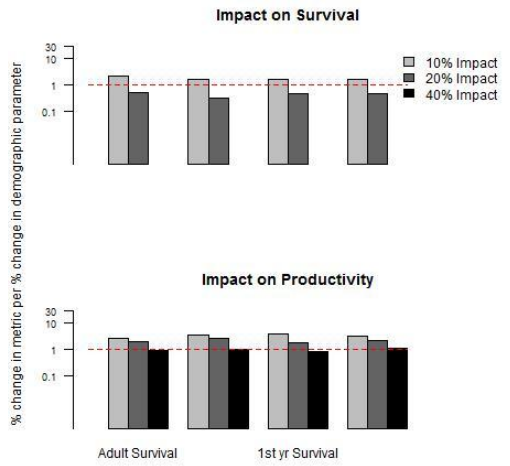

Figure 54. Percentage change in the probability of the impacted being 25% less than the impacted population after 25 years per percentage change in adult survival, immature survival, first year survival and productivity for offshore wind farm impacts on survival and productivity from stochastic models. Data tabulated in Tables 38 and 39.

Table 38. Influence of a 1% mis-specification of each demographic parameter on the probability of the impacted population being 25% less than the impacted population after 25 years (as assessed by % change in metric) for an r-selected seabird species estimated from a stochastic model assuming a 10%, 20% or 40% increase in mortality. Illustrated graphically in Figure 54.

	Stochastic		
	10%	20%	40%
Adult Survival	2.49	0.45	<0.01
Immature Survival	1.95	0.37	<0.01
Chick Survival	1.53	0.38	<0.01
Productivity	1.08	0.39	<0.01

Table 39. Influence of a 1% mis-specification of each demographic parameter on the probability of the impacted population being 25% less than the impacted population after 25 years (as assessed by % change in metric) for an r-selected seabird species estimated from a stochastic model assuming a 10% and 20% reduction in productivity. Illustrated graphically in Figure 54.

	Stochastic		
	10%	20%	40%
Adult Survival	3.84	2.29	0.89
Immature Survival	3.68	1.88	0.76
Chick Survival	2.52	2.43	0.98
Productivity	2.95	1.82	0.86

Sensitivity to form of density dependence

P(I<25) appears to be more sensitive to the form of density dependence assumed than previous metrics. As the maximum value allowable for productivity and the shape parameter increase, the probability of the impacted population being 25% less than the unimpacted population decreases (Fig. 55). This is likely to reflect the density dependent mechanisms acting to increase the growth rate of the impacted population, whilst the unimpacted population remains relatively stable.

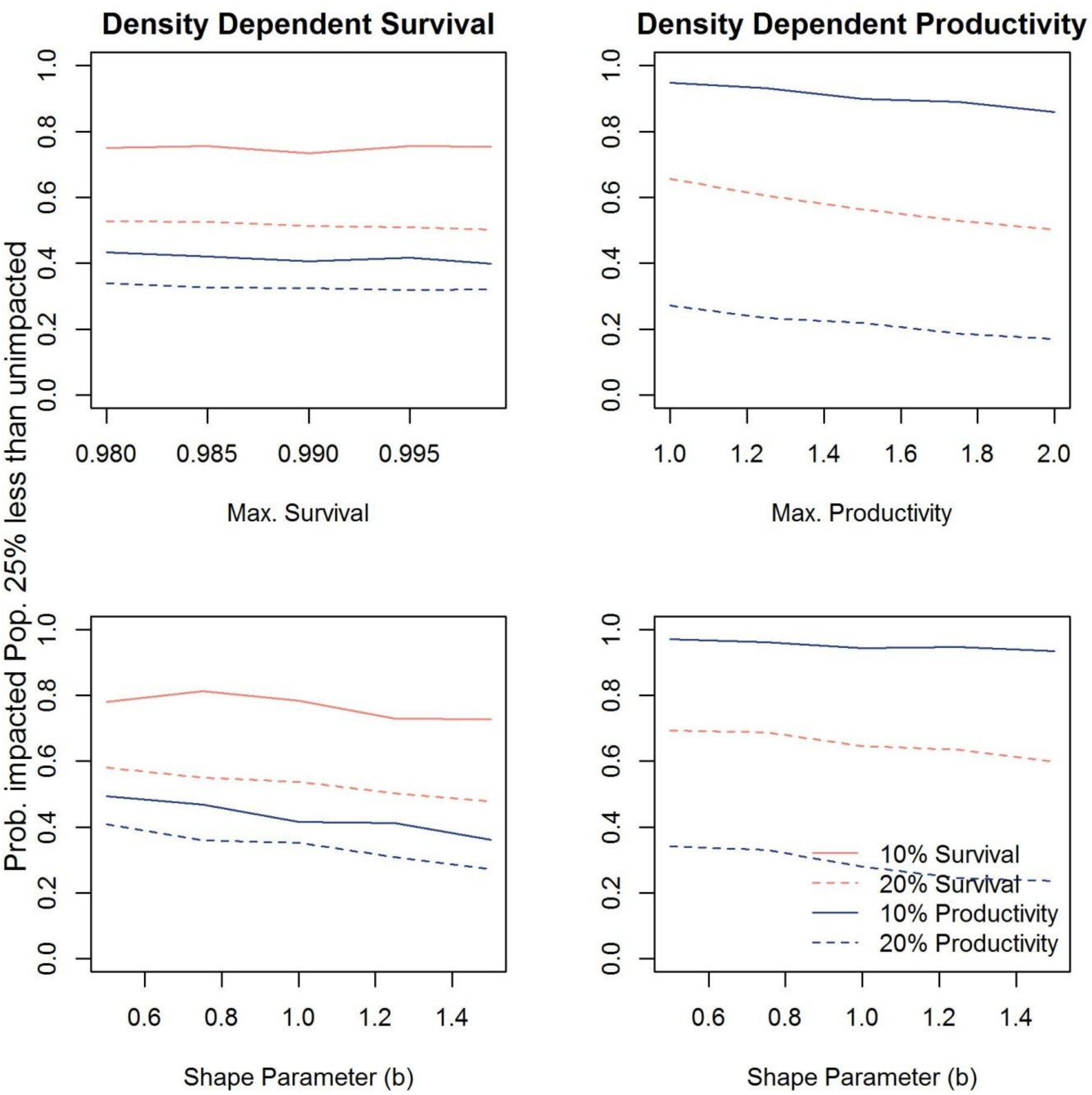

Figure 55. Impact of mis-specifying the shape parameter and maximum survival or productivity rate in a stable population of an r-selected seabird when using a stochastic model with density dependent regulation of survival or productivity.

Metric overview

P(I<25) reaches an asymptote, making it difficult to distinguish the population level effects of more severe impacts, particularly in relation to increases in mortality. The metric appears to be sensitive to assumptions about the underlying trend of the population concerned and to the demographic parameters used in the population models, particularly when impacts are on productivity. The metric is also sensitive to assumptions about the form of density dependence used in any models.

3.2.12 Probability that impacted population growth rate is 2.5, 5 or 10% less than unimpacted growth rate (P(IGR<2.5))

With growth rates simulated from stochastic models, it may be desirable to estimate a mean or median value for the unimpacted population and calculate the proportion of simulations in which the growth rate of the impacted population is lower, or a given percentage lower, than this value. This approach has the advantage of allowing a probabilistic forecast of the impact of the offshore wind farm on a population, e.g. there is a 50% chance that the wind farm will reduce the population growth rate by 10%.

The metric is on a scale from 0 – 1, with 0 indicating that none of the simulations from a stochastic model show the impacted population growth rate being a given magnitude below the unimpacted population (i.e. no population-level consequence) and 1 indicating that all simulations show the impacted population growth rate a given magnitude below the unimpacted population.

Initial Results

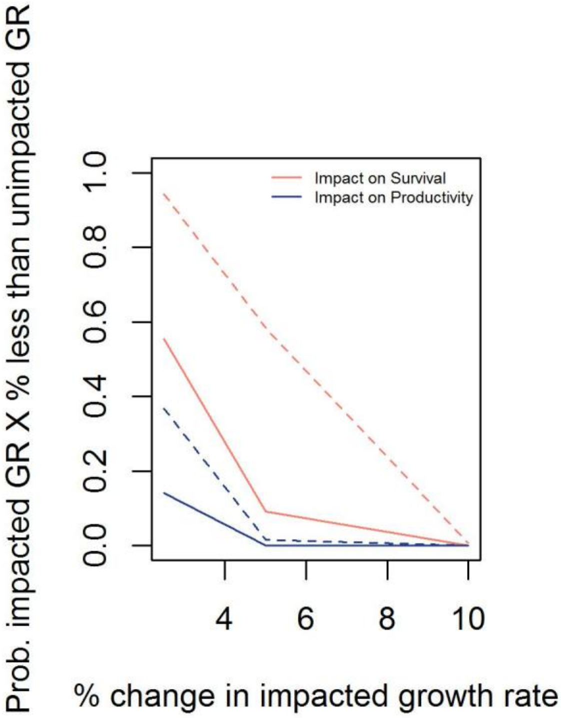

Figure 56. Probability of the growth rate of the impacted population being 2.5, 5 or 10% less than the growth rate of the unimpacted population assuming a stable population of an r-selected seabird species and a 10% (solid lines) or 20% (broken lines) increase in mortality or reduction in productivity.

When considering whether the growth rate of a population impacted by an offshore wind farm it is not possible to detect the population level effect of the impacts considered here when they are assessed against the probability of a 10% reduction in the impacted population growth rate (Fig. 56). Similarly, it is not possible to detect population level effects of the impacts on productivity considered when assessed against the probability of a 5% decline in the growth rate. For this reason, further analysis here focuses on the probability of a 2.5% decline in the population growth rate, which shows a range of values for the different impacts considered here (Fig. 56).

The probability of a change in population growth rate of 2.5% (P(IGR<2.5)) approaches an asymptote at 1 for a 25% increase in mortality. If the impact is on productivity, P(IGR<2.5) appears to have a more linear relationship with the magnitude of the impact under consideration, rising to 0.881 for a 40% reduction in productivity (Fig. 57). However, it should be noted that the confidence limits associated with the population growth rate (see section 3.2.3) may make it hard to detect a 2.5 percentage change in population growth rate with confidence.

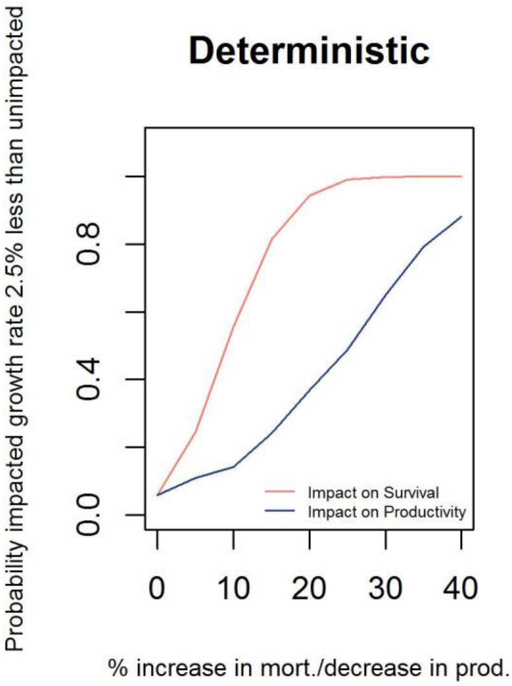

Figure 57. Impact of up to a 40% increase in mortality or up to a 40% decrease in productivity on the probability that the impacted population growth rate is 2.5% less than the unimpacted population growth rate for a population of seabirds with an r-selected life history strategy and a stable population trajectory.

Sensitivity to life history strategy, population trend and density dependence

P(IGR>2.5) is sensitive to population trend with different values recorded for low impacts depending on whether the population is declining or stable/increasing (Table 40). Using this metric, K-selected species are far less likely to reveal any population-level impact than r-selected species. As might be expected, incorporating density dependence into the model reduces the magnitude of the population level effect.

Table 40. Probability of a P(GR) decreasing >2.5% resulting from a 10%, 20% or 40% increase in mortality or a 10% decrease in productivity estimated from a stochastic model and assuming an increasing, stable or decreasing population of an r-selected seabird species, a stable population of a K-selected seabird species and a stable population of an r-selected species with density dependent regulation of survival or productivity.

	Survival			Productivity		
	10%	20%	40%	10%	20%	40%
Increasing	0.561	0.944	1.000	0.171	0.406	0.945
Stable	0.541	0.954	1.000	0.126	0.277	0.752
Decreasing	0.647	0.979	1.000	0.140	0.238	0.647
K selected	0.073	0.333	0.896	0.012	0.013	0.048
Density Dependent	0.289	0.556	0.917	0.145	0.200	0.361

Survival						
Density Dependent Productivity	0.207	0.664	0.995	0.029	0.049	0.150

Sensitivity to mis-specification of demographic parameters

P(IGR>2.5) is sensitive to the demographic parameters incorporated in the models (Fig. 58 and Tables 41 and 42). Sensitivity appears to be greater for more moderate impacts and also where productivity is affected.

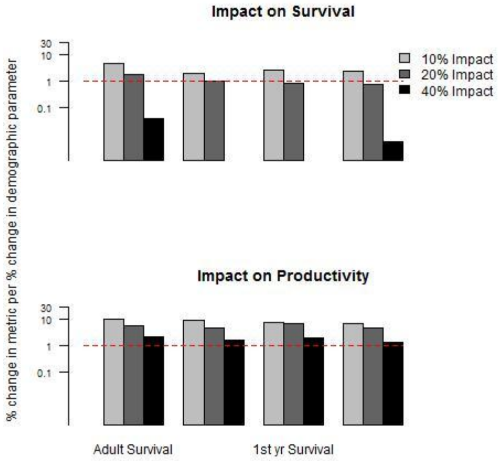

Figure 58. Percentage change in the probability of the impacted population growth rate being 2.5% less than the impacted population growth rate per percentage change in adult survival, immature survival, first year survival and productivity for offshore wind farm impacts on survival and productivity from deterministic (left) and stochastic (right) models. Note that the Y-axis is on a non-linear scale. Data tabulated in Tables 40 & 41.

Table 41. Influence of a 1% mis-specification of each demographic parameter on the probability of the impacted population growth rate being 2.5% less than the impacted population growth rate (as assessed by % change in metric) for an r-selected seabird species estimated from a stochastic model assuming a 10%, 20% or 40% increase in mortality. Illustrated graphically in Figure 58.

	Stochastic		
	10%	20%	40%
Adult Survival	5.02	1.72	0.04
Immature Survival	2.59	0.83	<0.01
Chick Survival	1.89	0.75	0.01
Productivity	2.28	0.99	0.01

Table 42. Influence of a 1% mis-specification of each demographic parameter on the probability of the impacted population growth rate being 2.5% less than the impacted population growth rate (as assessed by % change in metric) for an r-selected seabird species estimated from a stochastic model assuming a 10% and 20% reduction in productivity. Illustrated graphically in Figure 58.

	Stochastic		
	10%	20%	40%
Adult Survival	9.69	5.49	2.54
Immature Survival	6.88	2.84	1.65
Chick Survival	8.06	2.92	2.02
Productivity	6.06	3.82	1.43

Sensitivity to form of density dependence

P(IGR<2.5) appears to be more sensitive to the form of density dependence assumed than previous metrics. As the maximum value allowable for productivity and the shape parameter increase, the probability of the growth rate of the impacted population being 2.5% less than the unimpacted population decreases (Fig. 59). This is likely to reflect the density dependent mechanisms acting to increase the growth rate of the impacted population, whilst the unimpacted population remains relatively stable.

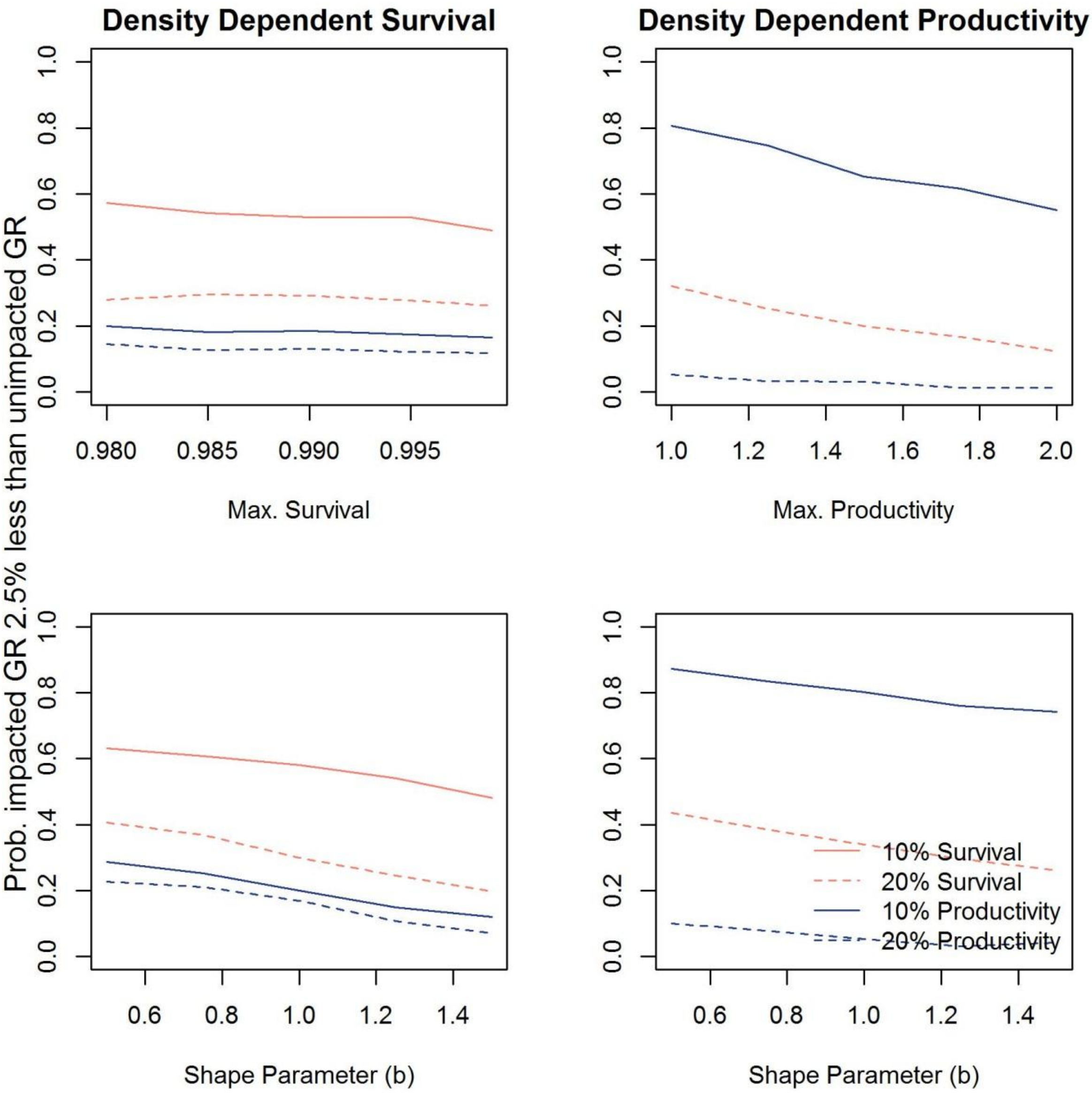

Figure 59. Impact of mis-specifying the shape parameter and maximum survival or productivity rate in a stable population of an r-selected seabird when using a stochastic model with density dependent regulation of survival or productivity.

Metric overview

Overall P(IGR>2.5) is sensitive to mis-specification of the input demographic parameters and assumptions about the underlying population trend. Furthermore, given the uncertainty which is associated with population growth rates as a result of stochasticity, it is likely to be difficult to determine whether a change of 2.5% in the population growth rate is actually statistically significant. Whilst, it may be possible to consider a similar metric, based on a greater change in the growth rate, initial simulations suggested that it was difficult to detect a population level effect using these values as the probability of detecting such a change declined dramatically (see Figure 56).

3.2.13 Overlap between impacted and unimpacted population (OI:U)

Using stochastic models, the population size at any fixed point in time (e.g. at the end of a project lifetime) may be expressed as a distribution. In these circumstances, it may be desirable to compare the distributions of the impacted and unimpacted population trajectories. Where there is greater overlap between the two trajectories, impacts may be deemed less significant. This metric may be expressed as a rule, for example Acceptable Biological Change (ABC, Bennet 2013), whereby impacts are deemed acceptable if the median impacted population after 25 years is predicted to be greater than the 33% quantile of the unimpacted population (i.e. >50% of the simulations from the model of an impacted population result in a population size that is equal to, or greater than, that predicted by the 33% quantile of the unimpacted population). For the purposes of this sensitivity analysis, we compared the overlap of the whole distribution of the impacted and unimpacted populations, without the use of confidence intervals.

The metric is on a scale from 0 – 1, with 0 indicating 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 unimpacted population.

Initial results

The percentage overlap between the impacted and unimpacted populations declines rapidly as impacts on mortality increase. Where mortality is predicted to increase by more than 35%, there is close to zero overlap between the two population sizes after 25 years (Figure 60). If the impact is on productivity, the population sizes after 25 years remain similar for reductions in productivity of up to 25%. A note of caution should be applied to this metric. From the initial analysis of decreases in productivity presented in Figure 60, following a decrease in the metric value between a 25 and 30% decrease in productivity, the metric value then increases again between a 30 and 35% decrease in productivity. This is likely to be an artefact of the number of simulations used to derive the metric and it is likely that an increase in the number of simulations used would remove this apparent discrepancy. Nevertheless, it is important to highlight the potential sensitivity of this metric to the number of simulations used in the population models.

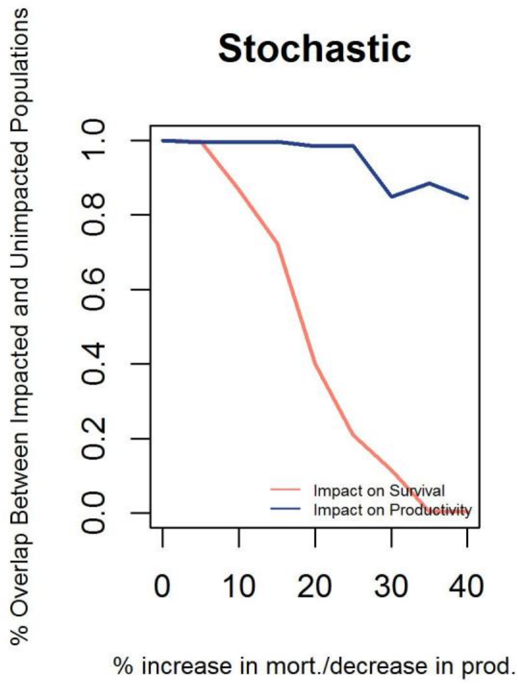

Figure 60. Impact of up to a 40% increase in mortality or up to a 40% decrease in productivity on the % overlap between the impacted and unimpacted population size after 25 years for a population of seabirds with an r-selected life history strategy and a stable population trajectory.

Sensitivity to life history strategy, population trend and density dependence

OI:U is sensitive to population trend with different values recorded for low impacts depending on whether the population is increasing or stable/declining (Table 43). Using this metric, K-selected species are far less likely to reveal any population-level impact than r-selected species. As might be expected, incorporating density dependence into the model reduces the magnitude of the population level effect.

Table 43. Overlap of the impacted and unimpacted population size after 25 years for a 10%, 20% or 40% increase in mortality or a 10%, 20% or 40% decrease in productivity estimated from a stochastic model and assuming an increasing, stable or decreasing population of an r-selected seabird species, a stable population of a K-selected seabird species and a stable population of an r-selected species with density dependent regulation of survival or productivity.

	Survival			Productivity		
	10%	20%	40%	10%	20%	40%
Increasing	0.976	0.497	0.003	0.998	0.991	0.829
Stable	0.840	0.488	0.008	0.999	0.991	0.960
Decreasing	0.840	0.600	<0.001	0.999	0.998	0.901
K selected	0.986	0.911	0.289	0.999	0.999	0.995
Density Dependent Survival	0.999	0.952	0.569	0.999	0.999	0.999
Density Dependent Productivity	0.997	0.827	0.079	0.999	0.999	0.999

Sensitivity to mis-specification of demographic parameters

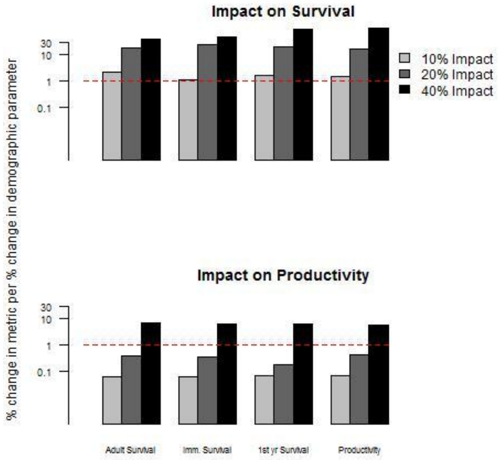

Figure 61. Percentage change in the overlap between the impacted and unimpacted populations per percentage change in adult survival, immature survival, first year survival and productivity for offshore wind farm impacts on survival and productivity. Note that the Y-axis is on a non-linear scale. Data tabulated in Tables 44 and 45.

OI:U is sensitive to the mis-specification of demographic parameters. (Figure 61, Tables 44 and 45). This sensitivity increases as magnitude of the predicted impact increases. This is particularly noticeable for impacts on chick survival and productivity.

Table 44. Impact of a 1% mis-specification of each demographic parameter on overlap between impacted and unimpacted population sizes after 25 years for an r-selected seabird species estimated from a stochastic model assuming a 10%, 20% or 40% increase in mortality.

	Stochastic		
	10%	20%	40%
Adult Survival	2.17	18.57	38.50
Immature Survival	1.13	23.60	48.31
Chick Survival	1.61	18.82	87.71
Productivity	1.40	15.91	100.00

Table 45. Impact of a 1% mis-specification of each demographic parameter on the change in probability of a population decreasing by 25% for an r-selected seabird species estimated from a stochastic model assuming a 10%, 20% or 40% reduction in productivity.

	Stochastic		
	10%	20%	40%
Adult Survival	0.06	0.37	0.68
Immature Survival	0.06	0.35	0.09
Chick Survival	0.07	0.17	0.49
Productivity	0.07	0.40	0.74

Sensitivity to form of density dependence

If density dependence is introduced into the models, OI:U does not appear to be particularly sensitive to either mis-specification of the shape parameter or mis-specification of the maximum survival or productivity rate (Fig. 62), if density dependence is assumed to influence survival. However, where density dependence is assumed to influence productivity, OI:U may be sensitive to the mis-specification of both the shape parameter and the maximum productivity rate where smaller impacts on survival are predicted.

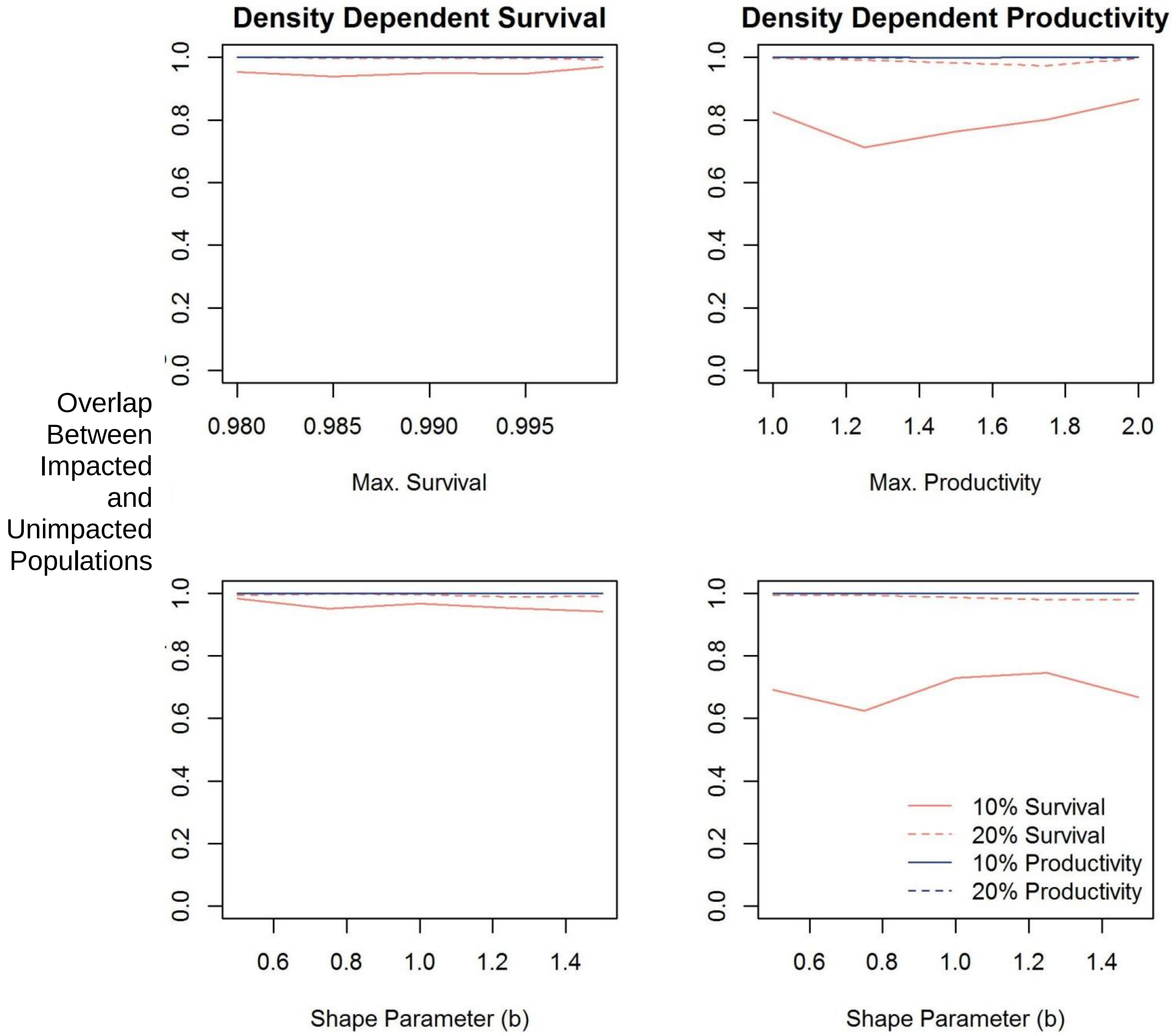

Figure 62. Impact of mis-specifying the shape parameter and maximum survival or productivity rate in a stable population of an r-selected seabird when using a stochastic model with density dependent regulation of survival or productivity.

Additional Analysis

Additional analyses revealed that OI:U showed sensitivity to the extent of uncertainty surrounding the demographic parameters used in population models (Fig. 62). These analyses show that in populations where there is greater uncertainty surrounding the demographic parameters, OI:U is higher, implying a lower population-level consequence, than is the case where there is less uncertainty surrounding the demographic parameters. This is because the wider confidence intervals surrounding the adult survival rate lead to simulations with a greater range of population sizes for both the impacted and unimpacted populations. As non-biological sources (e.g. variability due to sampling error) of variability can constitute a significant proportion of the total observed variability (Gould & Nicholls 1998), ideally, all four distributions should have a similar overlap between impacted and unimpacted populations. This is of concern as it leads to the possibility that an impact might be deemed acceptable simply as a result of a lack of knowledge of the demography of the population concerned, which may be a common occurrence in relation to seabird populations (Maclean *et al* 2007; Robinson & Ratcliffe 2010; Horswill & Robinson 2015).

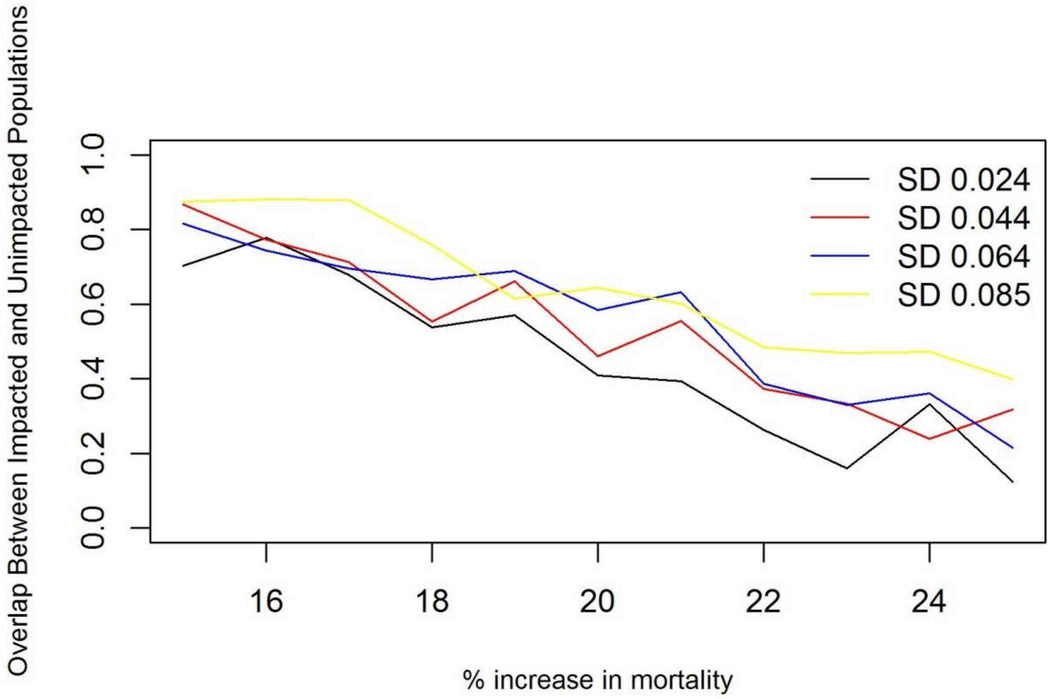

Figure 62. Impact of 15% to 25% increase in mortality on the overlap of impacted and unimpacted population sizes after 25 years for a population of seabirds with an r-selected life history strategy and a stable population trajectory, assuming an adult survival rate of 0.89 and standard deviations of 0.024, 0.044, 0.064 and 0.085.

Metric overview

It appears that OI:U is sensitive to both the demographic parameters used in the population models and to the underlying trend of the population concerned. It is less sensitive to mis-specification of the form of density-dependence, at least where this affects survival rates. Given the relatively high sensitivity of this metric, greater consideration of the value applied by a rule may be required because of its potential influence on the conclusions. There are some signs that OI:U may be sensitive to the number of simulations used in demographic models from which it is derived. In addition, this metric appears to be sensitive to the degree of uncertainty surrounding the parameters of the population models from which it is derived. This may reduce the ability to identify wind farm impacts on a population if knowledge of the demographic parameters is poor.

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Appendix 5 - Jitlal, M, Burthe, S, Freeman, S and Daunt, F (2017) Testing and validating metrics of change produced by population viability analysis (PVA).

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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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Executive Summary

- The aim of this research project was to review the use of Population Viability Analysis (PVA) metrics in the context of assessing the effect of offshore renewable developments on seabirds and to test PVA metric sensitivity to mis-specification of input parameters. The most useful metrics in this context are those that are least sensitive to such mis-specification, enabling more robust assessment of offshore renewable effects.
- Recent work has tested PVA metric sensitivity using a simulation approach. To complement these findings, the objective in this project was to test metric sensitivity using real-world data. This approach is useful where one wishes to understand a specific region where real data are available, or where one wishes to address generic questions with real data. If the same metrics show low sensitivity in models of real world data as in simulation models, then this would provide re-assurance that these metrics are the most appropriate for use in assessments.
- Five study species were selected: black-legged kittiwake *Rissa tridactyla*; common guillemot *Uria aalge*; razorbill *Alca torda*; herring gull *Larus argentatus* and European shag *Phalacrocorax aristolelis*. Of these, the first four were considered in population modelling in the Forth/Tay region in a previous Marine Scotland Science project (Freeman *et al.* 2014). Similar models have, in the interim, also been fitted for shags in this region so this species was also considered. The SPAs considered in this report were Buchan Ness to Collieston Coast SPA, Fowlsheugh SPA, Forth Islands SPA and St Abb's Head to Fastcastle SPA.
- Data on abundance, survival and productivity were collated from a variety of sources. Regular or sporadic counts were available from all sites, based on whole colony or plot counts. Productivity was available from all four SPAs for kittiwakes, and for European shags at two SPAs, otherwise data on demographic rates was limited to the Isle of May in the Forth Islands SPA.

- All models were fitted using a Bayesian approach in the software R/WinBUGS. Model fitting was in 'state-space' form, which allows for 'observation error' and environmental stochasticity simultaneously within the same model. Models forecasted the population size for each species at each SPA, for 25 years from 2016 to 2041. Adult survival was set to decline by one of a range of specified rates equating to offshore renewable effects, namely 0% (i.e. no change), 0.5%, 1%, 2% and 3%. Annual productivity was set to decline by 0%, 1%, 2%, 3% and 5%.
- Previous work has indicated that ratio PVA metrics are less sensitive than probabilistic PVA metrics. Accordingly, we tested the sensitivity of six PVA metrics, comprising two ratio metrics (median of the ratio of impacted to un-impacted annual growth rate; median of the ratio of impacted to un-impacted population size); two metrics related to the ratio metrics (median difference in impacted and un-impacted annual growth rates; median difference between impacted and un-impacted population size) and two probabilistic metrics (probability of a population decline exceeding 10%, 25% or 50%; centile for un-impacted population which matches the 50th centile for the impacted population).
- Sensitivity of the six PVA metrics was assessed in relation to mis-specification of input parameters. We considered adult mortality (the complement of survival, since survival is high in seabirds and % increases are limited by the constraint of lying below a survival rate of 1) and productivity to differ from those of the baseline by: -30%, -20%, -10%, 10%, 20% and 30%. We then assessed sensitivities in relation to population status, combining data from all species/SPAs for which we achieved model convergence. Finally, we assessed PVA sensitivities in relation to scenarios of change resulting from the renewables development (i.e. the effect size).
- The state-space modelling approach proved extremely powerful in forecasting population sizes, in particular where censuses were regular. Even in cases where censuses were sporadic, the models generally performed well, though for three species/SPA populations the models would not converge successfully.
- The two ratio metrics were least sensitive to mis-specification in input parameters. They performed well in populations of different status, and under different scenarios of change. The two difference metrics were not readily interpretable, but proved useful when growth rates or population size estimates were small. The probabilistic metrics were more sensitive to mis-specification to input parameters than the ratio PVA metrics. The 'probability of a population decline' metric has been widely used in assessments but proved highly sensitive to mis-specification. The metric representing the

centile from the un-impacted population size equal to the 50th centile of the impacted population size at the end of the wind farm showed moderately low sensitivity to mis-specification of survival and productivity. It performed considerably better than the other probabilistic metric with markedly lower sensitivity to mis-specification, population status and renewables effect size. However, it was more sensitive than ratio metrics, and in some cases showed unstable sensitivity which was less apparent in ratio PVA metrics.

- We recommend that those undertaking assessments consider the relative performance of different metrics with respect to sensitivity to mis-specification of input parameters. Of the two ratio and two probabilistic metrics, the ratio metric 'median of the ratio of impacted to un-impacted annual growth rate' was least sensitive, followed by the ratio metric 'median of the ratio of impacted to un-impacted population size' and then the probabilistic metric 'centile for un-impacted population which matches the 50th centile for the impacted population'. If these are used in assessments in future, we recommend that interpretation should factor in their relative sensitivities. Furthermore, a priority for future research would be to analyse the probabilistic metric using simulations, to assess whether the same results are found as in this study. The probabilistic PVA metric 'probability of a population decline' was much more sensitive than the other three and is not recommended for use in this context. Finally, we recommend that the two PVA metrics related to the ratio metrics (median difference in impacted and un-impacted annual growth rates; median difference between impacted and un-impacted population size) are used since they are estimable when ratios are being calculated and are useful in some circumstances.

1. Introduction

1.1 Policy Context

The Scottish Government has set a target of 100% of Scottish demand for electricity to be met by renewable sources by 2020. The Scottish Government has a duty to ensure that offshore renewable developments are achieved in a sustainable manner. Scottish Ministers have consented offshore renewable energy sites under Section 36 of the Electricity Act 1989. A licensing process was followed that included the examination of Environmental Statements (ES) which consider the potential impacts and mitigation strategies of the proposed developments.

Offshore renewable developments have the potential to impact on seabird populations that are protected by the EU Birds Directive [2009/147/EC], notably from collisions with turbine blades and through displacement from important habitat (Drewitt & Langston 2006; Larsen & Guillemette 2007; Masden *et al*. 2010; Grecian *et al*. 2010, Langton *et al*. 2011, Scottish Government 2011). Other factors of concern are barrier effects to the movement of migrating or commuting birds, disturbance during construction and operation, toxic and non-toxic contamination and negative effects of developments on the distribution and abundance of prey. Set against these, positive effects may be apparent, in particular if developments result in downstream changes to the physical environment that increase biomass of lower tropic levels (Inger *et al*. 2009). Further, they may act as Fish Aggregating Devices (FADs) creating foraging opportunities for seabirds (Inger *et al*. 2009), though attracting seabirds may increase their vulnerability to other effects such as collision and noise (Scottish Government 2011). Species differ in the sensitivity to disturbance, with auks of intermediate vulnerability and gulls and terns of low vulnerability (Garthe & Hüppop 2004; Langston 2010; Furness *et al*. 2013). These potential effects are predicted to be particularly important for breeding seabirds that, as central place foragers, are constrained to obtain food within a certain distance from the breeding colony (Daunt *et al*. 2002; Enstipp *et al.* 2006).

To aid the future development of offshore renewables, Marine Scotland have developed draft Sectoral Marine Plans for offshore Wind, Wave and Tidal Energy (Scottish Government 2013b) that have involved identifying the available resources and key constraints at a national and regional level, then applying social, economic and environmental assessments to inform the development of plan options. These plans have been subject to a Sustainability Appraisal and public consultation exercise (Scottish Government 2013e) and are underpinned by detailed technical assessments including a Strategic Environmental Assessment (SEA; Scottish

Government 2013d), Habitats Regulations Appraisal (HRA; Scottish Government 2013a) and Socio-economic Assessment (Scottish Government 2013c).

The above analyses have synthesised the potential sensitivities of internationally important seabird populations in Scotland and recognised areas of uncertainty associated with these effects. Therefore, in order to evaluate potential interactions between offshore renewables and marine wildlife in future, Marine Scotland believes that further marine science is required to continue to reduce uncertainty and apply the appropriate level of precaution.

Population Viability Analysis (PVA) provides a robust framework that uses demographic rates to forecast future population levels, either under baseline conditions or under scenarios of change resulting from, for example, an offshore development (Maclean *et al.* 2007; Freeman *et al*. 2014). A sensitivity analysis of PVA metrics to variation in demographic parameters would enable regulators and their advisers to assess the utility of each of these metrics in determining whether a predicted effect is unacceptably large. Demonstrating the validity of these metrics would also ensure that PVA outputs are presented and interpreted in the most suitable way. The outcomes could then be fed back into designing future monitoring requirements. Furthermore, the outputs could inform the establishment of thresholds of acceptable change by regulators, although such an approach has been heavily criticised (Green *et al*. 2016). Finally, they could improve assessments of risk and uncertainty with respect to population viability in environmental assessments and help to ensure that the level of precaution applied is appropriate.

1.2 Project Objectives

An important component of consenting of proposed offshore renewable developments is an assessment of the population consequences on seabirds. Population Viability Analysis (PVA) provides a robust framework that uses assumed or estimated demographic rates (principally survival and productivity) in a mathematical model to forecast future population levels of a wild animal population, either under currently prevailing circumstances or as a consequence of some perturbation to the system (Maclean *et al*. 2007; Freeman *et al*. 2014). Stochastic PVA models are run many times selecting from a distribution of input parameters, resulting in outputs representing the mean, confidence intervals and all quantiles including the 50% (median).

The range of PVA metrics which have the potential to describe the magnitude of a predicted effect on a population include population size by some target date, change in size or growth rate between pairs of consecutive years, trend in population size, counterfactual/ratio of population size or growth rate, probabilities of population decline to below a specific level or a specific percentage of the starting population size, excess probabilities of population decline to below a specific level or a specific percentage of the starting population size, population level predicted to be exceeded with predefined probability (e.g. ‘as likely as not’, Mastrandrea *et al*. 2010) and posterior probabilities (or quantiles derived from them) for any of the above.

This PVA framework allows the sensitivity of these metrics to changes in demographic parameters, notably due to estimation error, to be estimated. This is important as all demographic parameters are estimated with uncertainty, and population change and PVA metrics are disproportionately affected by changes in the magnitude of each. Accordingly, the aim of this project is to review the range of metrics available in PVA analysis and evaluate the sensitivity of these metrics in the context of decision making frameworks.

The report will first review the literature regarding the range of metrics available for use by PVA analysis in the context of renewable assessment frameworks of seabirds. It will then examine the relative sensitivity of a subset of these metrics to mis-specification of input parameters (adult survival and productivity) using PVAs developed on protected seabird populations at SPAs in the Forth/Tay region. It will also assess the impact of mis-specification in the context of population status and effect size of offshore renewable development. Finally, the project will make recommendations on the usefulness and application of the range of metrics within an assessment framework, and make recommendations to inform future assessments that use PVA analysis based on the conclusions of the study.

2. Literature Review

2.1 Introduction

Population Viability Analysis (PVA) uses life-history or population growth rate data to parameterise a mathematical population model to estimate population size and extinction risk of a species into the future (Norton 1995; Beissinger & Westphal 1998; Boyce 2001). Specifically, PVAs have been used for several purposes including predicting the future size of an animal population, estimating the probability of a population going extinct over time, evaluating management strategies most likely to maximise population persistence or exploring how different assumptions consequently alter the viability of small populations (see (Coulson *et al.* 2001)). PVAs have been widely used in conservation biology and wildlife management, aided by the development of intuitive, widely available and user-friendly software packages, particularly to forecast risks of extinction for species of conservation concern (Ludwig 1999). PVAs are a valuable tool because they facilitate the predictive modelling of animal populations under alternative environmental, management or harvesting scenarios and hence can be used to evaluate the effectiveness or consequences of different management decisions. Thus, PVAs can be considered to be a type of risk assessment of the long-term viability of animal populations.

A wide range of models can be considered to be PVAs (Ralls *et al*. 2002). However, in its most common form, PVA utilises life-history parameters (for example growth rates, juvenile and adult mortality, adult fecundity rates etc.) for individuals in a population projection matrix to estimate population size into the future (Boyce 1992). Models can either be deterministic (demographic rates such as survival and reproduction are constant or are determined in a predictable manner) or stochastic (vital rates vary unpredictably over time). Stochastic PVA models, can include demographic stochasticity (e.g. variation between individuals that affects whether a bird survives a given year) and environmental stochasticity (environmental change that would affect all individuals in a group), and hence the variability in the parameters is important, not just the mean values (Maclean, Frederiksen & Rehfisch 2007). PVAs have been developed for a wide range of species from different taxa, including plants (Maschinski *et al.* 2006), invertebrates (for example, sea-urchins (Pfister & Bradbury 1996) and insects (Bauer *et al.* 2013)), amphibians (Pickett *et al.* 2016), reptiles (Enneson & Litzgus 2009), fish (Sweka & Wainwright 2014), birds (Wootton & Bell 2014) and mammals (Pertoldi *et al.* 2013). Although difficult to assess due to the term “PVA” or “Population Viability Analysis” not commonly being included as a keyword, birds appear to be the taxonomic group where PVAs have most commonly been applied. A crude search of Web of Science including the search terms "PVA AND Population Viability Analysis" plus the group (e.g. "mammal") returned 25 citations for plants; 15 for fish; three for reptiles; 38 for birds and 20 for mammals. PVAs have been extensively used in conservation and management with studies focusing on a broad range of topics including: investigating risk of extinction and population viability in small populations (Grayson *et al.* 2014); assessing the impact of different harvesting levels (York *et al.* 2016), predicting population sizes after reintroductions and enhancements (Halsey *et al.* 2015), assessing impacts of threats such as habitat loss (Naveda-Rodriguez *et al.* 2016), climate change (Marrero-Gomez *et al.* 2007) and disease (Haydon, Laurenson & Sillero-Zubiri 2002), assessing effectiveness of alternative management strategies (Ferreras *et al.* 2001); establishing conservation status and strategies (Bevacqua *et al.* 2015); establishing the effectiveness of conservation strategies under a fixed budget (Duca *et al.* 2009); and evaluating which demographic parameters population growth is sensitive to in order to inform management (Mortensen & Reed 2016). As a crude indication, a search in Web of Science found the most published references for the search term "*PVA and management*" (320 references), followed by "*PVA and conservation*" (247), "*PVA and population size*" (167), with few references for "*PVA and renewable energy*" (7) or "*PVA and wind farm(s*)" (2, both on terrestrial wind farms; but note that the majority of studies on PVAs and wind farms are undertaken as part of the planning process e.g. Habitats Regulation Assessments (in Scotland, the law in England and Wales calls them Assessments) and are not published in peer-reviewed journals but within so called "grey- literature").

The outputs of PVAs consist of a predicted population trajectory through time. A suite of metrics have been used to predict the changes in the population of the focal species, both for conservation purposes and as a result of a particular threat or management scheme. Note that the term "metric" is not widely used outside the sphere of PVAs for seabirds and wind farms, where it has broadly been defined (Cook & Robinson 2016a, 2016b) as any value or rule upon which a decision about whether or not a population level effect associated with the impacts of an offshore wind farm is deemed acceptable. We consider the metric to be a value or unit of measurement, and not a rule, and hence cannot be used as an effective search term. A review of the model outputs from general literature in the last five years found that many studies simply reported estimated population sizes or population growth rate for particular time periods (Lopez-Lopez, Sara & Di Vittorio 2012; Wootton & Bell 2014; Naveda-Rodriguez *et al.* 2016). A commonly reported metric was that of quasi-extinction or extinction thresholds, whereby a probability is given for a population declining below a particular threshold (e.g. 10%) after a certain time (e.g. 10 years) or the predicted time to extinction (Blakesley *et al.* 2010; Alemayehu

2013; Hu, Jiang & Mallon 2013; Beissinger 2014; Robinson *et al.* 2016). The difference in extinction probability under different scenarios was reported when comparing management regimes e.g. management Scenario 1 resulted in an X% higher extinction probability than Scenario 2 (Bazzano *et al.* 2014). Susceptibility to quasi-extinction (SQE) has been used to assess whether or not a population is at risk of declining to a specified level (quasi-extinction threshold), a metric which supposedly integrates both parameter uncertainty and stochasticity. This method uses parametric bootstrapping to determine 95% confidence limits of quasi-extinction and then the SQE is defined as the proportion of the bootstrap that indicates a high probability of quasi-extinction (set arbitrarily as >= 0.9 in this paper; Snover and Heppell (2009)).

There are a number of sources of uncertainty that are incorporated within stochastic PVA models (Boyce 1992). There are two main components of uncertainty in time series of demographic variables or population counts: observation and process uncertainty (also called observation and process error or variation). Observation uncertainty (or sampling uncertainty) describes noise in the data that arises due to imprecise or biased empirical data collection methods, for example detection difficulties due to terrain, weather conditions or observer experience and human error. Process uncertainty describes noise that is related to the real variation in the parameter and comprises the real drivers of population fluctuations that are of interest (Bakker *et al.* 2009; Ahrestani, Hebblewhite & Post 2013). Methods for incorporating uncertainty are continuing to advance, including methods for separating out parameter uncertainty and process variation e.g. Heard *et al.* (2013). Therefore, the results of such PVAs are probabilistic, for example risks, probabilities or likelihoods of population decline or extinction. Sensitivity analysis, which determines the amount of change in the model results in response to changes in model parameters, is an important component of PVAs (Saltelli & Annoni 2010; Aiello-Lammens & Akçakaya 2016). Sensitivity analysis can be used to prioritise and inform empirical data collection by establishing the importance of parameters with imperfect knowledge and parameters where improved precision would enhance model predictions. Sensitivity analysis also facilitates understanding and identification of life-history parameters that are highly influential on population size and future viability in order to inform and prioritise conservation or management strategies. Sensitivity analysis is achieved by perturbing the life-history parameters either via a local (one at a time) or global sensitivity analysis (see Mccarthy, Burgman & Ferson (1995); Wisdom, Mills & Doak (2000); Cross & Beissinger (2001); Naujokaitis-Lewis *et al.* (2009); Aiello-Lammens & Akçakaya (2016) for details). Global sensitivity analysis is considered superior to local, because varying local analysis fails to account for the influence of interactions between parameters, but has rarely been applied in part due to computational difficulties and difficulties in quantifying interactions between parameters (Naujokaitis-Lewis *et al.* (2009); Coutts and Yokomizo (2014); but see Aiello-Lammens & Akçakaya (2016)).

Despite the wide application of PVAs to inform and make predictions including the impacts of management or developments, there have been a number of criticisms of their use and how well models can be used to inform management decisions, including how estimates of uncertainty are utilised (Coulson *et al.* 2001; Ellner *et al.* 2002; Reed *et al.* 2002; McCarthy, Andelman & Possingham 2003; Green *et al.* 2016). The quality of the life-history data used to parameterise models may determine how effectively PVAs are able to predict population changes, and for model predictions will only be valid at predicting extinction if the distribution of life-history parameters between individuals and years is stationary in the future (Coulson *et al.* 2001). There is a need to determine and understand how accurately PVAs can predict population size change but the predictions from PVAs are rarely tested against empirical data in the future to establish how well models performed. Criticism has been levied about how the model results can be difficult to understand, assess and interpret by stakeholders (Knight *et al.* 2008; Pe'er *et al.* 2013). Due to uncertainty and variability amongst the input parameters for the PVA models and hence uncertainty associated with the final metrics produced, decision makers may lack confidence in and may misinterpret model predictions (Addison *et al.* 2013; Green *et al.* 2016). Thus, it is critically important that steps are made to solve these challenges where possible (Masden *et al.* 2015; Green *et al.* 2016), since PVAs remain one of the most widely used tools for evaluating the impacts of anthropogenic developments, wildlife management or conservation strategies on focal populations.

2.2 Seabird PVAs and Marine Renewable Developments

One application of PVAs is as a tool to understand the likely impacts of offshore wind farms on seabird populations. The development of offshore wind farms has the potential to be an important anthropogenic intervention into marine habitats. The UK supports nationally and internationally important breeding and wintering populations of seabirds and the UK government has legal obligations to evaluate the effects such developments may have on such populations. The development of offshore wind farms may negatively impact seabird populations by increased mortality associated with direct collisions with turbines, by displacement of birds from suitable foraging areas; and by impeding movements of commuting or migrating birds (Garthe & Huppop 2004; Drewitt & Langston 2006; Everaert & Stienen 2007; Masden *et al.* 2009; Furness, Wade & Masden 2013; Searle *et al.* 2014; Cleasby *et al.* 2015; Vanermen *et al.* 2015; Busch & Garthe 2016). In the UK, a wide number of reports have used PVAs to assess the impact of wind farm developments on seabird populations and to inform the consenting process for approval of these wind farm developments (see Table 1 for examples). It should be noted that details of PVAs for evaluating the impacts of wind farms are largely available through so called “grey literature” (reports and assessments) rather than ISI published papers. PVAs have aimed to either compare the predicted population trajectory into the future with the wind farm development to that without the development, or to quantify the risk that the development poses by establishing probability of future population declines. Both deterministic and stochastic PVA models have been used for evaluating the impacts of wind farms and it has been argued that deterministic models are a more “honest” approach where there is significant uncertainty around demographic parameters because the presented confidence limits from stochastic models imply an unjustified level of precision in the underlying data (WWT 2012). However, stochastic models are more conservative (Lande, Engen & Sæther 2003) and deterministic models do not produce a distribution of results and hence cannot employ probabilistic metrics. A number of different metrics from the PVAs, for example the increase in the probability of a population decreasing by a fixed amount over time, have been used to provide assessments of the impact of wind farms on seabird populations. Metrics have been criticised for being sensitive to uncertainties both in the life-history parameters used to build the models and in the size of the impact of wind farms on the population (Masden *et al.* 2015; Green *et al.* 2016). Uncertainty in the demographic rates used to parameterise models can lead to uncertainty in whether the predicted magnitude of the impact (e.g. increased mortality or reduced productivity) will lead to an adverse effect on the focal population size (Masden *et al.* 2015). Uncertainty in the size of the impact of the wind farms on the population arises due to lack of empirical data on collision risk, displacement or barrier effects on seabird populations. Thus, there is concern that the metrics may not enable accurate predictions and good understanding of the impacts of offshore wind farms on seabird populations (Cook & Robinson 2016a; Green *et al.* 2016). This uncertainty has therefore led to a precautionary approach to assessments (see Thompson *et al.* (2013) for details).

A broad range of metrics have been derived from PVA population models in order to assess the population level effects of wind farm development on seabird populations (Cook & Robinson 2016a). Cook & Robinson (2016a, 2016b) identified 11 metrics that had been derived from population models as part of HRA undertaken for offshore wind farms that were within the planning process. These metrics were summarised from reports from 27 proposed sites at which the population level impacts of offshore wind farms on seabirds had been considered during assessment: Aberdeen Offshore Wind Farm, Beatrice, Burbo Bank Extension, Docking Shoal, Dogger Bank Creyke Beck A, Dogger Bank Creyke Beck B, Dogger Bank Teesside A, Dogger Bank Teesside B, Dudgeon, East Anglia One, Fife Wind Energy Park, Galloper, Hornsea Project One, Inch Cape, London Array Phase II, MORL (MacColl, Stevenson, Telford), Navitus Bay, Neart na Gaoithe, Race Bank, Rampion, Seagreen Alpha, Seagreen Bravo, Triton Knoll 3, Walney I & Walney Extension. The metrics derived from PVAs were split into two broad categories: i) probabilistic approaches (e.g. the probability of the population declining); or ii) ratio approaches (e.g. the ratio of the population size in the presence and absence of the wind farm). Cook & Robinson (2017) builds on this work, but for a reduced set of metrics from the reports, focusing on two PVA metrics (declines in probability difference for both growth rate and population size, equivalent to Metrics 4 and 7 in Table 2; and counterfactual of impacted and un-impacted populations for both growth rate and population size, equivalent to Metrics 2 and 3 in Table 2) and one rule (Acceptable Biological Change) derived from a PVA metric (Metric 15 in Table 2).

2.3 Review Aims

This review builds on the recent report from Cook & Robinson (2016b), which reviewed 11 metrics derived from population models used as part of the HRA undertaken for assessing the impacts of offshore wind farms on seabird populations, by considering a further range of published reports that did not form part of HRAs (see Table 1).

The purpose of our review was to:

1. Provide details of the metrics produced by PVAs;
2. To summarise any evaluations of how sensitive the metrics were to variation in the input parameters in order to recommend which metrics would be useful to pursue further.

In total we review 15 metrics, of which 11 were previously identified in the Cook & Robinson report (2016b). The four additional metrics that we identified were the difference in population growth rate, the difference in population size, the odds ratio of a decline and the centile for un-impacted population which matches the 50th centile for the impacted population (see No's.s 12-15 in Table 2). It should be noted that for stochastic models comparing impacted and un-impacted scenarios, metrics are derived using a "matched runs" approach (WWT 2012; Green *et al*. 2016). Stochasticity is applied to the population, but the same survival and productivity rates are incorporated for both the impacted and un-impacted populations at each time step prior to any impact from an offshore wind farm being applied.

Table 1

Additional reports reviewed for PVA modelling metrics which were recommended by the project steering group and were not included in the Cook & Robinson reviews (2016a and 2016b). N.B. population growth rate is defined as being the mean rate of growth across the period of interest (ratio of the population in year i+1 to that in year i; also known as the population multiplication rate).

Reference	Species considered	Metrics used	Equivalent metric No. and description if already included in Cook & Robinson 2016b (Table 2 in this report). Metrics in bold are not included.
Mackenzie, A. & Perrow, M.R. (2009) Population viability analysis of the north Norfolk Sandwich tern *Sterna sandvicensis* population. Report for Centrica Renewable Energy ltd and AMEC Power & Process. Mackenzie, A. & Perrow, M.R. (2011) Population viability analysis of the north Norfolk Sandwich tern *Sterna sandvicensis* population. Report for Centrica Renewable Energy ltd and AMEC Power & Process JNCC & NE (2012) Defining the level of additional mortality that the North Norfolk Coast SPA Sandwich tern population can sustain. JNCC & NE.	• Sandwich tern	• Probability of population decline: the probability of the simulated population falling below thresholds compared to the starting population • Change in probability of decline: the difference in probability of decline between impacted and un-impacted populations (also known as the Counterfactual of the probability of population decline; CPD)	• No. 7: Probability of a 10, 25 or 50% population decline • No 8: Change in probability of a 10, 25 or 50% population decline
Trinder, M. (2014) Flamborough and Filey Coast pSPA Seabird PVA Final Report: Appendix N to the response submitted for deadline V. Report for SMart Wind.	• Gannet • Kittiwake • Guillemot • Razorbill • Puffin	• Population growth rate • Predicted change in population growth rate i.e. the reduction in growth rate between un-impacted and impacted populations • Probability of population decline • Change in probability of population decline	• No. 1: Population growth rate • **Not included in Cook & Robinson** but similar to No 2: Ratio of median impacted to un-impacted growth rate • No 7: Probability of a 10, 25 or 50% population decline • No 8. Change in probability of a 10, 25 or 50% decline • No 7: Probability of a 10, 25 or 50% population decline (but considered in the final year)

Reference	Species considered	Metrics used	Equivalent metric No. and description if already included in Cook & Robinson 2016b (Table 2 in this report). Metrics in bold are not included.
		• Probability the population size in the final year for the impacted population will be less than a range of percentages of the un-impacted population size • Change in the probability of the population size in the final year for the impacted population will be less than a range of percentages of the un-impacted population size	• No. 8: Change in probability of a 10, 25 or 50% decline (but considered in the final year)
Trinder, M. (2015) Flamborough and Filey Coast pSPA Seabird PVA Report: Appendix M to the response submitted for deadline IIA. Report for SMart Wind.	• Gannet • Kittiwake • Guillemot • Razorbill • Puffin	• Predicted change in population growth rate i.e. the reduction in growth rate between un-impacted and impacted populations • Ratio of the impacted to un-impacted population size (Counterfactual of population size) at 5 year intervals up to 25 years	• **Not included in Cook Robinson** but similar to No 2: Ratio of median impacted to un-impacted growth rate • No. 3: Ratio of the impacted to un-impacted population size
Inch Cape Offshore Limited (2011) Inch Cape Offshore Wind Farm Environmental Statement: Appendix 15B Population Viability Analysis.	• Kittiwake • Guillemot • Razorbill • Puffin	• Change in probability of a population decline	• No. 8: Change in probability of a 10, 25 or 50% decline
Freeman, S., Searle, K., Bogdanova, M., Wanless, S. & Daunt, F. (2014) Population dynamics of Forth and Tay breeding seabirds: review of available models and modelling of key breeding populations. Final Report to Marine Scotland Science.	• Kittiwake • Guillemot • Razorbill • Puffin • Herring gull	• Probabilities of population decline to threshold percentages (25, 50, 75 and 100%) below the baseline in 2015 • Excess probabilities of population decline compared to that predicted by baseline in 2015 for threshold percentages (25, 50, 75 and 100%) i.e. probability of a decrease in the impacted population minus that for the un-impacted population	• No. 7: Probability of a 10, 25 or 50% population decline • No 8: Change in probability of a 10, 25 or 50% population decline

Reference	Species considered	Metrics used	Equivalent metric No. and description if already included in Cook & Robinson 2016b (Table 2 in this report). Metrics in bold are not included.
Moray Offshore Renewables Ltd (2013) Environmental Statement: Ornithology population viability analysis outputs and review.	• Gannet • Kittiwake • Guillemot • Razorbill • Puffin • Fulmar	• Probabilities of the population dropping below threshold percentages (quasi-extinction) of the baseline population size during the lifespan of the project (25 years or 25 years plus 10 year recovery) • Change in probabilities of the population dropping below threshold percentages (quasi-extinction) of the baseline population size during the lifespan of the project (25 years or 25 years plus 10 year recovery)	• No. 7: Probability of a 10, 25 or 50% population decline • No. 8: Change in probability of a 10, 25 or 50% population decline

Table 2

Description of metrics used to assess population responses to impacts of offshore wind farms. For each metric an indication is given of the scale over which the metric operates and a description of the metric. This table is adapted from Table 1 in Cook & Robinson 2016b and includes an additional four metrics (two based on our additional review of the reports listed in Table 1 and two requested to be included by Marine Scotland Science; additional metrics are numbers 12-15).

No.	Ratio or probabilistic	Can be used to distinguish wind farm effects?	Metric	Scale and meaning (N.B. the scale of 0-1 generally only applies if the impact of the wind farm is negative relative to the un-impacted scenario)	Description	Included in Cook & Robinson 2016b
1	**Neither**	**No**	**Population growth rate**	• Value of 1 indicates a stable population • <1 indicates a declining population • >1 indicates an increasing population	Calculation of population growth rate (calculated as mean rate over the study period; Final population size/Initial population size)$^{1/Nyears}$) in the presence of the wind farm enables evaluation of whether the population will remain stable, increase or decrease through the life time of the project.	Yes
2	**Ratio**	**Yes**	**Ratio of median impacted to un-impacted growth rate (counterfactual of population growth rate)**	• Scale from 0 – 1 • Value of 1 indicates the impacted population growth rate is the same as the un-impacted growth rate (no population-level consequence) • Values close to 0 indicate a large proportional difference between the impacted and un-impacted population growth rates (a strong population-level consequence)	Considering only the growth rate of a population (as in No. 1 above) in the presence of an offshore wind farm enables an assessment of whether the population will remain stable, increase or decrease over time, but it does not make it possible to quantify the impact of the wind farm on that growth rate. However, this is possible if the growth rate of the population in the presence of a wind farm is compared to that expected in the absence of a wind farm. This ratio is also known as the **COUNTERFACTUAL OF POPULATION GROWTH RATE**	Yes
3	**Ratio**	**Yes**	**Ratio of impacted to un-impacted population size (counterfactual of population size)**	• Scale from 0 – 1 • Value of 1 indicates the impacted population size is the same as the un-impacted size (no population-level consequence) • Values close to 0 indicate a large proportional difference between the impacted and un-impacted population size (a strong population-level consequence)	PVA models can be used to estimate population size through time both with and without the offshore wind farm. Comparing the ratio of these two population sizes gives a statistic that can be used to assess the population level impact of the offshore wind farm. Cook and Robinson state that the ratio could be derived either from a simple deterministic model or taken from the mean or median values simulated using a more complex stochastic model. We advocate that the ratio should be obtained from the median of x simulations of matched pairs; or in a Bayesian context the median will come from the posterior distribution of	Yes

No.	Ratio or probabilistic	Can be used to distinguish wind farm effects?	Metric	Scale and meaning (N.B. the scale of 0-1 generally only applies if the impact of the wind farm is negative relative to the un-impacted scenario)	Description	Included in Cook & Robinson 2016b
					the ratios. The ratio of population sizes could be estimated either at a fixed point in time, for example at the end of a project, or at a series of intervals throughout the life time of a project. This ratio is also known as the **COUNTERFACTUAL OF POPULATION SIZE (CPS).** For example,CPS_{25} = Predicted population size at 25 years (with wind farm)/ predicted population size at 25 years (no wind farm)	
4	**Probabilistic**	**No**	**Probability that growth rate <1, 0.95, 0.8**	• Scale from 0 – 1 • 0 indicates that none of the simulations from a stochastic model result in a growth rate <1 (decreasing population) • 1 indicates that all of the simulations from a stochastic model result in a growth rate <1	Calculated from a stochastic model based on the proportion of simulations where the population declines (has a growth rate <1). The probability of a population declining is typically assessed over the lifetime of the project, but other time scales could be selected. The metric could consider the probability of the growth rate being below other values (e.g. 0.95 or 0.8) which could be selected with reference to the status of the population concerned. Referred to as the Decline Probability Difference (DPDλ) in Cook & Robinson (2017)	Yes
5	**Probabilistic**	**Yes**	**Change in probability that growth rate <1, 0.95, 0.8 (linked to No. 4)**	• Scale from 0 – 1 • 0 indicates that there is no likely change in the probability of the growth rate being <1 between impacted and un-impacted populations (no population-level consequence) • values approaching 1 indicate there is an almost certainly change in the probability of the growth rate being <1 between the impacted and un-impacted populations (i.e. a population-level consequence)	Quantifying the probability of a population decline in the presence of an offshore wind farm may not be meaningful if simulations show that the population may already be at risk of declining in the absence of a wind farm, for example if >50% of simulations have a growth rate <1. To assess the population level impact of a wind farm it is necessary in this case to determine how much greater the probability of a decline is in the presence of an offshore wind farm than in the absence of an offshore wind farm. This can be done either at a single fixed point in time, or at intervals throughout the life time of the project.	Yes
6	**Probabilistic**	**No**	**Probability that population is below initial size at any point in time**	• Scale from 0 – 1 • 0 indicates that none of the simulations from a stochastic model result in a population below its initial size at any point in time • 1 indicates that all of the simulations	After an initial impact, environmental stochasticity and density dependence may mean a population is able to recover throughout the life time of a project. This recovery would mean that over 25 years the final population size may not be smaller than starting population size.	Yes

No.	Ratio or probabilistic	Can be used to distinguish wind farm effects?	Metric	Scale and meaning (N.B. the scale of 0-1 generally only applies if the impact of the wind farm is negative relative to the un-impacted scenario)	Description	Included in Cook & Robinson 2016b
				from a stochastic model result in a population below its initial size at any point in time		
7	**Probabilistic**	**No**	**Probability of a 10, 25 or 50% population decline**	• Scale from 0 – 1 • 0 indicates that none of the simulations from a stochastic model show the impacted population declining by a given magnitude (no population-level consequence) • 1 indicates that all simulations show the impacted population declining by at least the given magnitude • The probability thresholds are also known as quasi-extinction or pseudo-extinction thresholds	A metric to assess the population level impact of a development could be derived by estimating the proportion of simulations for a population in the presence of a wind farm in which a decline of a given magnitude was recorded. Referred to as the Decline Probability Difference (DPDn) in Cook & Robinson (2017)	Yes

No.	Ratio or probabilistic	Can be used to distinguish wind farm effects?	Metric	Scale and meaning (N.B. the scale of 0-1 generally only applies if the impact of the wind farm is negative relative to the un-impacted scenario)	Description	Included in Cook & Robinson 2016b
8	**Probabilistic**	**Yes**	**Change in probability of a 10, 25 or 50% decline (Linked to No. 7; also known as Counterfactual of the probability of population decline)**	• Scale from 0 – 1 • 0 indicates that there is no likely change in the probability of the population decreasing by a given magnitude between the impacted and un-impacted populations (no population-level consequence) • Values approaching 1 indicate there is a large change in the probability of the population decreasing by a given magnitude between the impacted and un-impacted populations (a population-level consequence)	Seabird populations are already declining at many UK colonies (JNCC 2013). Hence, the presence of a wind farm may not substantially increase the probability of the population size at these colonies being <1, if all simulations from the baseline scenario already have a population size less than the starting population size. However, the presence of the wind farm may cause a further reduction in population size. It may, therefore, be more meaningful to consider the change in probability of population size decreasing by a given magnitude, for example a X% increase in the probability of a Y% decline. Also referred to as the Counterfactual of the probability of population decline (CPD), for example the $CPD_{25,10}$ is the difference in the probability of a decline from the starting population size of 10% occurring 25 years after the wind farm construction between impacted and un-impacted populations. CPD can be calculated relative to the change from the starting population after a set time, or relative to the median population. Risk to the population concerned based on the changes in probability can be assessed using IPCC based likelihoods (see Mastrandrea *et al*. 2010). Such likelihoods simply convert the probabilities of the population dropping below the starting population into more accessible language for stakeholders according to boundaries	Yes
9	**Probabilistic**	**Yes**	**Probability of a population being a given magnitude below the median size predicted in the**	• Scale from 0 – 1 • 0 indicates that none of the simulations from a stochastic model show the impacted population size being a given magnitude below the un-impacted population size (no population-level consequence)	The metric to assess the population level impacts of a wind farm may be derived by estimating a median size for a population in the absence of an offshore wind farm and calculating the proportion of simulations for a population in the presence of a wind farm which were either below this median population size, or a given magnitude below this median population size.	Yes

No.	Ratio or probabilistic	Can be used to distinguish wind farm effects?	Metric	Scale and meaning (N.B. the scale of 0-1 generally only applies if the impact of the wind farm is negative relative to the un-impacted scenario)	Description	Included in Cook & Robinson 2016b
			absence of an impact	• 1 indicates that all simulations show the impacted population is a given magnitude below the un-impacted population size (population level consequence)		
10	**Probabilistic**	**Yes**	**Probability that impacted population growth rate is 2.5, 5 or 10% less than un-impacted growth rate**	• Scale from 0 – 1 • 0 indicates that none of the simulations from a stochastic model show the impacted population growth rate being a given magnitude below the un-impacted population growth rate (no population-level consequence) • 1 indicates that all simulations show the impacted population growth rate is a given magnitude below the un-impacted population growth rate (population level consequence)	With growth rates simulated from stochastic models, it may be desirable to estimate a mean or median value for the un-impacted population and calculate the proportion of simulations in which the growth rate of the impacted population is lower, or a given percentage lower, than this value. This approach has the advantage of allowing a probabilistic forecast of the impact of the offshore wind farm on a population, e.g. there is a 50% chance that the wind farm will reduce the population growth rate by 10%.	Yes
11	**Probabilistic**	**Yes**	**Overlap of Impacted and Un-impacted Populations**	• Scale from 0 – 1 • 0 indicates that none of the simulated population sizes after 25 years from the stochastic model of the impacted population overlap with the simulated population sizes after 25 years from the un-impacted population • 1 indicates that all of the simulated population sizes after 25 years from the stochastic model of the impacted population overlap with the simulated population sizes after 25 years from the un-impacted population	Using stochastic models, the population size at a fixed point in time (i.e. at the end of a project lifetime e.g. 25 years) may be expressed as a distribution. In these circumstances, it may be desirable to compare the distributions of the impacted and un-impacted populations. Where there is greater overlap between the two populations, impacts may be deemed less significant.	Yes
12	**Closely related to ratio approaches**	**Yes**	**Difference in population growth rate i.e. the reduction in growth rate between un-**	• Similar to No. 2 (Ratio of median impacted to un-impacted growth rate) but absolute not ratio values (one growth rate subtracted from the other) • The magnitude of the value relates to the magnitude of the difference	Considering only the growth rate of a population (as in No. 1) in the presence of an offshore wind farm enables an assessment of whether the population will remain stable, increase or decrease over time, but it does not make it possible to quantify the impact of the wind farm on that growth rate. However, as with No. 2, this is possible if the growth rate of the population in	No; closely related to No. 2

No.	Ratio or probabilistic	Can be used to distinguish wind farm effects?	Metric	Scale and meaning (N.B. the scale of 0-1 generally only applies if the impact of the wind farm is negative relative to the un-impacted scenario)	Description	Included in Cook & Robinson 2016b
			impacted and impacted populations	between the two growth rates	the presence of a wind farm is compared to that expected in the absence of a wind farm.	
13	**Closely related to ratio approaches**	**Yes**	**Difference in population size i.e. the reduction in population size between un-impacted and impacted populations**	• Similar to No. 3 (Ratio of impacted to un-impacted population size) but absolute not ratio values (one population size subtracted from the other) • The magnitude of the value relates to the magnitude of the difference between the two population sizes	PVA models can be used to estimate population size through time both with and without the offshore wind farm. Comparing these two population sizes by looking at the difference between them enables assessment of the population level impact of the offshore wind farm. As with No 3, the metric of population sizes could be estimated either at a fixed point in time, for example at the end of a project, or at a series of intervals throughout the life time of a project.	No; closely related to No. 3
14	**Probabilistic**	**Yes**	**Odds Ratio of a threshold population decline comparing impacted to un-impacted populations**	• An odds ratio of 1 implies that the presence of the wind farm has no effect on the probability of an event (e.g. a threshold population decline) • An odds ratio >1 implies that the wind farm leads to an increase in the probability of the event	Odds ratios are a way of quantifying the odds of an event happening and provide an additional way of reporting the impacts of a wind farm on seabird populations. However, we did not find any instances where odds ratios were used as metrics for PVAs associated with wind farms in the literature examined in Table 1. The odds ratio essentially provides a summary of the difference between the probabilities for impacts and un-impacted populations so is an alternative way of quantifying the difference between the raw probabilities. For example: - If a decline of 50% in the population (N.B. the level of the decline is not actually relevant to the calculation of the odds ratio) has been estimated to have a probability of 0.2 in the absence of a wind farm, but 0.5 when the wind farm is present - then the odds ratio for the effect of the wind farm is: (0.5 / (1 - 0.5)) / (0.2 / (1 - 0.2)) = 4 - the wind farm has the effect of multiplying the odds of the event (a 50% decline) by four.	No; closely related to No. 8

No.	Ratio or probabilistic	Can be used to distinguish wind farm effects?	Metric	Scale and meaning (N.B. the scale of 0-1 generally only applies if the impact of the wind farm is negative relative to the un-impacted scenario)	Description	Included in Cook & Robinson 2016b
15	**Probabilistic**	**Yes**	**Centile for un-impacted population which matches the 50th centile for the impacted population**	• Related to No. 11 • Values between 0 and 100	This metric is the centile for the un-impacted population which matches the 50th centile of the impacted population. The centile values are taken from the distributions of the impacted and un-impacted populations. The metric from which Acceptable Biological Change (Marine Scotland 2015) is derived.	No; closely related to No. 11

2.4 Sensitivity of PVA Metrics

The second aim of our review was to **summarise any evaluations of how sensitive the metrics were to variation in the input parameters in order to recommend which metrics would be useful to pursue further.** Metrics have been criticised as being sensitive to uncertainties in the demographic parameters used in the modelling process and in the magnitude of the impact predicted on populations (Green *et al.* 2016). In order to evaluate this, Cook & Robinson (2016b) conducted analyses to quantify how sensitive the conclusions drawn from each model were to uncertainty in the demographic parameters used in the population models, the structure of the population models used to derive the metrics and the magnitude of the impact considered. Cook & Robinson (2017) built on this sensitivity analysis by comparing model sensitivity for the counterfactual metrics (No's.s 2 and 3 in Table 2) between models run using a matched runs approach and those without (i.e. where base demographic rates within a stochastic population model vary between un-impacted and impacted populations).

Overall, Cook & Robinson evaluated the metrics according to whether the metric responses were **clear** (the metric shows a noticeable change in response to impacts of increasing magnitude) and **consistent** (the shape of the relationship between the metric and the magnitude of the impact was linear). A clear response would make it easier to distinguish between population level changes associated with differing magnitudes of the impact. Thus, if metrics are not clear then it may be difficult to distinguish impacts arising as a result of the wind farm from natural variation in the population. The shape and consistency of the response are also important because if the response is consistent then it is easier to understand and predict the relationship between the metric and the population level impacts and to understand the consequences of under- or over-estimating the magnitude of impacts. Curved relationships between metrics and the magnitude of the impact are more difficult to interpret than linear relationships because the effects on the population will depend on the magnitude of the impact and hence conclusions are more vulnerable to mis-specification of model parameters. Cook & Robinson concluded that none of the 11 metrics they considered showed both a clear and consistent response to impacts of increasing magnitude, and that none of the probabilistic approaches gave responses that were clear or consistent. Of the 11 metrics, population growth rate, ratio of impacted to un-impacted population growth rate and ratio of impacted to un-impacted population size were the most promising (see Cook & Robinson 2016b; Cook & Robinson 2017). Population growth rate and ratio of impacted to un-impacted population growth rate were promising because of a consistent linear relationship with the magnitude of the impact. However, due to overlap in the confidence limits for these metrics and the range over which they operate, distinguishing population level effects with and without the wind farm would be difficult unless the magnitude of the impact was very large. The ratio of impacted to un-impacted population size was promising because it was the only metric that showed a clear response to the range of impacts considered in the analysis.

Cook & Robinson specifically tested sensitivity to the following:

1. **Population trend**: whether the metric was sensitive to the population trend prior to wind farm construction increasing, decreasing or being stable.
2. **Mis-specification of the demographic parameters:** whether the metrics are sensitive to changes in the demographic parameters (i.e. a large change in the metric arises from a small change in the demographic parameter; for:

    i. Adult survival;
    ii. Immature survival;
    iii. Chick survival;
    iv. Productivity.

3. **Density dependence:** whether the metric is sensitive to inclusion of density dependence on productivity and breeding adult survival in the models.
4. **The form of density dependence:** whether the metric is sensitive to the form of density dependence in the models i.e. how quickly the adult survival rate changes as the population approaches or moves away from the carrying capacity (rather than whether this is compensatory i.e. population growth rate will reduce with increasing density or depensatory i.e. population growth rates will reduce with decreasing density).
5. **Whether stochastic or deterministic:** whether the metric is sensitive to the inclusion of stochasticity (i.e. is modelled from input parameters over a range of values rather than a fixed value).

The most promising metrics for use in assessing the population level effects of wind farms on seabirds were considered to be the ratio of impacted to un-impacted population growth rate (No. 2 in Table 2) and the ratio of impacted to un-impacted population size (No. 3 in Table 2). Cook & Robinson (2017) recommended that stochastic models using a matched run approach are used because this is likely to reflect the most precautionary approach. The median values of the decision criteria predicted for the counterfactual metrics (Metrics 2 and 3) were greater when a matched run approach was used than when models were run without (see Cook & Robinson 2017). See Table 3 for a full summary of sensitivity of all metrics to the five criteria listed above and a summary of how clear and consistent the metrics were. Table 4 summarises the main strengths and weaknesses of each metric and how the metric should be used and interpreted if being used to assess the impacts of wind farms.

Table 3

Sensitivity of metrics used to determine the impacts of offshore wind farms on seabird populations to variation in the input parameters (adapted from Table 5 in Cook & Robinson (2016b)). Shading indicates how well each metric performs: light grey indicates good, dark grey moderate and black poor performance. The two main criteria (highlighted with a thick black line) are whether there was a clear and consistent relationship between the magnitude of the effect and the metric. N.B. probabilistic metrics cannot be calculated from deterministic models, so the comparison between stochastic and deterministic models is not applicable. No's.12-14 from Table 2 were not included as these were not included in the sensitivity analysis from Cook & Robinson (2016b).

No.	Metric	**Clear** • **Green**: clear difference between metrics for impacts of increasing magnitude • **Amber**: metric varies over a very narrow range • **Red**: the metric reaches an asymptote with impacts of increasing magnitude	**Consistent** • **Green**: linear relationship • **Amber**: non-linear curved relationship • **Red**: stepped relationship	**Insensitive to population trend** • **Green**: identical regardless of population trend • **Amber**: <10% change in metric in relation to population trend • **Red**: >10% change in metric in relation to population trend	**Insensitive to adult survival** • **Green**: sensitivity to misspecification <1% for 10% impacts on survival or productivity • **Amber**: sensitivity to misspecification <5% for 10% impacts on survival or productivity • **Red**: sensitivity to misspecification >5% for 10% impacts on survival or productivity	**Insensitive to immature survival** • **Green**: sensitivity to misspecification <1% for 10% impacts on survival or productivity • **Amber**: sensitivity to misspecification <5% for 10% impacts on survival or productivity • **Red**: sensitivity to misspecification >5% for 10% impacts on survival or productivity	**Insensitive to chick survival** • **Green**: sensitivity to misspecification <1% for 10% impacts on survival or productivity • **Amber**: sensitivity to misspecification <5% for 10% impacts on survival or productivity • **Red**: sensitivity to misspecification >5% for 10% impacts on survival or productivity	**Insensitive to productivity** • **Green**: sensitivity to misspecification <1% for 10% impacts on survival or productivity • **Amber**: sensitivity to misspecification <5% for 10% impacts on survival or productivity • **Red**: sensitivity to misspecification >5% for 10% impacts on survival or productivity	**Insensitive to incorporation of density dependence** • **Green**: median values the same with or without density dependence • **Amber**: <10% change with density dependence vs independent • **Red**: >10% change with density dependence vs independent	**Insensitive to the form of density dependence incorporated** • **Green**: straight line regardless of shape of density dependence or max value of pructivity/adult survival • **Amber**: wavy line • **Red**: clear relationship with density dependence	**Insensitive to stochastic/deterministic model** • **Green**: median values same for both model types • **Amber**: <10% change between models • **Red**: >10% change between models
1	Population growth rate										
2	Ratio of median impacted to un-impacted growth rate										
3	Ratio of impacted to un-impacted population size after 25 years										
4	Probability that growth rate <1										
5	Change in probability that growth rate <1										
6	Probability that population is below initial size at any point in time										
7	Probability of a 25% population decline										
8	Change in probability of a 25% decline										
9	Probability of a population being 50% below un-impacted population										
10	Probability that impacted population growth rate is 2.5% less than un-impacted growth rate										
11	Overlap of Impacted and Un-impacted Populations										

Table 4

Overview of the strengths and weaknesses of the different metrics and information on how the metric should be used to assess the impacts of wind farms. Table adapted from Table 6 in Cook & Robinson (2016b) with the addition of numbers 12 and 13 which were not included in the sensitivity analysis from Cook & Robinson (2016b). We have not included metrics 14 or 15 since sensitivity of these metrics to input parameter specification has not been assessed, so it is not possible to synthesise their strengths and weaknesses.

No.	Metric	Strengths	Weaknesses	How to use and interpret the metrics
1	**Population growth rate**	• Easy to interpret • Consistent relationship between metric and magnitude of impact: easier to make predictions about likely impacts • Relatively insensitive to misspecification of the input parameters	• On its own can't be used to assess wind farm impacts due to lack of comparison with un-impacted population • Variability around the estimates mean it can be difficult to distinguish between variation in the baseline population growth rate and the impacts from the wind farm	• Not a meaningful metric on its own- need to compare the population growth rate of the un-impacted population with that of the impacted population in order to understand then population level effect associated with a wind farm • Lack of a significant difference between impacted and un-impacted populations does not necessarily mean that there would be no population level consequences of the wind farm (due to overlapping confidence intervals)
2	**Ratio of median impacted to un-impacted growth rate**	• Consistent relationship between metric and magnitude of impact: easier to make predictions about likely impacts • Insensitive to misspecification of the input parameters and relatively insensitive to uncertainty in parameter estimates • Insensitive to population trend: metric reflects impact of wind farm and not population status	• Metric varies over a limited range, with the overlapping confidence limits this makes it hard to determine likely population level effects from different magnitudes of effect • Hard to assess effects of the wind farm in a population context due to this limited range	• Metric can be used regardless of population status or trend • Metric should be presented as a median value with 95% confidence limits • Thresholds for determining a wind farm impact are subjective but could be set in reference to the status or trend of the population • Models should be run with a matched run approach
3	**Ratio of impacted to un-impacted population size**	• Easy to interpret in context of a population effect • Clear relationship between metric and magnitude of impact: easier to make	• Sensitive to population declines • More sensitive to misspecification of the demographic parameters than population growth rate or ratio of impacted to un-	• Metric can be used for stable or increasing populations on its own • May be useful context for the ratio of impacted to un-impacted population

No.	Metric	Strengths	Weaknesses	How to use and interpret the metrics
	after 25 years	predictions about likely impacts • Relatively insensitive to uncertainty in the demographic parameters	impacted population growth rate	growth rate regardless of trend • Metric should be presented as a median value with 95% confidence limits • Thresholds for determining a wind farm impact are subjective but could be set in reference to the status or trend of the population • Models should be run with a matched run approach
4	**Probability that growth rate <1**	• Easy to understand, intuitive	• On its own can't be used to assess wind farm impacts due to lack of comparison with un-impacted population • Sensitive to misspecification of adult survival rate • Sensitive to population trends: if population is stable/declining then metric only varies over limited range and so it is difficult to identify population level effects associated with different impacts • True variation in parameters and that based upon observation error are usually not distinguished • Measures are sensitive to any change in conditions in the future	• Not a meaningful metric on its own- need to compare the population growth rate of the un-impacted population with that of the impacted population in order to understand then population level effect associated with a wind farm • Can only be used when the population was increasing prior to the wind farm construction • Requires robust measures of site-specific adult survival • Thresholds for determining a wind farm impact are subjective but could be set in reference to the status or trend of the population
5	**Change in probability that growth rate <1**	• Easy to understand, intuitive: metric quantifies the change in probability of a population declining as a result of a wind farm	• Sensitive to population trend • Sensitive to misspecification of demographic parameters • True variation in parameters and that based upon observation error are usually not distinguished • Measures are sensitive to any change in	• Should not be used when the populations were declining prior to wind farm construction where the change in probability of growth rate is already close to 1 • Requires robust, site specific data on demographic parameters

No.	Metric	Strengths	Weaknesses	How to use and interpret the metrics
			conditions in the future	• Thresholds for determining a wind farm impact are subjective but could be set in reference to the status or trend of the population
6	**Probability that population is below initial size at any point in time**	• Accounts for the fact that populations may recover over the lifetime of the wind farm	• On its own can't be used to assess wind farm impacts due to lack of comparison with un-impacted population • Sensitive to population trends prior to wind farm construction • Sensitive to misspecification of the demographic parameters • True variation in parameters and that based upon observation error are usually not distinguished • Measures are sensitive to any change in conditions in the future	• Not a meaningful metric on its own - need to compare the population growth rate of the un-impacted population with that of the impacted population in order to understand then population level effect associated with a wind farm • Can only be used when the population was increasing prior to the wind farm construction • Requires robust measures of site-specific adult survival • Thresholds for determining a wind farm impact are subjective but could be set in reference to the status or trend of the population
7	**Probability of a 25% population decline**	• Easy to understand • Can be related to established conservation assessments (e.g. (Eaton *et al.* 2015)	• On its own can't be used to assess wind farm impacts due to lack of comparison with un-impacted population • Sensitive to population trends prior to wind farm construction • Sensitive to misspecification of the demographic parameters • True variation in parameters and that based upon observation error are usually not distinguished • Measures are sensitive to any change in conditions in the future	• Not a meaningful metric on its own - need to compare the population growth rate of the un-impacted population with that of the impacted population in order to understand then population level effect associated with a wind farm • Can only be used when the population was increasing prior to the wind farm construction • Requires robust measures of site-specific adult survival • Thresholds for determining a wind farm impact are subjective but could be set in reference to the status or trend of the population

No.	Metric	Strengths	Weaknesses	How to use and interpret the metrics
8	**Change in probability of a 25% decline**	• Easy to understand, intuitive: metric quantifies the change in probability of a population declining by 25% as a result of a wind farm	• Sensitive to population trends prior to wind farm construction • Sensitive to misspecification of the demographic parameters • True variation in parameters and that based upon observation error are usually not distinguished • Measures are sensitive to any change in conditions in the future	• Should not be used when the populations were declining prior to wind farm construction where the change in probability of growth rate is already close to 1 • Requires robust, site specific data on demographic parameters
9	**Probability of a population being 25% below un-impacted population**	• Easy to understand, intuitive comparison of impacted and un-impacted populations • Can be related to established conservation assessments (e.g. (Eaton *et al.* 2015)	• Some sensitivity to population trends prior to wind farm construction • Sensitive to misspecification of the demographic parameters • True variation in parameters and that based upon observation error are usually not distinguished • Measures are sensitive to any change in conditions in the future	• The 25% threshold is subjective and may not be appropriate. Consideration needs to be given to whether to whether alternative thresholds may be more appropriate considering the status and importance of the focal population • Requires robust, site specific data on demographic parameters • Sensitivity to the form and inclusion of density dependence means that models with density dependence should only be used where there is robust evidence for it occurring within the population
10	**Probability that impacted population growth rate is 2.5% less than un-impacted growth rate**	• Relates the impacted population growth rate to that of the un-impacted population	• Difficult to understand in a population context • May be statistically difficult to detect a 2.5% difference in growth rate. Could use higher levels of change but more severe impacts would be required to detect them • Sensitive to population trends prior to wind farm construction • Sensitive to misspecification of the demographic parameters • True variation in parameters and that based upon observation error are usually	• Should not be used when the populations were declining prior to wind farm construction where the change in probability of growth rate is already close to 1 • Requires robust, site specific data on demographic parameters • Sensitivity to the form and inclusion of density dependence means that models with density dependence should only be used where there is robust evidence for it occurring within the population

No.	Metric	Strengths	Weaknesses	How to use and interpret the metrics
			not distinguished • Measures are sensitive to any change in conditions in the future	
11	**Overlap of Impacted and Un-impacted Populations**	• Straightforward comparison that looks at how similar the model outputs are for impacted and un-impacted populations	• Sensitive to population trends prior to wind farm construction • Sensitive to misspecification of the demographic parameters • Sensitive to estimates of uncertainty surrounding the demographic parameters • Value can depend on the number of simulations used in the modelling to obtain the metric • True variation in parameters and that based upon observation error are usually not distinguished • Measures are sensitive to any change in conditions in the future	• Sensitive to population trends means the metric should only be used where there is good understanding of the status of the focal population • Requires robust, site specific data on demographic parameters and the uncertainty surrounding them • Sensitivity to the form and inclusion of density dependence means that models with density dependence should only be used where there is robust evidence for it occurring within the population • Needs careful analysis to ensure enough simulations are used in the models
12	**Difference in population growth rate i.e. the reduction in growth rate between un-impacted and impacted populations**	• consistent relationship between metric and magnitude of impact: easier to make predictions about likely impacts • Insensitive to misspecification of the input parameters and relatively insensitive to uncertainty in parameter estimates • Insensitive to population trend: metric reflects impact of wind farm and not population status	• Metric varies over a limited range, with the overlapping confidence limits this makes it hard to determine likely population level effects from different magnitudes of effect • Hard to assess effects of the wind farm in a population context due to this limited range • Provides absolute values of difference between population growth rate rather than ratios and may need to be interpreted also in the context of No. 2	• Metric can be used regardless of population status or trend • Metric should be presented as a median value with 95% confidence limits • Thresholds for determining a wind farm impact are subjective but could be set in reference to the status or trend of the population
13	**Difference in population size i.e. the reduction in size between un-impacted and**	• consistent relationship between metric and magnitude of impact: easier to make predictions about likely impacts • Insensitive to misspecification of the input parameters and relatively insensitive to	• Provides absolute values of difference between populations rather than ratios and may need to be interpreted also in the context of No. 3	• Metric can be used regardless of population status or trend • Metric should be presented as a median value with 95% confidence limits • Thresholds for determining a wind farm

No.	Metric	Strengths	Weaknesses	How to use and interpret the metrics
	impacted populations	uncertainty in parameter estimates • Insensitive to population trend: metric reflects impact of wind farm and not population status		impact are subjective but could be set in reference to the status or trend of the population

2.5 Criticisms of PVA Metrics in Assessing Wind Farm Impacts

A number of criticisms have been levied against using the metrics derived from PVAs to assess the impact of wind farms (Cook & Robinson 2016a; Green *et al.* 2016). The main criticisms (some of which e.g. No. 1 are equally applicable to broader modelling contexts) were as follows:

1. Lack of empirical data to provide robust estimates and associated confidence limits of collision, barrier and displacement effects on seabirds.
2. Due to this lack of robust estimates of impact levels, probabilistic methods for assessing the risk of population impacts from wind farms are not scientifically robust or defensible - this includes metrics from PVAs that estimate e.g. the difference in probability of a decline between impacted and un-impacted populations.
3. Thresholds are subjective and it should not be claimed that these have been set based on scientific evidence.

Green *et al.* 2016 makes a number of recommendations for providing a scientifically robust and defensible method of assessing population-level impacts of wind farms on seabirds. In the context of PVA modelling the ratio of the expected population size with the wind farm to that without it (No. 3 in Table 2; also termed the so-called Counterfactual of Population size (CPS)) is recommended as a robust metric because this metric is relatively insensitive to uncertainties about demographic rates because they apply to both impacted and un-impacted scenarios. Cook & Robinson (2016b) also advocate the use of this metric, which in conjunction with the ratio of population growth rate (No. 2 in Table 2), is considered to score well in the assessments of sensitivity in Table 3. However, it should be noted that the ratio of impacted to un-impacted population size was sensitive to incorporation and the form of density dependence (see Table 3). Uncertainty can be incorporated, as in Cook & Robinson 2016b, if metrics are derived from a stochastic model or across a range of impact levels. Bayesian approaches, such as those utilised by Freeman *et al.* (2014) and a potential method for conducting Global Sensitivity Analysis developed by Aiello-Lammens & Akçakaya (2016) show promise in being able to separate out the uncertainty associated with input parameter values used in the modelling with that from scenarios of impact on a population (for example different levels of collision mortality or displacement risk), and thus have potential to help address the criticisms levied by Green *et al.* (2016). It has been highlighted that the strength of PVAs lies not in predicting absolute values of viability or costs of management but rather in evaluating the relative effects of different management scenarios (Perkins, Vickery & Shriver 2008). Green *et al.* (2016) is highly critical of interpreting effects based on arbitrary boundaries, which includes probabilistic approaches including probabilities and changes in probabilities of population declines below quasi-extinction thresholds (No. 7 and No. 8 in Table 2), and interpretation of such boundaries advocated for species conservation using IPCC based approaches detailed in Mastrandrea *et al.* (2011) where, for example, an effect is considered to be 'moderate-high' if there is a > 5 % increase in the likelihood of a 20 % population reduction.

2.5.1 Density Dependence

Green *et al.* (2016) also recommends that PVAs should be constructed using density-independent matrix models because such models would be more precautionary in their assessments of population impacts than models including density dependence (as compensatory density dependence, widely assumed to be the most common form, would tend to reduce the impact on population size). However, density-dependent processes may be depensatory, thus slowing the rate of population growth at lower population densities rather than at high densities. Establishing whether compensatory or depensatory density-dependent processes are occurring for species that are the focus of PVAs for wind farms is important: if depensatory processes are operating and are ignored in PVAs then a population decline arising from a wind farm could have larger consequences on the population than are predicted by the models, accelerating population decline and delaying population recovery. Recent work has identified depensation occurring due to increased anti-predator vigilance or colonial defence decreasing rates of productivity in smaller populations in eight species of seabird and seaduck, including species that have been the focus of PVAs for wind farms (Arctic skua, kittiwake, black-headed gull, sandwich tern, common tern, guillemot, puffin and herring gull; Horswill & Robinson 2015; Horswill *et al.* 2016). Indeed, depensation was reported almost twice as often as compensation as a mechanism regulating productivity rates and the authors highlight that this positive feedback mechanism on population size has the potential to be highly destabilising. However, density-dependent effects can vary significantly between colonies in relation to local conditions. Cook & Robinson (2016b) concluded from their sensitivity analyses that density dependent processes operating on the population would mitigate any impacts arising from the wind farm and hence that assuming no density dependence is present is likely to be the most precautionary approach unless depensatory density dependence is known to be operating. Furthermore, Cook & Robinson (2016a) recommend that density-dependence could be incorporated within models where careful consideration has deemed this appropriate, but that density independent models are likely to represent a more precautionary approach in many cases.

2.5.2 Consideration of the Time-Span used to Assess Impacts

Consideration needs to be given to the time-span over which metrics are used to determine whether the wind farm is likely to have an impact on seabird populations, for example whether the assessment is made at time increments from the construction period of the wind farm or at the end of the wind farm operating period e.g. 25 years. The time period selected needs to consider that there will be increasing uncertainty for both impacted and un-impacted scenarios with extrapolation in to the future and hence increased risk of false conclusions on the predicted magnitude of population level effects, but conversely short time windows do not reflect the duration of the lifespan of the wind farm licence (typically 25 years).

2.6 Knowledge Gaps

Cook & Robinson (2016b) adopted a conventional PVA approach whereby they assumed values for demographic parameters (specifically survival, varying between ages, and productivity) and projected simulated population predictions forward in time from a specified starting point (typically at an 'equilibrium' age-structure). No data were directly used, so no models were fitted and the results could be assumed valid for any species with demography approximately similar to that adopted in the simulations. With such an approach, since values appropriate for a given species will often be unknown with accuracy, a range of values tend to be considered, and this is the approach the BTO adopted. The advantage of this approach is that since no data fitting is required, there is a considerable reduction in computational demands. The second advantage is that it is possible to model a range of seabird life history strategies. As such, one can construct an analysis that is potentially relevant to all species and regions. However, this approach is less desirable where one wishes to understand a specific region where real data are available, or where one wishes to address generic questions with real data. One example of the latter is the need for a generic solution to the common situation where there are non-local empirical data that are relevant to the focal colony which itself lacks data. Another feature of these models is that the confidence intervals can be unrealistically narrow. A further consideration is that although the Cook & Robinson (2016b) sensitivity analysis undertook a comprehensive assessment of metric sensitivity using simulation approaches, a key knowledge gap is that metric sensitivity has not been comprehensively examined using real data. A project that focussed on this would be complementary to the work undertaken by the BTO. If the same metrics show low sensitivity in models of real world data as in simulation models, then this would provide re-assurance that these metrics are the most promising. Furthermore, such an approach would enable generic questions to be addressed with real data. One example which is very common with UK seabird populations is where data are absent from the focal colony but available from an adjacent colony, thereby offering a natural, informative prior. We would recommend that such approaches are undertaken so that sensitivity of metrics can be tested using real-world data.

2.7 Recommendations from Literature Review

- The two metrics that have been recommended for use in establishing the impact of a wind farm on seabird populations are the **Ratio of median impacted to un-impacted growth rate** and the **ratio of impacted to un-impacted population size (also known as counterfactual of population size).**
- The two metrics of the **difference in population growth rate between impacted and un-impacted populations** and the **difference in population size** should also be considered as these may be more useful if the growth rates or population size estimates being compared are small (ratios may be misleading in this context).
- Metrics should be obtained from stochastic models using a matched run approach because this is likely to reflect the most precautionary approach.
- Should probabilistic metrics be used, based on the rationale that they have been widely used in the past within published conservation science literature, and may still be used extensively in the future, it should be acknowledged that these have received criticism in Green *et al.* (2016) and Cook & Robinson (2016b).
- Density dependence should only be included where there is evidence that this may be occurring in the population of interest, otherwise use of density-independent models, or a range of density dependent structures, is advised.
- Global Sensitivity Analysis approaches detailed in Aiello-Lammens & Akçakaya (2016) and Bayesian approaches utilised by Freeman *et al.* (2014) to separate model outcome uncertainty that arises due to uncertainty in the parameter estimates used to build the models from the uncertainty in the effects of the management action (in this case wind farms) should be considered.

3. Population Modelling: Methods

3.1 Modelling Approach

A key early decision by the Steering Group was to agree which population modelling approach to use. Conventionally, PVA have been applied by assuming values for demographic parameters (specifically survival, varying between ages, and productivity) and projecting simulated population predictions forward in time from a specified starting point (typically at an 'equilibrium' age-structure). No data are directly used, so no models are fitted and the results can be assumed valid for any species with demography approximately similar to that adopted in the simulations. In practice, since values appropriate for a given species will rarely be known with much accuracy, a range of values tend to be considered. The advantage of this approach is that since no data fitting is required, there is a considerable reduction in computational demands. The second advantage is that it is possible to model a range of seabird life history strategies. As such, one can construct an analysis that is potentially relevant to all species and regions. This approach is less desirable where one wishes to understand a specific region where real data are available, or where one wishes to address generic questions with real data. One example of the latter is the need for a generic solution to the common situation where there are non-local empirical data that are relevant to the focal colony which itself lacks data (see next section). Another feature of these models is that the confidence intervals can be unrealistically narrow.

In the previous population modelling contract CEH undertook for Marine Scotland Science, we fitted state-space models using Bayesian techniques via WinBUGS to data from four SPAs for five species in the Forth/Tay region (Freeman *et al.* 2014). Here, no parameter values were specified beforehand; all were estimated from the data prior to projecting the population predictions forwards to beyond the period of the data. In these models, the population is assumed to change stochastically (the 'state process') and the counts to be equal in expectation to the population level (or part of it), subject also to sampling variability (the 'observation process'). Using this method, sampling co-variances of parameter estimates are naturally accommodated. In Freeman *et al.* (2014), demographic parameters were assumed to vary about a mean value, with a specified variance, where these were estimated from models applied at sites with more substantial data (generally the Isle of May). While the need for defining parameter configurations *a priori* are reduced in such models, the results are dependent upon the data used (precision, for example, will depend in part upon the likely representativeness of the data from the well-studied colony). One advantage of this approach is in the case where there is interest in specific colonies/study areas, thereby providing a rationale for fitting the model to real data. Of the various methods that can be used to fit models to data, we consider this approach to be the most robust because of greater realism in the estimating of credibility intervals, in particular due to the partitioning of observation and process error, in cases where there are empirical data (counts and/or demography data) or informative priors (see Freeman *et al.* 2014 for a discussion of this). A second advantage of this approach is in addressing generic questions with real-world data. One example has been addressed above that we think is particularly relevant in this context, where data are absent from the focal colony but available from an adjacent colony, thereby offering a natural, informative prior. However, considerable thought is required before adopting this approach since information from another colony cannot automatically be assumed to apply elsewhere, to other species and/or regions, and any assumptions should be clearly specified. Two more advantages arise from this approach within the specific context of this project: a) Cook & Robinson (2016) have undertaken a comprehensive sensitivity analysis of PVA metrics using simulations in a traditional framework, so there would be a benefit in testing the performance of the same suite of metrics in an empirical analysis, with confidence gained if the same metrics perform well using both approaches; b) there is continuity with the previous report (Freeman *et al.* 2014). The main disadvantage of this approach is the analytical and computational demands. Furthermore, if there is no interest in specific colonies/regions, or if the generic questions that can be addressed using real-world data, then a simulation approach is the logical way forward.

The Steering Group decided that there was such interest, and that it would be complementary to the recent work by Cook & Robinson (2016), so this was the method that was undertaken. Further, the decision was to focus on the three main issues emerging from past work and stakeholder interest: sensitivity in a range of PVA metrics including a comparison of ratio and probabilistic types, effect of population status on sensitivity, and effect of renewables effect size on sensitivity. Finally, it was agreed following consideration of the literature that density dependence would not be included in the models (see literature review).

3.2 Modelling Methods

3.2.1 Input Data

Five study species were selected: black-legged kittiwake, common guillemot, razorbill, herring gull and European shag. Of these kittiwake, guillemot, razorbill and herring gull were considered in Freeman *et al.* (2014). As similar models have, in the interim, also been fitted for shags we also consider this extra species. We accumulated data sets on abundance, survival and productivity from four SPAs (Buchan Ness to Collieston Coast SPA; Fowsheugh SPA; Forth Islands SPA; St Abb's Head to Fastcastle SPA).

New data were added up to 2016 where available (Freeman *et al.* 2014 modelled data up to 2012). Data include colony counts, in full if possible but often such data are available only in a limited number of years, or else have been made only in smaller parts of the main colony (i.e. plots). Demography is estimated from ringing data (survival) or nest record data (productivity per nest/pair). Such data have long been gathered by CEH at the Isle of May in the Forth Islands SPA, but are often missing elsewhere in the region. Data availability and sources for the species considered are given in Tables 5 and 6, respectively.

Counts and demographic parameter estimates can be found in Appendix 1.

Table 5

Data availability for each species at each SPA. Regular census means annual or near-annual. Sporadic census is less regular – typically every four to seven years. Sources: [a]Seabirds Monitoring Programme online database; [b]Vicky Anderson/Edward Grace, RSPB, pers comm; [c]Roddy Mavor, JNCC pers comm.; [d]Harris *et al.* 2009, 2013; [e]Frederiksen *et al.* 2004 updated; [f]Lahoz-Monfort *et al.* 2011, 2014; [g]Newell *et al.* 2012; [h]Lahoz-Monfort *et al.* 2013; [i]BTO ringing and recovery data, purchased for Freeman *et al.* 2014

Species	SPA	Counts	Survival (Adult birds)	Productivity
Kittiwake	Forth Islands	Regular census[a]	Regular survey[e]	Regular census[a,g]
	St Abb's Head	Regular census[a]	No	Regular census[a]
	Fowlsheugh	Sporadic census[a]	No	Regular census[a]
	Buchan Ness	Sporadic census[a]	No	Regular census[a]
Guillemot	Forth Islands	Regular census[a]	Regular survey[f]	Regular census[a,g,h]
	St. Abb's Head	Sporadic census [a] Regular sub-plot survey[a]	No	No
	Fowlsheugh	Sporadic census[a] Regular sub-plot survey[b]	No	No
	Buchan Ness	Sporadic census[a] Sporadic sub-plot survey[c]	No	No
Razorbill	Forth Islands	Regular census[a]	Regular survey[f]	Regular census[a,g,h]
	St Abb's Head	Sporadic census[a] Regular sub-plot survey[a]	No	No
	Fowlsheugh	Sporadic census[a] Regular sub-plot survey[b]	No	No
Herring gull	Forth Islands	Regular census[a]	Historical survey[i]	Regular census[a]
	St Abb's Head	Regular census[a]	No	No
Shag	Forth Islands	Regular census[a]	Regular survey[a]	Regular census[a]
	St Abb's Head	Regular census[a]	No	Regular census[a]
	Buchan Ness	Sporadic census[a]	No	No

Table 6

Data source for each species at each SPA.

Species	SPA	Counts	Adult survival	Productivity
Kittiwake	Forth Islands	Forth Islands	Isle of May	Isle of May
	St Abb's Head	St Abb's Head	Isle of May	St Abb's Head
	Fowlsheugh	Fowlsheugh	Isle of May	Fowlsheugh
	Buchan Ness	Buchan Ness	Isle of May	Buchan Ness
Guillemot	Forth Islands	Forth Islands	Isle of May	Isle of May
	St. Abb's Head	St. Abb's Head	Isle of May	Isle of May
	Fowlsheugh	Fowlsheugh	Isle of May	Isle of May
	Buchan Ness	Buchan Ness	Isle of May	Isle of May
Razorbill	Forth Islands	Forth Islands	Isle of May	Isle of May
	St Abb's Head	St Abb's Head	Isle of May	Isle of May
	Fowlsheugh	Fowlsheugh	Isle of May	Isle of May
Herring gull	Forth Islands	Forth Islands	Isle of May	Isle of May
	St Abb's Head	St Abb's Head	Isle of May	Isle of May
Shag	Forth Islands	Forth Islands	Isle of May	Isle of May
	St Abb's Head	St Abb's Head	Isle of May	St Abb's Head
	Buchan Ness	Buchan Ness	Isle of May	Isle of May

3.2.2 Population Models

The models adopted for these data are as described in Freeman *et al.* (2014) and we provide only a brief overview here. A state-space model for the annual counts was adopted, with the expected number of breeding pairs of a population in year t given by N_t, where, for a species such as shag that begins breeding at age three is:

$$N_t = Nr_t + Na_t$$

$$Nr_t \sim Poisson\left(N_{t-3}\left(\frac{f_{t-3}}{2} \times \varphi_{j,t-3}\varphi_{j,t-2}\varphi_{j,t-1}\right)\right)$$

$$Na_t \sim Binomial\left(N_{t-1}, \varphi_{a,t-1}\right)$$

Where Nr_t and Na_t are respectively the numbers of new recruits, and survivors of the previous breeding population, in year t. The model is straightforwardly amended to accommodate those species that do not begin breeding until aged five or six. Juvenile survival probabilities $\varphi_{j,t}$ are assumed to take a constant value φ_j , unknown but estimable from the data; those for adults $\varphi_{a,t}$ are assumed normally distributed mean values and variance estimated from a set of ringing data at the Isle of May. Completing the model the annual numbers of chicks per pair f_t are estimated with means and variance from nest record data gathered at the site in question, where available, or using the data from the Isle of May where site-specific productivity data are unavailable. Due to the paucity of kittiwake counts at Fowlsheugh and Buchan Ness to Collieston Coast, these were modelled simultaneously in a single (multivariate) state-space model, with a common juvenile survival rate. As in Freeman *et al.* (2014) there were problems modelling the Kittiwakes at the Forth Islands SPA; this was due to low counts in 1994, which subsequently recovered for a few years, and so the 1994 counts were omitted from the data that informed the state space model.

Models were fitted using Bayesian techniques using the software JAGS (Plummer 2013). As in Freeman *et al.* (2014), multiple projections for 25 future years (2016 to 2041) of wind farm impact under various scenarios (given below) are made by repeatedly sampling from the distributions above, effectively generating posterior distributions for the abundance in future years. Using the model above, we thus produce 'baseline' predictions, under the assumption that prevailing conditions apply in future years. We then produced a series of alternative 'impacted' population trajectories assuming that adult survival, productivity or both were negatively affected by some 'perturbation', equating to the effect of an offshore wind farm. This enables a comparison of future predictions following perturbation with those under the 'status quo' assumptions, known as the baseline. In consultation with the Steering Group, adult survival was set to decline by one of a range of specified rates, namely 0% (i.e. no change), 0.5%, 1%, 2% and 3%. Declines in annual productivity were set to 0%, 1%, 2%, 3% and 5%. Finally, combined effects of survival and productivity were set to, respectively, 0%/0%, 1%/1%, 2%/2%, 3%/3% and 0.5%/5%. Note that these are percentage point changes, as requested by the Steering Group, which differs from the approach taken in Cook & Robinson (2016b) where percentage changes were investigated. In all models, an additional five years were projected with no change in survival or productivity, representing a post-wind farm decommissioning period.

3.3 PVA Metric Sensitivity

The above modelling framework allowed us to examine the population changes under various levels of impact upon the demographic parameters, given that these take the values of the model. It is, of course, plausible that the average values of adult survival and productivity experienced by the populations may differ from those implied by the demographic data used, especially where these are 'borrowed' from adjacent sites for those without such data of their own (for survival, this is all sites apart from the Forth Islands; even there, all ringing data are from a single study at the Isle of May). Therefore, we also repeated the entire procedure with demographic parameters "mis-specified" to varying degrees. Specifically, we considered median adult mortality (the complement of survival, since survival is generally high in seabirds and percentage increases are greatly limited by the constraint of lying below a survival rate of one) and productivity to differ from those of the baseline by, in turn -30%, -20%, -10%, 10%, 20% and 30%. The consequences of uncertain adoption of demographic parameters could then be examined by plotting a suite of PVA metrics against this rate of mis-specification, under a range of renewables effect sizes.

The Steering Group, having considered the findings of the literature review, requested that we examine the sensitivity of five PVA metrics, and Marine Scotland Science requested that we include a sixth metric (PVA F):

1) Median of the ratio of impacted to un-impacted (=baseline) annual growth rate (PVA A; Metric No. 2 in Table 2).
2) Median of the ratio of impacted to un-impacted population size after 25 years (PVA B; Metric No. 3 in Table 2).
3) Median difference in impacted and un-impacted annual growth rates (PVA C; Metric No. 12 in Table 2.)

4) Median difference between impacted and un-impacted population size after 25 years (PVA D; Metric No. 13 in Table 2).
5) Probability of a population decline over 25 years exceeding a) 10% b) 25% and c) 50% (PVA E1, E2 and E3 respectively; Metric No. 7 in Table 2).
6) Centile for un-impacted population which matches the 50th centile for the impacted population after 25 years (PVA F; Metric No. 15 in Table 2).

PVAs A and B are ratio metrics, PVAs C and D are metrics related to ratio metrics and PVAs E and F are probabilistic metrics. All of these metrics are readily estimable from the repeated simulations above, with posterior distributions of the ratios/differences arising from a "matched runs" approach, as recommended (WWT 2012; Green *et al.* 2016; Cook & Robinson 2017) i.e. the parameters defining the expected annual counts in each replicate are identical, except insofar as the expected impacted figures are adjusted to reflect the level of the impact. Plotting these metrics against alternative levels of adult survival or productivity used gives a visual assessment of the sensitivity of these metrics to the choice of demographic parameters.

Note that for the models of razorbills at Fowlsheugh, two of the thirteen models exhibited formal warnings via the Brooks-Gelman statistic values regarding convergence for juvenile survival. However, the estimates of the PVA metrics from these models appear to be consistent with the pattern as shown by other species/SPAs and so these are retained in the plots.

However, for three species/SPA combinations there were inherent problems with the "baseline" model (with no mis-specification). This was for shags at Buchan Ness, having a baseline model which "converged", but not to anything sensible (the observation error was greater than the counts) and for herring gull at both sites, which had problems with the convergence of key parameters, adult survival and juvenile survival. Therefore, we considered these three species/SPAs to be unreliable and did not use them in the assessment of the sensitivity of the PVA metrics.

3.4 Structure of the Results

The Steering Group requested that we examine the sensitivity of these PVA metrics to mis-specification in adult mortality and productivity, and investigate to what extent this sensitivity varied with predicted population status and size of renewables effect. Accordingly, the results section is split into three parts.

First, we provide the full results of population modelling, including retrospective data fitting, population forecasts and PVA sensitivities for one species/SPA population: kittiwakes at Forth Islands. It was considered by the Steering Group necessary to show this comprehensive output for one population only, although models presented were undertaken on all populations. Combining the mis-specifications in adult mortality or productivity with the scenarios of annual decline in adult survival or productivity provides four graphical outputs:

1. Mis-specification in adult mortality with scenarios of renewables-induced change in productivity;
2. Mis-specification in adult mortality with scenarios of change in adult survival;
3. Mis-specification in productivity with scenarios of renewables-induced change in productivity;
4. Mis-specification in productivity with scenarios of change in adult survival.

Second, we present PVA sensitivities in relation to population status, combining data from all species/SPAs for which we achieved model convergence. We estimated the projected population growth rate as follows:

$$\lambda = \left(\frac{Estimated\ median\ total\ population\ in\ 2041}{Estimated\ median\ total\ population\ in\ 2016}\right)^{1/25}$$

Lambda is calculated for the baseline model and takes the values for the various species/SPA combinations shown in Table 7. Populations were classed as increasing (lambda > 1) or decreasing (lambda < 1). Of the four combinations outlined above, we only show results from the analysis of mis-specification in adult mortality with the maximum scenario of change in adult survival (3%), to maximise clarity.

Third, we present PVA sensitivities in relation to scenarios of change resulting from the renewables development (i.e. the effect size). Of the four combinations outlined above, we only show results from the analysis of mis-specification in adult mortality with scenarios of change in adult survival.

Table 7

Projected population growth rates over the period 2016-2041 for Species/SPA populations.

Species/SPA population	**Lambda**
Kittiwakes:	
Forth Islands	0.964
St Abb's Head	0.937
Fowlsheugh	0.969
Buchan Ness to Collieston Coast	0.967
Guillemots:	
Forth Islands	1.012
St Abb's Head	1.018
Fowlsheugh	0.997
Buchan Ness to Collieston Coast	1.022
Razorbills:	
Forth Islands	1.023
St Abb's Head	0.991
Fowlsheugh	1.040
Shags:	
Forth Islands	1.004
St Abb's Head	0.980

4. Population Modelling: Results

4.1 Population Modelling and PVA Sensitivity in Forth Islands Kittiwakes

The data available for Forth Island kittiwakes, the population for which we present the full set of outputs, ranges from 1984 to 2016. The annual variation in the median adult survival and productivity as well as the posterior distribution of juvenile survival and the observation error are given in Figure 1. The latter two parameters approximate a normal distribution, with a mean juvenile survival of 0.685. The model suggests that Kittiwakes at the Forth SPA have declined from an initial abundance of just over 10,000 to about 4,000 in 2016. Future projections indicate further declines (Figures 2a-c), though note the wide credible intervals, broadening as time passes, as uncertainty increases in these estimates.

For the sensitivity analysis, the median population size after 25 projected years (2041) was estimated under a range of mis-specifications in adult mortality or productivity and scenarios of annual decline in adult survival or productivity (Figure 3). The estimated population size when adult survival or productivity does not change and there is no mis-specification in the Bayesian model results in an estimate of approximately 1,300 birds. As expected, population size under all effect size scenarios declines with increasing mortality and increases with increasing productivity (Figure 3). These relationships are non-linear, and different scenarios of annual decline diverge as the overall effect of mis-specification strengthens, because percentage point changes in mis-specification have a relative, not absolute effect on population size.

The outputs of PVA metric sensitivity can be found in Figures 4a-h for PVA A, B, C, D, E1, E2, E3 and F, respectively (see Section 3.3 of the methods for a definition of each metric). We estimated the PVA metrics using seven model runs for changes in adult mortality (-30% to +30% at 10% increments) and seven runs for productivity (-30% to +30% at 10% increments). The model run of no change in adult mortality or productivity is shared by both, hence a total of thirteen models were run.

The ratio of impacted to un-impacted annual growth rate (PVA A; Figure 4a) was very close to one for the full range of scenarios and, matching theory and past evidence using simulations, was insensitive to mis-specification in demographic parameters. One possibility for the low sensitivity of PVA A is the scale of values, with all values being close to one, and, therefore, sensitivity potentially appearing low in a visual assessment even in cases where it is not. However, we show that this is not the case in Appendix 2, where we consider a 25 year growth rate, whereby lines deviate markedly from one and low sensitivity is still apparent.

Estimates for the ratio of impacted to un-impacted population size after 25 years (PVA B; Figure 4b) showed a range of values with respect to scenarios of change in productivity and, in particular, mortality, but it was also insensitive to mis-specification in demographic parameters. The PVA metric representing the difference in impacted and un-impacted growth rates (PVA C; Figure 4c) was also comparatively insensitive. In contrast, the PVA metric representing the difference in impacted and un-impacted population size (PVA D; Figure 4d) was considerably more sensitive, and showed non-linear patterns of change which were dependent on the effect size scenario, associated with the relationship between absolute and relative changes in population size (as with Figure 3).

As regards the probabilistic metrics, the metric presenting the probability of a population decline over 25 years exceeding 10%, 25% and 50% (PVAs E1, E2 and E3; Figure 4e, f and g respectively) showed high sensitivity to mis-specification both in mortality and reproduction. Each shows a non-linear pattern of change in line with expectations and past use of these metrics, including the expected variation between PVAs E1, E2 and E3 in relation to the stated exceedance thresholds of 10%, 25% and 50%. In contrast, the metric representing the centile for un-impacted population which matches the 50% centile for the impacted population after 25 years (PVA F; Figure 4h) showed moderately low sensitivity to mis-specification of survival and productivity. It was less sensitive than PVA E with and more sensitive than ratio metrics PVA A and B.

Graphical presentation of sensitivity of PVA metrics for all 13 species/SPA combinations can be found in Appendix 3.

Figure 1: Diagnostics plot from the Bayesian state space model for adult survival, productivity, juvenile survival and observation error.

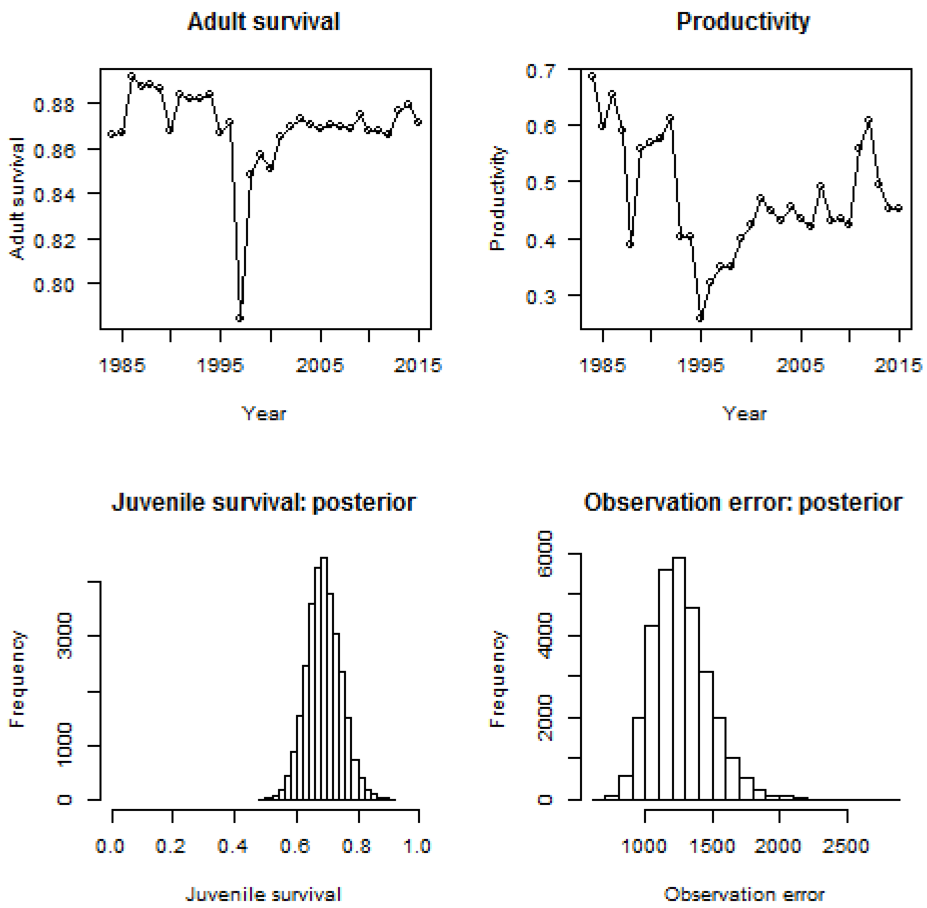


Figure 2a: Estimated total abundance from 1984 to 2016, with an additional 25 years of projections with various declines in productivity and a final five years of projections with no decline in productivity.

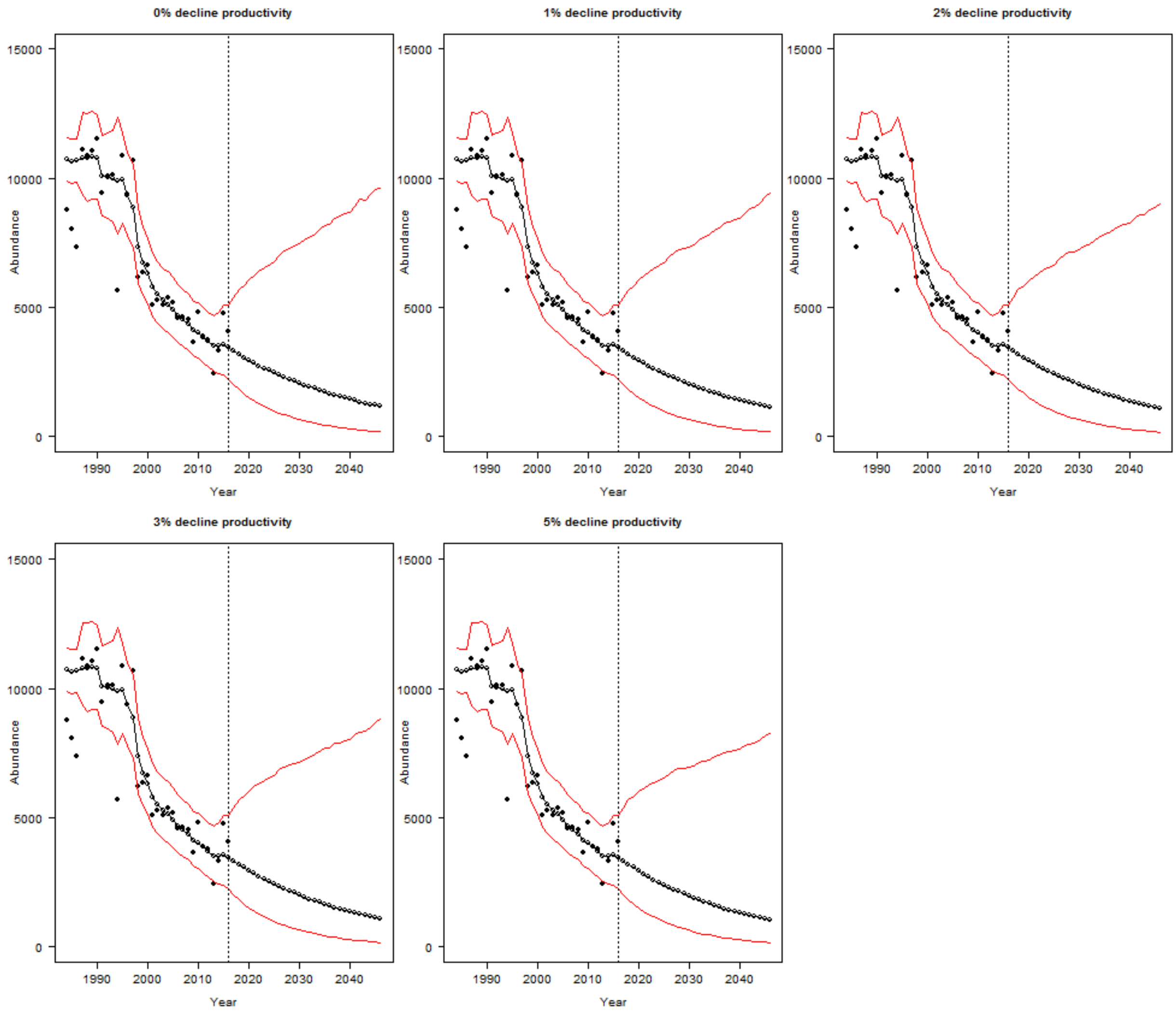


Figure 2b: Estimated total abundance from 1984 to 2016, with an additional 25 years of projections with various declines in adult survival and a final five years of projections with no decline in adult survival.

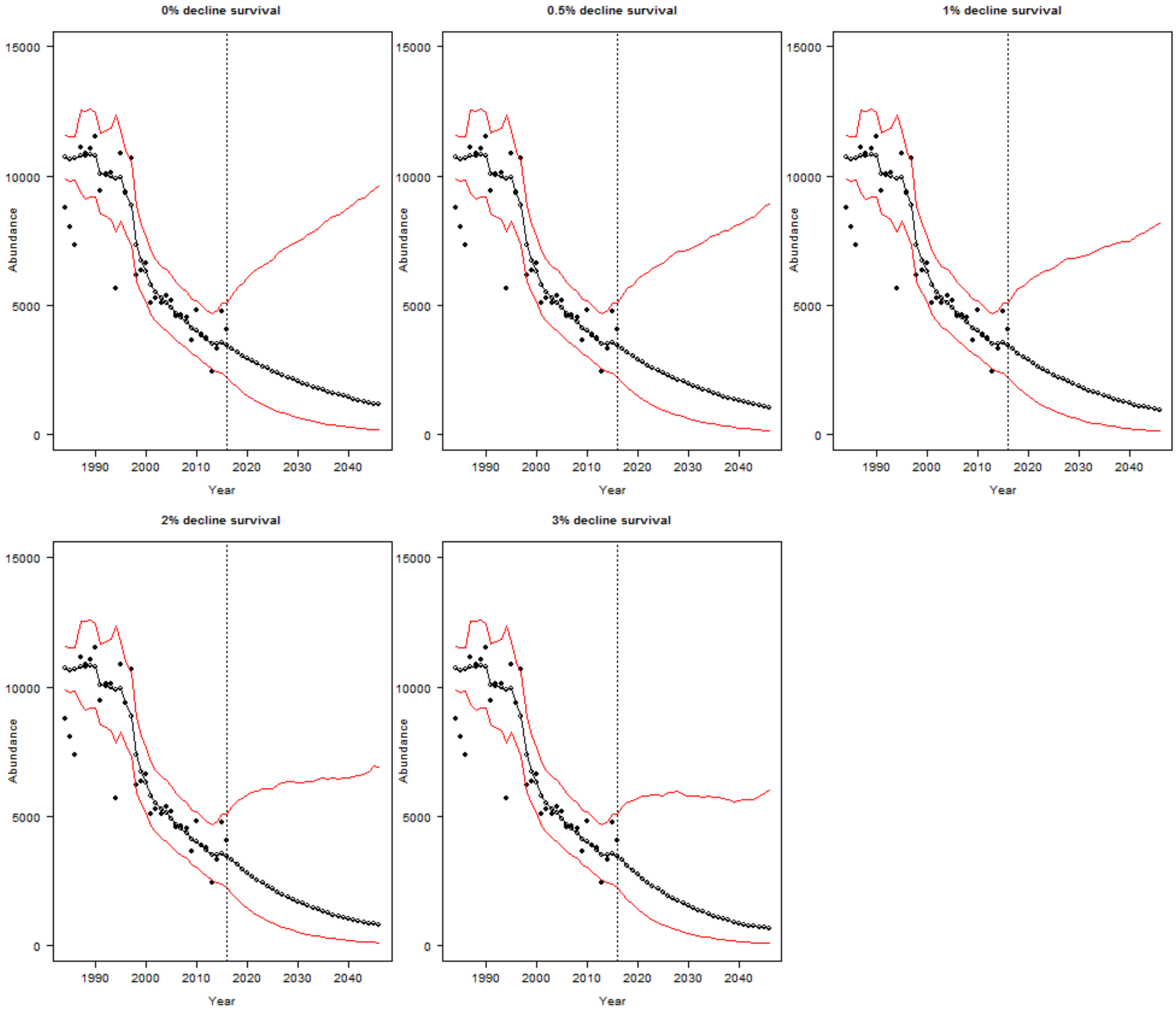


Figure 2c: Estimated total abundance from 1984 to 2016, with an additional 25 years of projections with various declines in both productivity and adult survival and a final five years of projections with no decline in either productivity or adult survival.

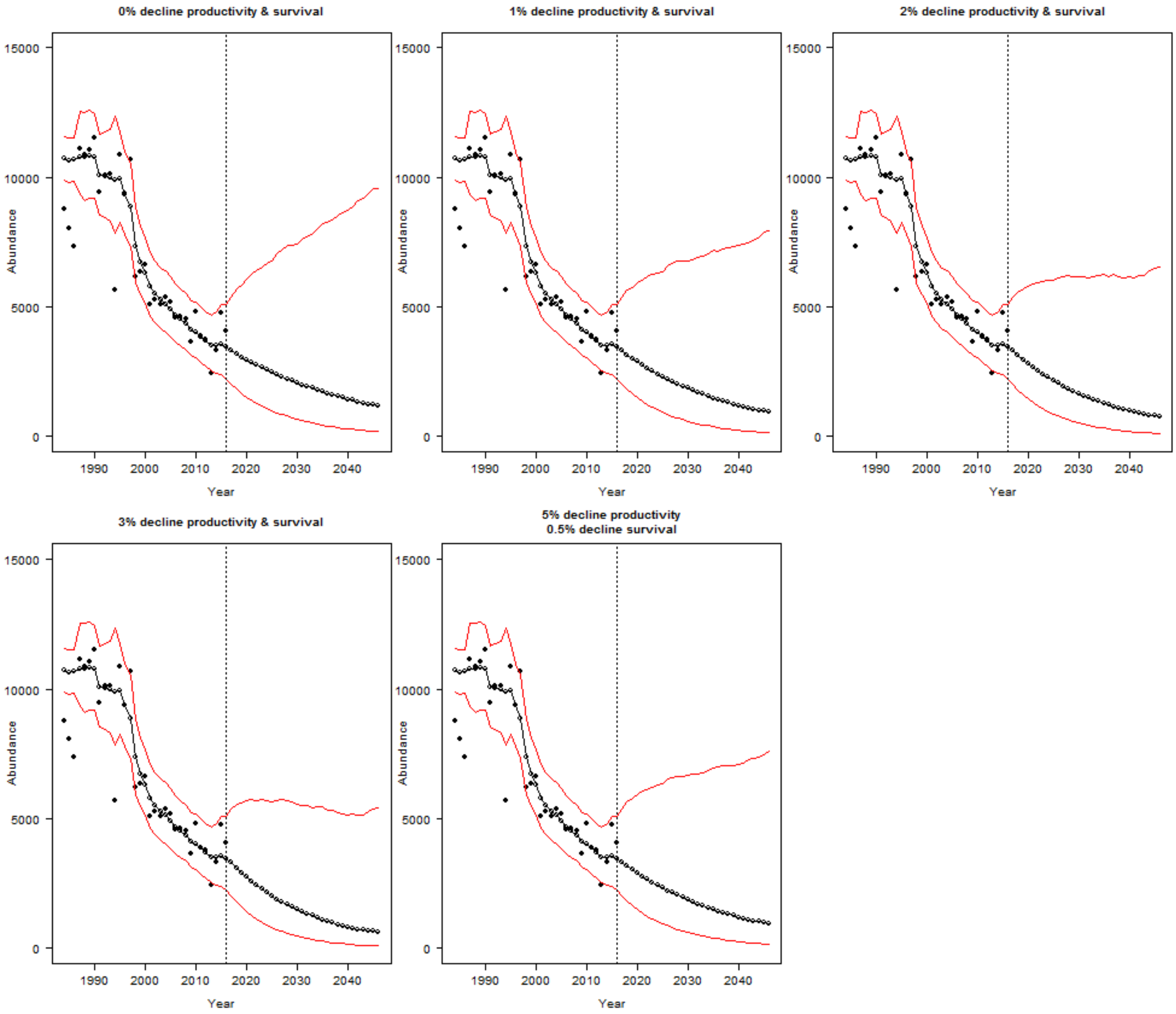


Figure 3: Median impacted population size after 25 years of projections under various scenarios of mis-specification in productivity and adult mortality. Adult mortality mis-specification is illustrated in the upper panels and productivity mis-specification in the lower panels. Mis-specification was varied from -30% to +30% (with 0% representing no mis-specification). The five coloured lines represent the different levels of potential impact on annual productivity (left panels) or annual adult survival (right panels) over the hypothetical 25 year lifetime of the wind farm (2016-2041).

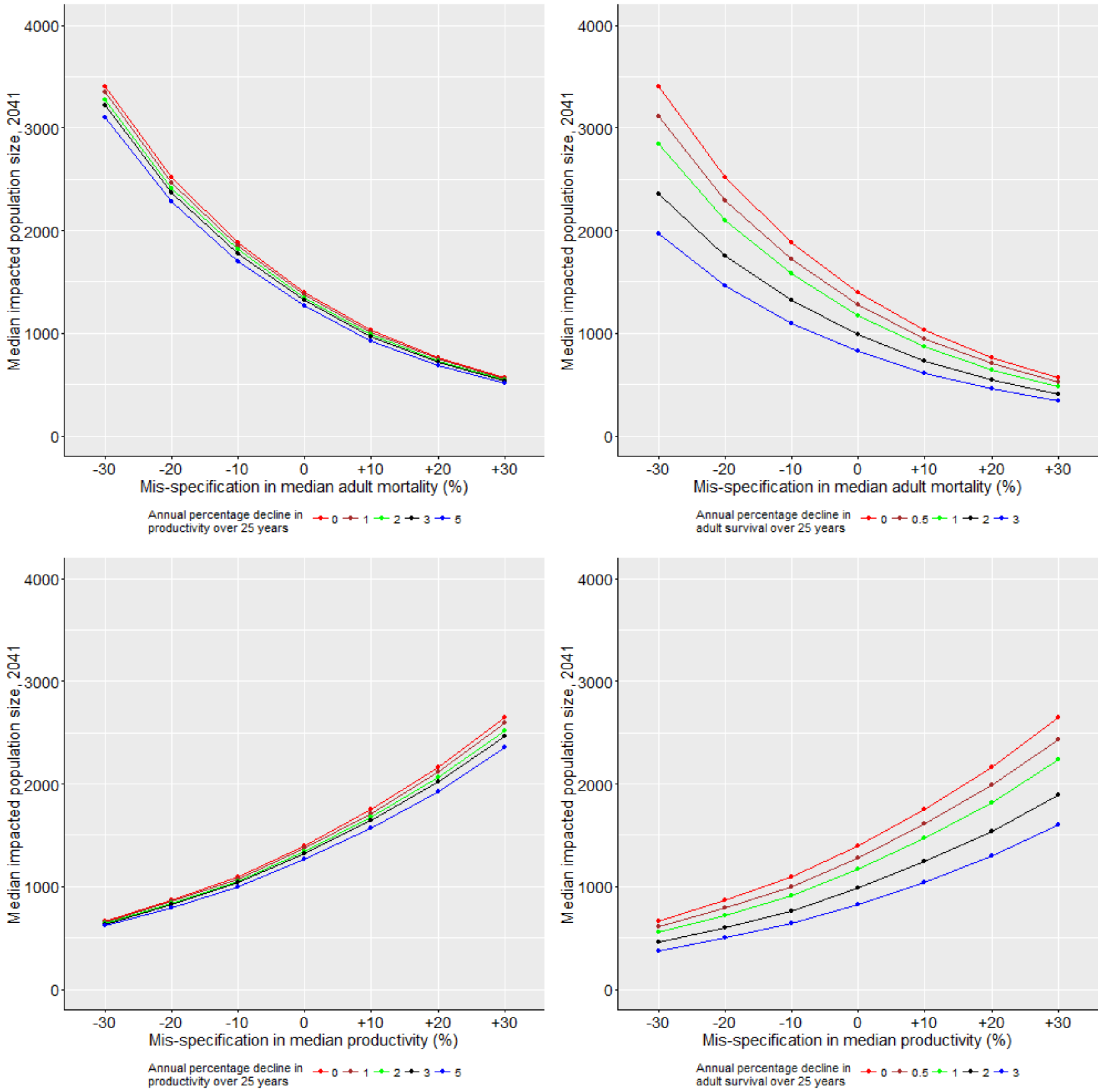


Figure 4a: PVA Metric A – ratio of population growth rate from 2016-2041, comparing impacted population vs. un-impacted population. Adult mortality mis-specification is illustrated in the upper panels and productivity mis-specification in the lower panels. Mis-specification was varied from -30% to +30% (with 0% representing no mis-specification). The five coloured lines represent the different levels of potential impact on annual productivity (left panels) or annual adult survival (right panels) over the hypothetical 25 year lifetime of the wind farm (2016-2041).

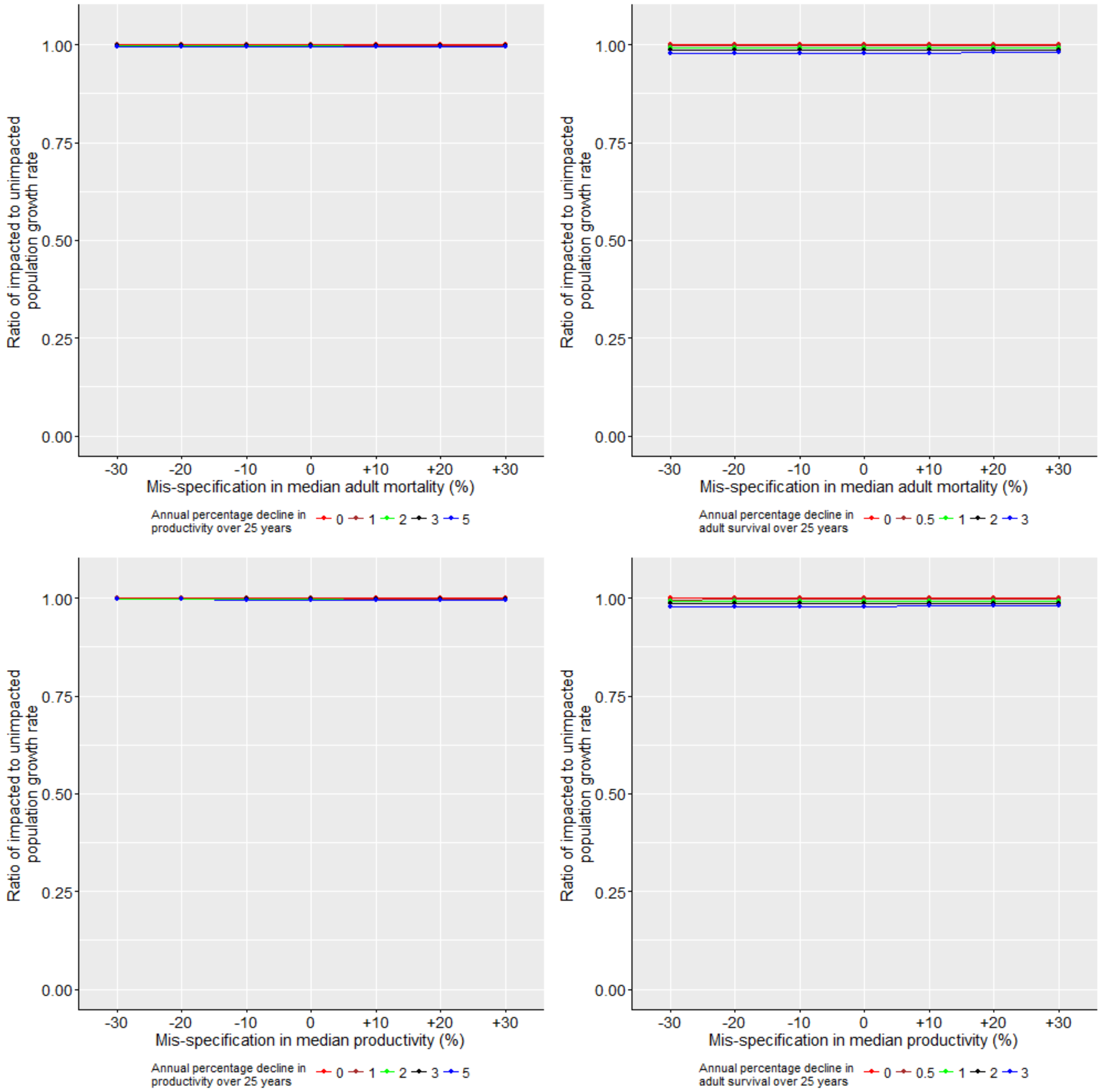


Figure 4b: PVA Metric B – ratio of population size at 2041, comparing impacted population vs. un-impacted population. Adult mortality mis-specification is illustrated in the upper panels and productivity mis-specification in the lower panels. Mis-specification was varied from -30% to +30% (with 0% representing no mis-specification). The five coloured lines represent the different levels of potential impact on annual productivity (left panels) or annual adult survival (right panels) over the hypothetical 25 year lifetime of the wind farm (2016-2041).

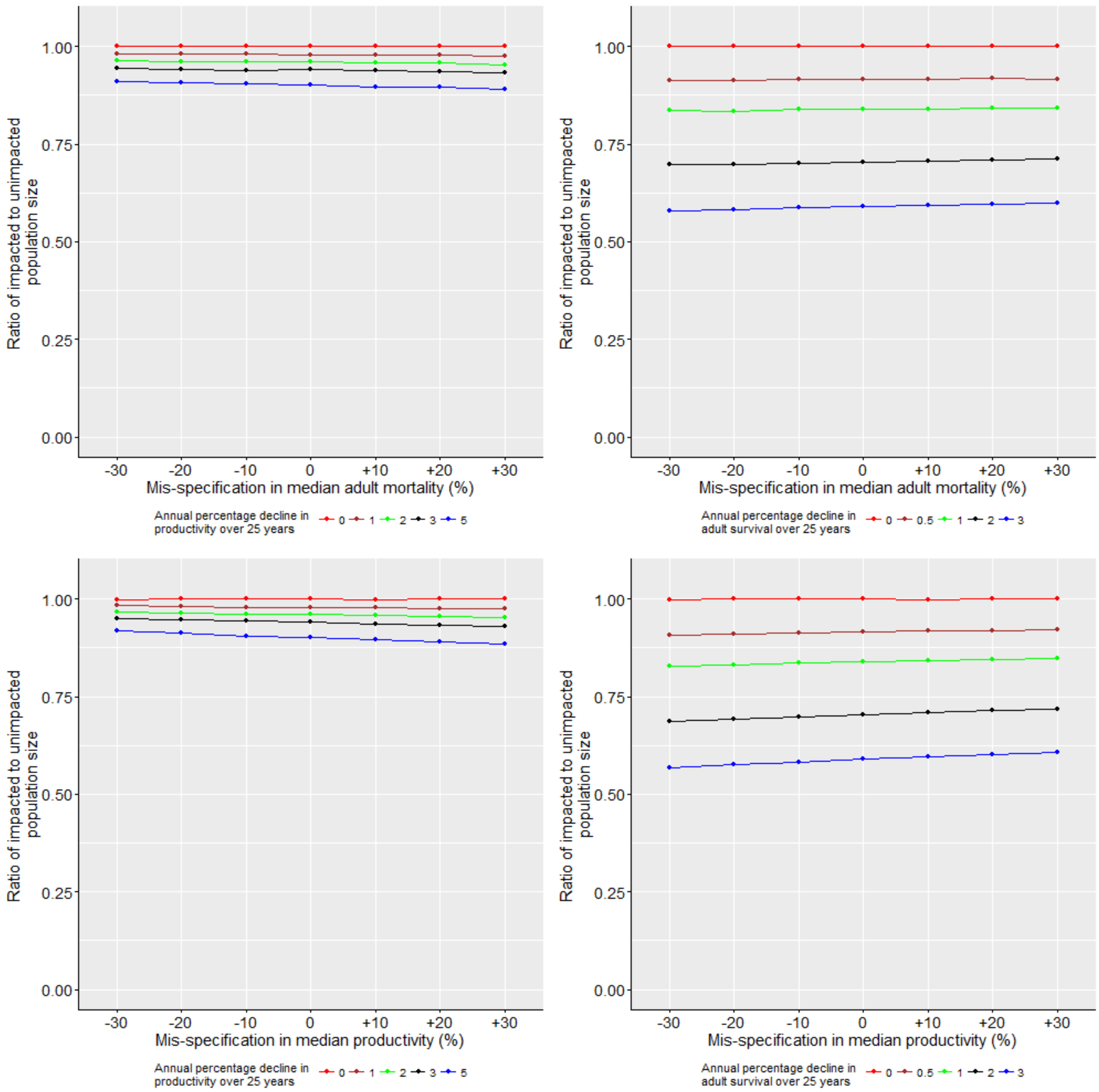


Figure 4c: PVA Metric C – difference in population growth rate from 2016-2041, comparing impacted population vs. un-impacted population. Adult mortality mis-specification is illustrated in the upper panels and productivity mis-specification in the lower panels. Mis-specification was varied from -30% to +30% (with 0% representing no mis-specification). The five coloured lines represent the different levels of potential impact on annual productivity (left panels) or annual adult survival (right panels) over the hypothetical 25 year lifetime of the wind farm (2016-2041).

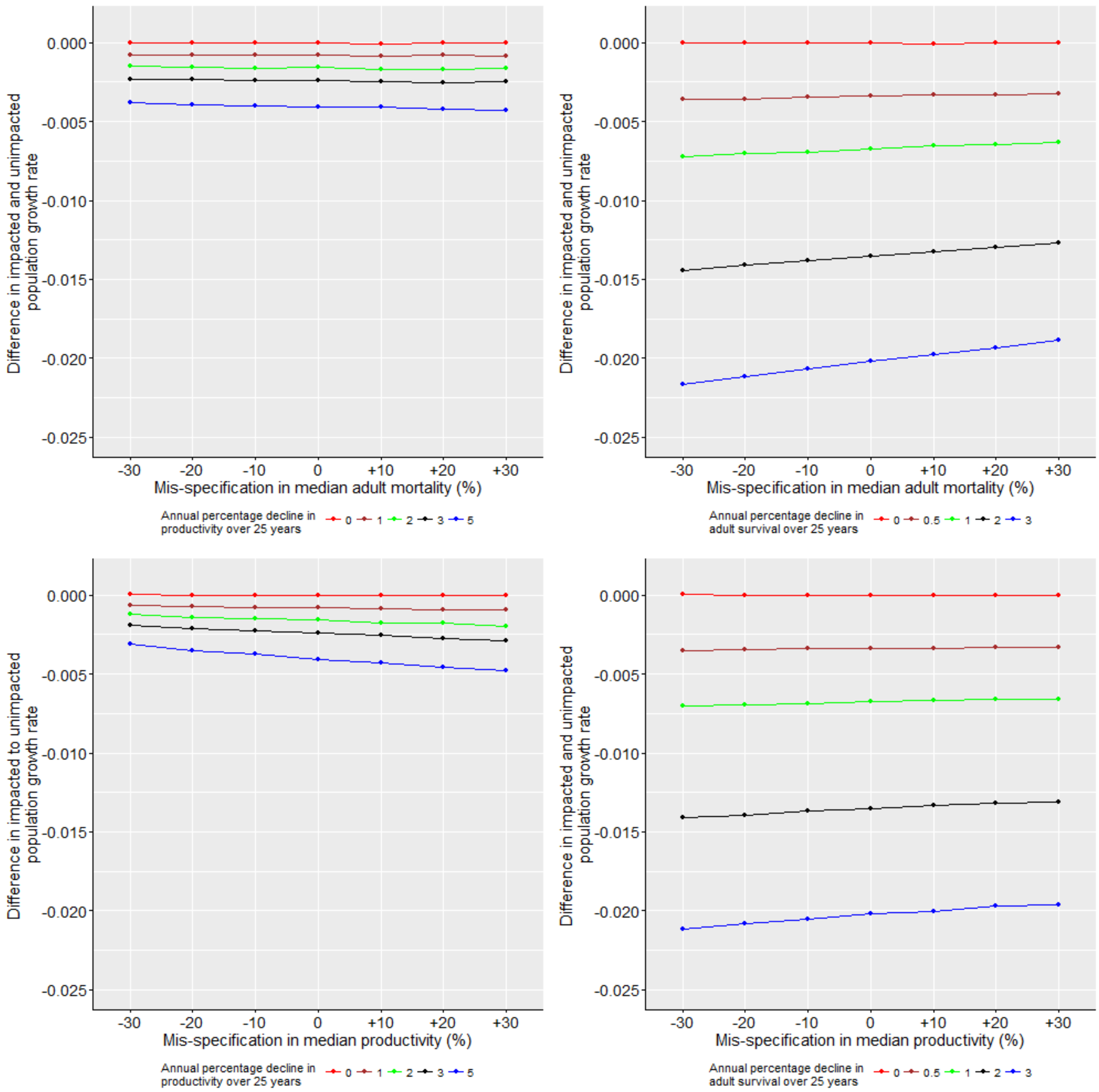


Figure 4d: PVA Metric D – difference in population size at 2041, comparing impacted population vs. un-impacted population. Adult mortality mis-specification is illustrated in the upper panels and productivity mis-specification in the lower panels. Mis-specification was varied from -30% to +30% (with 0% representing no mis-specification). The five coloured lines represent the different levels of potential impact on annual productivity (left panels) or annual adult survival (right panels) over the hypothetical 25 year lifetime of the wind farm (2016-2041).

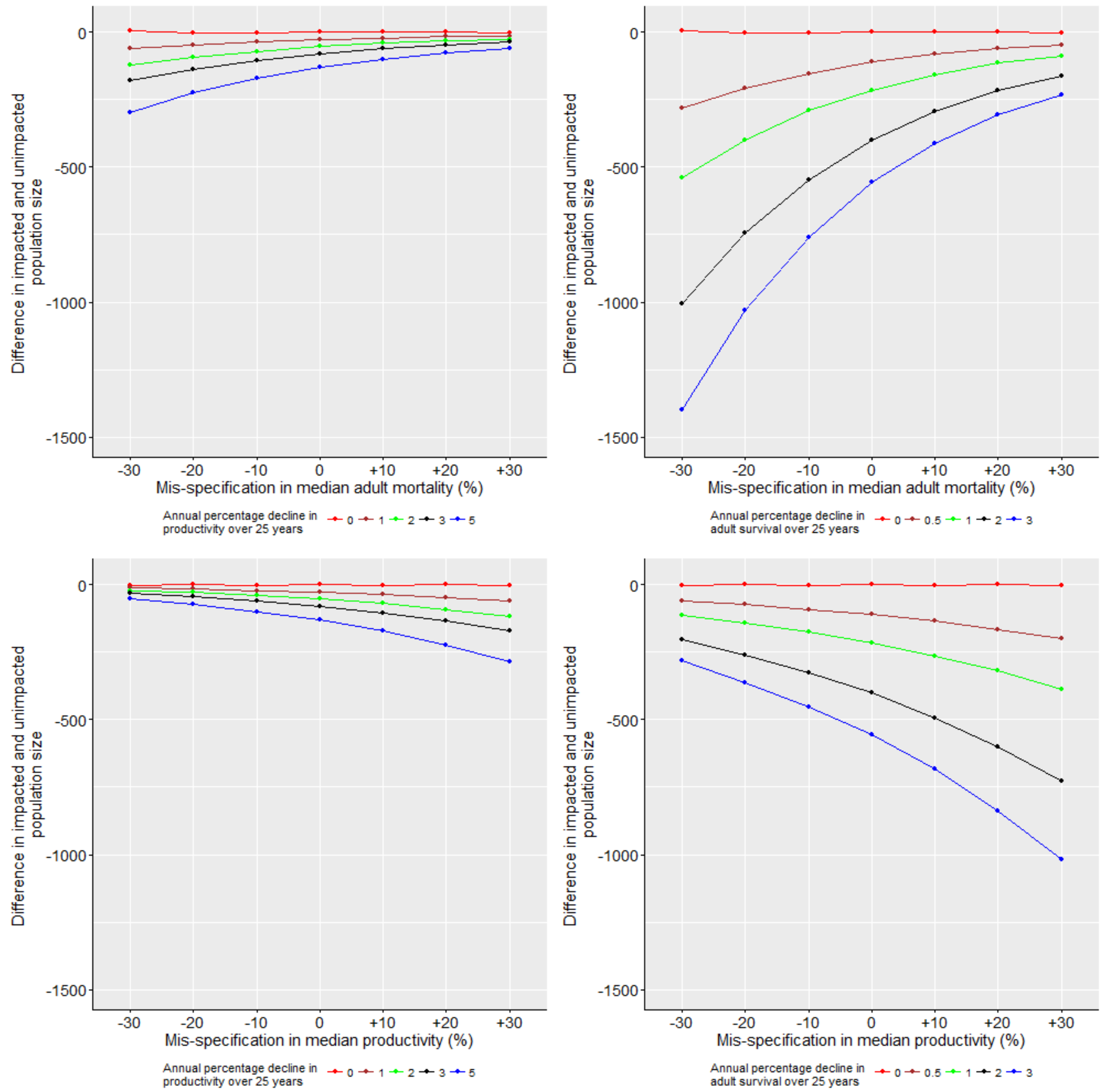


Figure 4e: PVA Metric E1 – probability of population decline greater than 10% from 2016-2041. Adult mortality mis-specification is illustrated in the upper panels and productivity mis-specification in the lower panels. Mis-specification was varied from -30% to +30% (with 0% representing no mis-specification). The five coloured lines represent the different levels of potential impact on annual productivity (left panels) or annual adult survival (right panels) over the hypothetical 25 year lifetime of the wind farm (2016-2041).

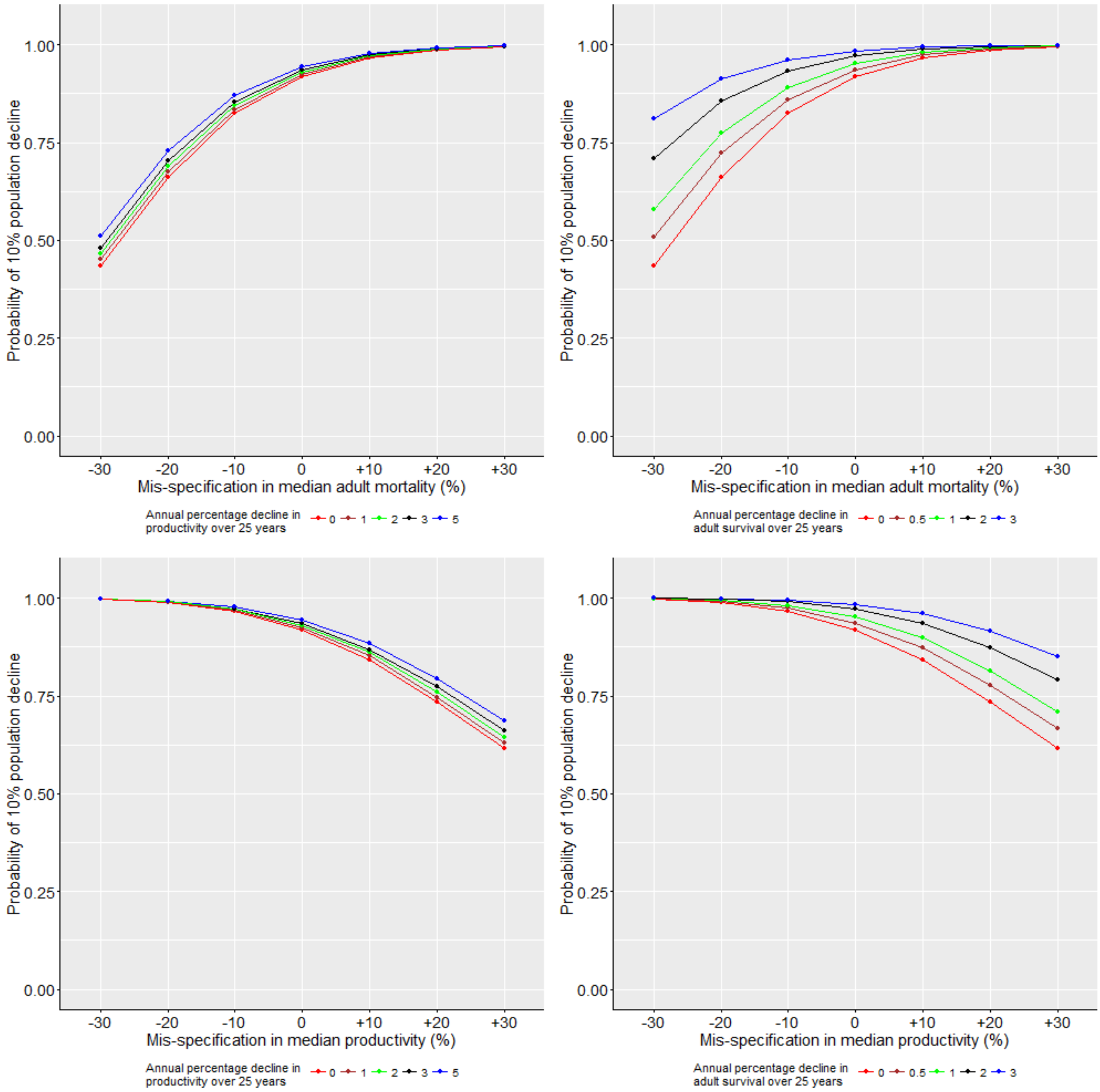


Figure 4f: PVA Metric E2 – probability of population decline greater than 25% from 2016-2041. Adult mortality mis-specification is illustrated in the upper panels and productivity mis-specification in the lower panels. Mis-specification was varied from -30% to +30% (with 0% representing no mis-specification). The five coloured lines represent the different levels of potential impact on annual productivity (left panels) or annual adult survival (right panels) over the hypothetical 25 year lifetime of the wind farm (2016-2041).

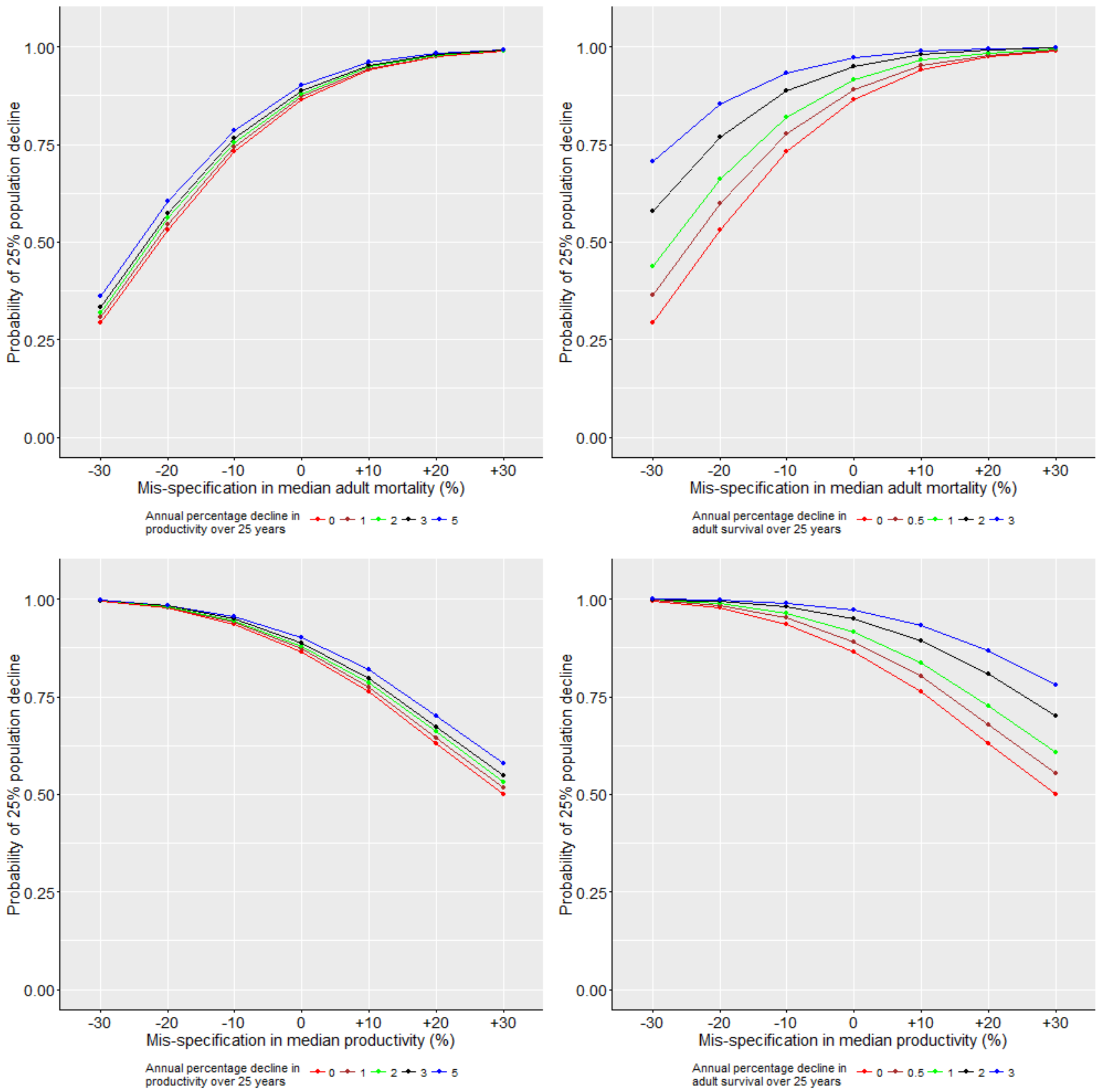


Figure 4g: PVA Metric E3 – probability of population decline greater than 50% from 2016-2041. Adult mortality mis-specification is illustrated in the upper panels and productivity mis-specification in the lower panels. Mis-specification was varied from -30% to +30% (with 0% representing no mis-specification). The five coloured lines represent the different levels of potential impact on annual productivity (left panels) or annual adult survival (right panels) over the hypothetical 25 year lifetime of the wind farm (2017-2041).

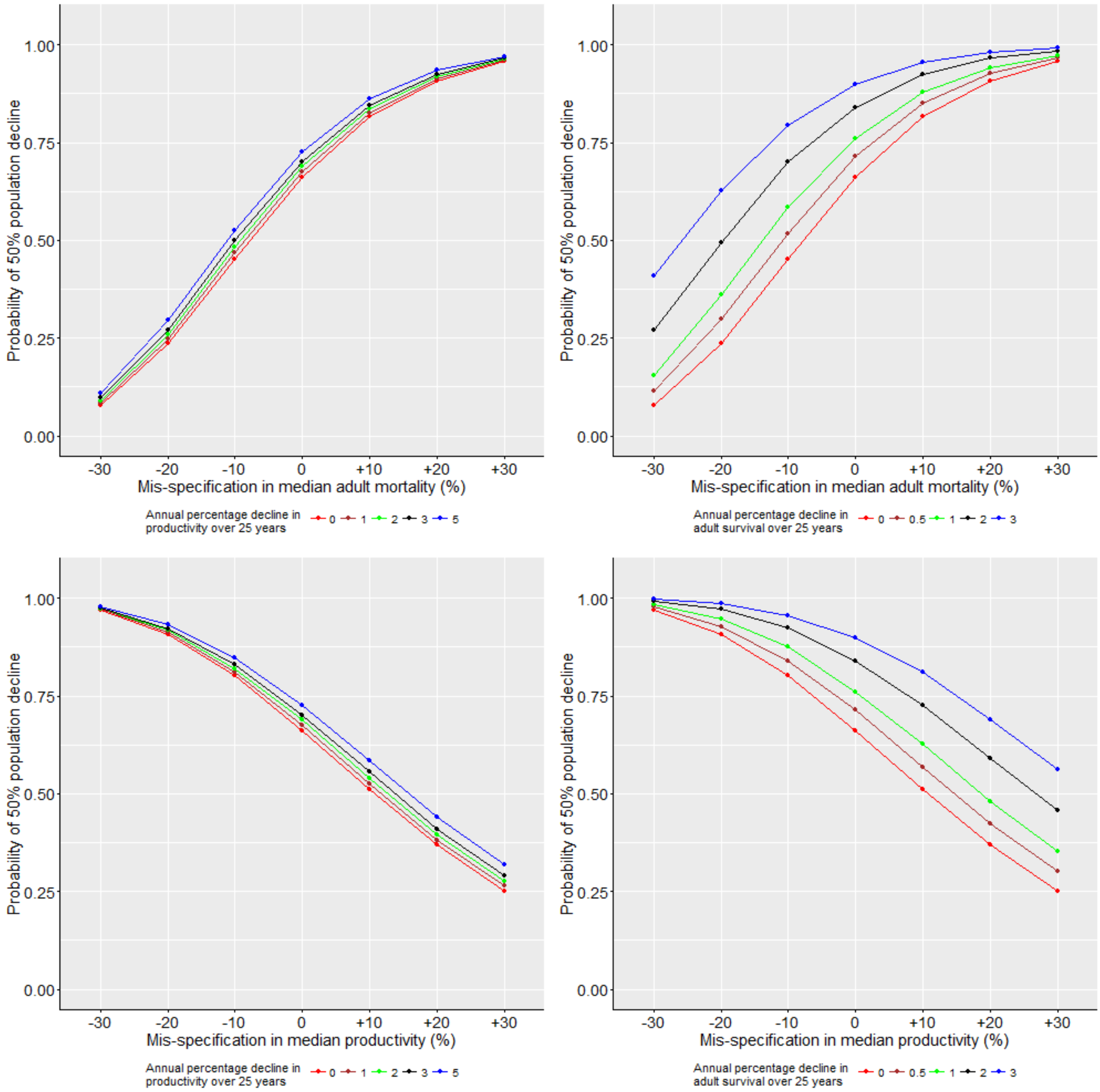
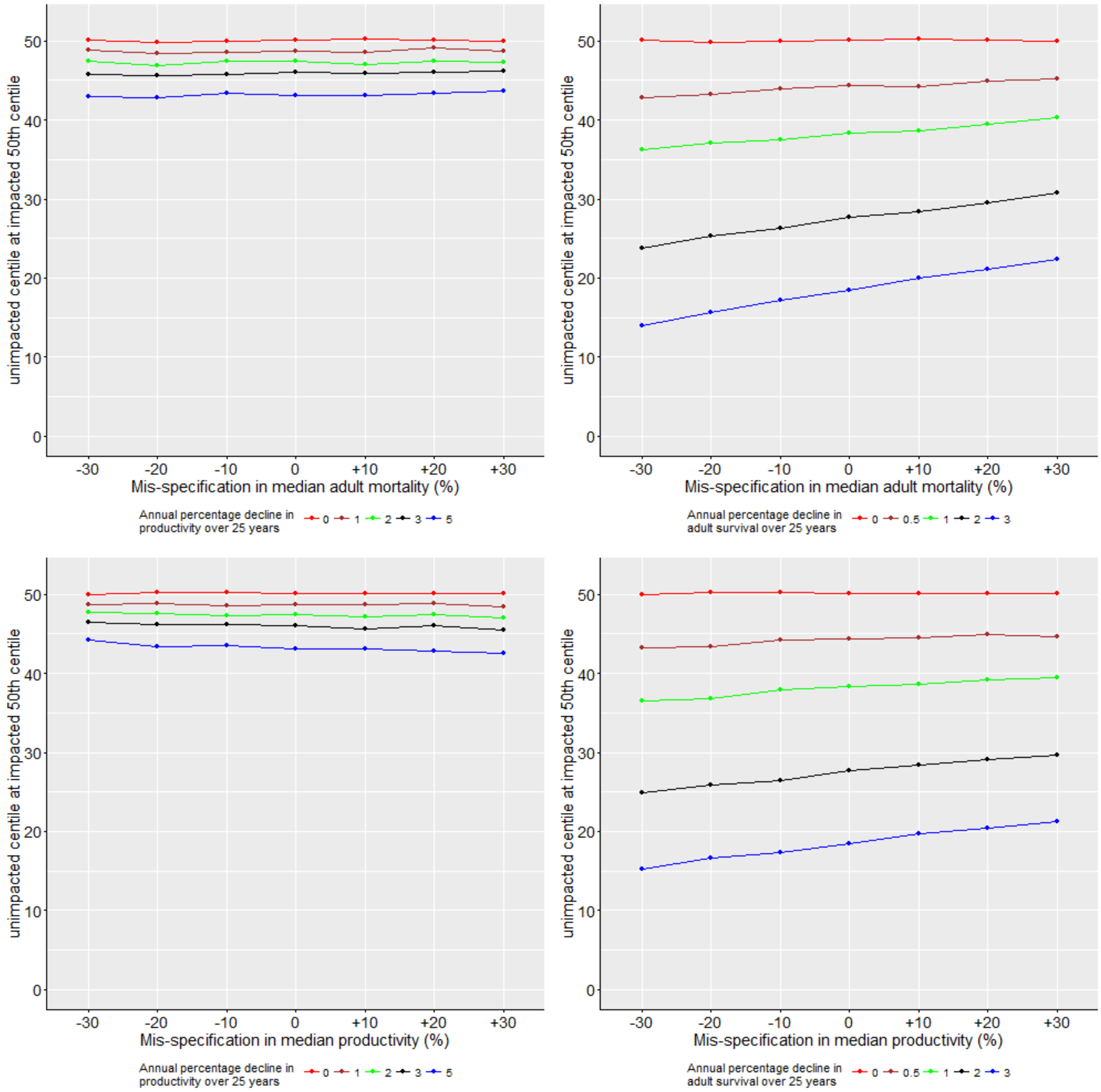


Figure 4h: PVA Metric F – centile from un-impacted population size equal to the 50th centile of the impacted population size, at 2041. Adult mortality mis-specification is illustrated in the upper panels and productivity mis-specification in the lower panels. Mis-specification was varied from -30% to +30% (with 0% representing no mis-specification). The five coloured lines represent the different levels of potential impact on annual productivity (left panels) or annual adult survival (right panels) over the hypothetical 25 year lifetime of the wind farm (2016-2041).



4.2 PVA Sensitivity in Relation to Population Status and Renewables Effect Size

To examine the effects of population status and renewables effect size, we integrated the results for the 13 SPA species/combinations for which we had good model convergence at the time of writing:

- Kittiwakes: Forth Islands; St Abb's Head; Fowlsheugh; Buchan Ness to Collieston Coast
- Guillemots: Forth Islands; St Abb's Head; Fowlsheugh; Buchan Ness to Collieston Coast
- Razorbills: Forth Islands; St Abb's Head; Fowlsheugh
- Shags: Forth Islands; St Abb's Head

Six of the thirteen indicated increasing abundance over time. These are guillemots at Forth Islands, St Abb's Head and Buchan Ness to Collieston Coast, razorbills at Forth Islands and Fowlsheugh and shags at Forth Islands, while the remainder showed a decrease (Table 7), providing a comparatively even balance facilitating this comparison. Results for differences in sensitivity in decreasing and increasing populations can be found in Figures 5a-h for PVA A, B, C, D, E1, E2, E3 and F, respectively. These plots show results from the analysis of mis-specification in adult mortality with the maximum scenario of change in adult survival (3%).

We present PVA sensitivities in relation to scenarios of renewables effect size in Figures 6a-h for PVA A, B, C, D, E1, E2, E3 and F, respectively. Of the four combinations shown in Figures 3 and 4, we only show results from the analysis of mis-specification in adult mortality with scenarios of change in adult survival, with effect sizes of 0.5%, 1%, 2% and 3%.

For PVA A, values approximate one (range 0.977-1) and there was no discernible difference in sensitivity between decreasing and increasing populations or with respect to renewables effect size (Figures 5a and 6a). Note that although annual growth rates are close to one, 25 year growth rates will show a discernible difference. For example, an annual growth rate of 0.977, results in a 25 year growth rate of 0.559.

For PVA B, there was also no discernible difference in sensitivity between decreasing and increasing species (Figure 5b). There was an increase in sensitivity with increasing effect sizes, with slopes flatter at 0.5% effect size compared with 3%

effect size, though the effect was small and the metric can be considered comparatively insensitive to all scenarios of effect size (Figure 6b).

PVAs C and D had higher sensitivity than PVAs A and B overall, but showed a similar response to population status and renewables effect size to PVA B, such that there was no clear difference between decreasing and increasing species in slope (Figure 5c and 5d), and a slight increase in gradient with increasing effect size from 0.5% to 3% (Figure 6c and 6d).

For PVA E, increasing populations showed greater sensitivity to probability of population decline greater than 10% than decreasing populations (Figure 5e), whereas the converse was true for a probability of population decline greater than 50% (Figure 5g). Similar sensitivities were apparent at 25% (Figure 5f). These differences reflect the pattern of probabilities of thresholds of change in population size relative to population status, with mis-specification having a smaller effect on probability of a smaller change in population size (10%) in a decreasing population since probability of this outcome is very high in most circumstances, and a smaller effect on probability of a larger change in population size (50%) in an increasing population (where probability of this outcome is very low in most circumstances). There was no clear difference in sensitivity with respect to renewables effect size, being comparatively high and variable in all scenarios at all three thresholds (Figures 6e-g).

PVA F showed a similar response to PVAs A, B, C and D with respect to population status and effect size. Thus, there was no clear difference in sensitivity between decreasing and increasing species in slope, with sensitivity overall being moderately low, higher than ratio metrics but lower than PVA E (Figure 5h). Sensitivity was also comparatively unaffected by effect sizes (Figure 6h).

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

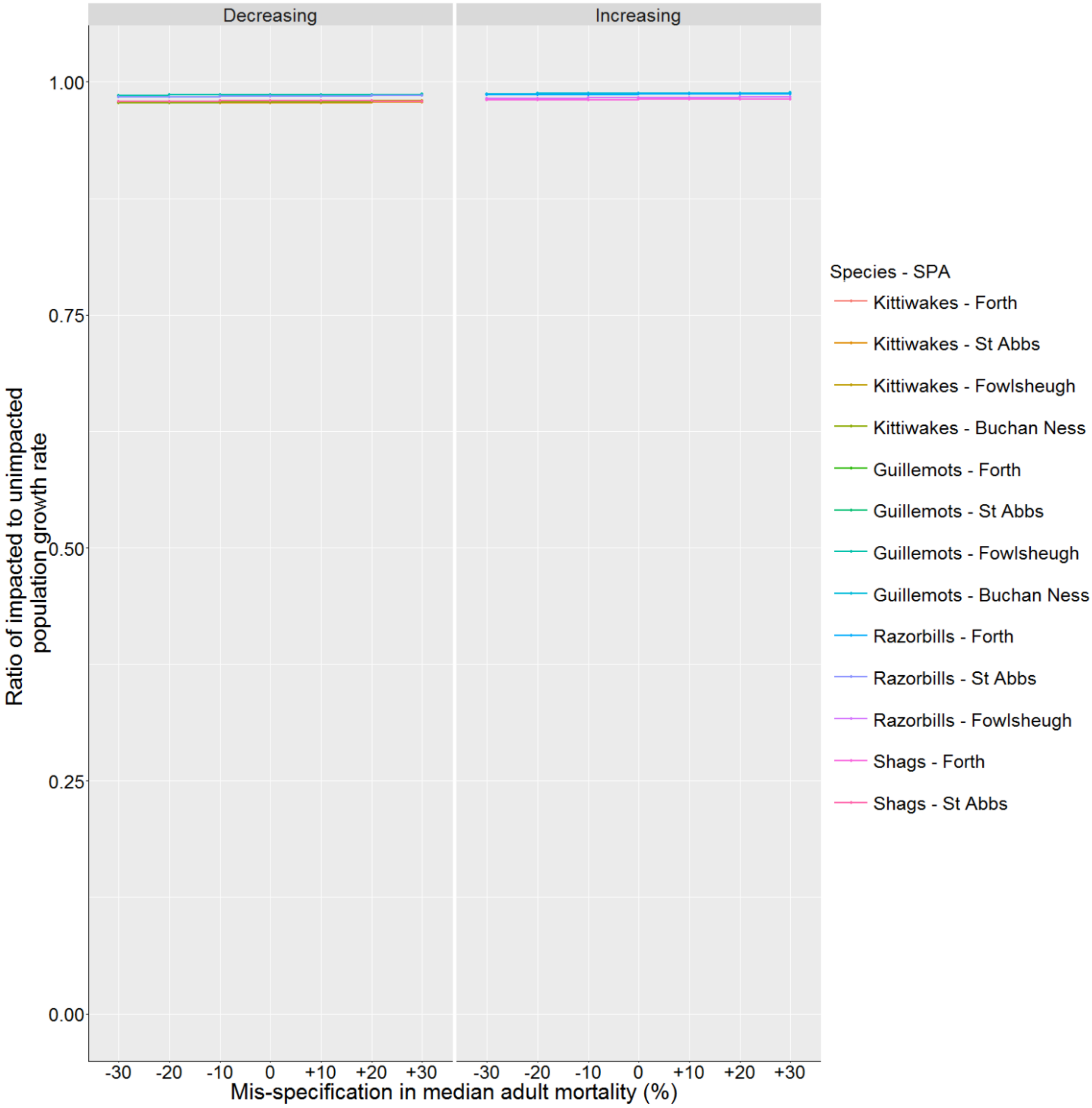


Figure 5b: PVA Metric B – ratio of population size at 2041, comparing impacted population vs. un-impacted population, for changing adult mortality and a 3% decrease in adult survival, across decreasing populations and increasing populations.

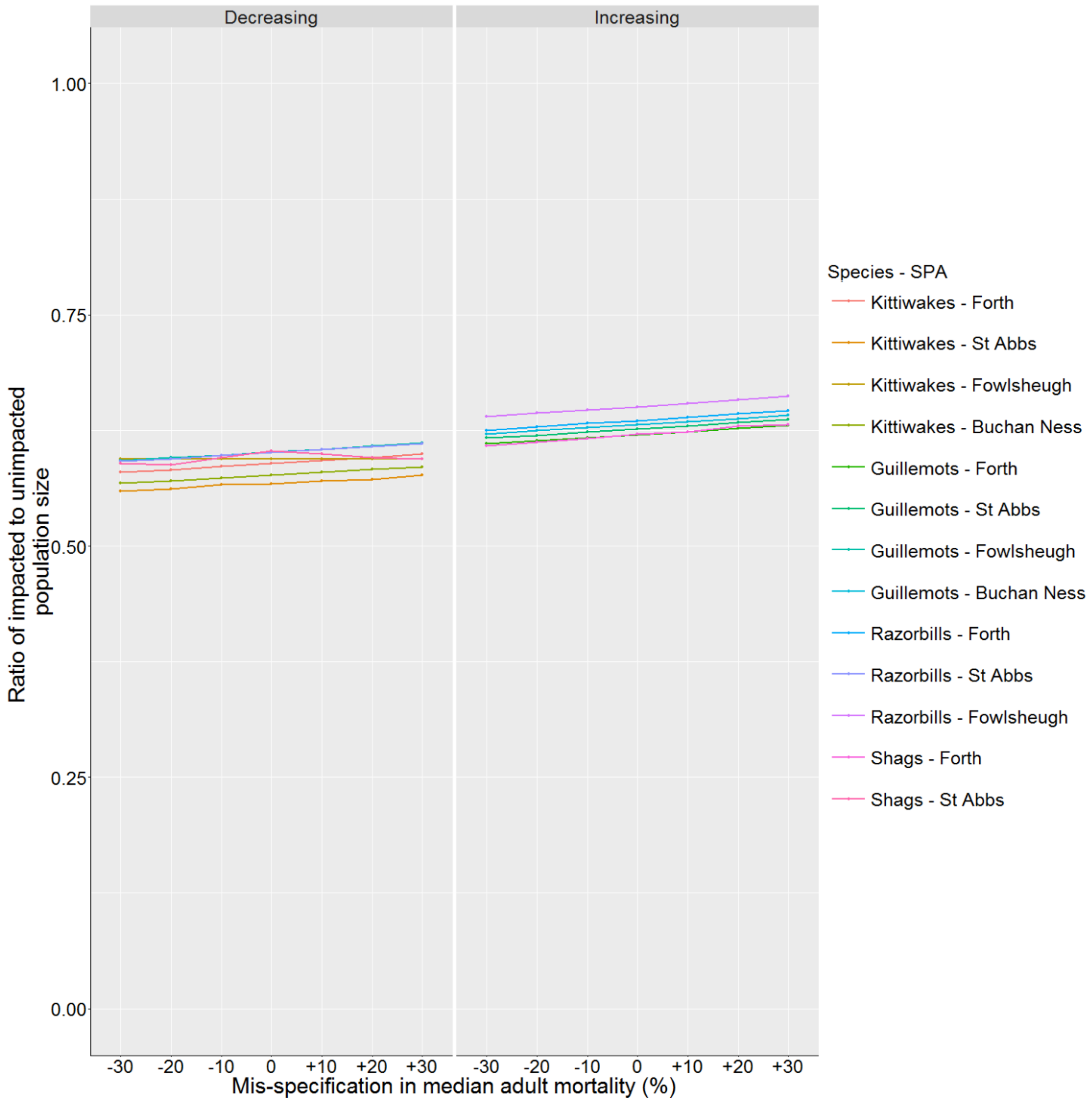


Figure 5c: PVA Metric C – difference in population growth rate from 2016-2041, comparing impacted population vs. un-impacted population, for changing adult mortality and a 3% decrease in adult survival, across decreasing populations and increasing populations.

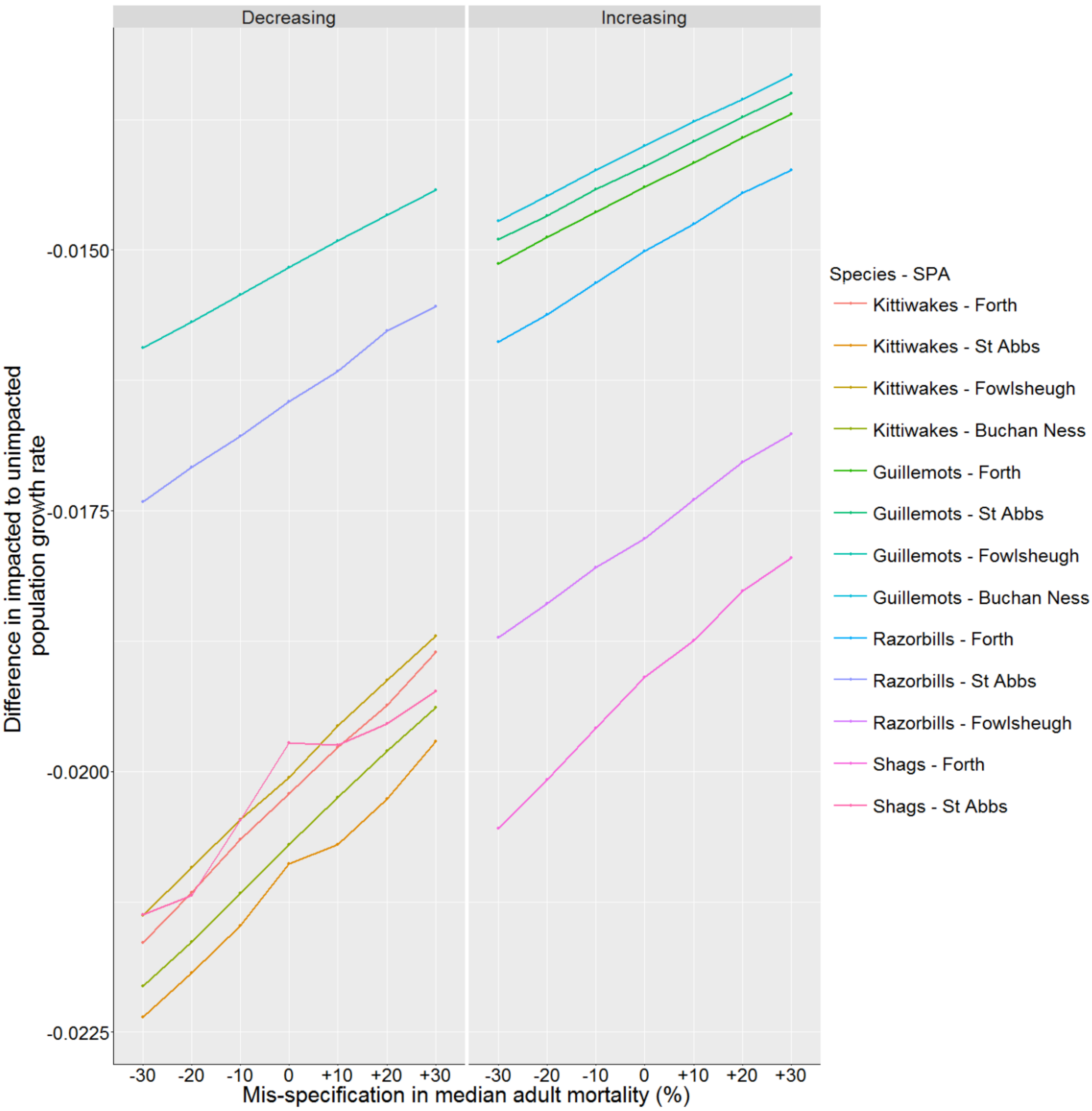


Figure 5d: PVA Metric D – difference in population size at 2041, comparing impacted population vs. un-impacted population, for changing adult mortality and a 3% decrease in adult survival, across decreasing populations and increasing populations.

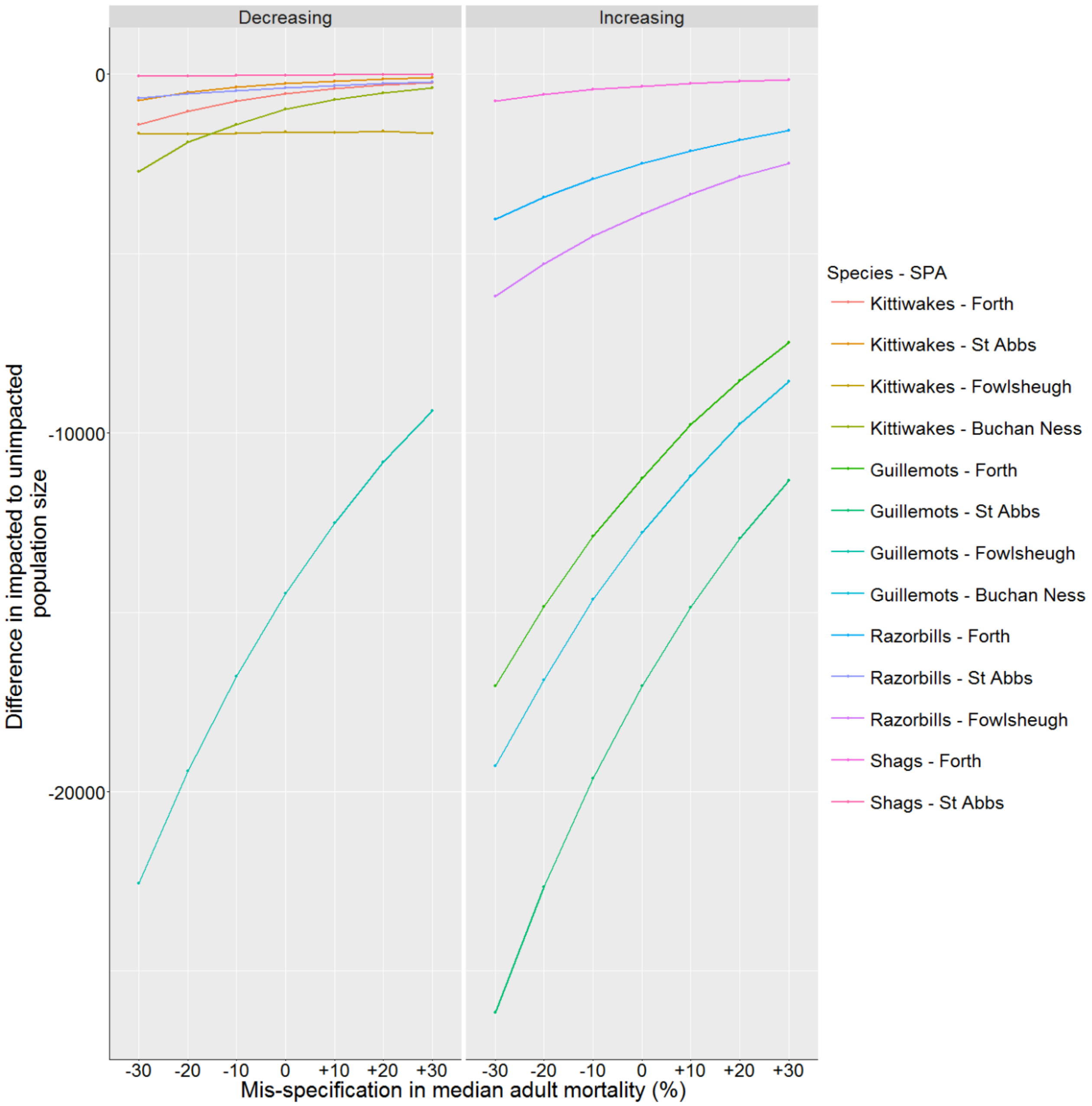


Figure 5e: PVA Metric E1 – probability of population decline greater than 10% from 2016-2041, for changing adult mortality and a 3% decrease in adult survival, across decreasing populations and increasing populations.

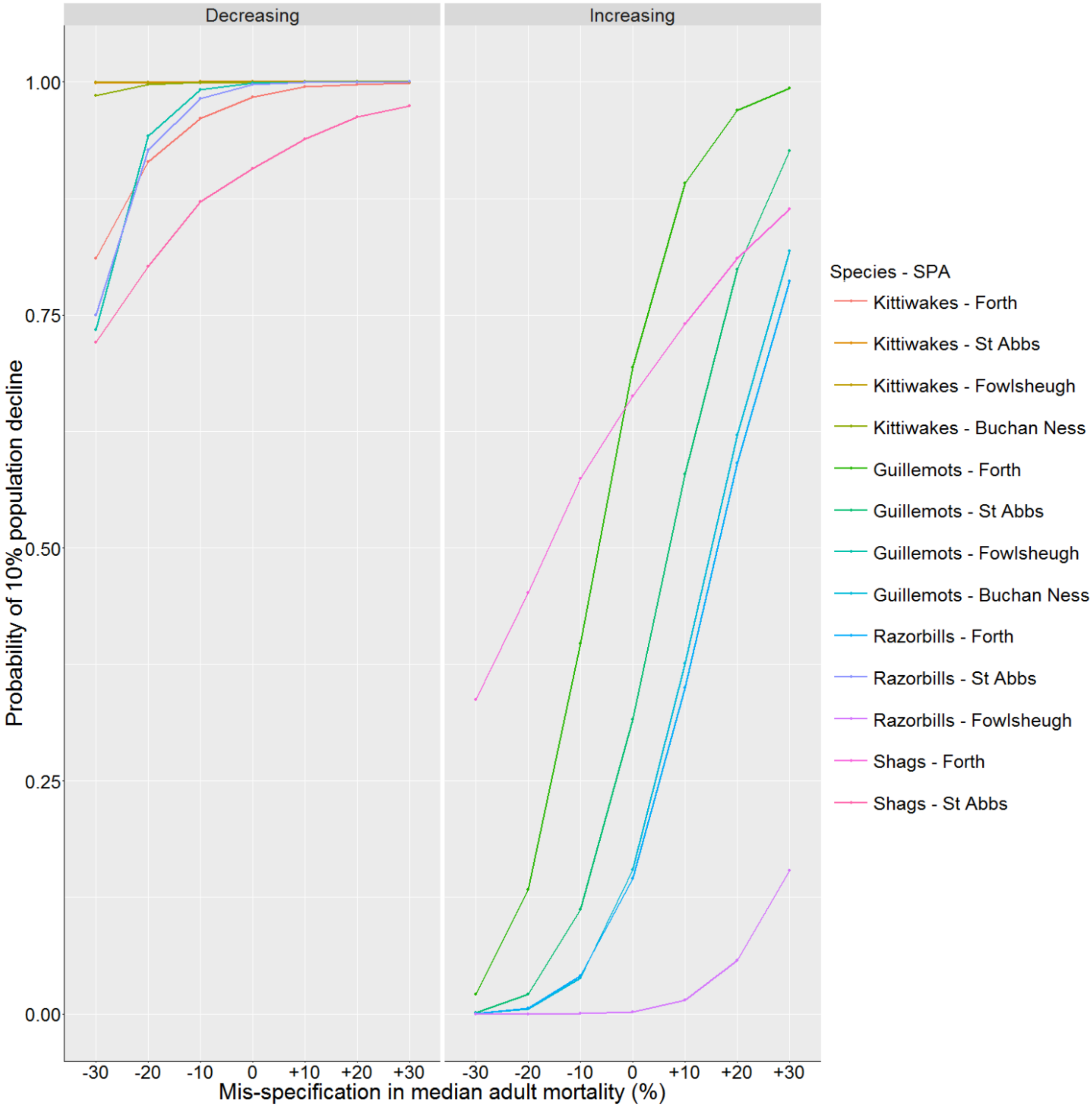


Figure 5f: PVA Metric E2 – probability of population decline greater than 25% from 2016-2041, for changing adult mortality and a 3% decrease in adult survival, across decreasing populations and increasing populations.

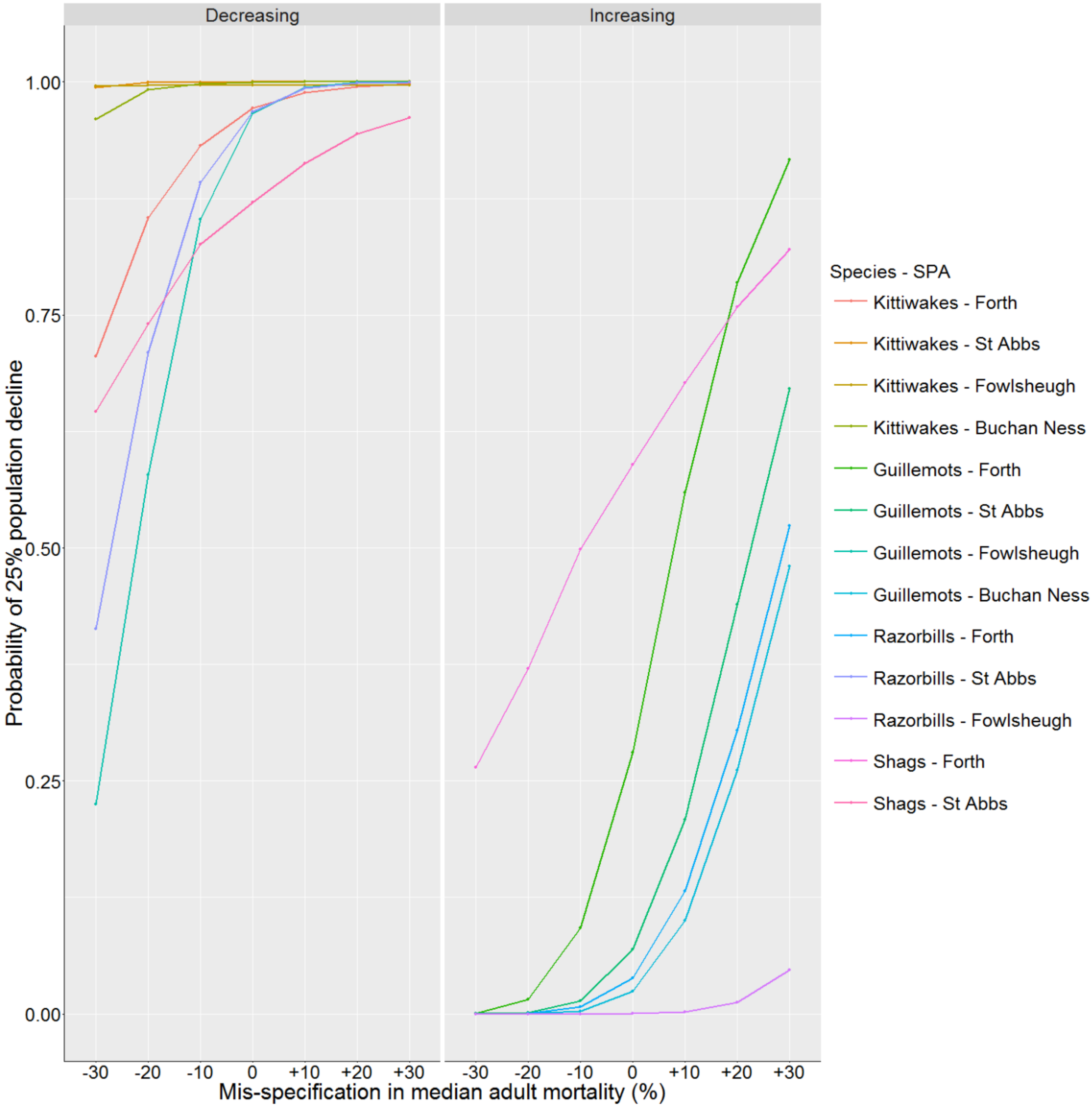


Figure 5g: PVA Metric E3 – probability of population decline greater than 50% from 2016-2041, for changing adult mortality and a 3% decrease in adult survival, across decreasing populations and increasing populations.

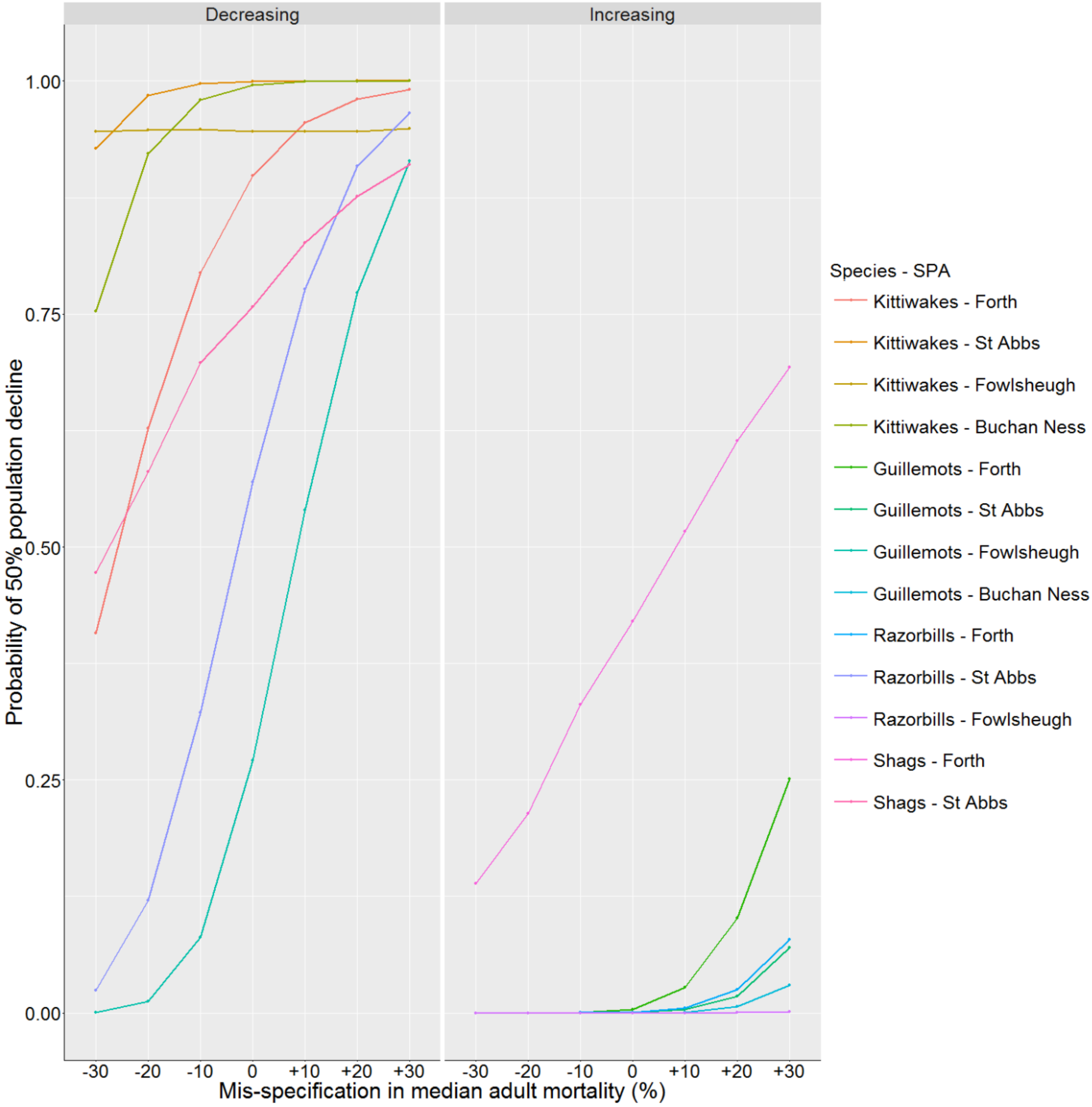


Figure 5h: PVA Metric F – centile from un-impacted population size equal to the 50th centile of the impacted population size, at 2041, for changing adult mortality and a 3% decrease in adult survival, across decreasing populations and increasing populations.

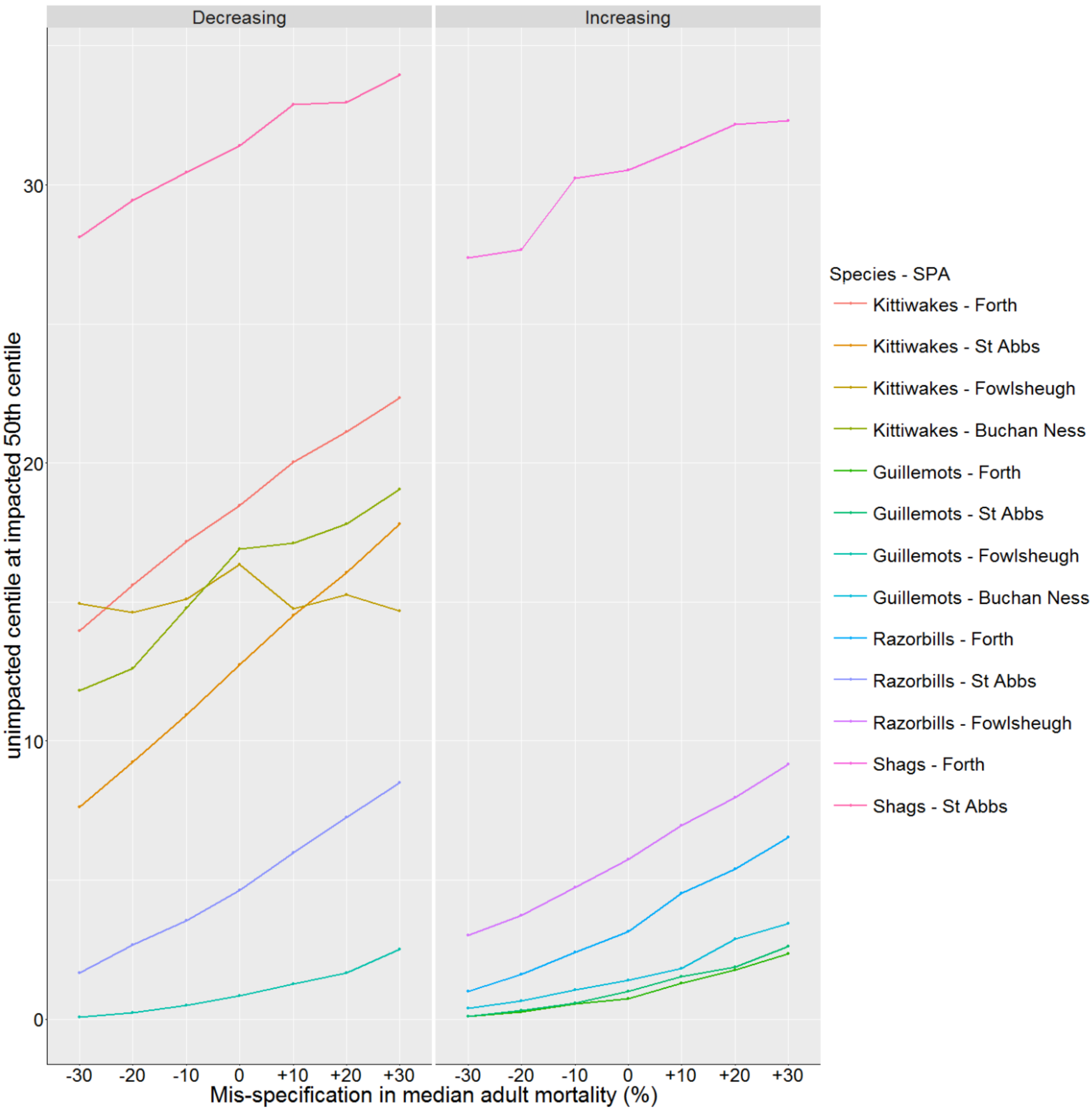


Figure 6a: PVA Metric A – ratio of population growth rate from 2016-2041, comparing impacted population vs. un-impacted population, for changing adult mortality and various decreases in adult survival, across all populations.

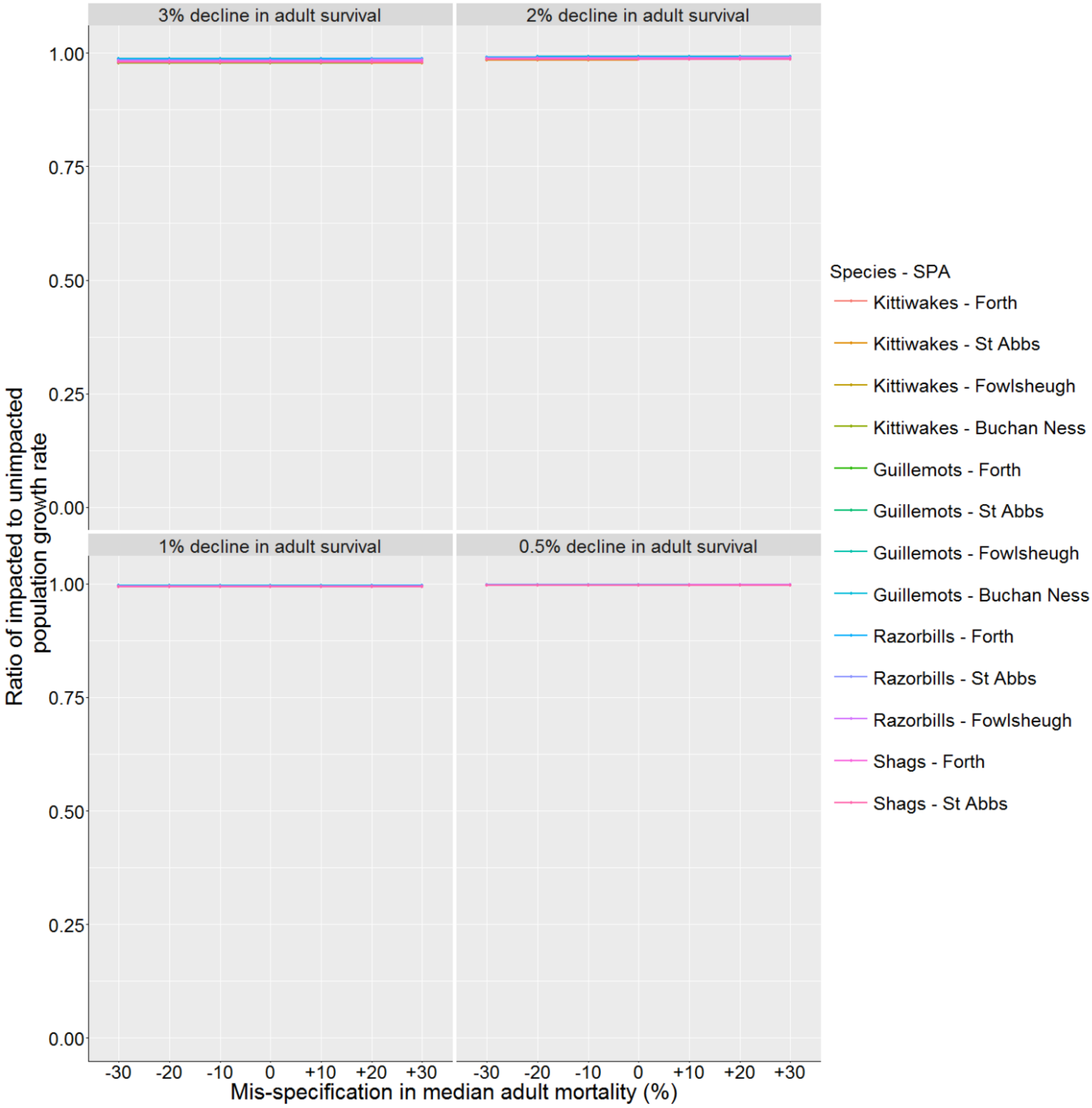


Figure 6b: PVA Metric B – ratio of population size at 2041, comparing impacted population vs. un-impacted population, for changing adult mortality and various decreases in adult survival, across all populations.

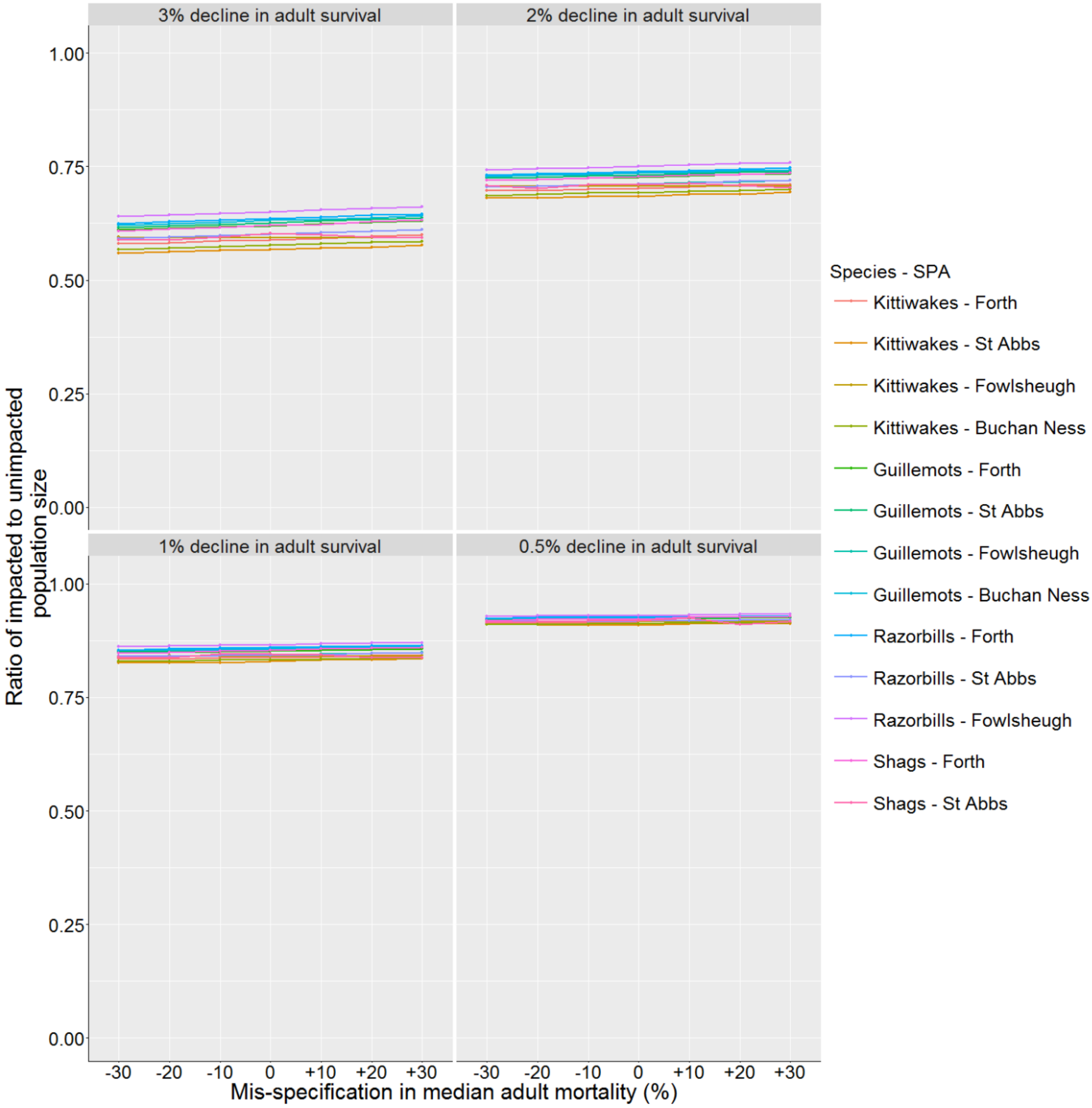


Figure 6c: PVA Metric C – difference in population growth rate from 2016-2041, comparing impacted population vs. un-impacted population, for changing adult mortality and various decreases in adult survival, across all populations.

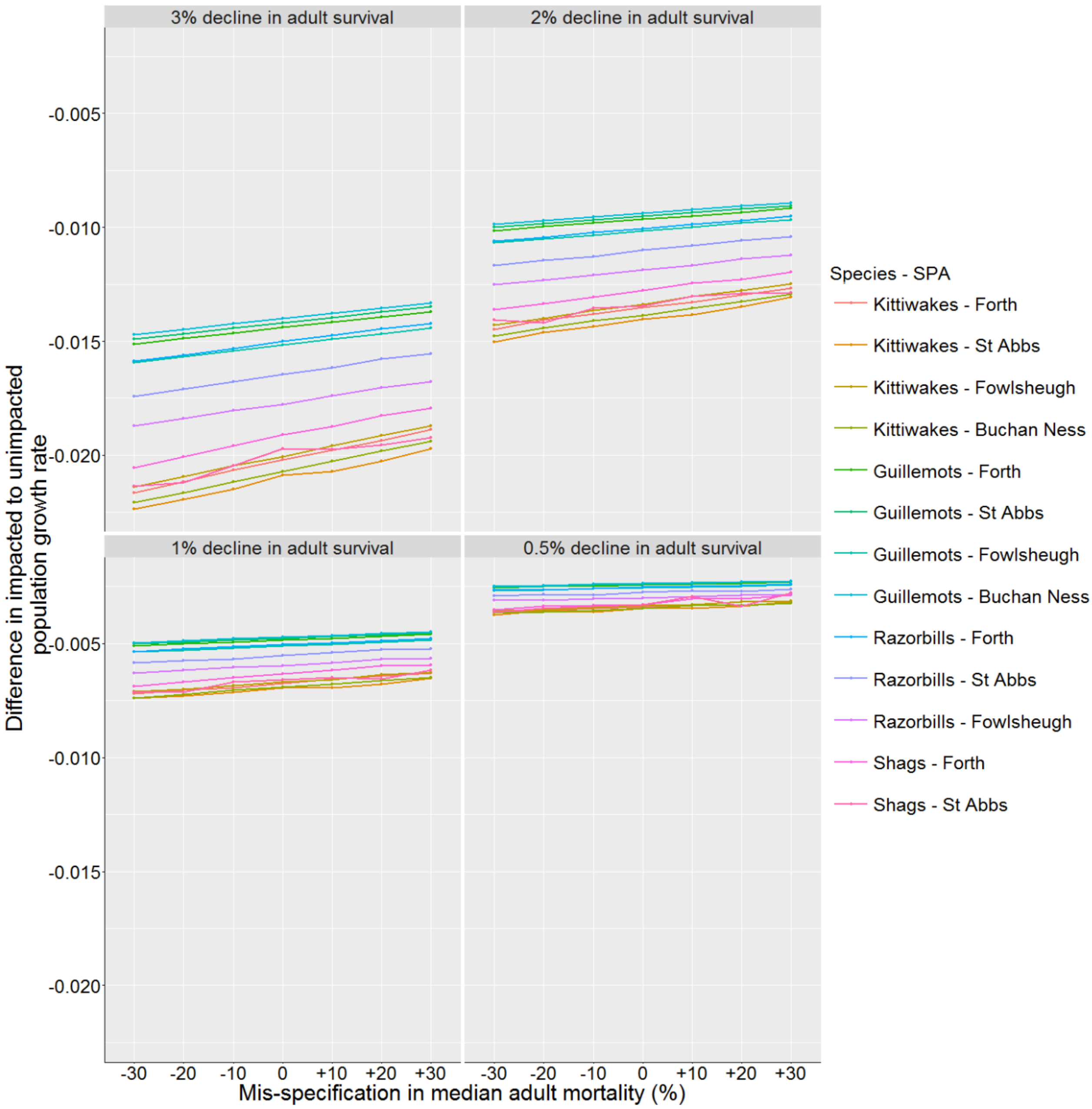


Figure 6d: PVA Metric D – difference in population size at 2041, comparing impacted population vs. un-impacted population, for changing adult mortality and various decreases in adult survival, across all populations.

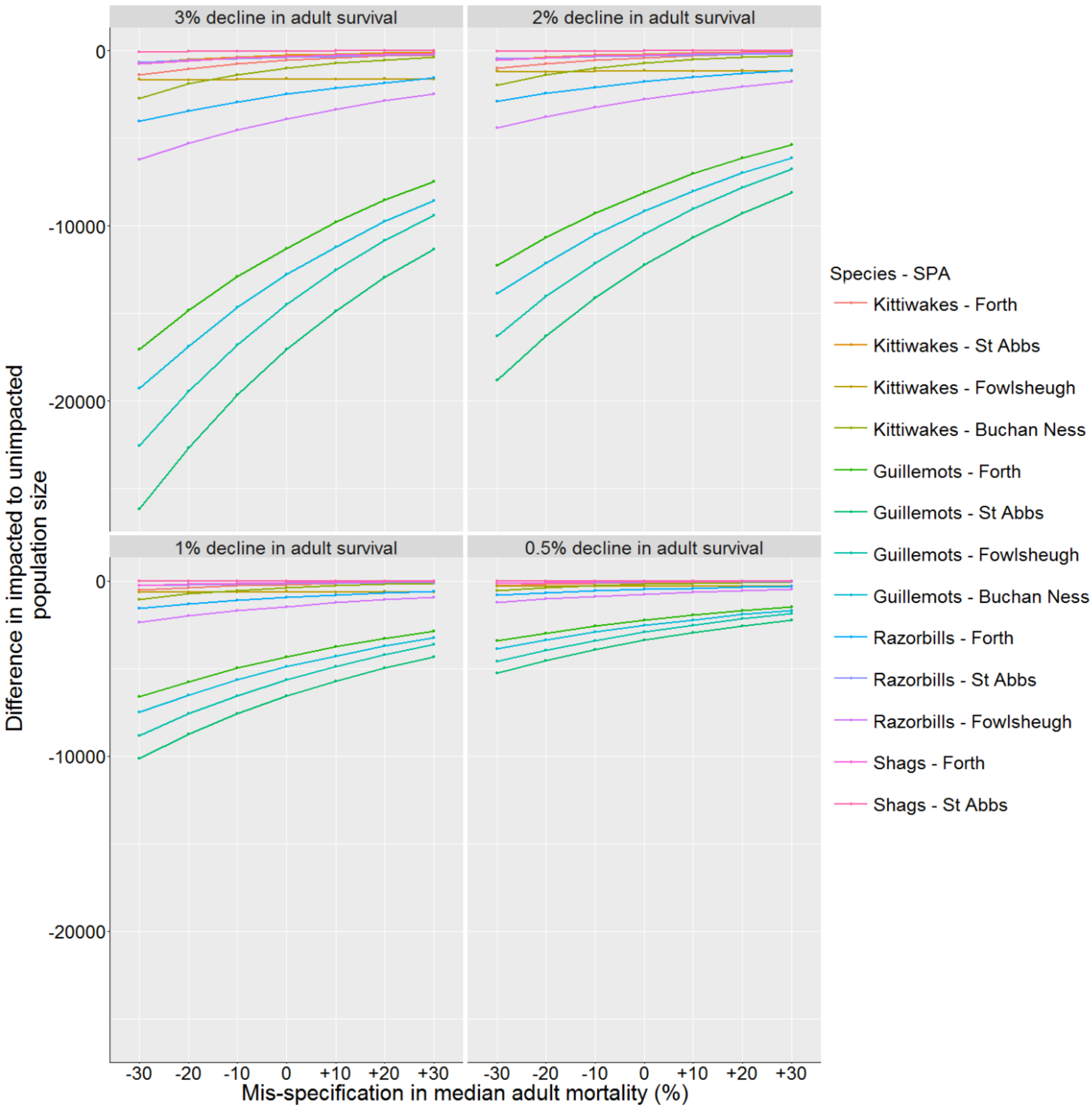


Figure 6e: PVA Metric E1 – probability of population decline greater than 10% from 2016-2041, for changing adult mortality and various decreases in adult survival, across all populations.

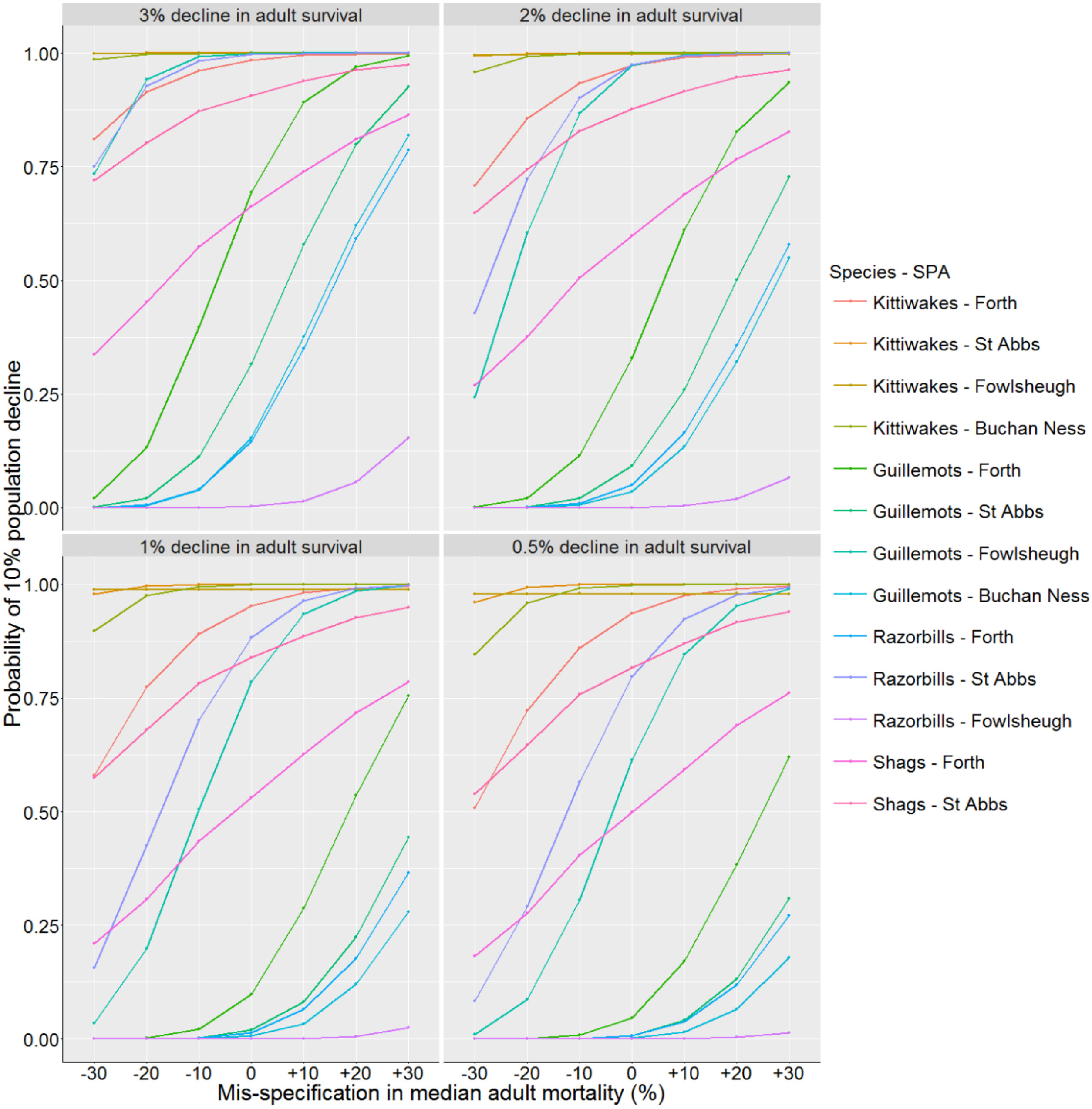


Figure 6f: PVA Metric E2 – probability of population decline greater than 25% from 2016-2041, for changing adult mortality and various decreases in adult survival, across all populations.

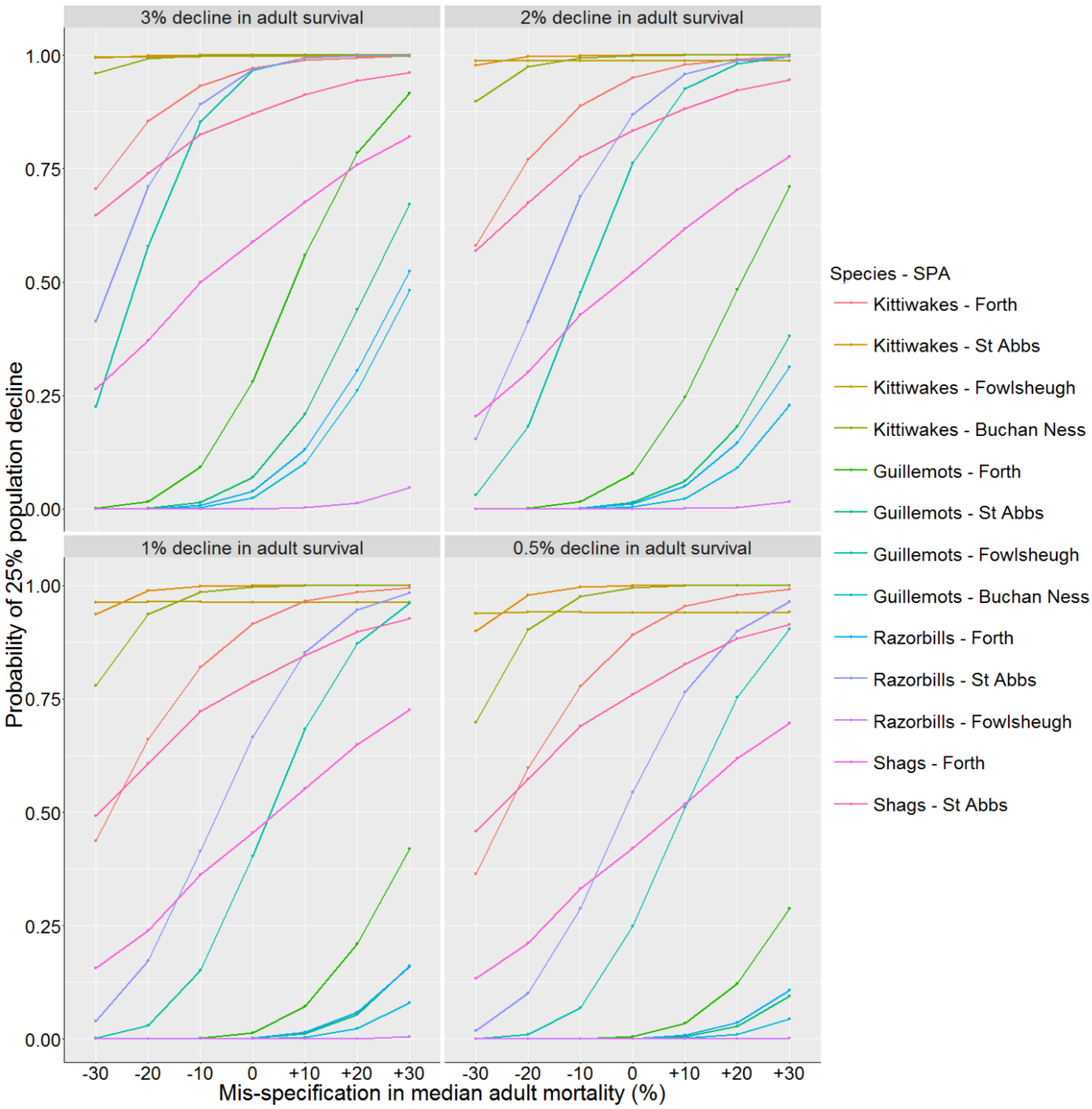


Figure 6g: PVA Metric E3 – probability of population decline greater than 50% from 2016-2041, for changing adult mortality and various decreases in adult survival, across all populations.

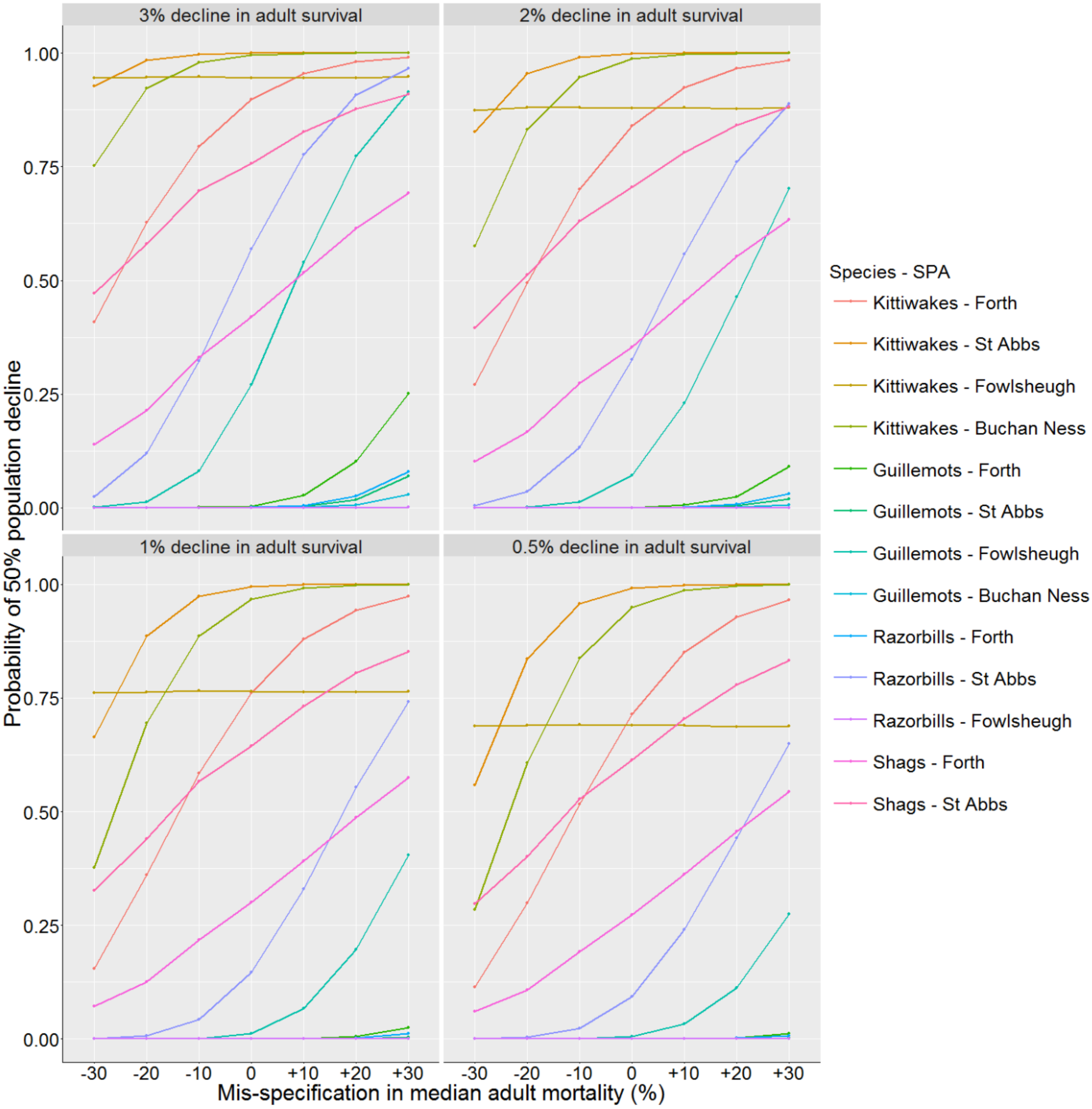
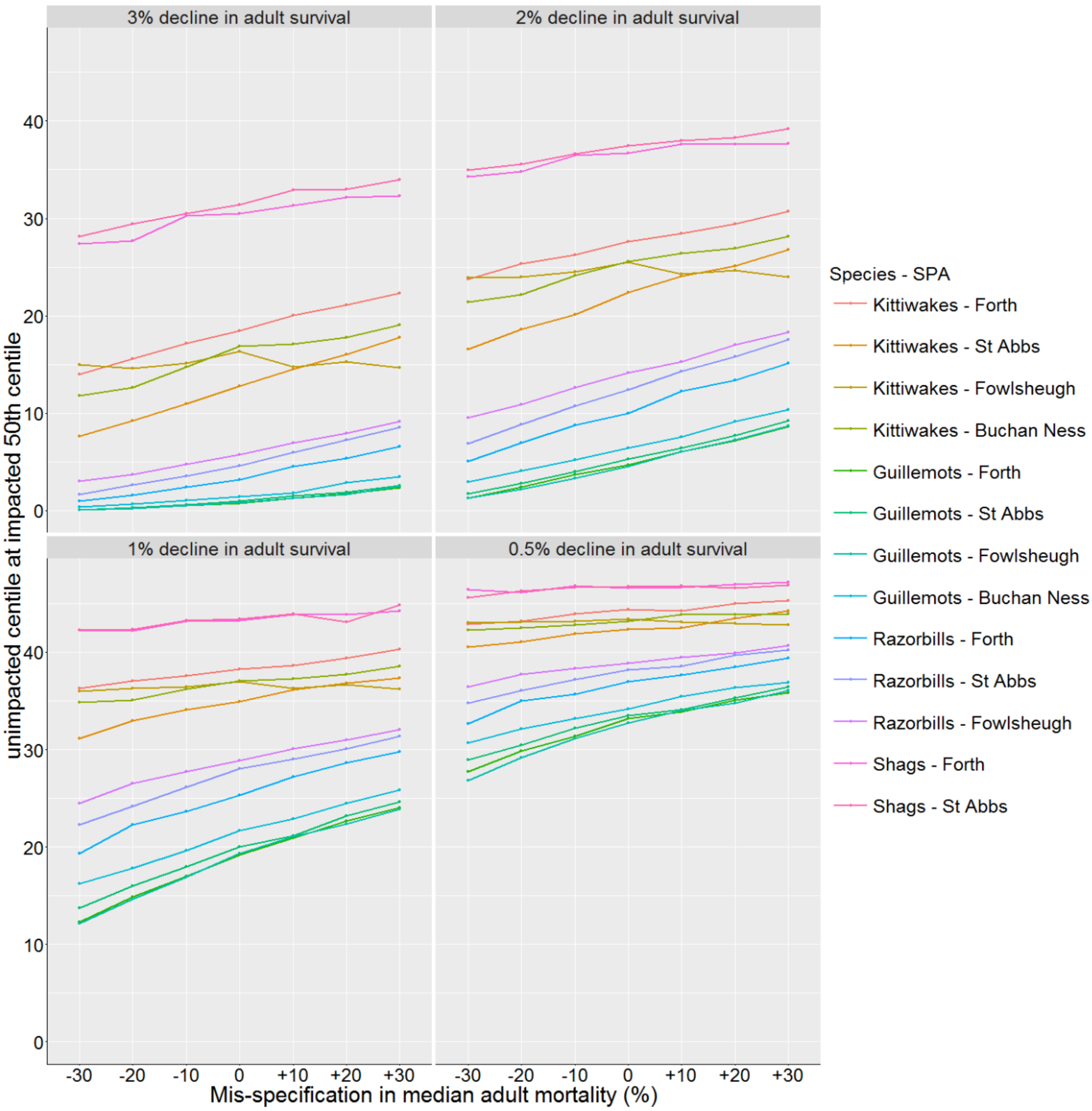


Figure 6h: PVA Metric F – centile from un-impacted population size equal to the 50th centile of the impacted population size, at 2041, for changing adult mortality and various decreases in adult survival, across all populations.



5. Discussion and Recommendations

5.1 PVA Metric Sensitivity

This study represents the most comprehensive assessment of PVA metric sensitivity to mis-specification of demographic rates in relation to population status and perturbation effect sizes in the seabird/marine renewable context using real-world data. Using available data on abundance, survival and productivity in a well-studied region of the UK and Bayesian population modelling approaches, we compared the sensitivity to mis-specification of input demographic parameters of six PVA metrics, comprising two ratio metrics (PVAs A and B), two metrics related to ratio metrics (PVAs C and D) and two probabilistic metrics (PVAs E and F).

By undertaking an analysis of real-world data sets, our work provides a useful complement to recent work on sensitivity of PVA metrics to input parameter uncertainty using simulation modelling of generic seabird species with varying life histories (Cook & Robinson 2016b, 2017). The close accordance in findings provides confidence on choice of PVA metrics that are least sensitive to such mis-specification, and, therefore, most suitable for use in wind farm assessments.

5.2 Recommendations on PVA Metrics

The two ratio metrics performed best among the six metrics considered with respect to sensitivity to mis-specification in input parameters. The ratio of impacted to un-impacted annual growth rate (PVA A) and ratio of impacted to un-impacted population size after 25 years (PVA B) both showed low sensitivity to demographic input mis-specification, in accordance with findings from other studies (Green *et al.* 2016; Cook & Robinson 2016b, 2017), with PVA A performing consistently better than PVA B.

The calculations of difference in impacted and un-impacted annual growth rates (PVA C) and between impacted and un-impacted population size after 25 years (PVA D) were not so readily interpretable but they are useful when growth rates or population size estimates are small.

In keeping with other work, we found that the probability PVA metric (PVA E) was highly sensitive and we, therefore, caution against using it in this context, in accordance with recommendations by other authors (Green *et al.* 2016; Cook & Robinson 2016b, 2017). We were not tasked with testing the sensitivity of counterfactual probabilistic metrics, in particular Metric 8 in Table 2 (“Change in probability of a 10, 25 or 50% decline", also known as "Counterfactual of the probability of population decline", and linked to Metric 7 in Table 2/PVA E in this report), a metric that has been used frequently in assessments, often in association with PVA E. However, a visual examination of the figures presenting PVA metric E shows in almost all cases, a clear divergence between the lines across the range of values of mis-specification, and this change in the difference between values across effect sizes represents sensitivity to mis-specification of demographic rates in the excess probability referred to here. Good examples where this is clear are Figure 4e (all four panels) and Figure 4f (all four panels). It is not clear in all cases – see for example Figure 4g (top left panel). However, overall we can conclude that this counterfactual is comparatively more sensitive to mis-specification than ratio metrics.

Finally, the metric representing the centile from the un-impacted population size equal to the 50th centile of the impacted population size at the end of the wind farm (PVA F) showed moderately low sensitivity to mis-specification of survival and productivity. It performed considerably better than the other probabilistic metric (PVA E - probability of a population decline) with markedly lower sensitivity to mis-specification, population status and renewables effect size. However, it was more sensitive than ratio metrics, and in some cases showed unstable sensitivity which was less apparent in PVA metrics A and B (see Figures 5 a, b and h; Figures 6 a, b and h).

We recommend that those undertaking assessments consider the relative performance of different metrics with respect to sensitivity to mis-specification of input parameters. To summarise, of the two ratio and two probabilistic metrics considered here, the order with respect to sensitivity to mis-specification of input parameters was PVA A; PVA B; PVA F; PVA E. PVA E was much more sensitive than the other three and is not recommended for use in this context. If the first three are used in assessments in future, we recommend that interpretation should factor in their relative sensitivities. We also recommend that PVA metrics (C and D) are used since they are estimable when ratios are being calculated.

Note that we do not make recommendations on appropriate thresholds in relation to the above metrics, which is a societal choice and a matter for regulators.

5.3 Recommendations on PVA Analysis in Assessments of Renewables on Seabirds

We believe that Population Viability Analysis is a robust framework for forecasting future population change of seabirds under baseline conditions and under conditions of varying perturbations on demographic rates caused by renewable developments.

Furthermore, we believe that Bayesian state-space models have considerable potential in Population Viability Analysis using real data. Forecasts are made straightforward by the adoption of this approach, since posterior distributions are naturally generated. Furthermore, these methods do not suffer from the same criticism aimed at traditional methods that confidence intervals are unrealistically narrow. In addition, the study region has some of the most comprehensive demographic data available on seabirds in the UK, collected by CEH at their long term field site on the Isle of May, which has proved extremely valuable in carrying out this work. However, the restricted availability of high quality data left us with no alternative but to use these data on other populations where no such data exist. Despite this, the models of these other populations generally performed well. Exceptions were where population counts were sparse and variable, a particular issue at the Buchan Ness to Collieston Coast SPA.

5.4 Future Research and Monitoring Priorities

A fruitful avenue for future research would be extension to more complex models that incorporate environmental covariates or density dependence. Although there remains a lack of empirical evidence linking environmental covariates and seabird demography (Daunt *et al.* 2017), examples do exist (e.g. Frederiksen *et al.* 2004) and could form the rationale for future modelling including covariates. Evidence for density dependence in UK seabird populations is emerging (Horswill *et al.* 2016) and could be included where there is strong evidence for its occurrence including, crucially, whether the form of density dependence is compensatory or depensatory.

It would also be beneficial to estimate PVA metric sensitivity across a broader range of real world examples, comprising more species with differing life histories than we could consider here. This approach would enable a more comprehensive assessment of ratio and probabilistic metrics. Furthermore, it would be useful to test PVA F using a simulation modelling approach (Cook & Robinson 2016b, 2017) to establish whether a similar sensitivity to mis-specification of input parameters was apparent using that method. Another future priority would be to test sensitivity of different metrics using different population modelling methods: in addition to

Bayesian state-space models, other methods that may be more suited to sparse data could be incorporated, such as age-structured population growth models.

It is encouraging to note the value of plot counts, since these can be maintained on an annual or near- annual basis much more readily than full colony counts. However, we would recommend that full counts continue to be undertaken regularly to ensure that plots continue to be representative. Local data on survival and productivity add significantly to the ability to model populations effectively. However, our study demonstrates that PVA metrics, and their sensitivity to mis-specification, can be estimated where data are absent from the focal colony but available from an alternative, ideally nearby colony, thereby offering a natural, informative model prior. However, considerable thought is required before adopting this approach since information from another colony cannot automatically be assumed to apply elsewhere to other species and/or regions.

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Appendix 1

Input Parameters to the Bayesian State Space Models

This appendix details the input values for the population models.

Input parameters for adult survival and productivity are provided at two scales. In the Bayesian models, they are on the logit or the log scale (Table A1.1). However, these can be somewhat difficult to understand, so we have back transformed those that are on the log scale (productivity for kittiwakes and shags), using the mean and variance on the log scale to estimate the mean and variance of the untransformed productivity, which is log-normally distributed; these estimates can be verified with simulations. The two approaches matched. We, therefore, ran simulations for the parameters on the logit scale and estimate the mean and variance for the remaining untransformed survival and productivity parameters (Table A1.2).

Population counts are provided for all populations that were successfully modelled in this project in Tables A1.3 (kittiwakes), A1.4a and A1.4b (guillemots), A1.5a and A1.5b (razorbills) and A1.6 (shags).

Table A1.1

Input parameters into the Bayesian state space models for kittiwakes, guillemots, razorbills and shags at Forth Island, St Abbs, Buchan Ness and Fowlsheugh SPAs. Note that adult survival is on the logit scale and productivity is on the log scale for kittiwakes and shags, and on the logit scale for guillemots and razorbills (see Table A1.2 for values on the untransformed scale).

Species	**SPA**	**Adult survival: mean (sd)**	**Productivity: mean (sd)**
Kittiwake	Forth Islands	1.875 (0.546)	-0.790 (0.898)
	St Abb's Head	1.875 (0.546)	-0.615 (0.679)
	Fowlsheugh	1.875 (0.546)	-0.313 (0.492)
	Buchan Ness to Collieston Coast	1.875 (0.546)	-0.678 (0.699)
Guillemot	Forth Islands	2.705 (0.634)	1.041 (0.583)
	St Abb's Head	2.705 (0.634)	1.041 (0.583)
	Fowlsheugh	2.705 (0.634)	1.041 (0.583)
	Buchan Ness to Collieston Coast	2.705 (0.634)	1.041 (0.583)
Razorbill	Forth Islands	2.494 (0.685)	0.552 (0.350)
	St Abb's Head	2.494 (0.685)	0.552 (0.350)
	Fowlsheugh	2.494 (0.685)	0.552 (0.350)
Shag	Forth Islands	2.147 (1.215)	-0.052 (0.637)
	St Abb's Head	2.147 (1.215)	0.170 (0.590)

Table A1.2

Input parameters into the Bayesian state space models for kittiwakes, guillemots, razorbills and shags at Forth Island, St Abbs, Buchan Ness and Fowlsheugh SPAs. Note that adult survival and productivity are on the untransformed scale.

Species	SPA	Adult survival: mean (sd)	Productivity: mean (sd)
Kittiwake	Forth Islands	0.855 (0.067)	0.679 (0.755)
	St Abb's Head	0.855 (0.067)	0.681 (0.521)
	Fowlsheugh	0.855 (0.067)	0.825 (0.432)
	Buchan Ness to Collieston Coast	0.855 (0.067)	0.648 (0.515)
Guillemot	Forth Islands	0.927 (0.045)	0.725 (0.111)
	St Abb's Head	0.927 (0.045)	0.725 (0.111)
	Fowlsheugh	0.927 (0.045)	0.725 (0.111)
	Buchan Ness to Collieston Coast	0.927 (0.045)	0.725 (0.111)
Razorbill	Forth Islands	0.910 (0.058)	0.631 (0.080)
	St Abb's Head	0.910 (0.058)	0.631 (0.080)
	Fowlsheugh	0.910 (0.058)	0.631 (0.080)
Shag	Forth Islands	0.847 (0.145)	1.163 (0.823)
	St Abb's Head	0.847 (0.145)	1.410 (0.909)

Table A1.3

Kittiwake breeding population sizes used in population models for each SPA. Values represent number of breeding pairs.

SPA	Forth Islands	Forth Islands	Forth Islands	Forth Islands	Forth Islands	St Abbs to Fast Castle SPA	Fowlsheugh SPA	Buchan Ness to Collieston Coast SPA
Site	**Bass Rock**	**Craigleith**	**Fidra**	**Isle of May**	**The Lamb**	**St Abb's Head NNR**	**Fowlsheugh**	**Boddam to Collieston**
1981				6115				
1982								
1983								
1984				6012				
1985				5510				
1986		725	532	4801	167	13940	22051	19498
1987	2400		726	6765	214	15182		
1988		770	610	7638	175	16200		
1989		840	705	7564	250	19066		
1990		850	598	8129	187	17642		
1991			494	6535	106	16183	23522	
1992			489	6916	223	16524	34872	
1993		1028	452	7009	84	15268		
1994		564	330	3751	160	13007		
1995		951	435	7603	210	13670		24957
1996	2142	509	314	6269	143	13437		
1997	3044	714	298	6518	119	13393		
1998			243	4306		8044		
1999	1307	511	225	4196	115	9576	18800	
2000	1000	539	343	4618	132	11077		
2001	670	440	243	3639	117	8028		14091
2002	774	383	315	3666	139	8890		
2003	910	450	273	3335	124	6642		
2004	660	501	217	3876	126	6239		13330
2005	563	492	257	3790	94	7239		
2006	505	444	275	3167	202	6288	11140	
2007	377	508	244	3424	96	6463		12542
2008	323	513	222	3354	110	5298		
2009	425	594	237	2316	82	4616	9454	
2010	440	600	232	3422	133	4744		
2011	313	542	204	2685	140	4688		
2012	395	620	191	2465	95	4314	9388	
2013	270	293	128	1712	47	3403		
2014	324	300	167	2464	84	3625		
2015	441	537	275	3433	99	4209	9655	
2016	325	468	259	2912	101	2779		

Table A1.4a

Guillemot breeding population sizes used in population models for Forth Islands SPA. Values represent number of breeding pairs. Counts of individuals were converted to pairs using k-values from the Isle of May (Harris *et al.* 2015a, updated). Count type WCC = whole colony count.

SPA	Forth Islands SPA	Forth Islands SPA	Forth Islands SPA	Forth Islands SPA	Forth Islands SPA
Site	**Bass Rock**	**Craigleith**	**Fidra**	**Isle of May**	**The Lamb**
Count type	**WCC**	**WCC**	**WCC**	**WCC**	**WCC**
1981				11250	
1982					
1983				14750	
1984				13000	
1985				13000	
1986		1404	126	13700	1967
1987	1797		53	11680	572
1988		969	88	11223	1604
1989		1181	101	12736	2502
1990		1167	67	12632	1807
1991			134	11440	1631
1992			161	11511	2136
1993		981	143	12418	2287
1994		1400	219	13843	2309
1995		1263	172	15326	1887
1996	1911	1112	153	14500	2163
1997	2682	507	173	17340	2829
1998			207	17384	2063
1999	1890	1333	293	16933	2935
2000	2373	1913	427	17979	1677
2001	2395	2087	448	18442	1431
2002	2452	1291	506	20185	820
2003	2057	1546	434	19519	1449
2004	1966	1549	492	20332	1517
2005	1547	1208	583	18858	1313
2006	2346	1215	333	15578	1268
2007	1030	1058	541	15536	1283
2008	1402	1347	353	15036	2541
2009	2136	1512	439	14143	1842
2010	1329	919	429	15029	1806
2011	1906	1625	316	14955	1944
2012	1328	1371		14100	
2013	1546	1347	372	13349	2224
2014	1759	2498	550	14248	2403
2015	2385	2254	467	15945	2289
2016	1562	1798	325	16132	2150

Table A1.4b

Guillemot breeding population sizes used in population models for St Abbs Head to Fast Castle SPA, Fowsheugh SPA and Buchan Ness to Collieston Coast SPA. Values represent number of breeding pairs. Counts of individuals were converted to pairs using k-values from the Isle of May (Harris *et al.* 2015a, updated). Count type WCC = whole colony count; PC = mean of plot means.

SPA	**St Abbs to Fast Castle SPA**	**St Abbs to Fast Castle SPA**	**Fowlsheugh SPA**	**Fowlsheugh SPA**	**Buchan Ness to Collieston Coast SPA**	**Buchan Ness to Collieston Coast SPA**
Site	**St Abb's Head NNR**	**St Abb's Head NNR**	**Fowlsheugh**	**Fowlsheugh**	**Boddam to Collieston**	**Boddam to Collieston**
Count type	**WCC**	**PC**	**WCC**	**PC**	**WCC**	**PC**
1981						
1982						
1983						
1984		142		198		
1985		119		209		
1986	16443	157	37453	173	9225	
1987	17775	156		208		
1988	18667	143		194		
1989	21394	165		232		
1990	21790	172		206		
1991		174				
1992		167	39381	240		126
1993	20036	180		217		
1994		190		216		
1995		199		237	16602	137
1996		177		244		
1997		240		244		
1998	26254	219		234		148
1999		232	48651	295		
2000		272		234		
2001		248		253	19286	185
2002		318				
2003	29502	264		296		
2004		255		300		202
2005		283		243		
2006		238	39370	216		
2007		270		225	17876	153
2008	29079	252		305		
2009		304	42339	244		
2010		229		240		163
2011		299		285		
2012		232	37277	233		
2013	29828	253		221		158
2014		265		199		
2015		223	40979	236		
2016		236		201		194

Table A1.5a

Razorbill breeding population sizes used in population models for Forth Islands SPA. Values represent number of breeding pairs. Counts of individuals were converted to pairs using k-values from the Isle of May (Harris *et al.* 2015b, updated). Count type WCC = whole colony count. Unrealistic k-values were recorded in 2005 so population counts were excluded.

SPA	Forth Islands SPA	Forth Islands SPA	Forth Islands SPA	Forth Islands SPA	Forth Islands SPA
Site	**Bass Rock**	**Craigleith**	**Fidra**	**Isle of May**	**The Lamb**
count type	**WCC**	**WCC**	**WCC**	**WCC**	**WCC**
1988		79	120	1903	26
1989		74	91	2075	33
1990		38	48	1508	21
1991		70	79	1425	28
1992		34	53	1909	30
1993		41	44	2052	9
1994		56	62	2227	26
1995		79	59	3108	34
1996	165	64	65	2989	64
1997	138	66	81	2719	19
1998			86	3126	
1999	71	114	147	3429	92
2000	65	157	86	3105	68
2001	111	111	72	3346	78
2002	180	131	111	2844	90
2003	64	117	63	2233	81
2004	128	138	82	2677	85
2005					
2006	169	175	123	2975	62
2007	119	181	128	2735	77
2008	85	147	95	2591	80
2009	70	117	127	2400	70
2010	63	136	123	2557	42
2011	94	185	108	2705	70
2012	106	157	70	3068	66
2013	105	129	109	2879	59
2014	124	110	170	2987	65
2015	144	193	139	3202	46
2016	91	186	122	3570	82

Table A1.5b

Razorbill breeding population sizes used in population models for St Abbs Head to Fast Castle SPA and Fowsheugh SPA. Values represent number of breeding pairs. Counts of individuals were converted to pairs using k-values from the Isle of May (Harris *et al.* 2015b, updated). Count type WCC = whole colony count; PC = mean of plot means.

SPA	**St Abbs to Fast Castle SPA**	**St Abbs to Fast Castle SPA**	**Fowlsheugh SPA**	**Fowlsheugh SPA**
Site	**St Abb's Head**	**St Abb's Head NNR**	**Fowlsheugh**	**Fowlsheugh**
count type	**WCC**	**PC**	**WCC**	**PC**
1988	1343	21		
1989	1398	23		
1990	1072	18		
1991		29		
1992		24	6827	
1993	1187	21		
1994		25		
1995		29		
1996		23		
1997		33		
1998	1793	29		
1999		28	5808	
2000		30		
2001		26		
2002		32		
2003	1595	20		
2004		15		9
2005		29		
2006		20	3341	20
2007		21		19
2008	1262	18		
2009		23	3696	18
2010		18		14
2011		24		
2012		23	4883	21
2013	1269	22		14
2014		20		18
2015		16	5180	22
2016		18		20

Table A1.6

Shag breeding population sizes used in population models for each SPA. Values represent number of breeding pairs.

SPA	Forth Islands SPA	Forth Islands SPA	Forth Islands SPA	Forth Islands SPA	Forth Islands SPA	Forth Islands SPA	St Abbs to Fast Castle SPA	Buchan Ness to Collieston Coast SPA
Site	**Bass Rock**	**Craig-leith**	**Fidra**	**Inch-mickery**	**Isle of May**	**The Lamb**	**St Abb's Head NNR**	**Boddam to Collieston**
1973		164	17		1076	244		
1974		225	27		933	255		
1975	180	214	25		644	233		
1976	213	201	20	8	497	210	187	
1977	201	186	18	12	921	156	193	
1978	202	208	23	14	769	143	134	
1979	188	215	25	14	966	160		
1980	191	198	25	11	1041	143		
1981	154	252	43	14	1163	220		
1982	194	344	59	22	1425		209	
1983	170	356	66	42	1567	283		
1984	193	379	64	22	1639	284		
1985	101	345	55	29	1524	303	268	
1986	75	388	67	24	1310	301	364	440
1987	162	465	64	24	1916		396	
1988	93	435	86	24	1290	250	318	
1989	111	544	124	29	1703	286	366	
1990	121	522	116	28	1386	290	338	
1991		646	242	33	1487	305	463	
1992		665	255	36	1634	318	450	
1993	20	155	88	28	715	65	300	
1994	13	106	73	10	403	36	115	
1995		171	84	20	503	81	173	223
1996	47	159	81	18	512	77	175	
1997	41	180	107	28	502	65	160	
1998			86	25	621		196	
1999	30	131	61	33	259	76	165	
2000	28	208	123	32	541	46	233	
2001	39	237	139	41	734	99	300	415
2002	25	233	186	52	676	102	296	
2003	24	197	254	70	968	124	365	
2004	46	324	272	78	687	111	369	594
2005	18	131	115	52	281	49	131	
2006	36	118	198	57	485	65	162	
2007	28	199	169	57	399	73	132	331
2008	22	133	146	55	427	97	131	
2009	15	200	159	54	465	75	138	
2010	16	207	204	55	492	114	157	
2011	25	281	191	62	540	66	160	
2012	11	258	172	71	648	77	171	
2013	31	117	153	59	322	44	94	363
2014	12	137	162	65	338	49	107	

Appendix 2

Ratio of Impacted to Un-Impacted 25 Year Population Growth Rate

One possibility for the low sensitivity of PVA metric A (median of the ratio of impacted to un-impacted annual growth rate) is the scale of values, with all values being close to one, and, therefore, sensitivity potentially appearing low in a visual assessment even in cases where it is not. However, here we consider a 25 year growth rate, where lines deviate markedly from 1 and sensitivity is more discernible. This analysis shows that low sensitivity is still apparent (Figure A2.1).

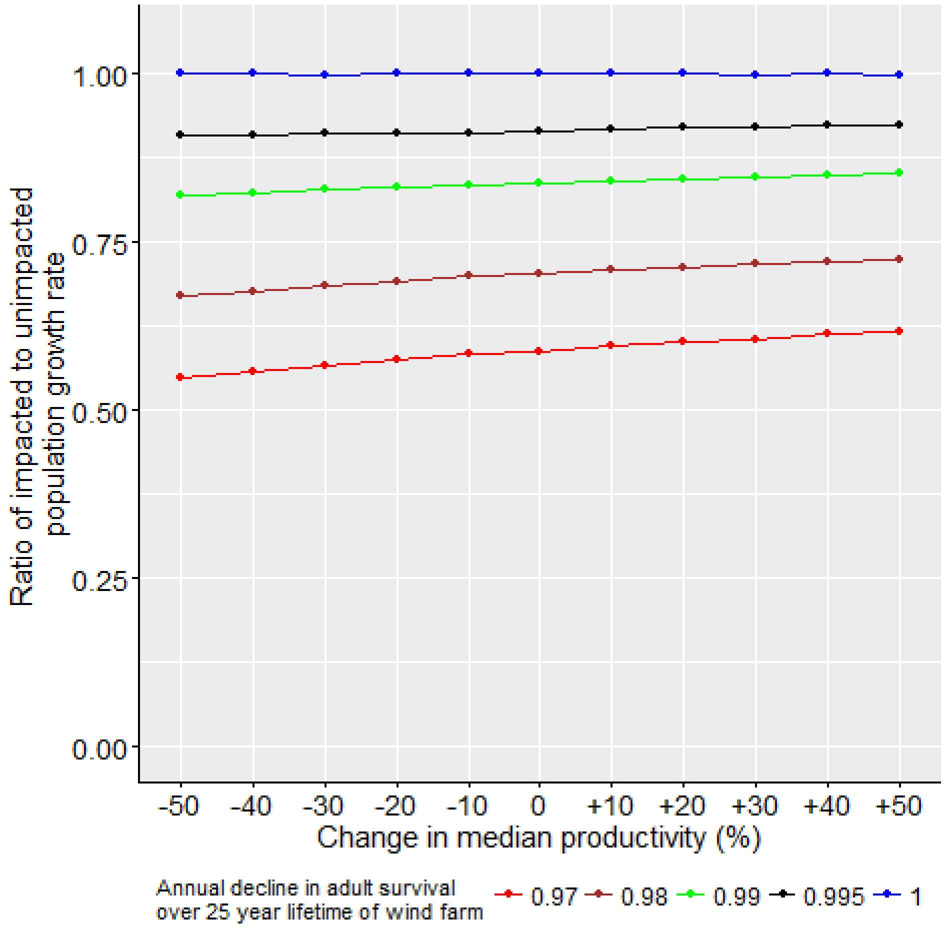


Figure A2.1: PVA Metric A – ratio of 25 year population growth rate, comparing impacted population vs. un-impacted population, showing productivity mis-specification varied from -50% to +50% (with 0% representing no mis-specification) in Forth Islands kittiwakes. The five coloured lines represent the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

Appendix 3

PVA Metric Sensitivity for all Populations

This Appendix presents graphical output of PVA metric sensitivity for the 13 populations considered in this project. For each species, the sequence of figures is as presented in Figure 4 of the main report for Forth Islands kittiwakes. For completeness, we include Forth Islands kittiwakes here.

In all figures, adult mortality mis-specification is illustrated in the upper panels and productivity mis-specification in the lower panels. Mis-specification was varied from -30% to +30% (with 0% representing no mis-specification). The five coloured lines represent the different levels of potential impact on annual productivity (left panels) or annual adult survival (right panels) over the hypothetical 25 year lifetime of the wind farm (2017-2041).

1. **Kittiwakes at Forth Islands SPA**:

Figure A2.1a: PVA Metric A for Forth Kittiwakes – ratio of population growth rate from 2016-2041, comparing impacted population vs. un-impacted population.

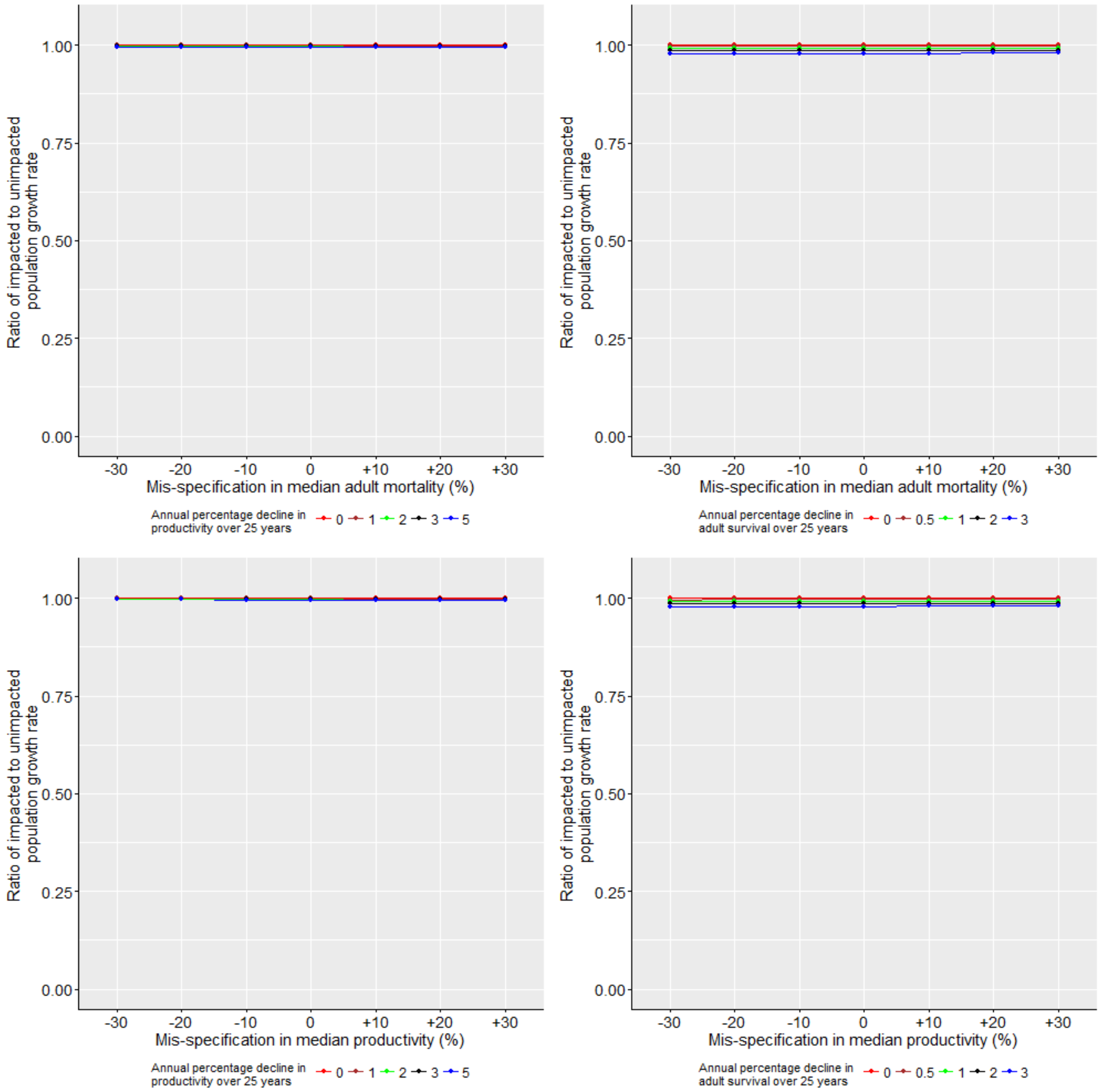


Figure A2.1b: PVA Metric B for Forth Kittiwakes – ratio of population size at 2041, comparing impacted population vs. un-impacted population.

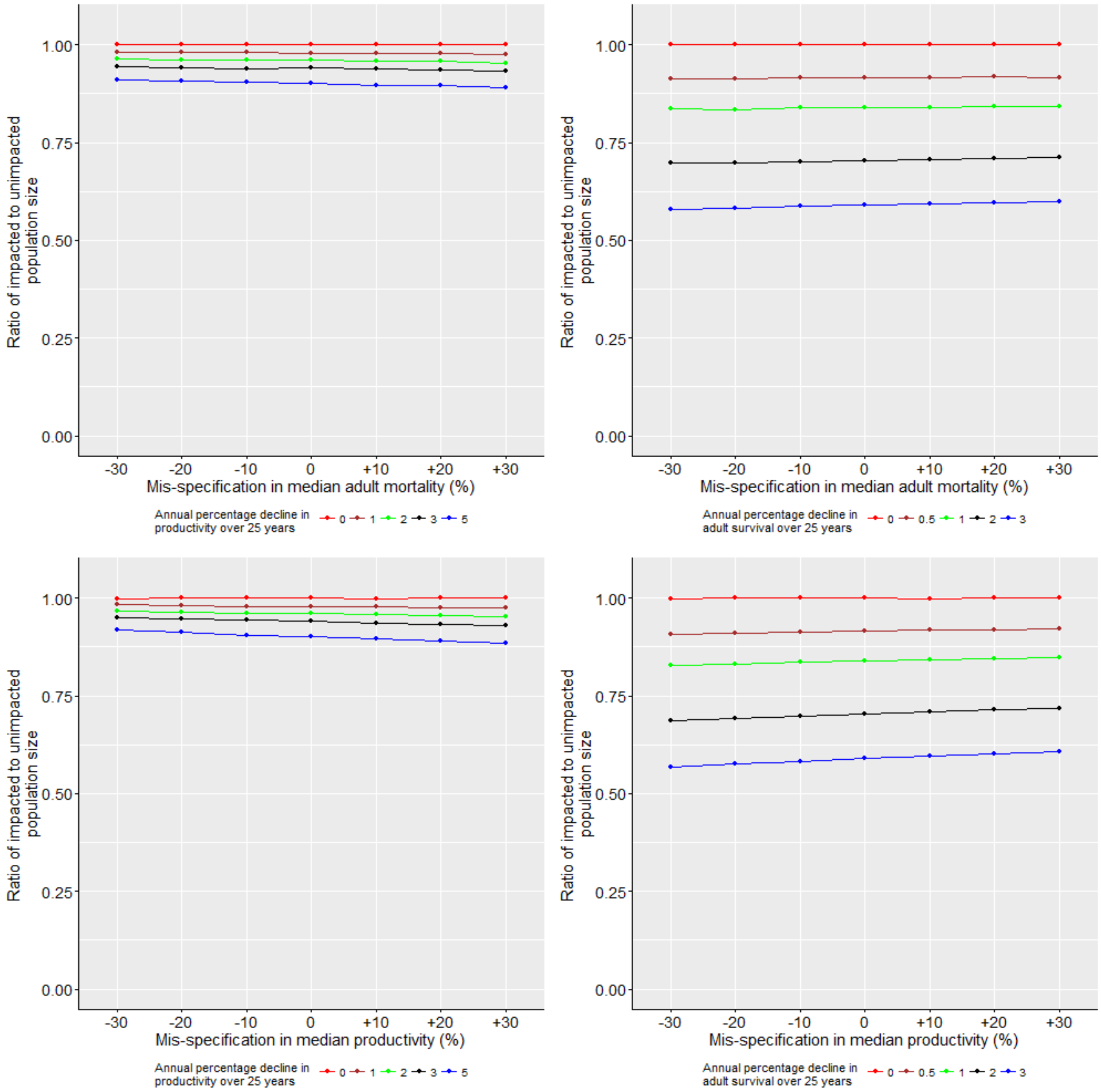


Figure A2.1c: PVA Metric C for Forth Kittiwakes – difference in population growth rate from 2016-2041, comparing impacted population vs. un-impacted population.

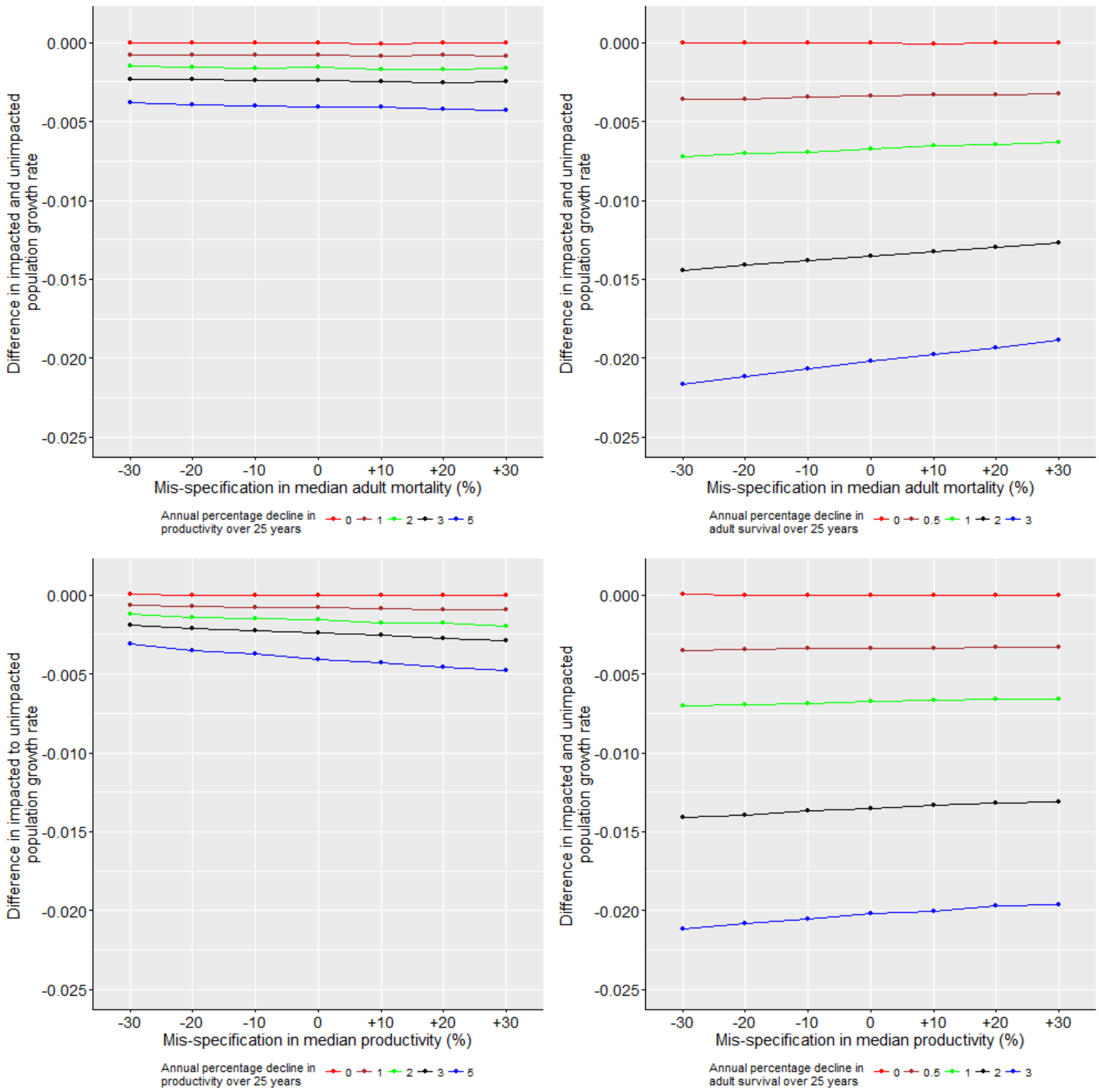


Figure A2.1d: PVA Metric D for Forth Kittiwakes – difference in population size at 2041, comparing impacted population vs. un-impacted population.

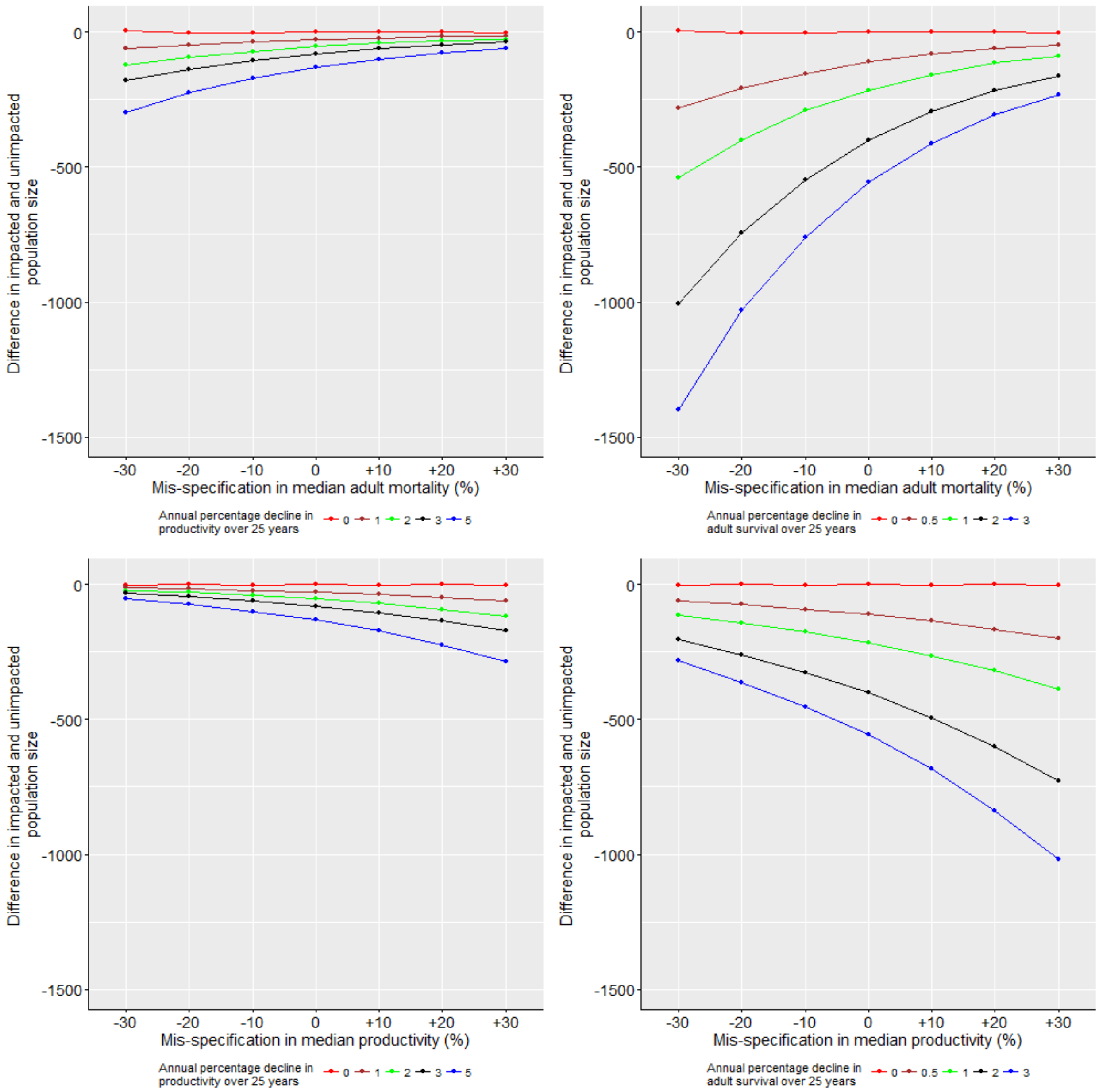


Figure A2.1e: PVA Metric E1 for Forth Kittiwakes – probability of population decline greater than 10% from 2016-2041.

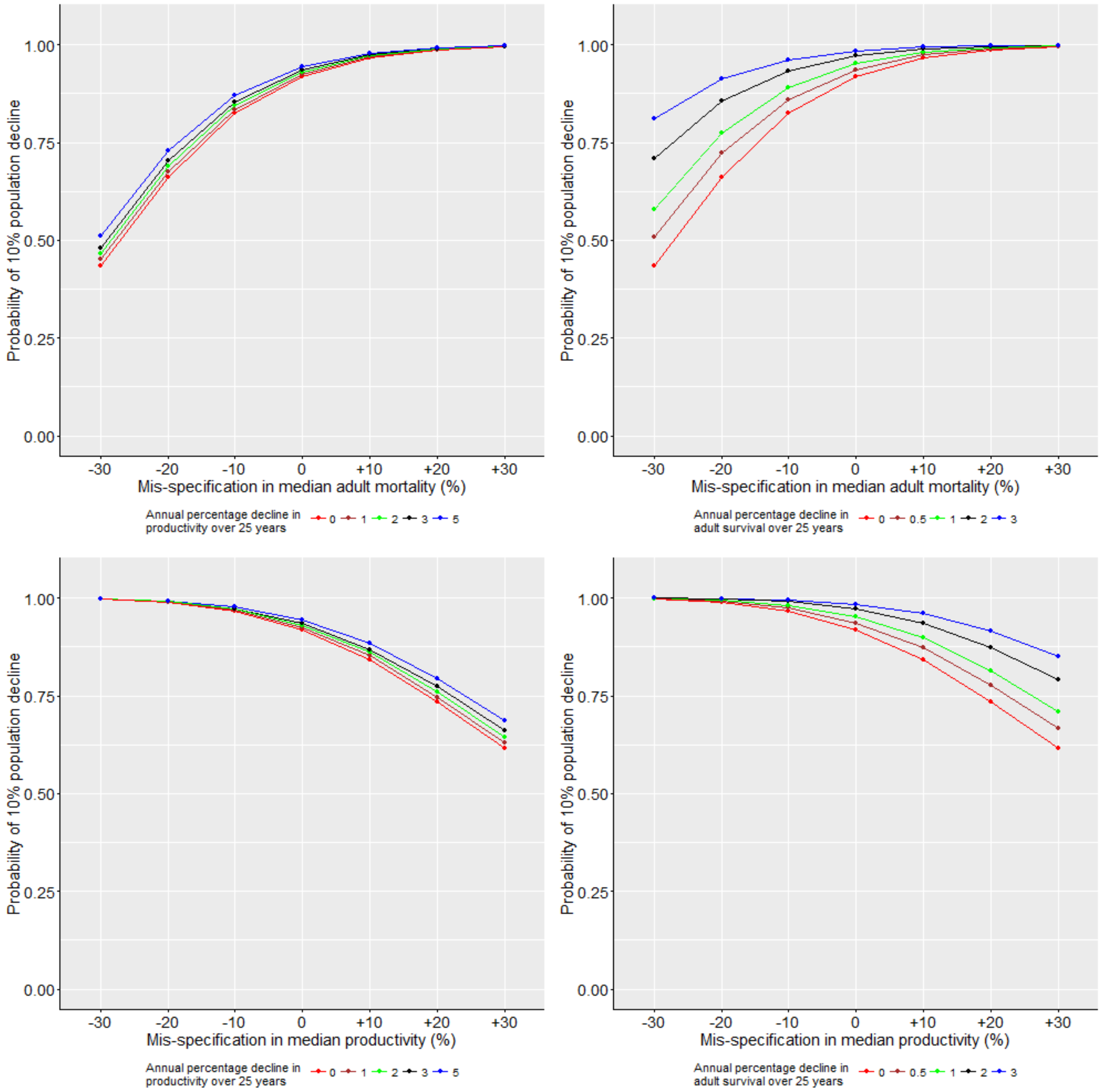


Figure A2.1f: PVA Metric E2 for Forth Kittiwakes – probability of population decline greater than 25% from 2016-2041.

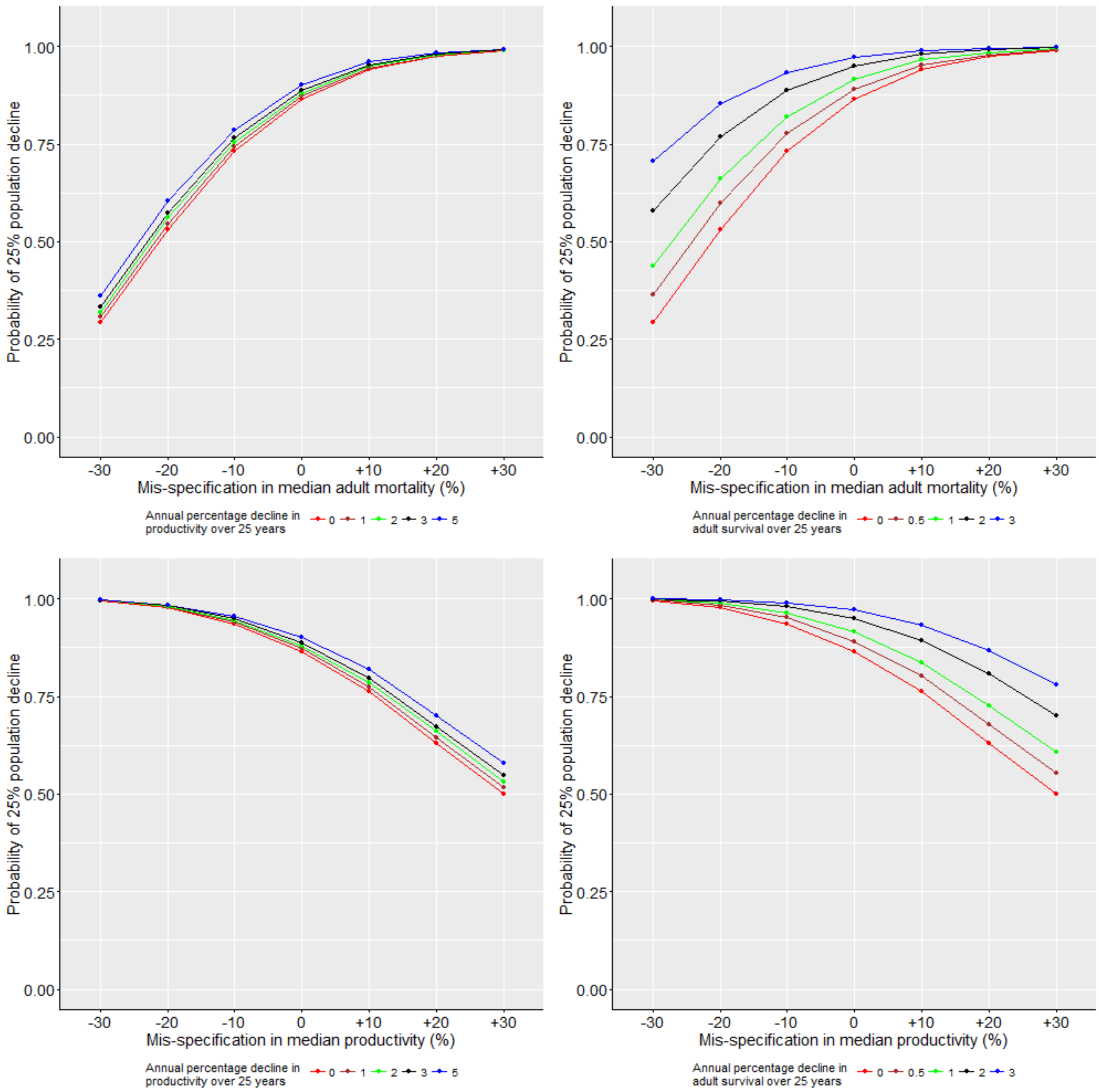


Figure A2.1g: PVA Metric E3 for Forth Kittiwakes – probability of population decline greater than 50% from 2016-2041.

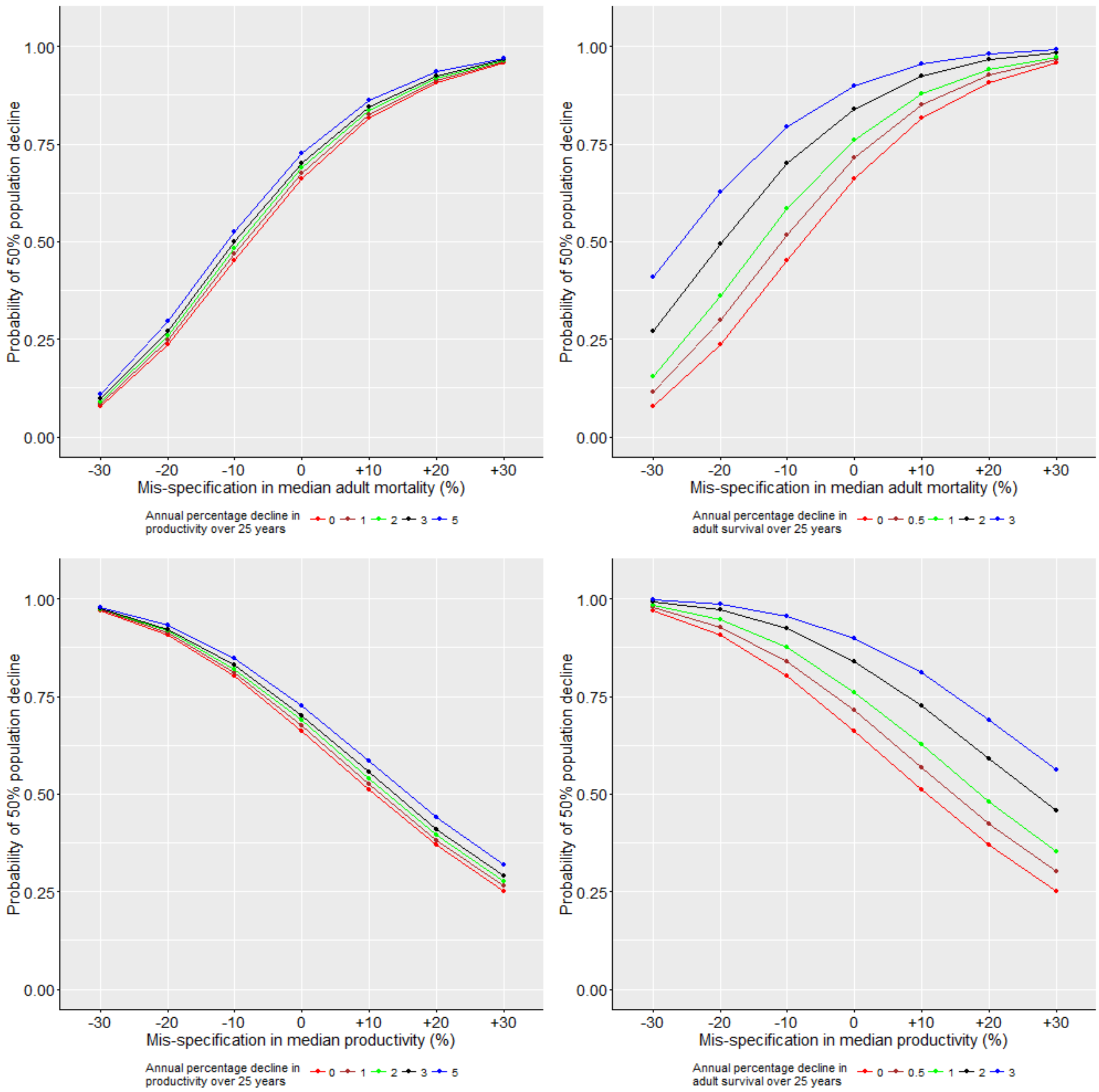
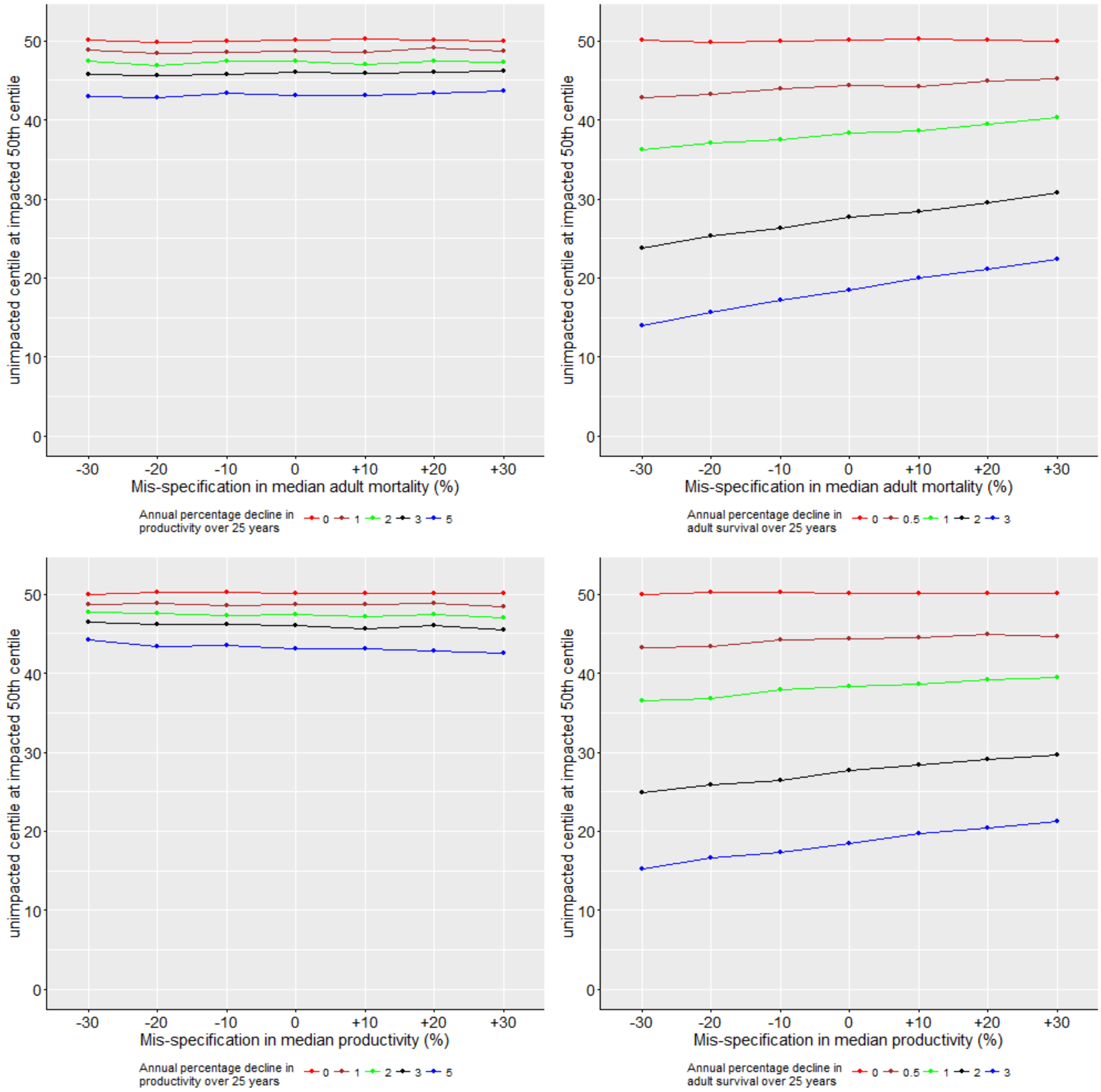


Figure A2.1h: PVA Metric F for Forth Kittiwakes – centile from un-impacted population size equal to the 50th centile of the impacted population size, at 2041.



2. **Kittiwakes at St Abb's Head SPA**:

Figure A2.2a. PVA Metric A for St Abb's Kittiwakes – ratio of population growth rate from 2016-2041, comparing impacted population vs. un-impacted population.

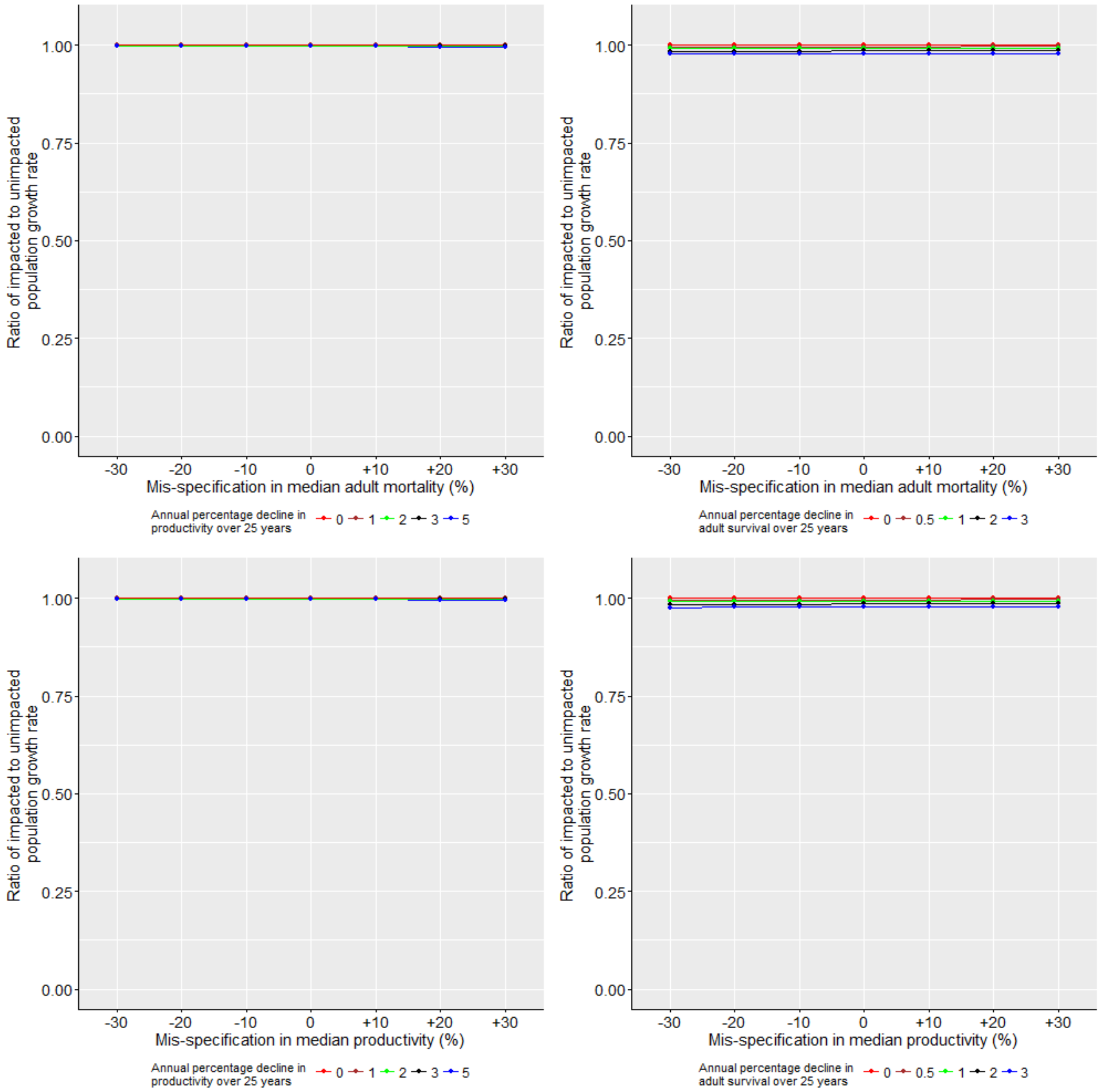


Figure A2.2b. PVA Metric B for St Abb's Kittiwakes – ratio of population size at 2041, comparing impacted population vs. un-impacted population.

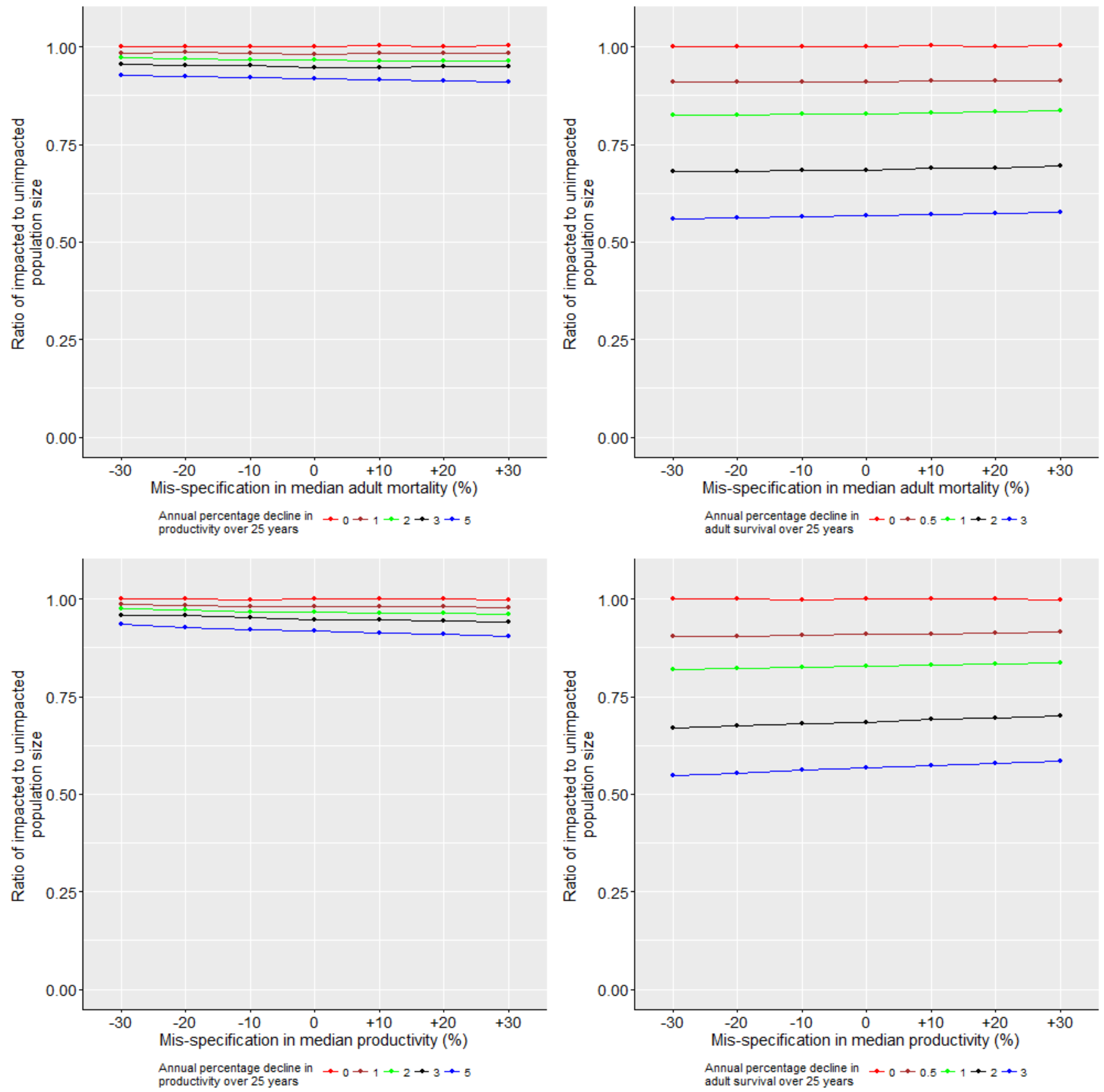


Figure A2.2c. PVA Metric C for St Abb's Kittiwakes – difference in population growth rate from 2016-2041, comparing impacted population vs. un-impacted population.

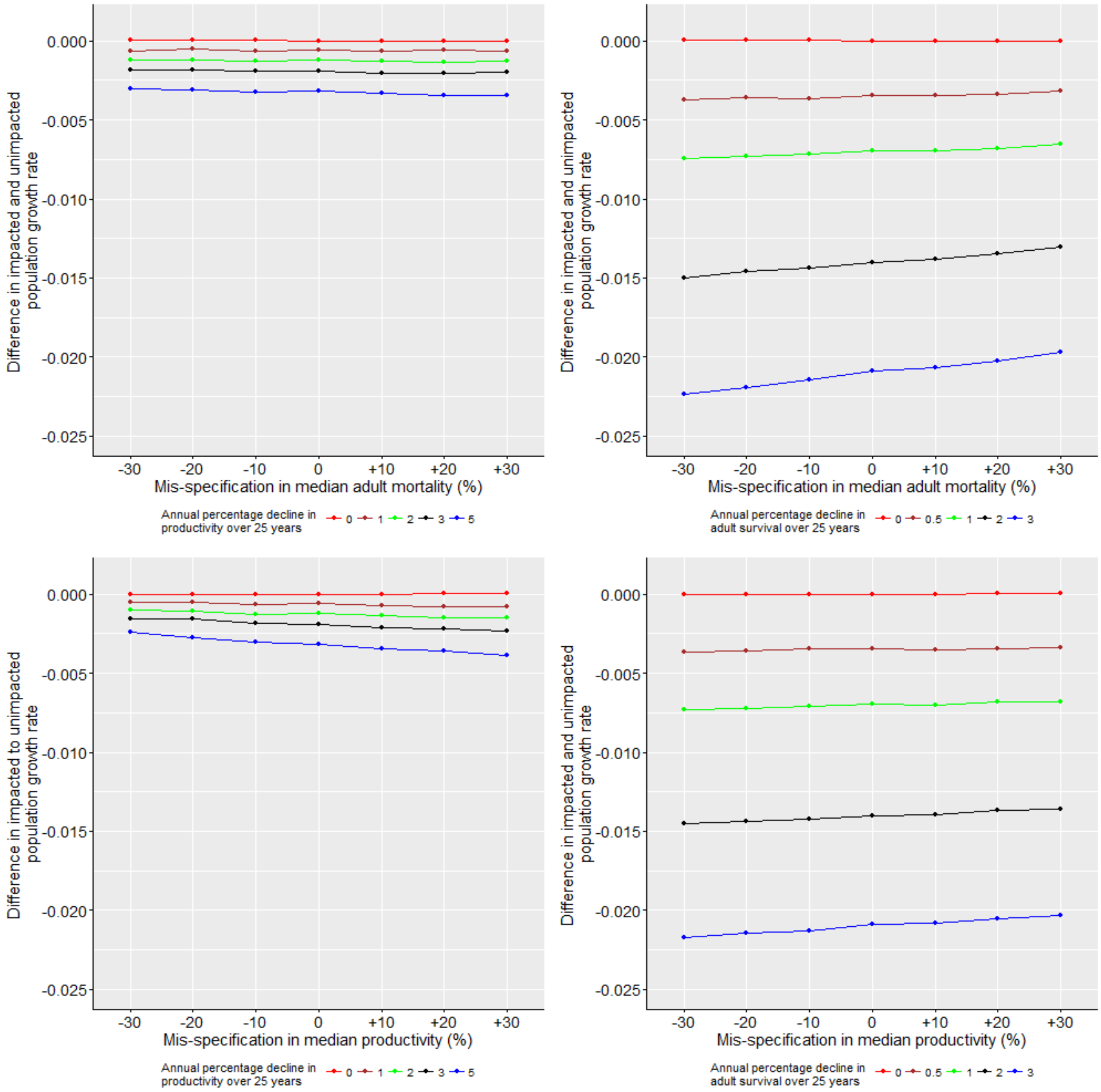


Figure A2.2d. PVA Metric D for St Abb's Kittiwakes – difference in population size at 2041, comparing impacted population vs. un-impacted population.

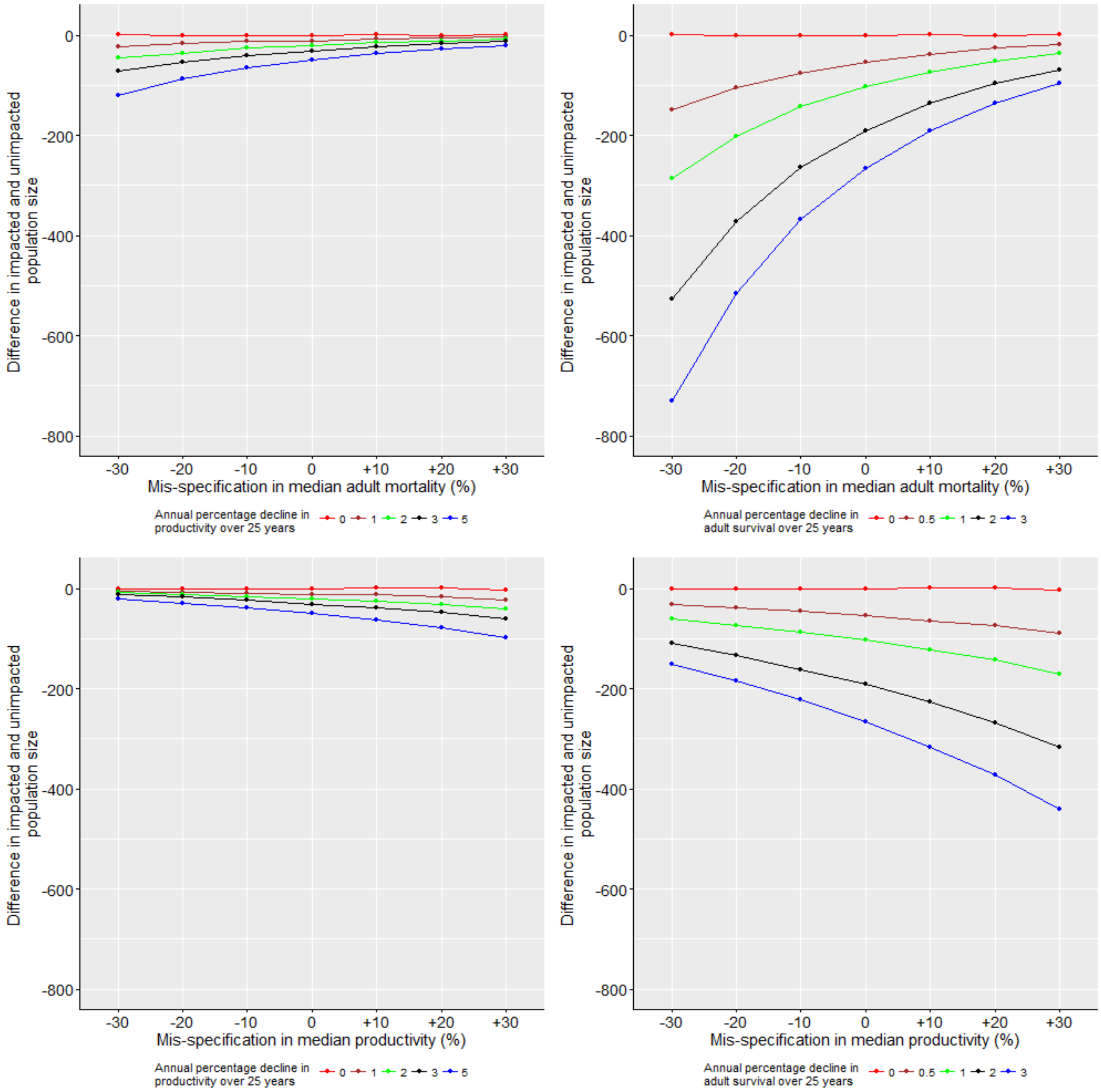


Figure A2.2e. PVA Metric E1 for St Abb's Kittiwakes – probability of population decline greater than 10% from 2016-2041.

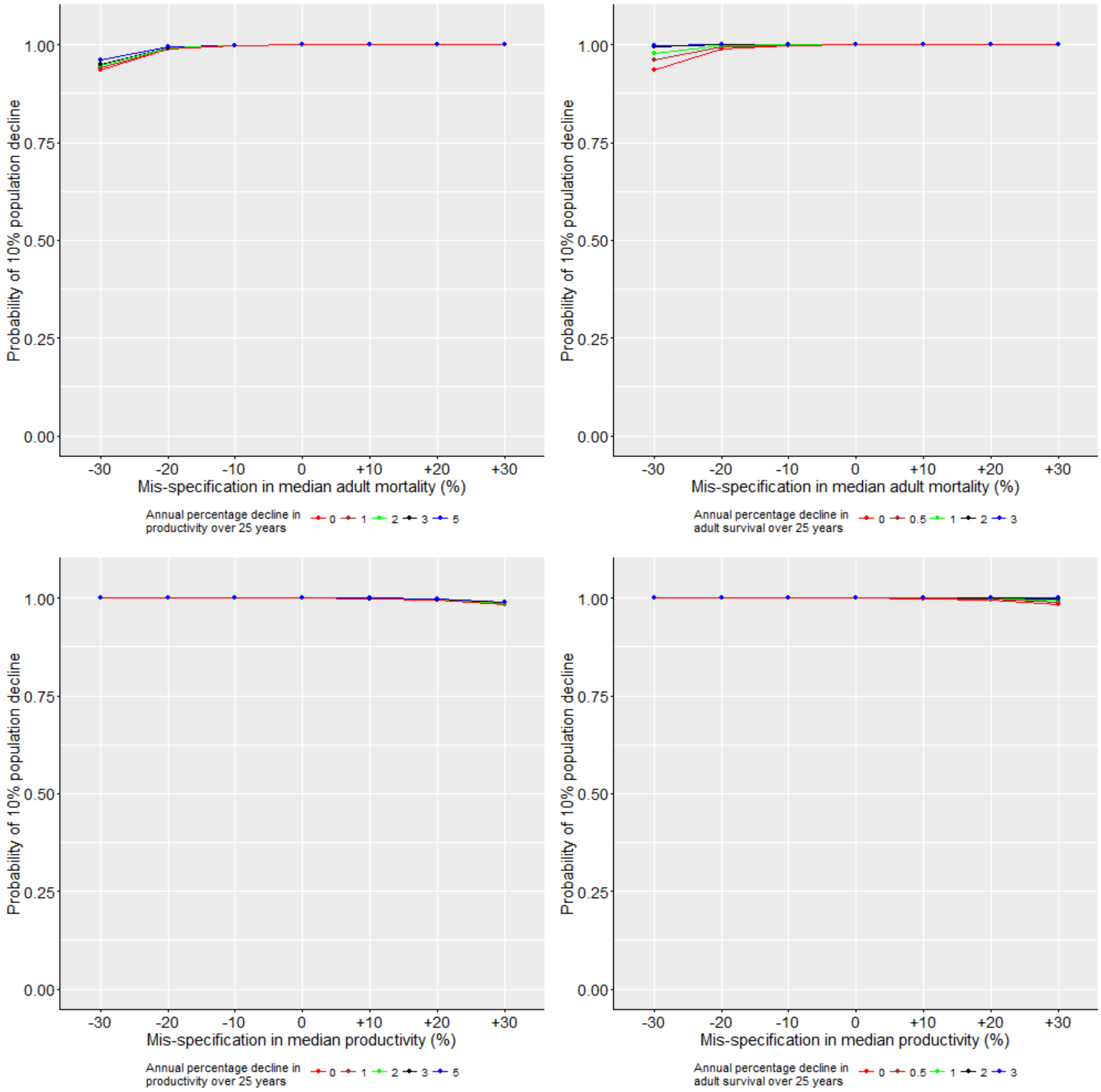


Figure A2.2f. PVA Metric E2 for St Abb's Kittiwakes – probability of population decline greater than 25% from 2016-2041.

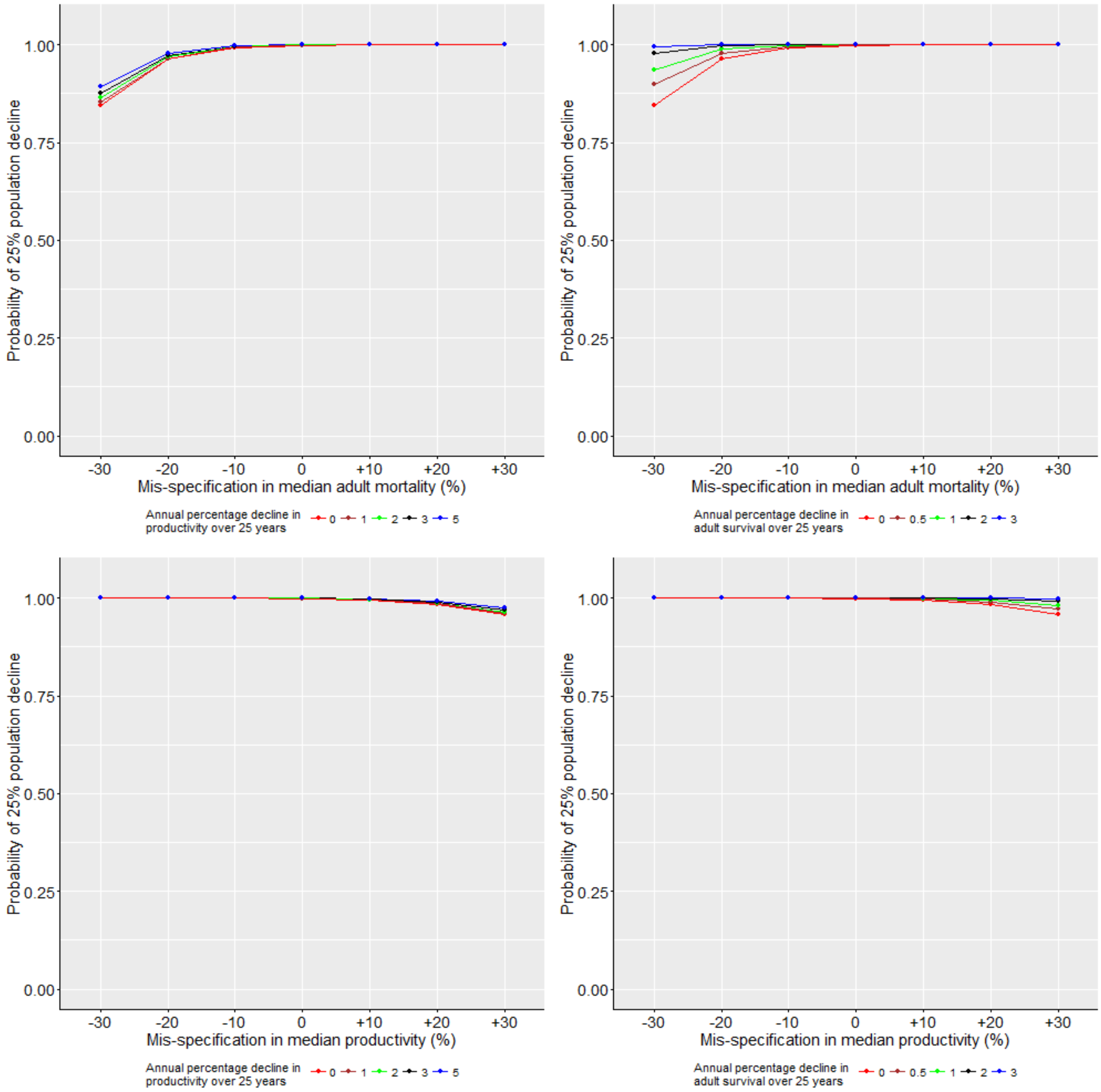


Figure A2.2g. PVA Metric E3 for St Abb's Kittiwakes – probability of population decline greater than 50% from 2016-2041.

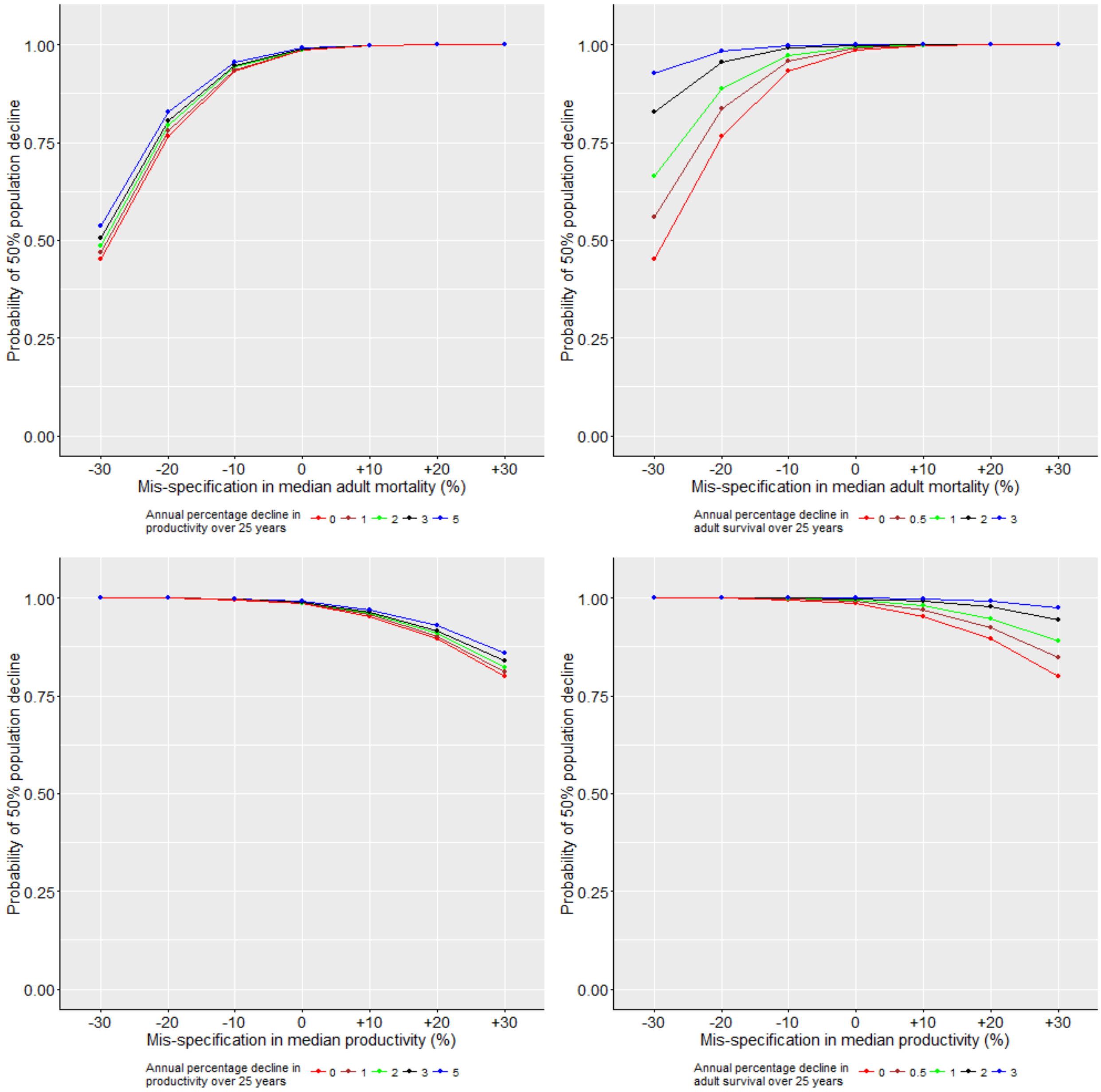
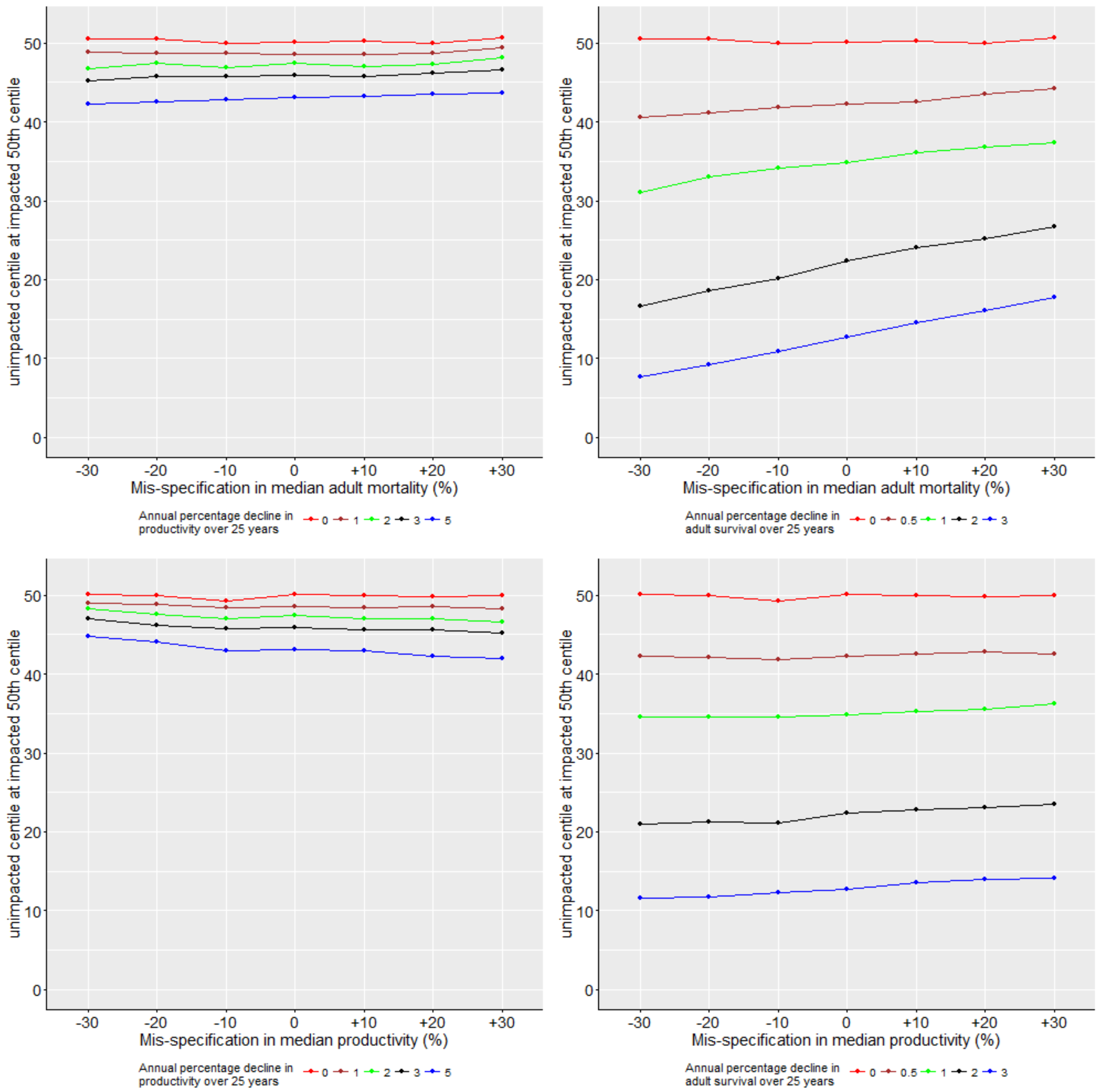


Figure A2.2h. PVA Metric F for St Abb's Kittiwakes – centile from un-impacted population size equal to the 50th centile of the impacted population size, at 2041.



3. **Kittiwakes at Fowlsheugh SPA**:

Figure A2.3a. PVA Metric A for Fowlsheugh Kittiwakes – ratio of population growth rate from 2016-2041, comparing impacted population vs. un-impacted population.

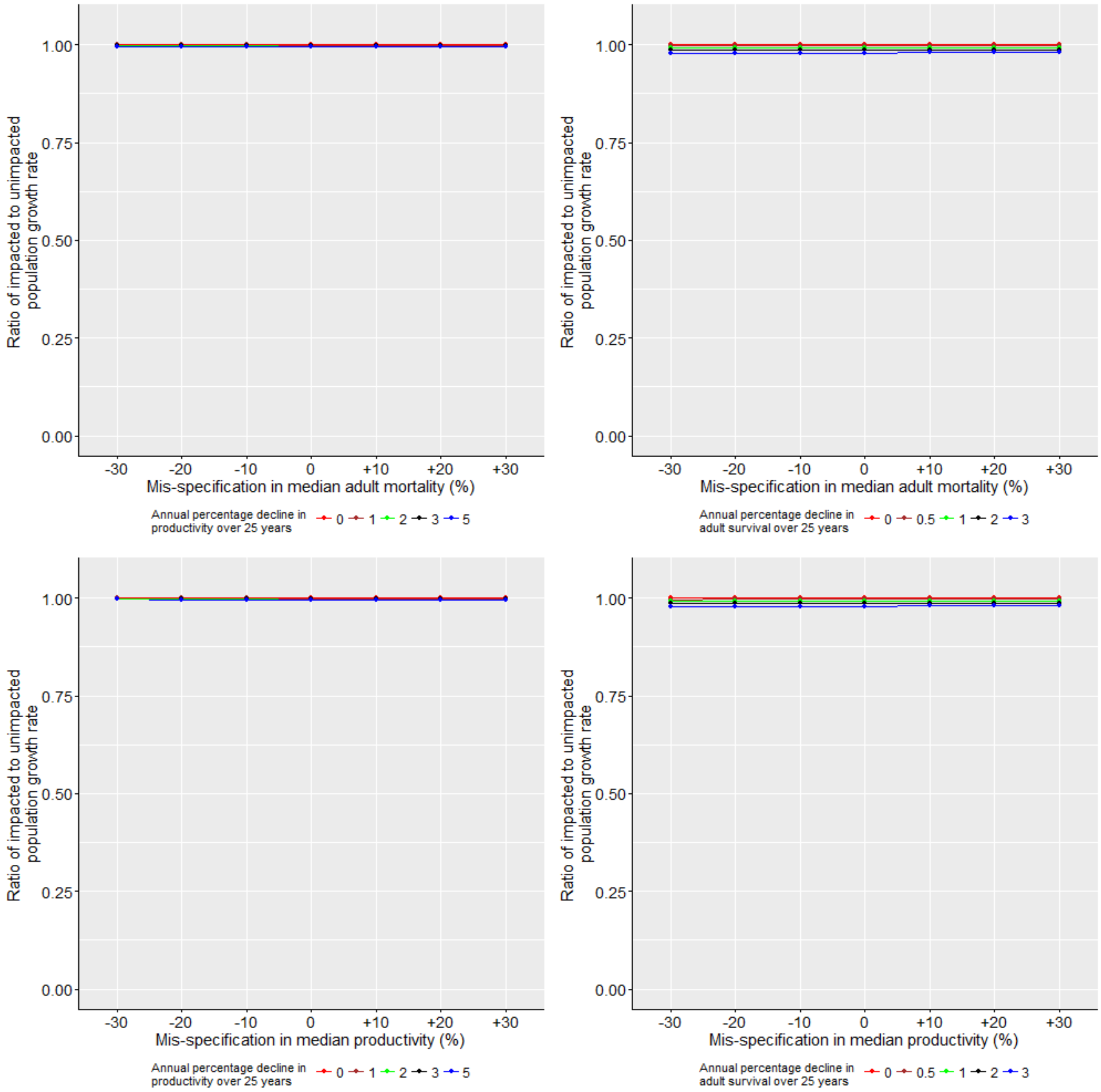


Figure A2.3b. PVA Metric B for Fowlsheugh Kittiwakes – ratio of population size at 2041, comparing impacted population vs. un-impacted population.

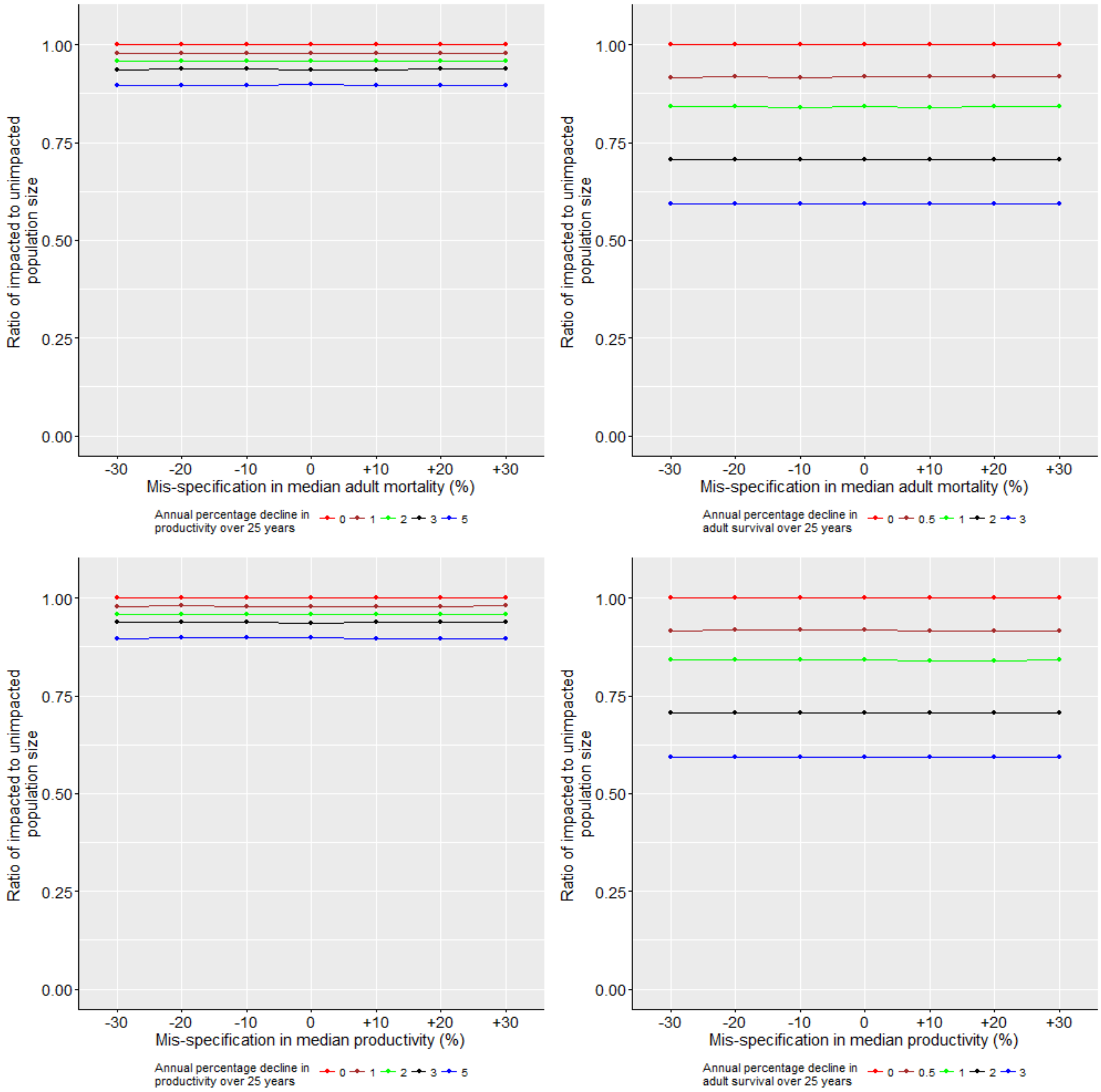


Figure A2.3c. PVA Metric C for Fowlsheugh Kittiwakes – difference in population growth rate from 2016-2041, comparing impacted population vs. un-impacted population.

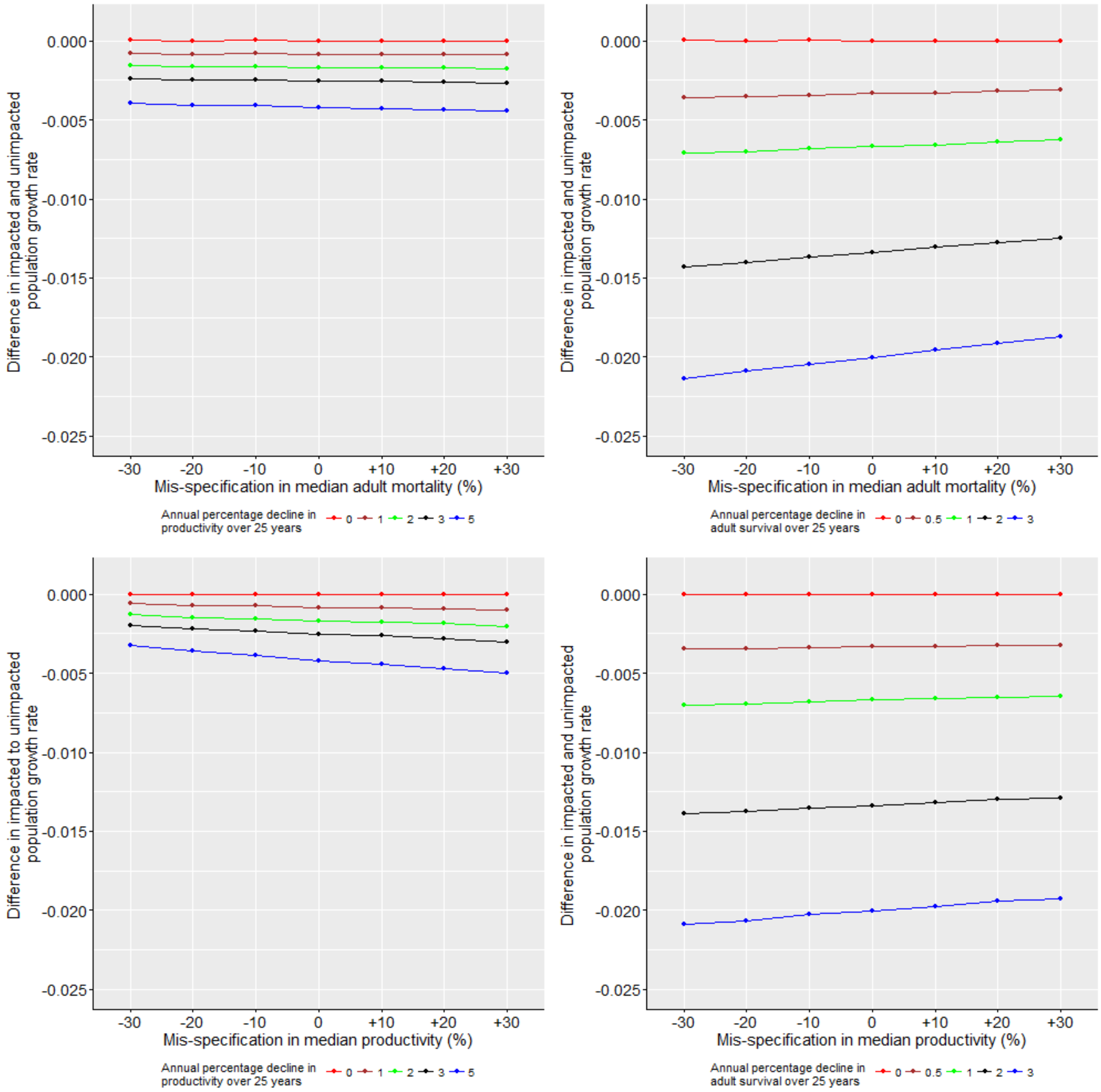


Figure A2.3d. PVA Metric D for Fowlsheugh Kittiwakes – difference in population size at 2041, comparing impacted population vs. un-impacted population.

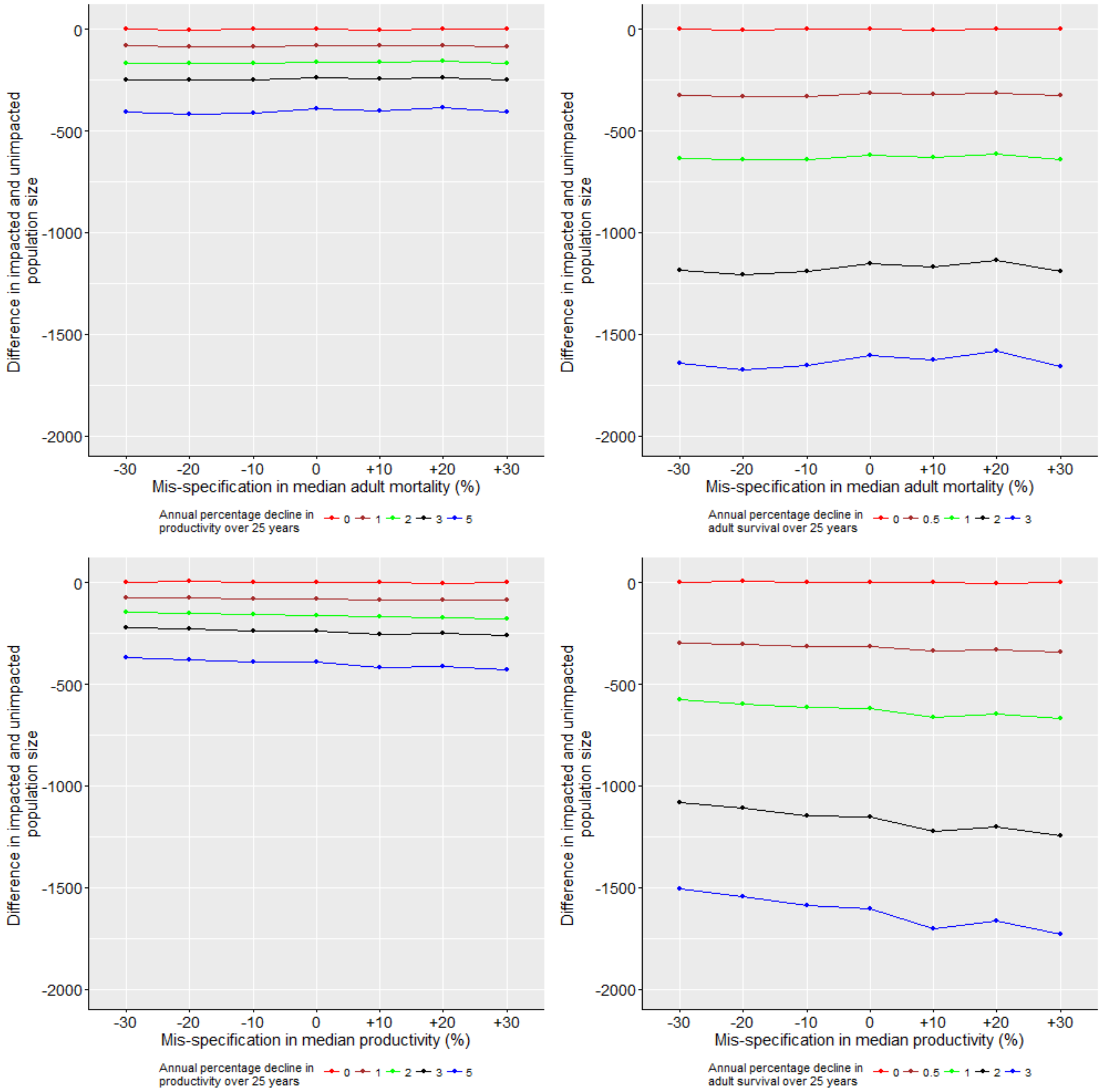


Figure A2.3e. PVA Metric E1 for Fowlsheugh Kittiwakes – probability of population decline greater than 10% from 2016-2041.

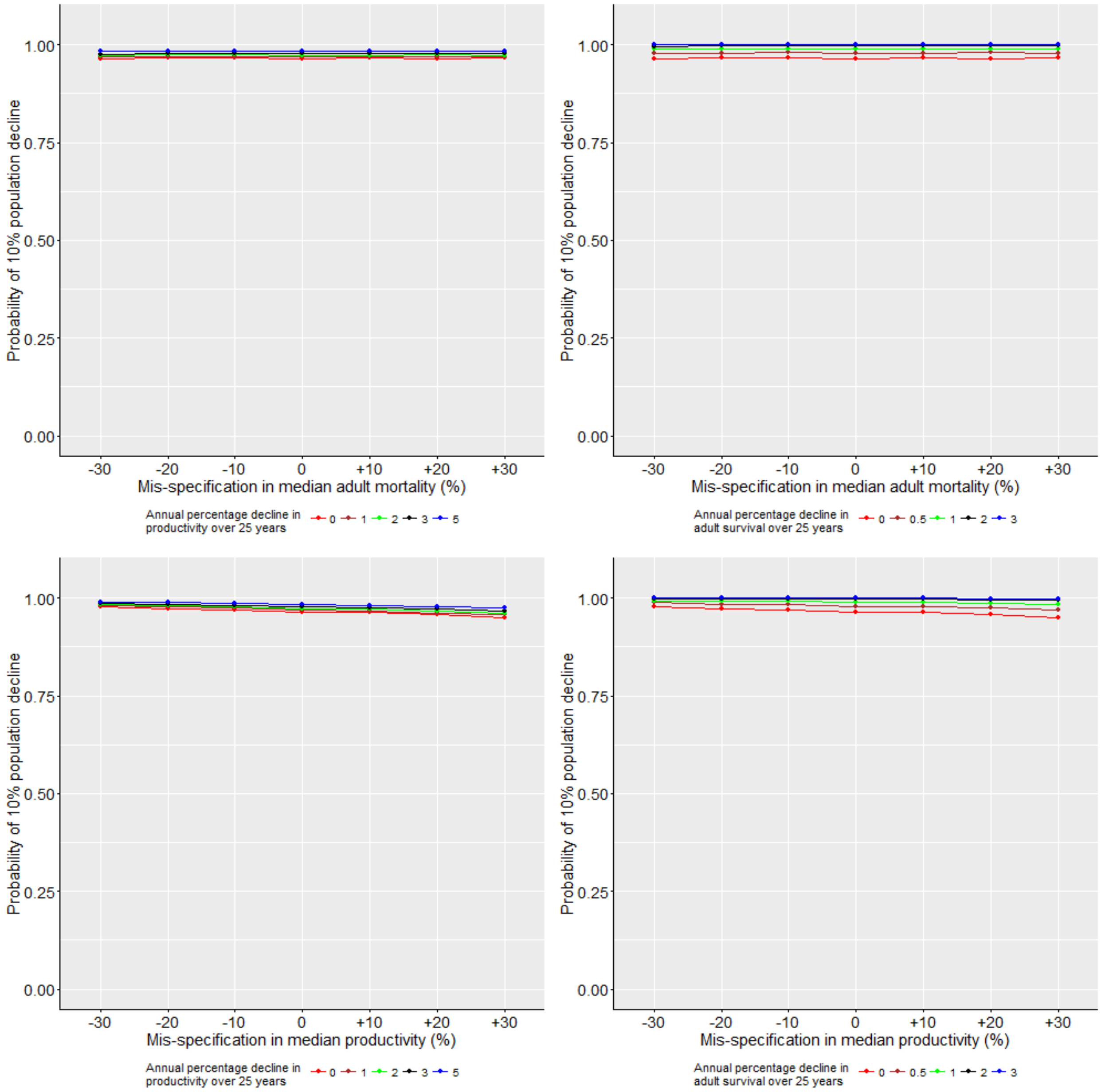


Figure A2.3f. PVA Metric E2 for Fowlsheugh Kittiwakes – probability of population decline greater than 25% from 2016-2041.

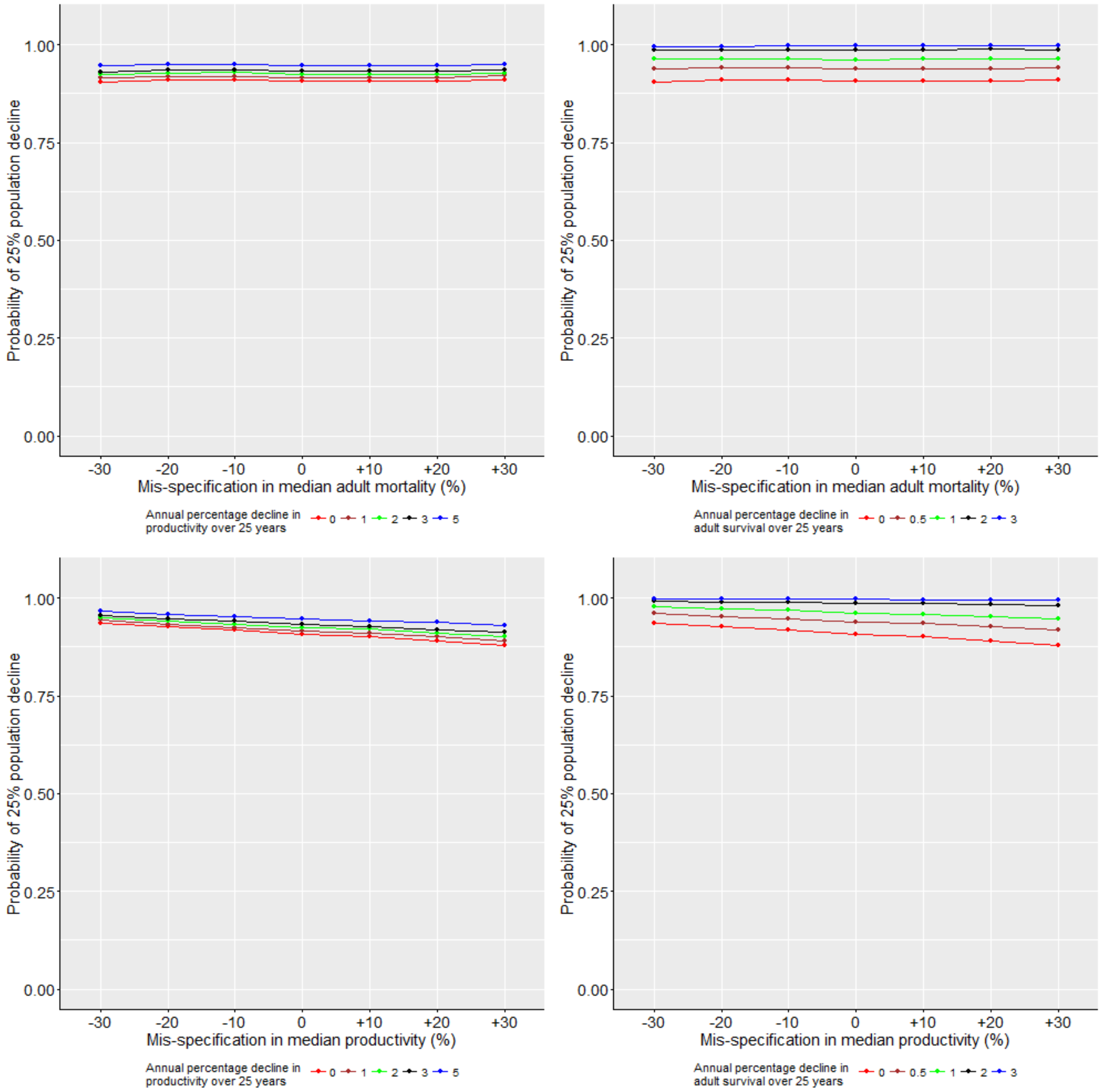


Figure A2.3g. PVA Metric E3 for Fowlsheugh Kittiwakes – probability of population decline greater than 50% from 2016-2041.

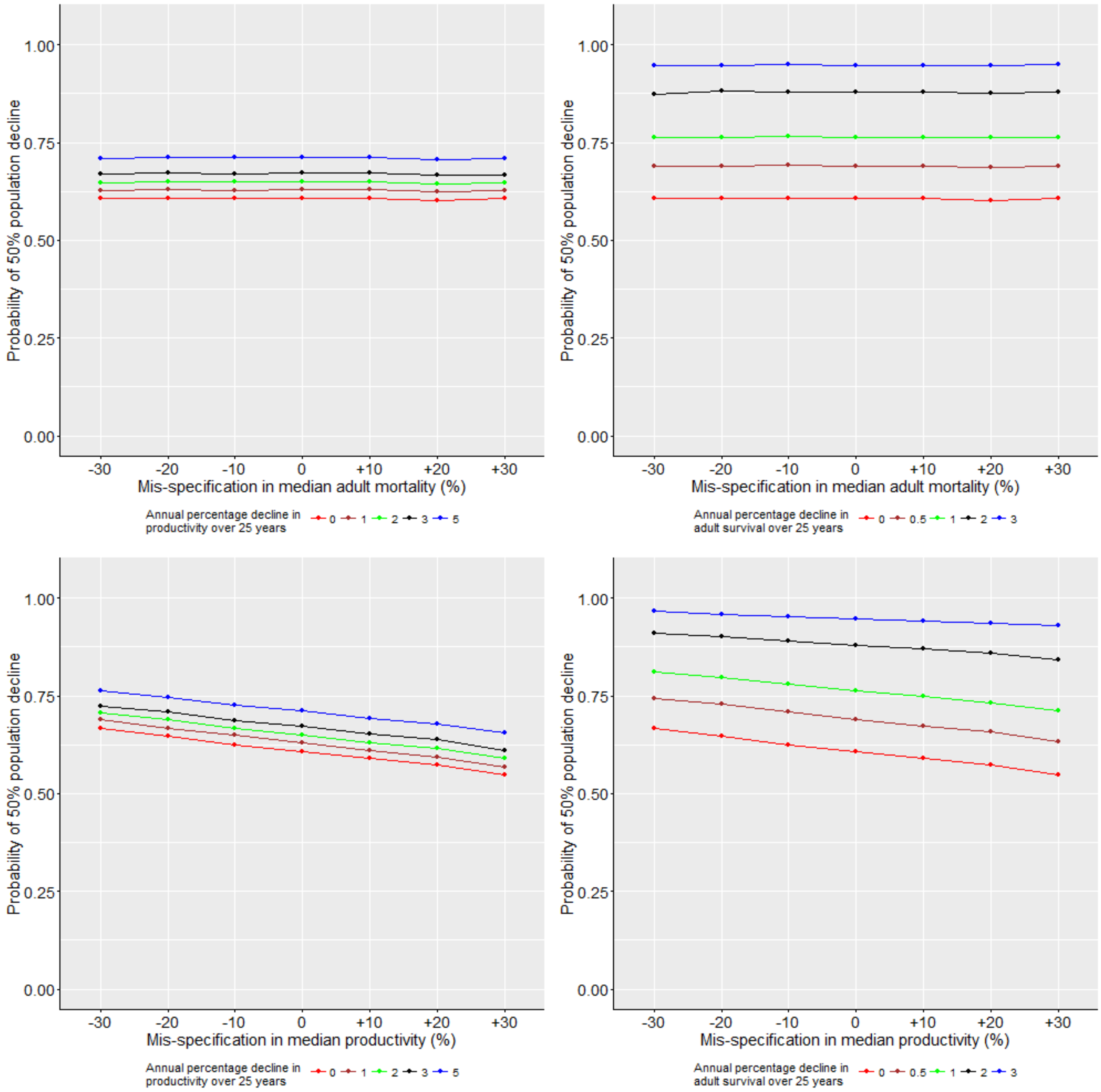
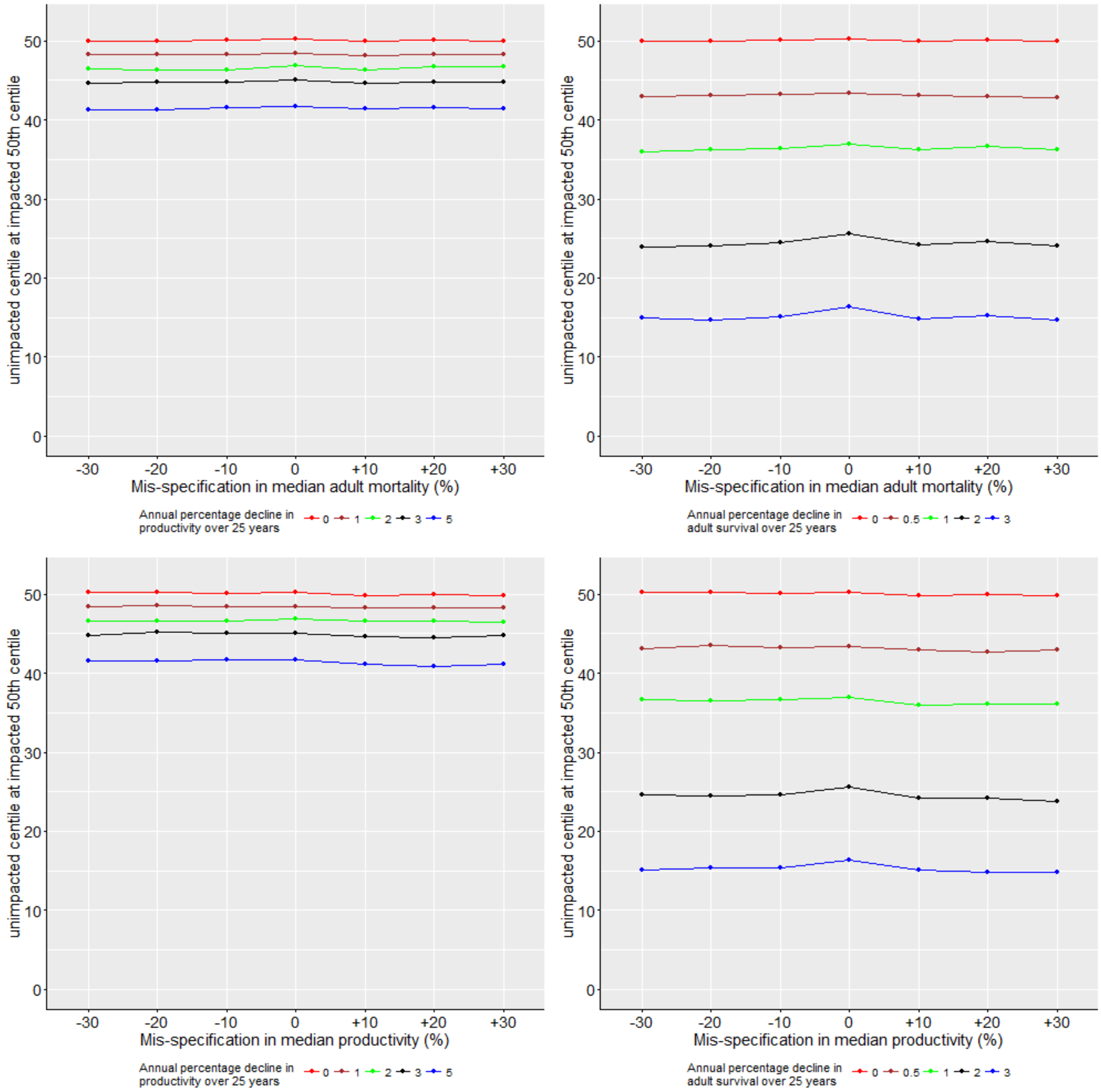


Figure A2.3h. PVA Metric F for Fowlsheugh Kittiwakes – centile from un-impacted population size equal to the 50th centile of the impacted population size, at 2041.



4. **Kittiwakes at Buchan Ness to Collieston Coast SPA**:

Figure A2.4a. PVA Metric A for Buchan Ness Kittiwakes – ratio of population growth rate from 2016-2041, comparing impacted population vs. un-impacted population.

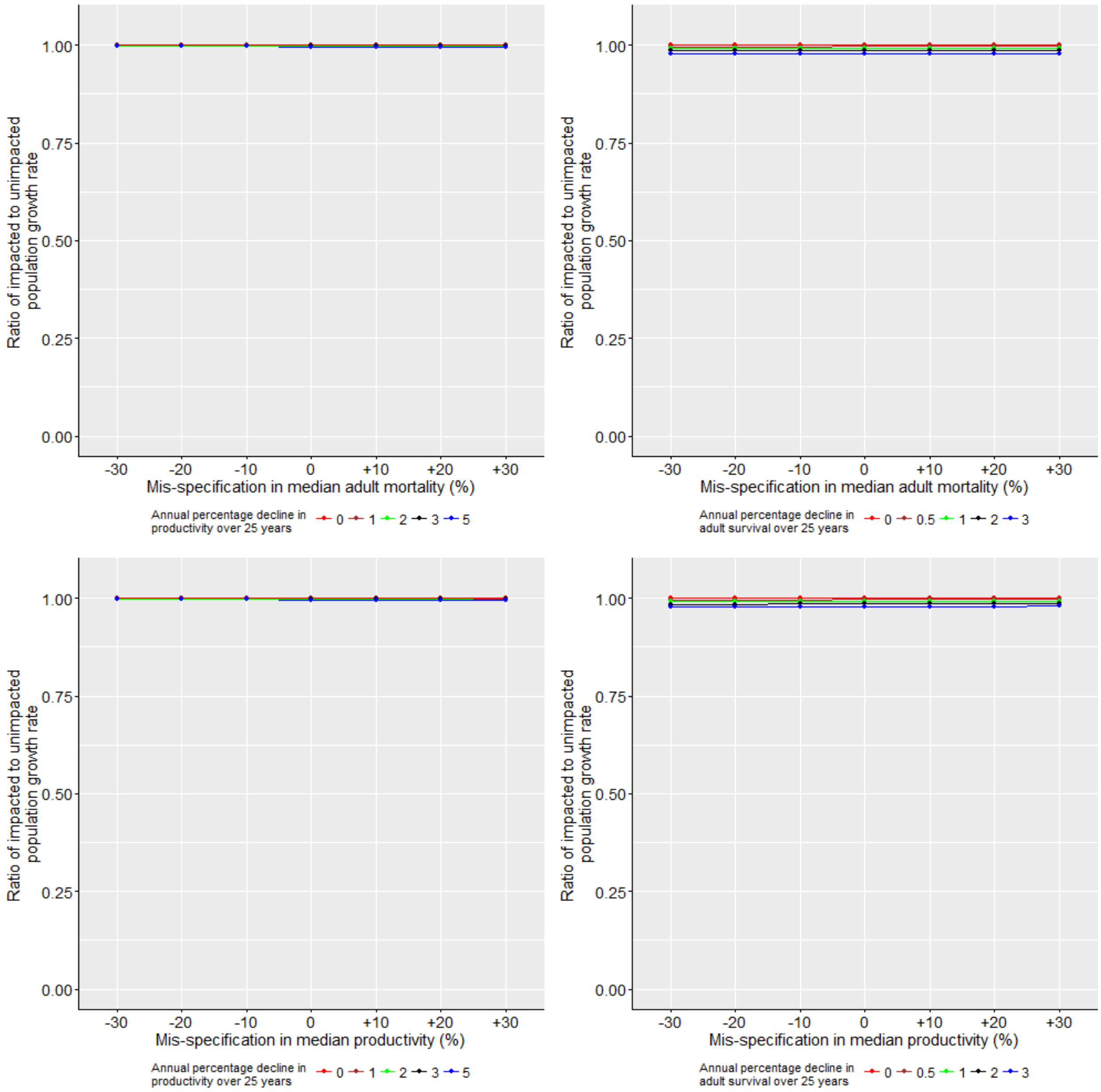


Figure A2.4b. PVA Metric B for Buchan Ness Kittiwakes – ratio of population size at 2041, comparing impacted population vs. un-impacted population.

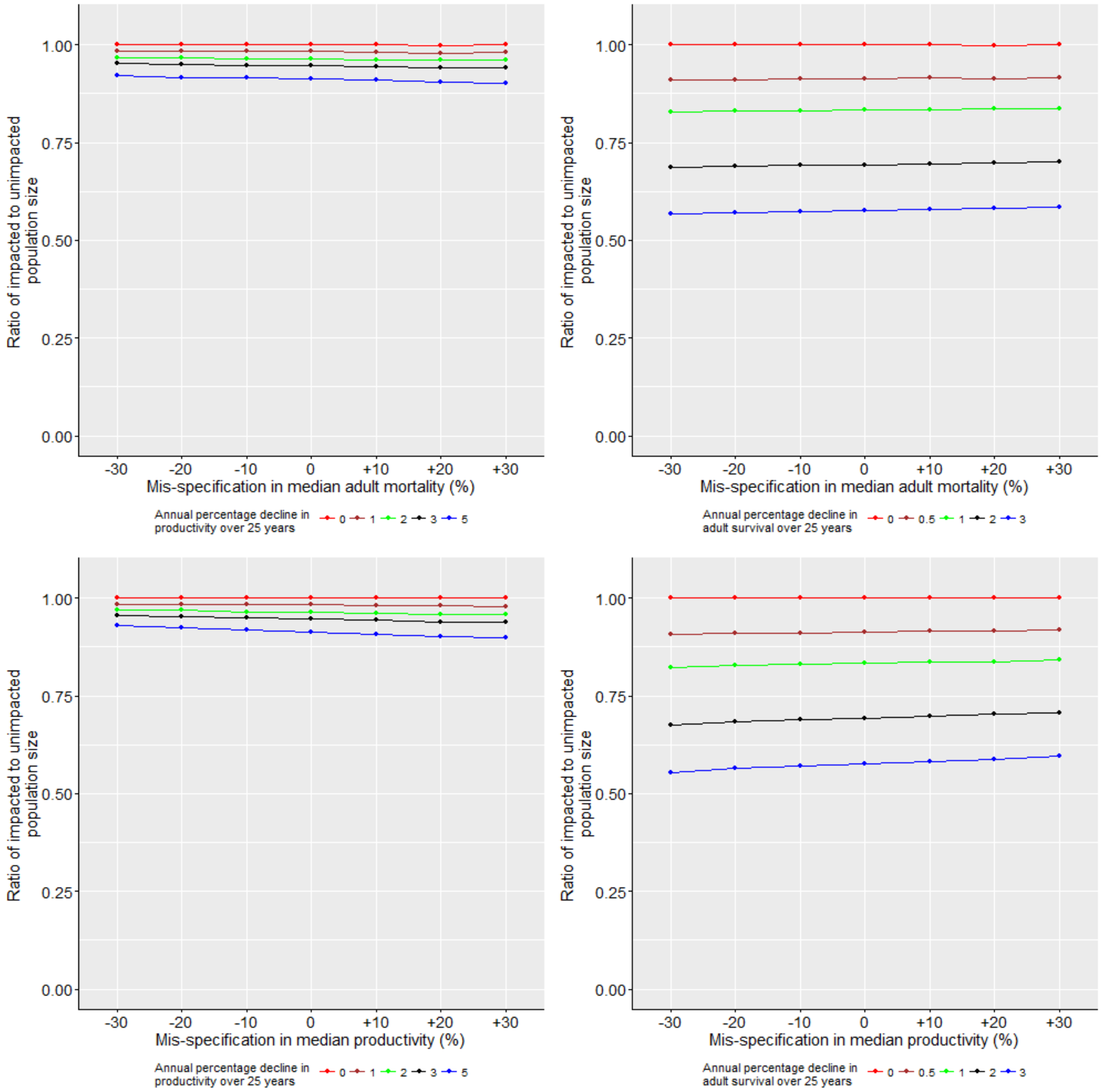


Figure A2.4c. PVA Metric C for Buchan Ness Kittiwakes – difference in population growth rate from 2016-2041, comparing impacted population vs. un-impacted population.

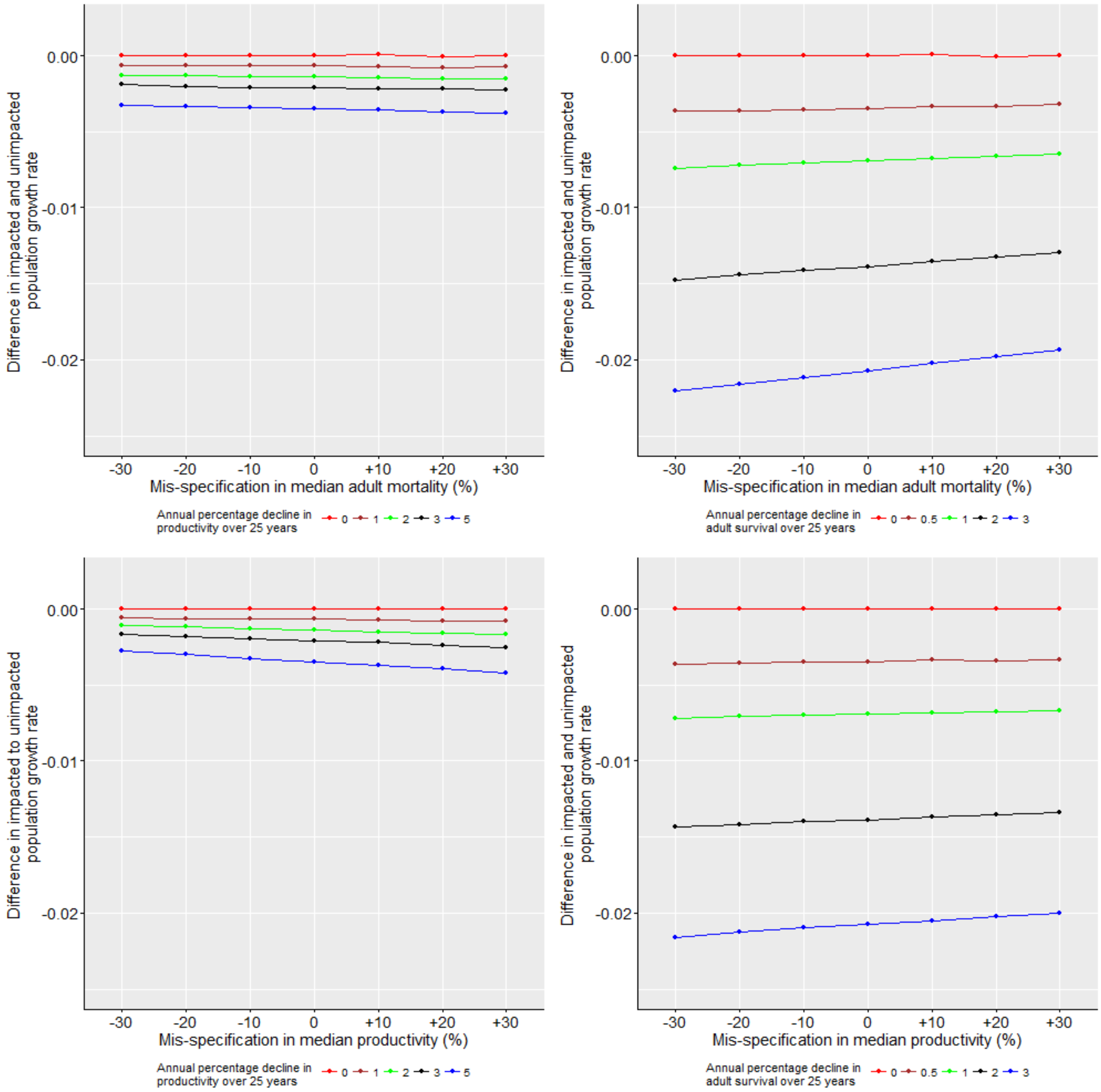


Figure A2.4d. PVA Metric D for Buchan Ness Kittiwakes – difference in population size at 2041, comparing impacted population vs. un-impacted population.

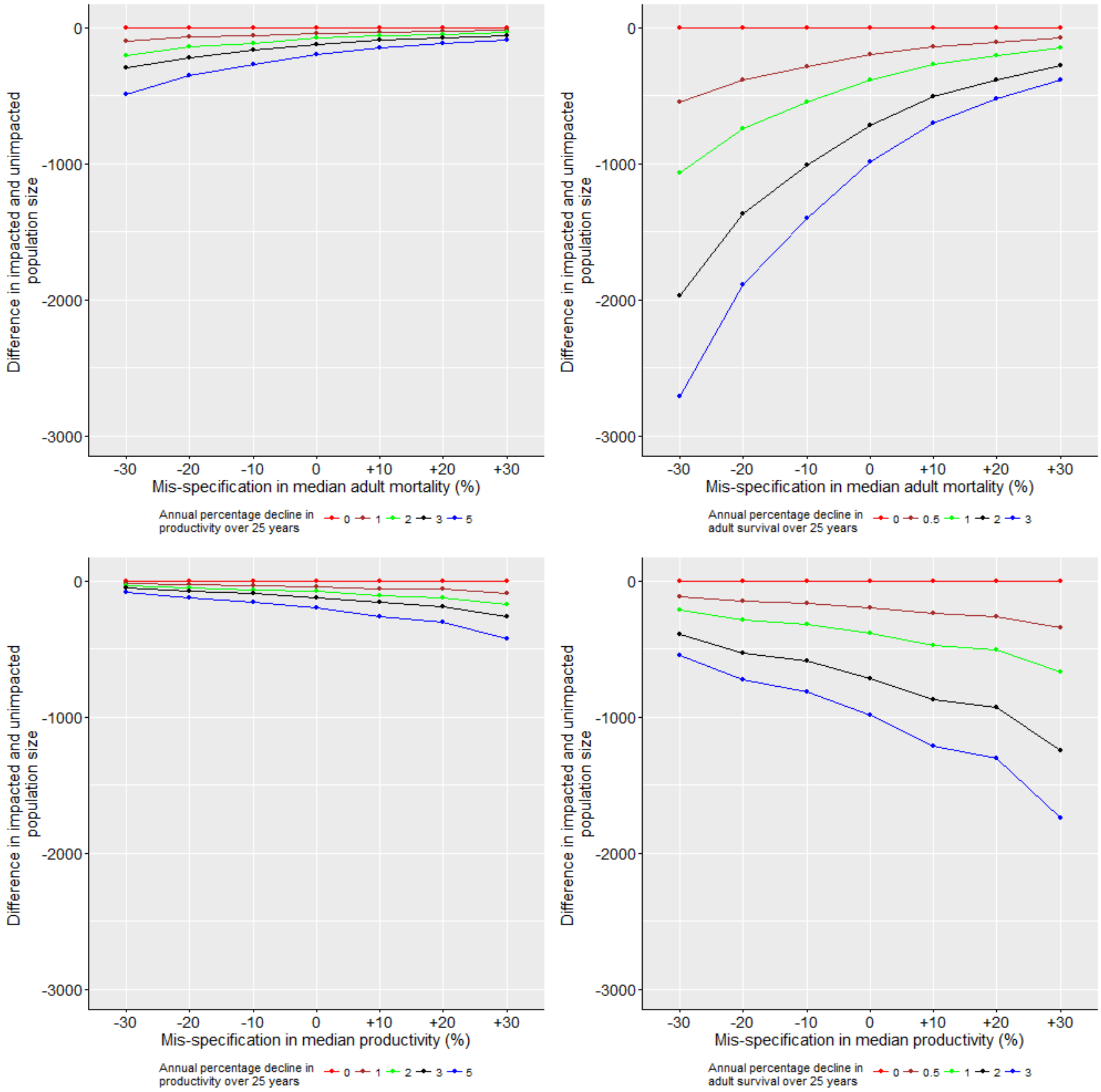


Figure A2.4e. PVA Metric E1 for Buchan Ness Kittiwakes – probability of population decline greater than 10% from 2016-2041.

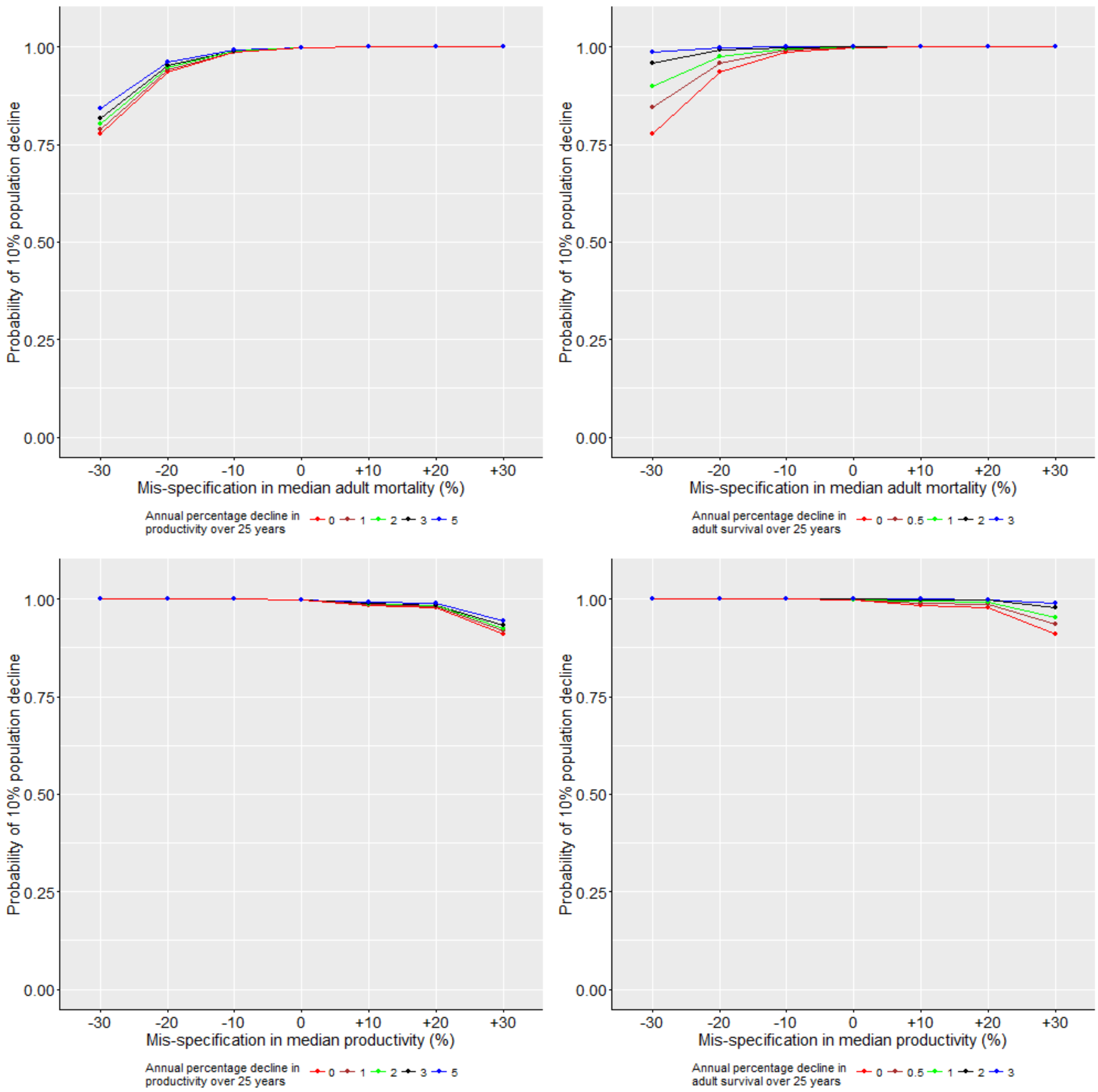


Figure A2.4f. PVA Metric E2 for Buchan Ness Kittiwakes – probability of population decline greater than 25% from 2016-2041.

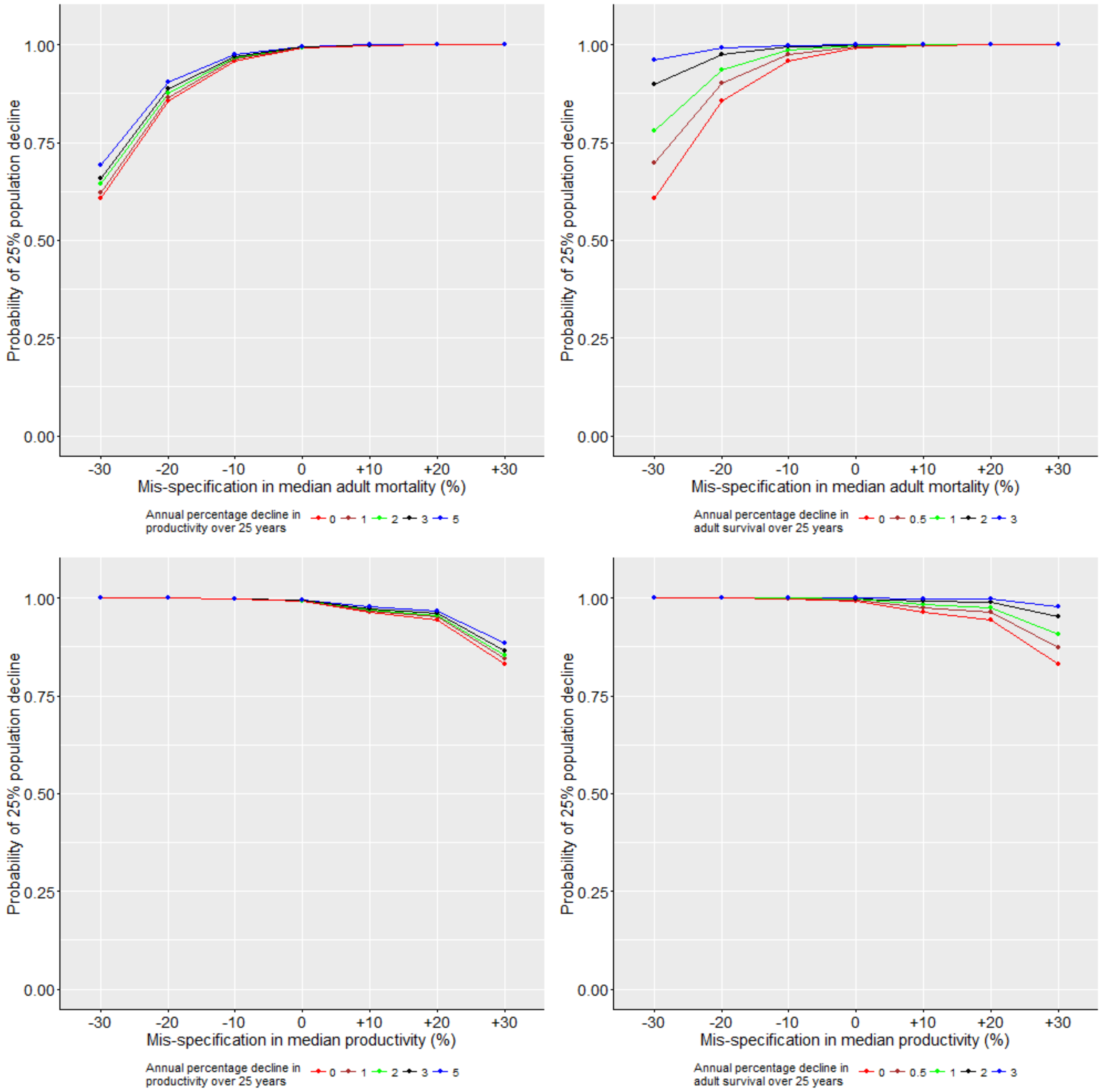


Figure A2.4g. PVA Metric E3 for Buchan Ness Kittiwakes – probability of population decline greater than 50% from 2016-2041.

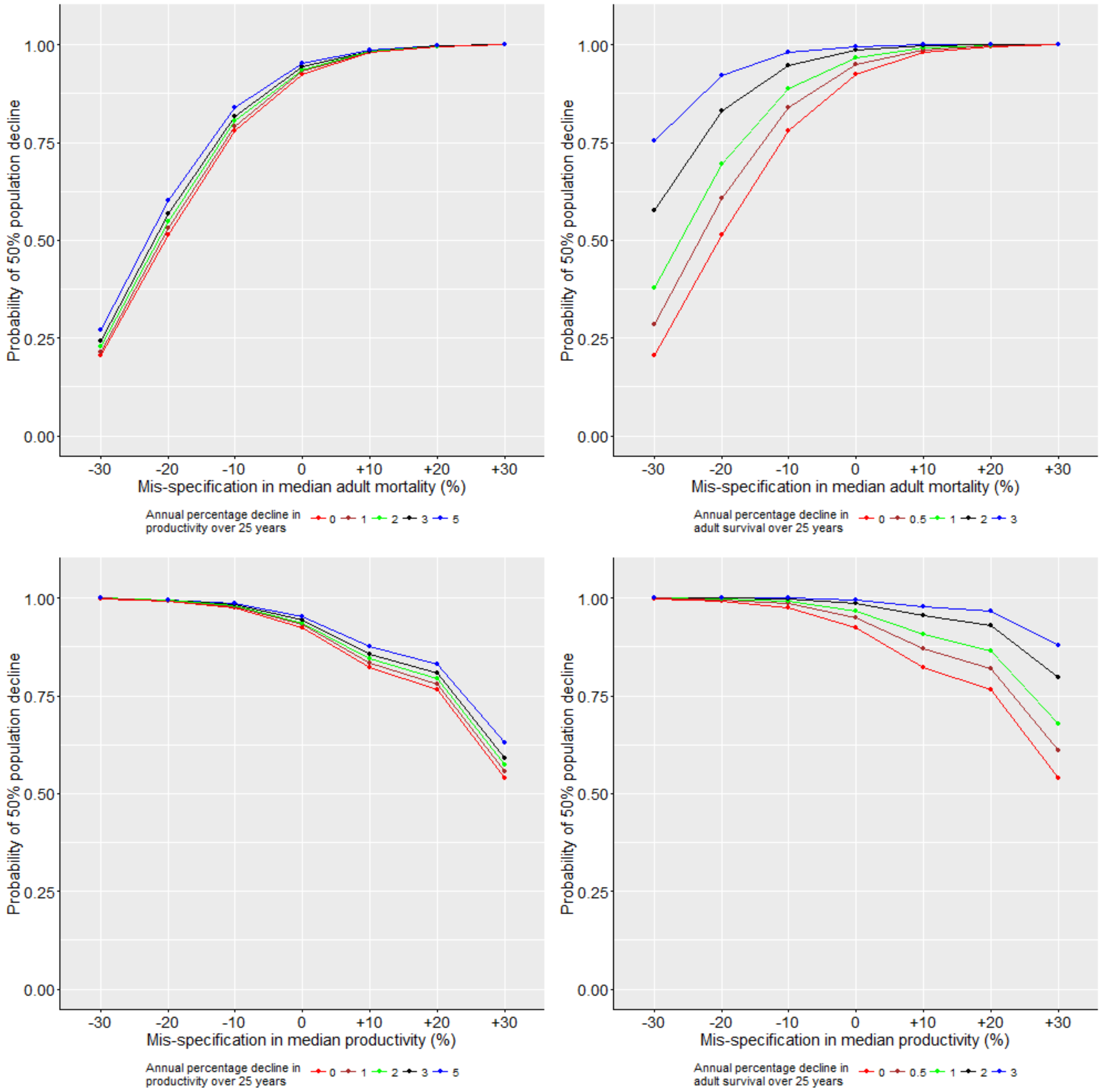
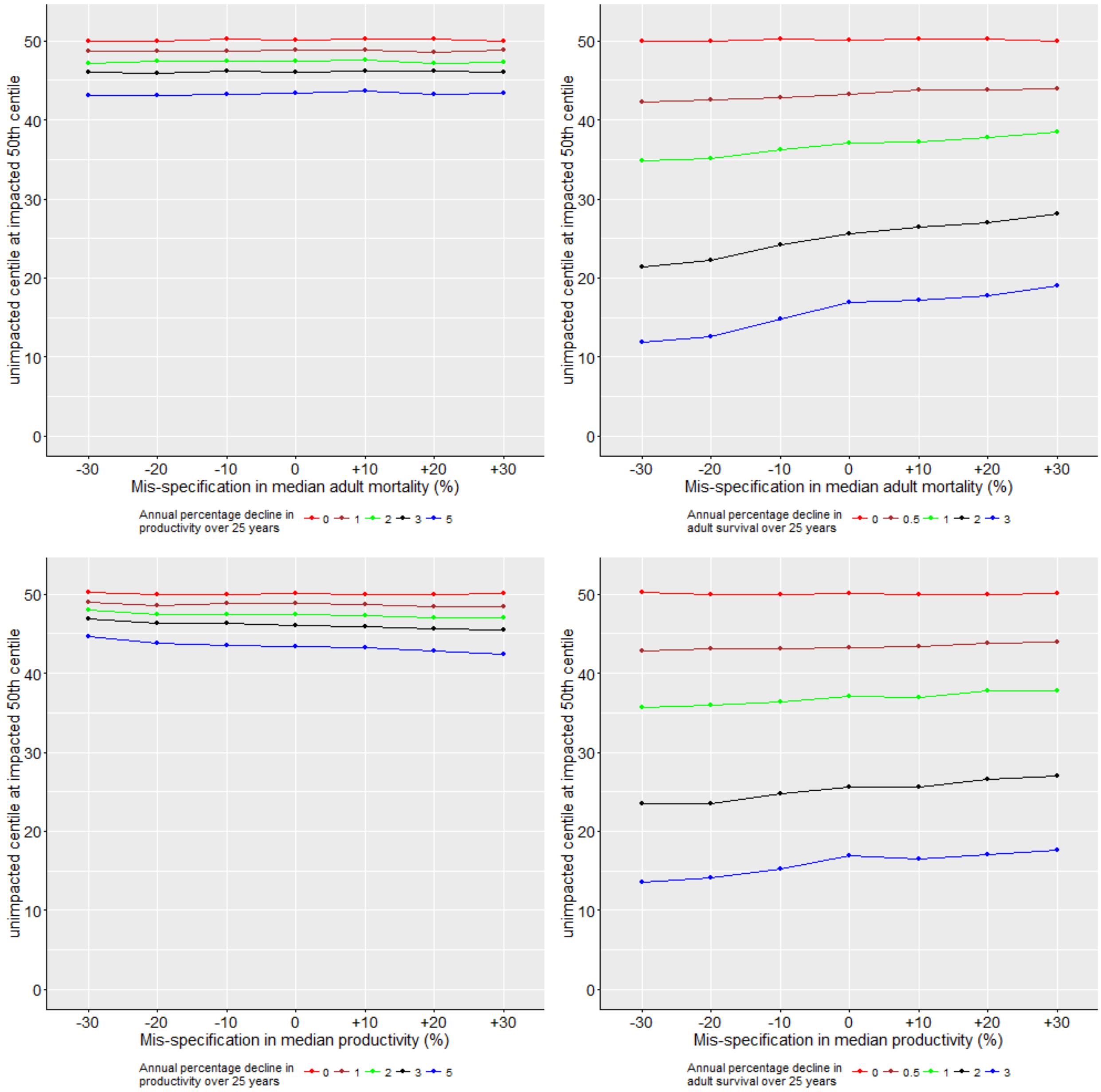


Figure A2.4h. PVA Metric F for Buchan Ness Kittiwakes – centile from un-impacted population size equal to the 50th centile of the impacted population size, at 2041.



5. Guillemots at Forth Islands SPA:

Figure A2.5a. PVA Metric A for Forth Guillemots – ratio of population growth rate from 2016-2041, comparing impacted population vs. un-impacted population.

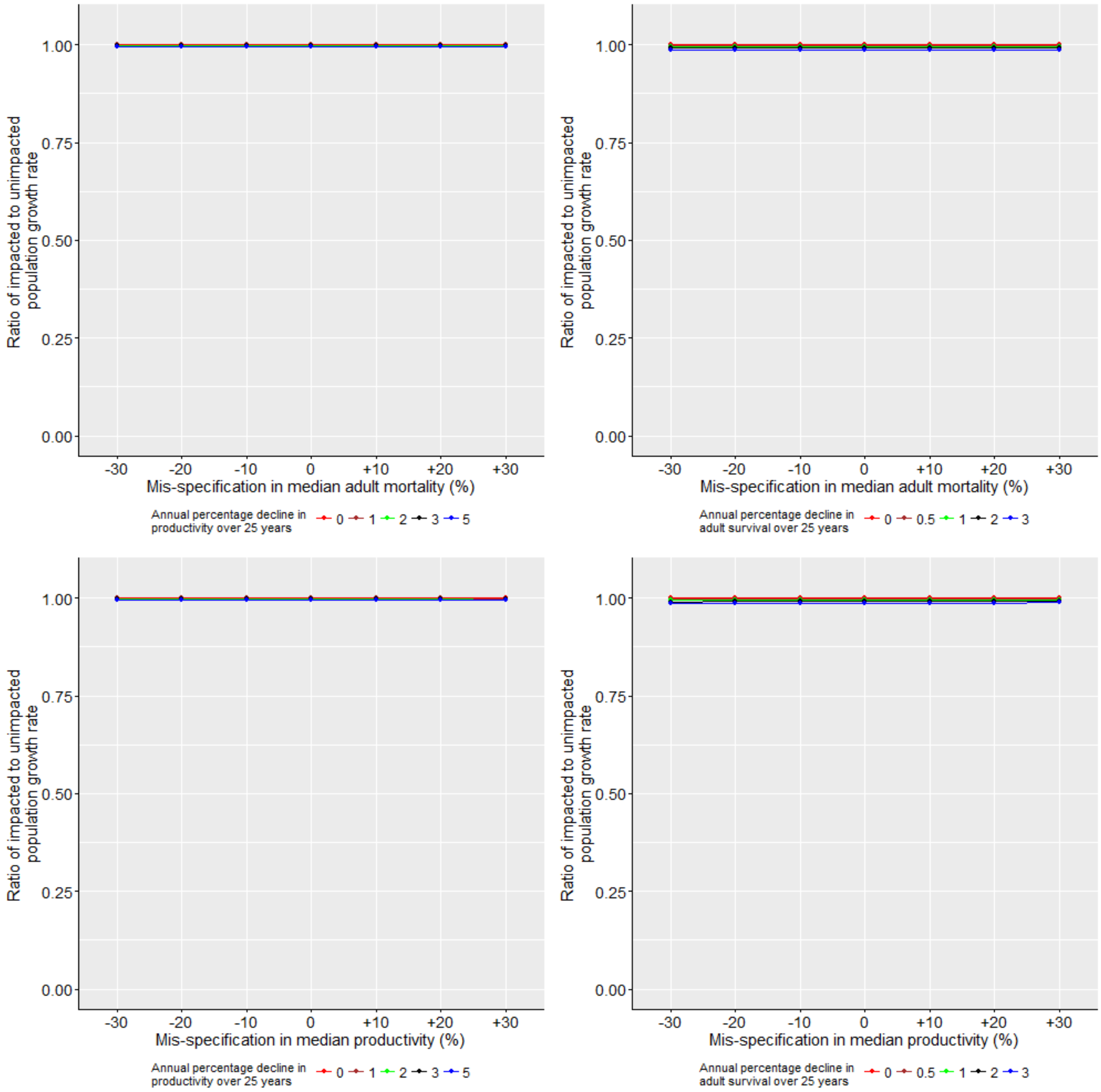


Figure A2.5b. PVA Metric B for Forth Guillemots – ratio of population size at 2041, comparing impacted population vs. un-impacted population.

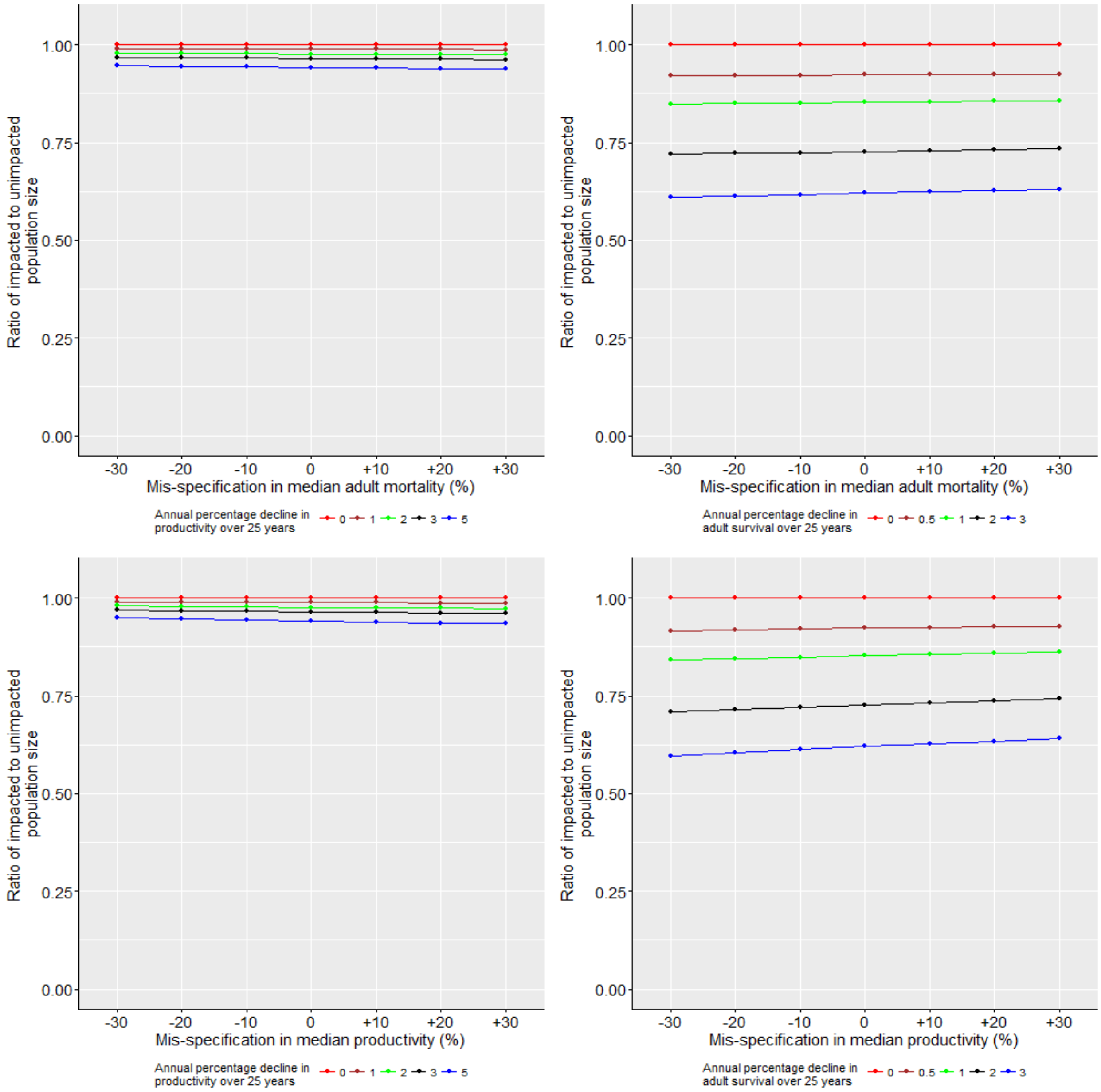


Figure A2.5c. PVA Metric C for Forth Guillemots – difference in population growth rate from 2016-2041, comparing impacted population vs. un-impacted population.

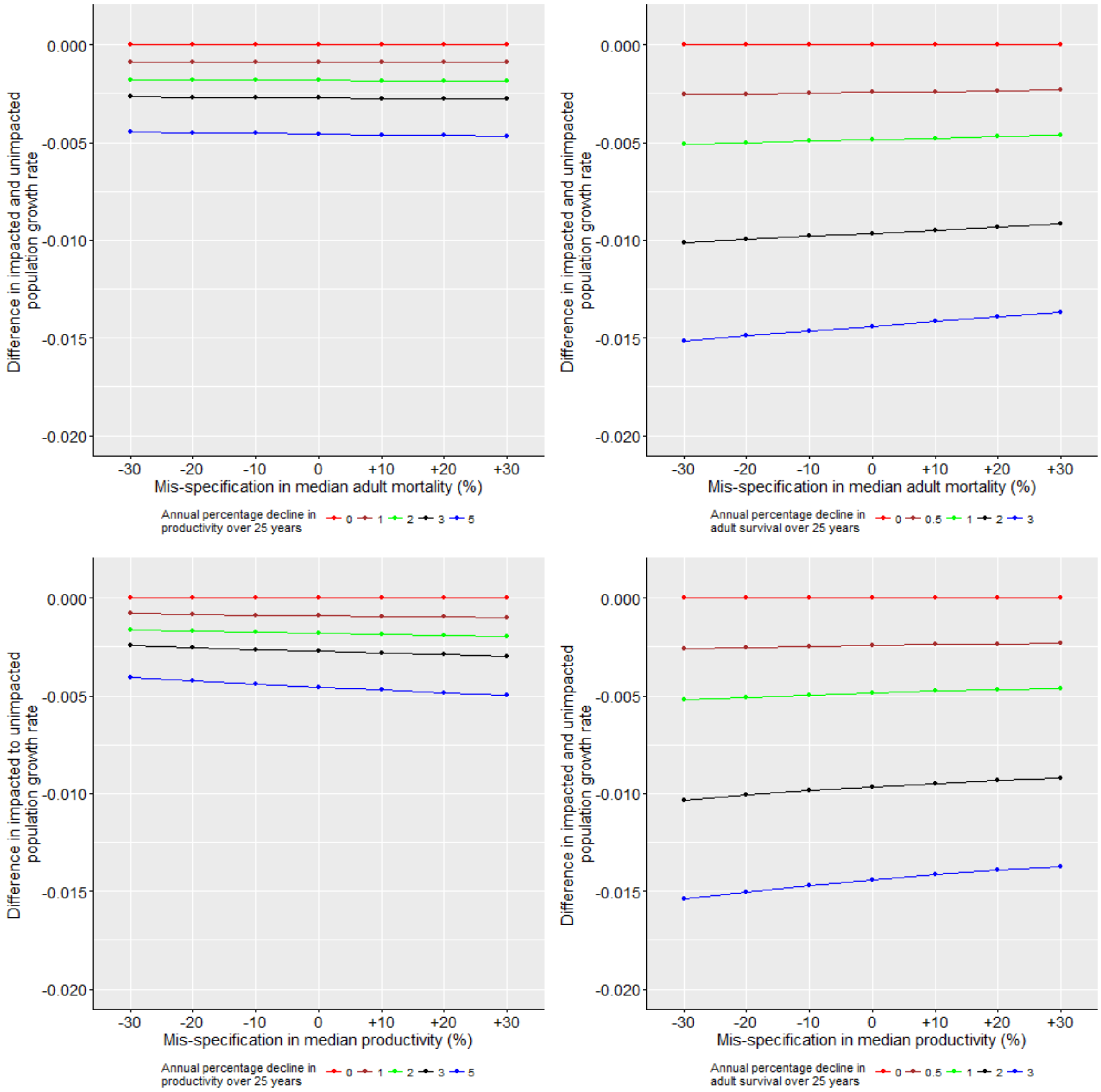


Figure A2.5d. PVA Metric D for Forth Guillemots – difference in population size at 2041, comparing impacted population vs. un-impacted population.

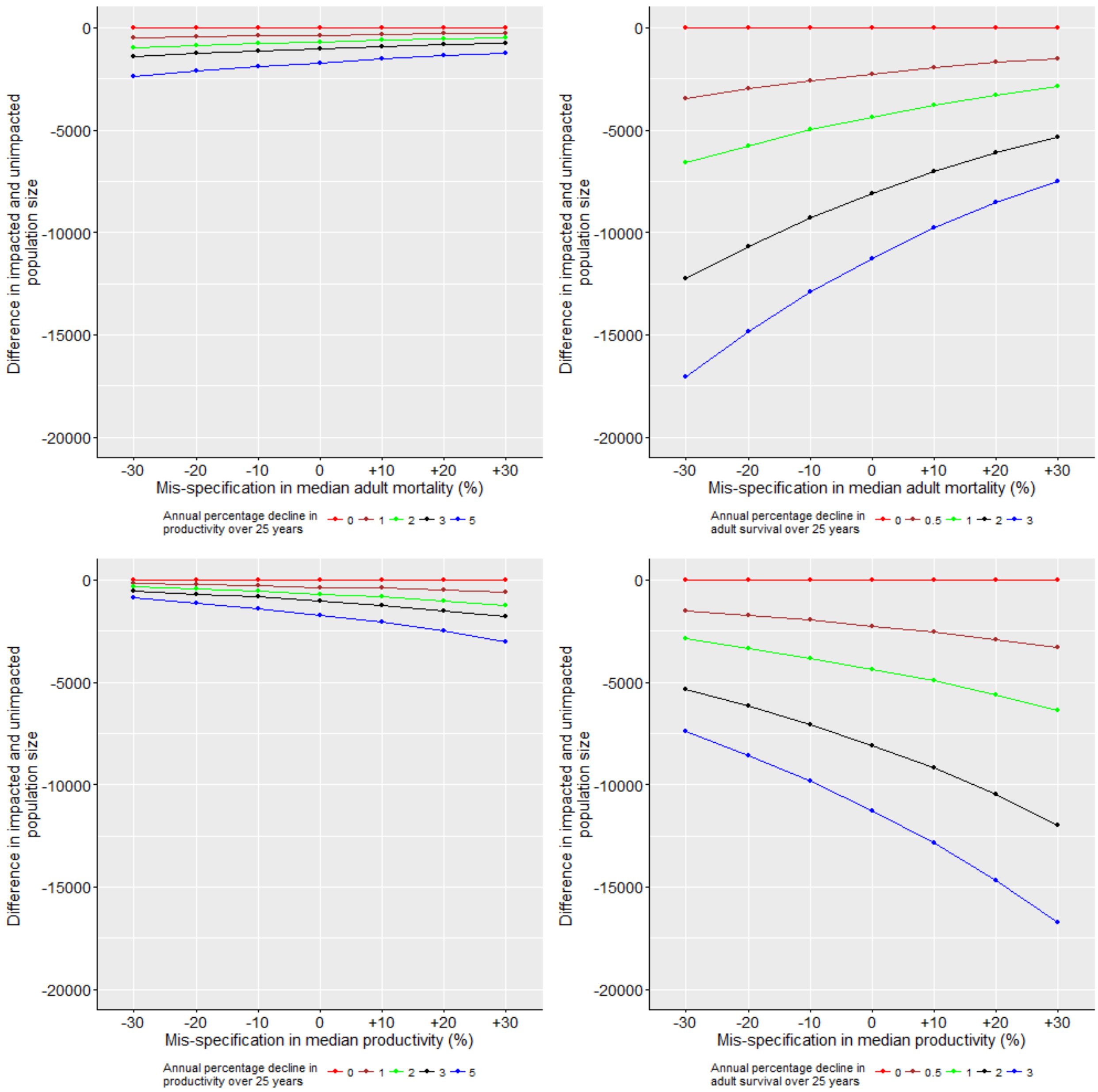


Figure A2.5e. PVA Metric E1 for Forth Guillemots – probability of population decline greater than 10% from 2016-2041.

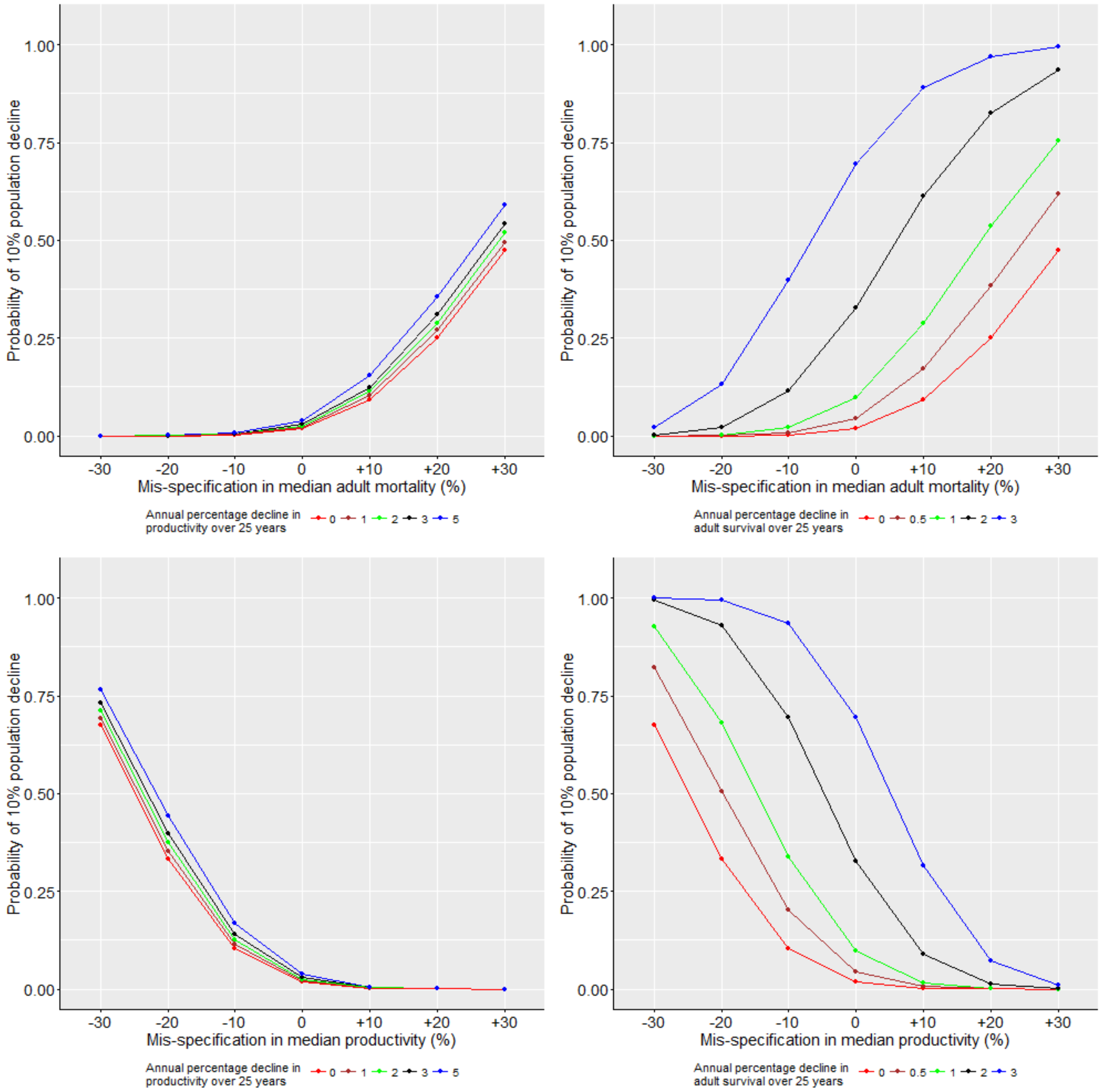


Figure A2.5f. PVA Metric E2 for Forth Guillemots – probability of population decline greater than 25% from 2016-2041.

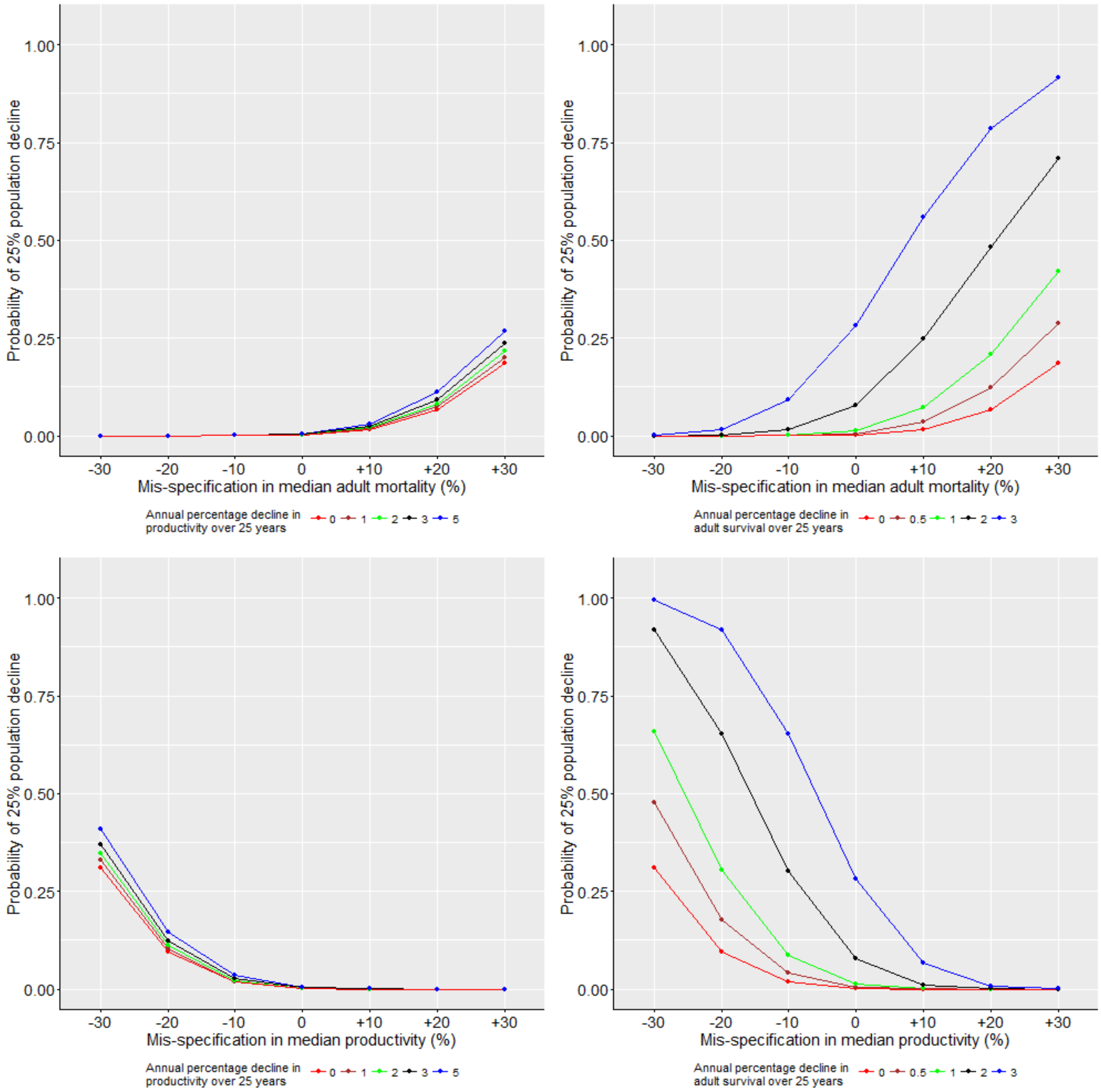


Figure A2.5g. PVA Metric E3 for Forth Guillemots – probability of population decline greater than 50% from 2016-2041.

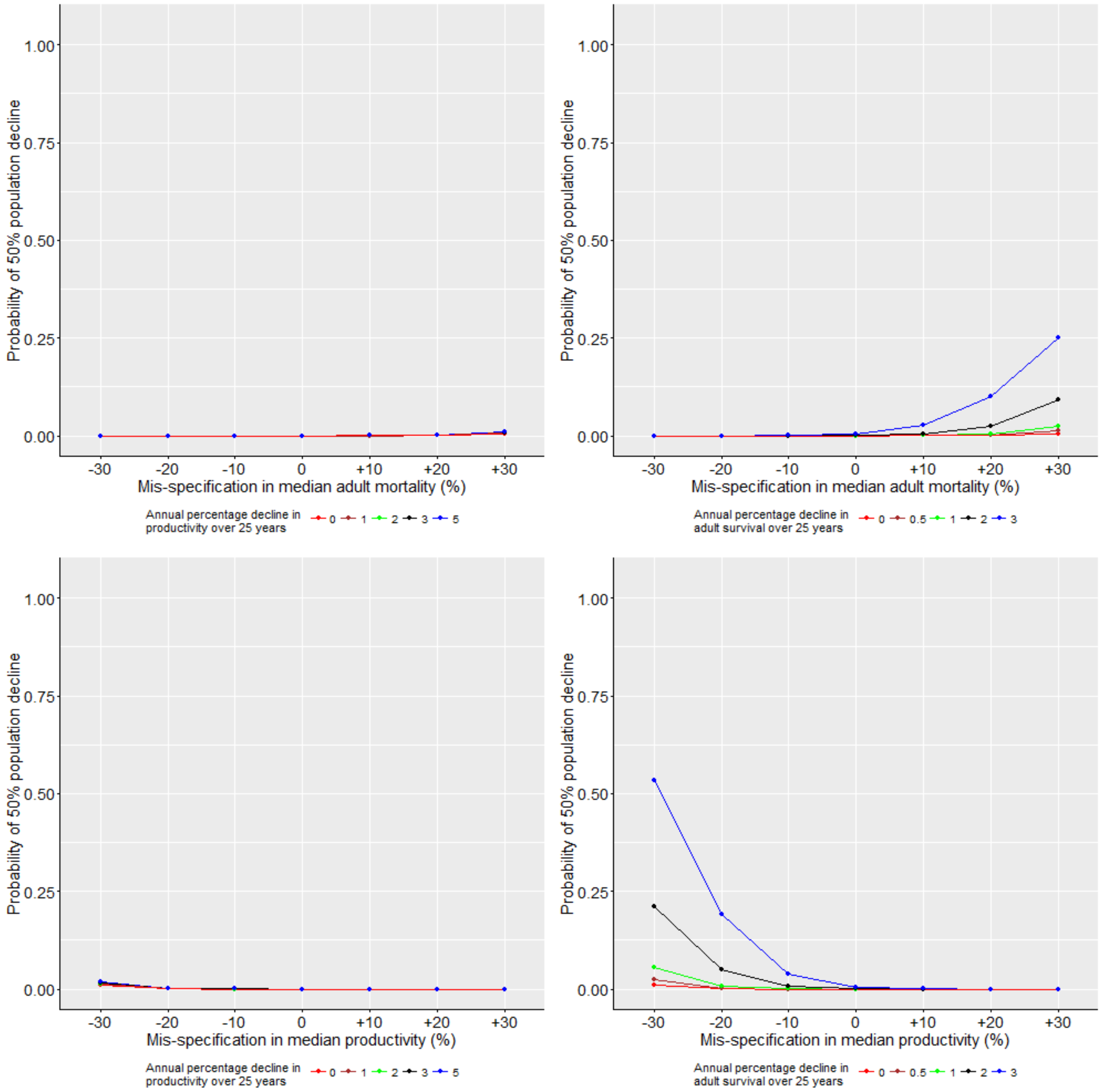
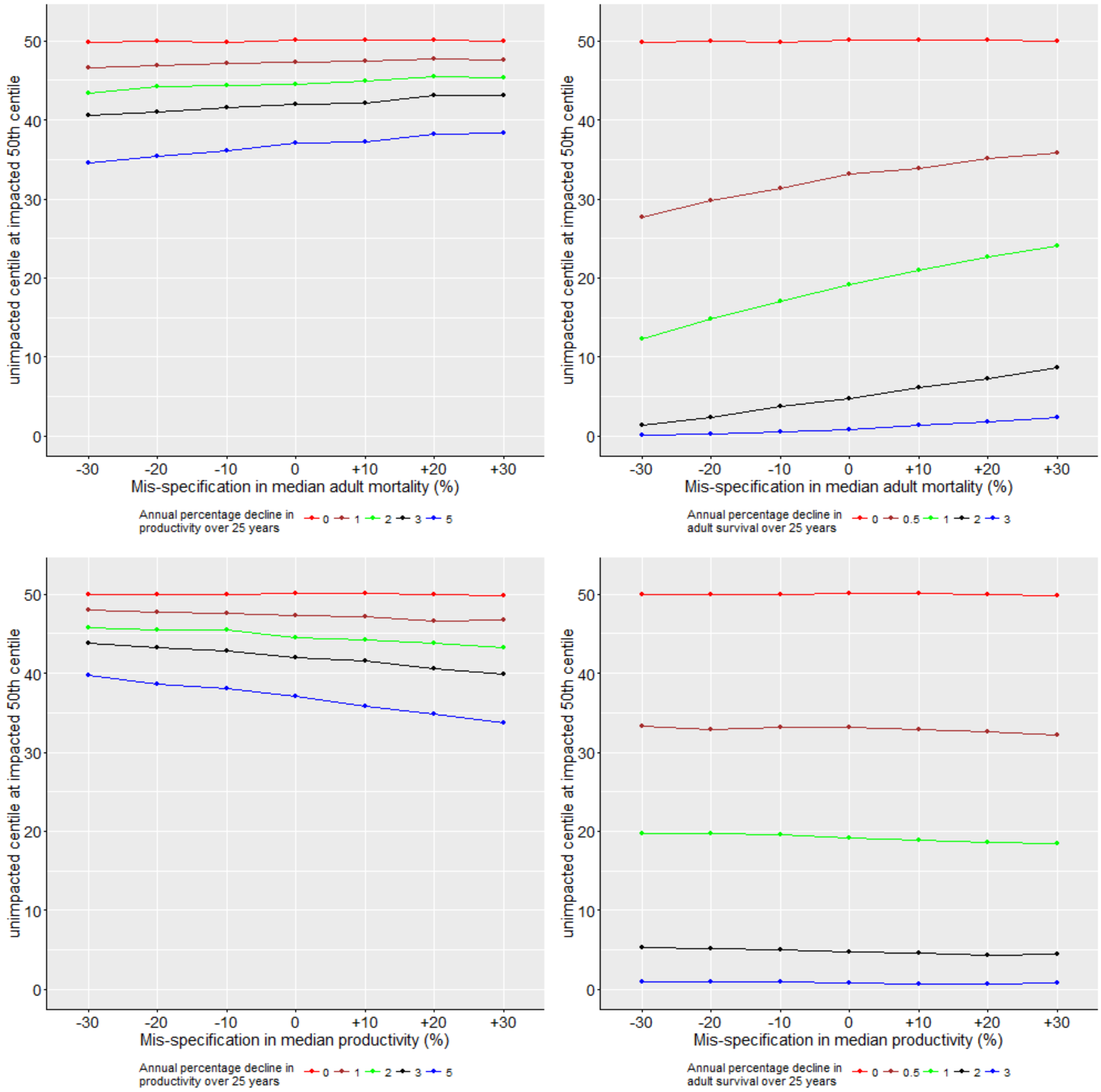


Figure A2.5h. PVA Metric F for Forth Guillemots – centile from un-impacted population size equal to the 50th centile of the impacted population size, at 2041.



6. **Guillemots at St Abb's Head SPA**:

Figure A2.6a. PVA Metric A for St Abb's Guillemots – ratio of population growth rate from 2016-2041, comparing impacted population vs. un-impacted population.

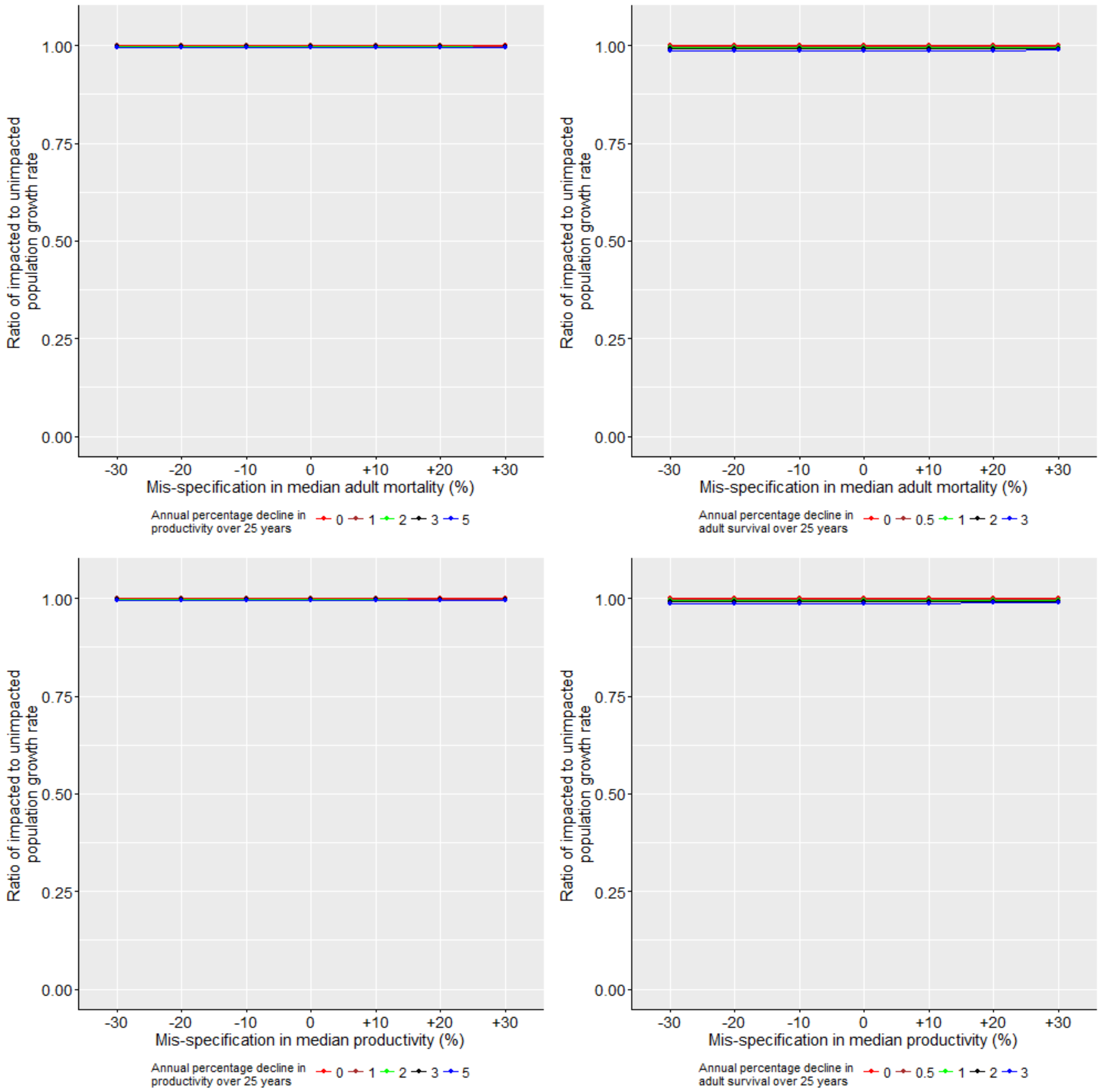


Figure A2.6b. PVA Metric B for St Abb's Guillemots – ratio of population size at 2041, comparing impacted population vs. un-impacted population.

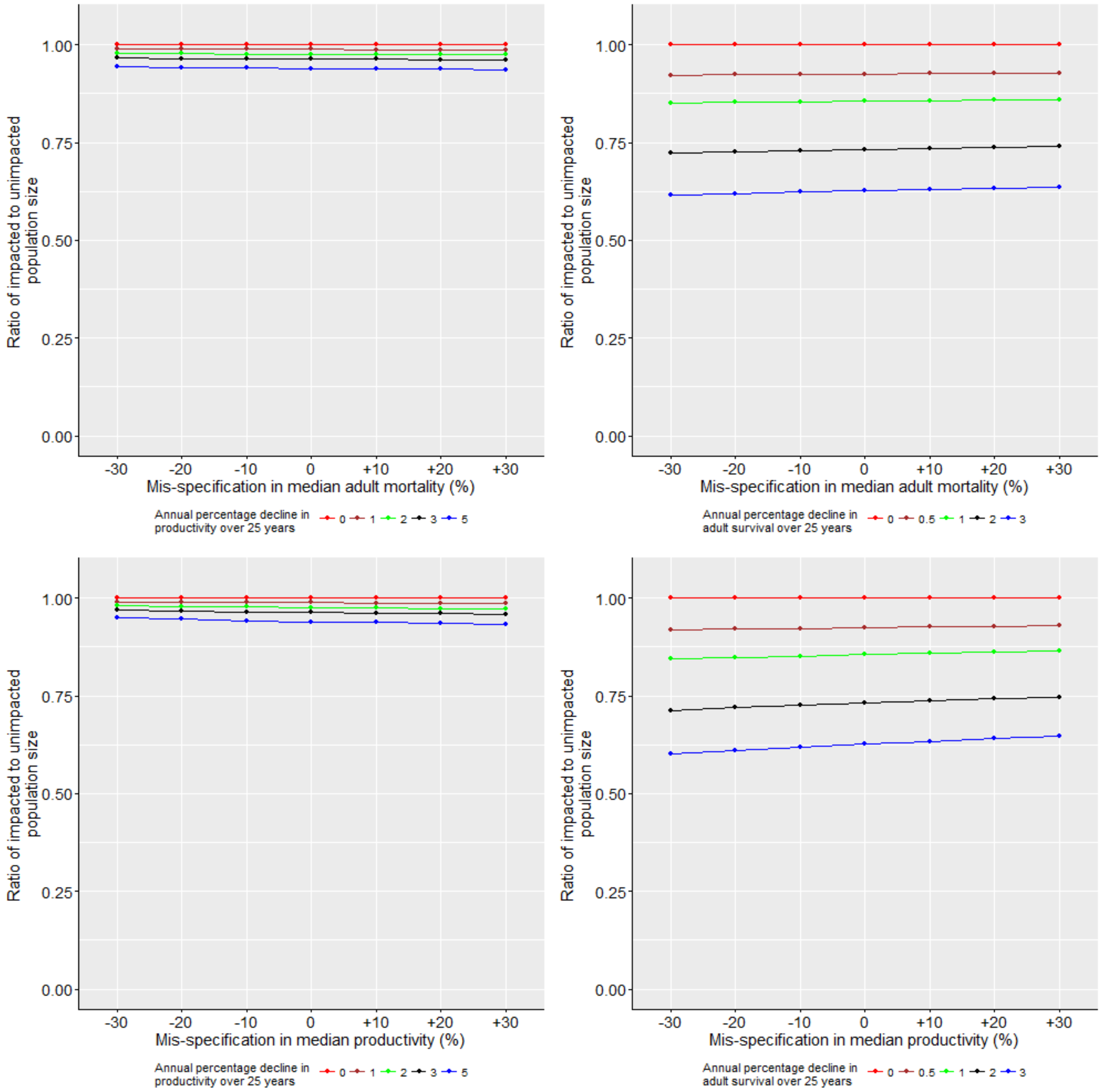


Figure A2.6c. PVA Metric C for St Abb's Guillemots – difference in population growth rate from 2016-2041, comparing impacted population vs. un-impacted population.

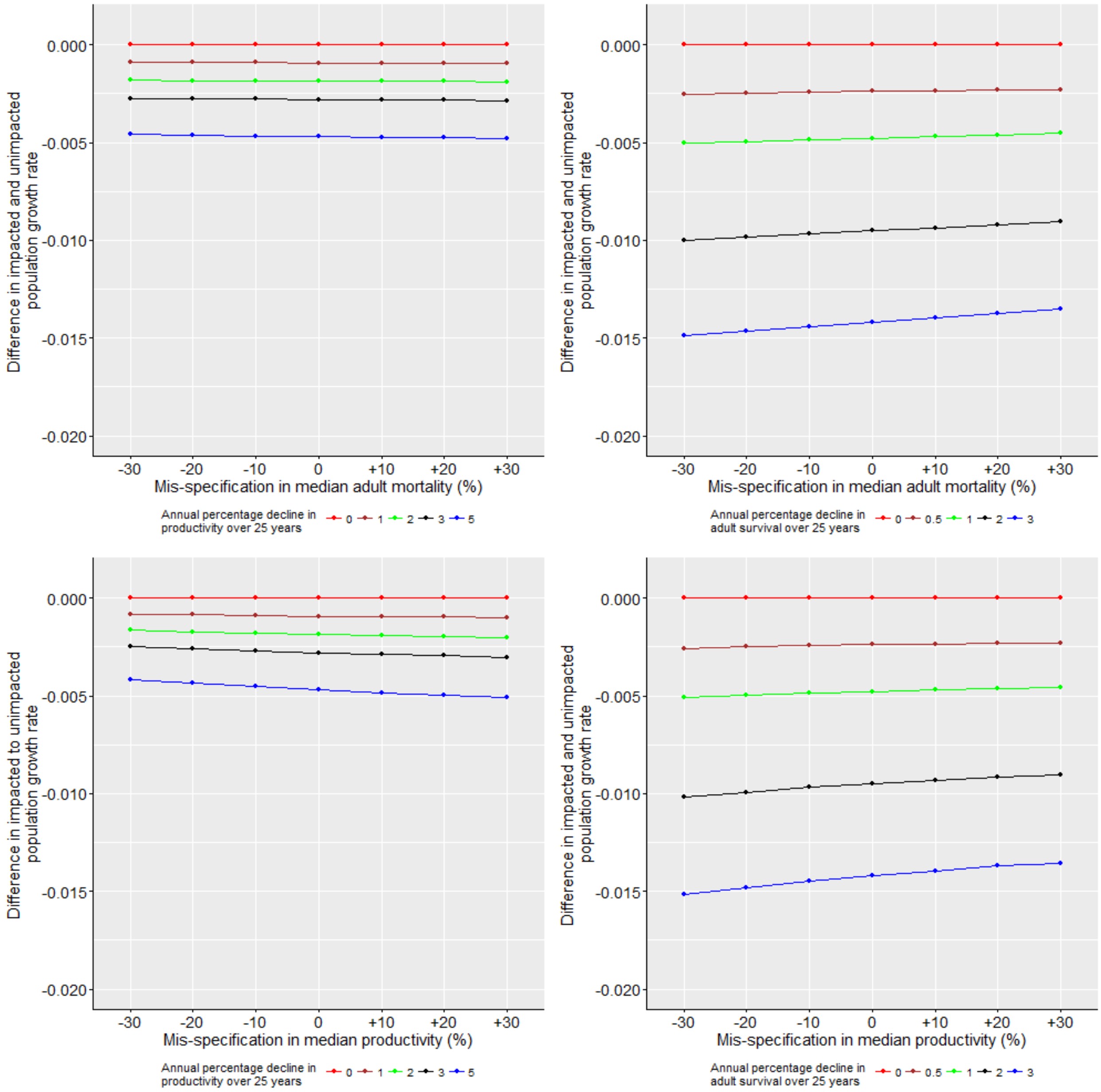


Figure A2.6d. PVA Metric D for St Abb's Guillemots – difference in population size at 2041, comparing impacted population vs. un-impacted population.

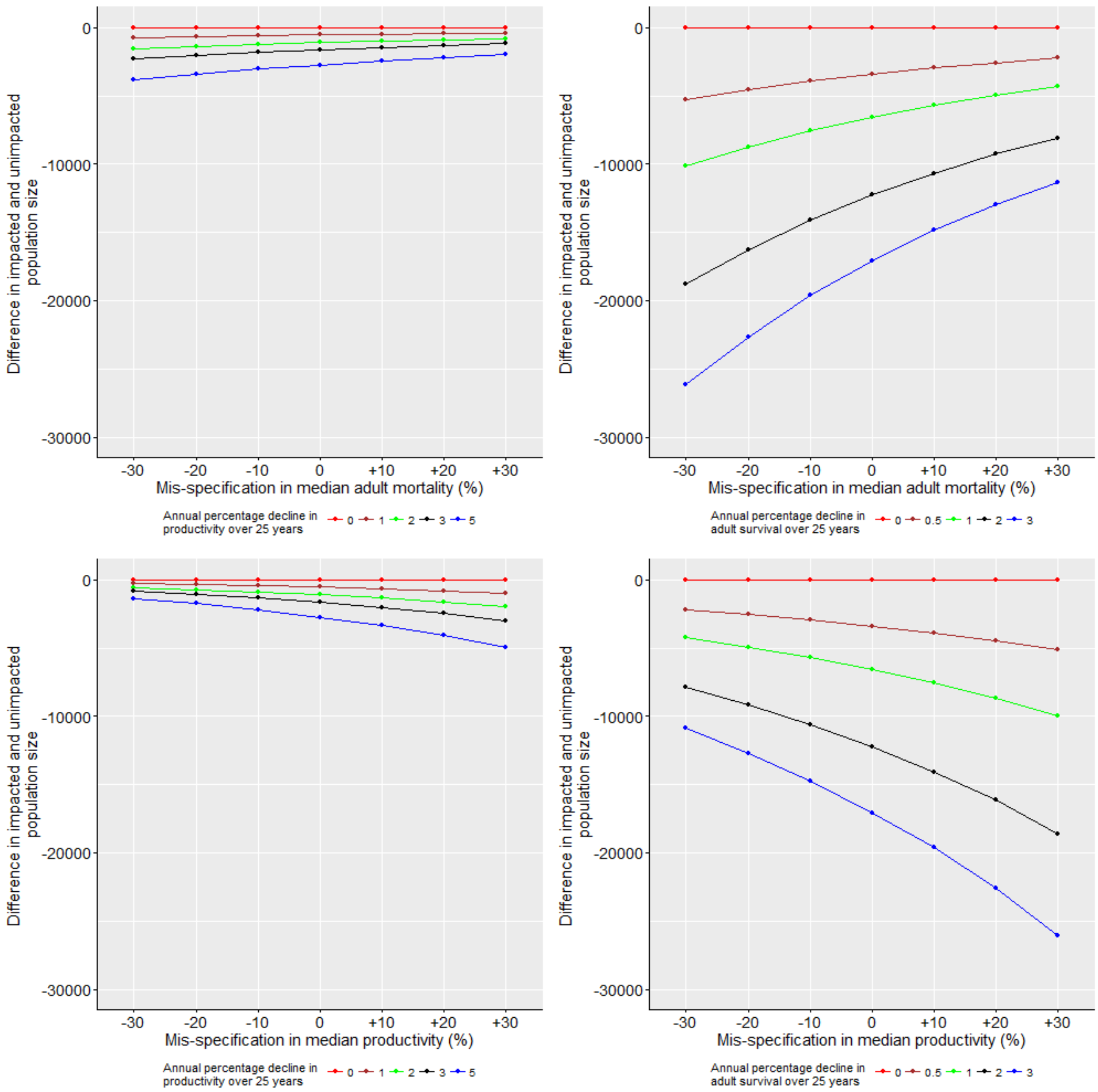


Figure A2.6e. PVA Metric E1 for St Abb's Guillemots – probability of population decline greater than 10% from 2016-2041.

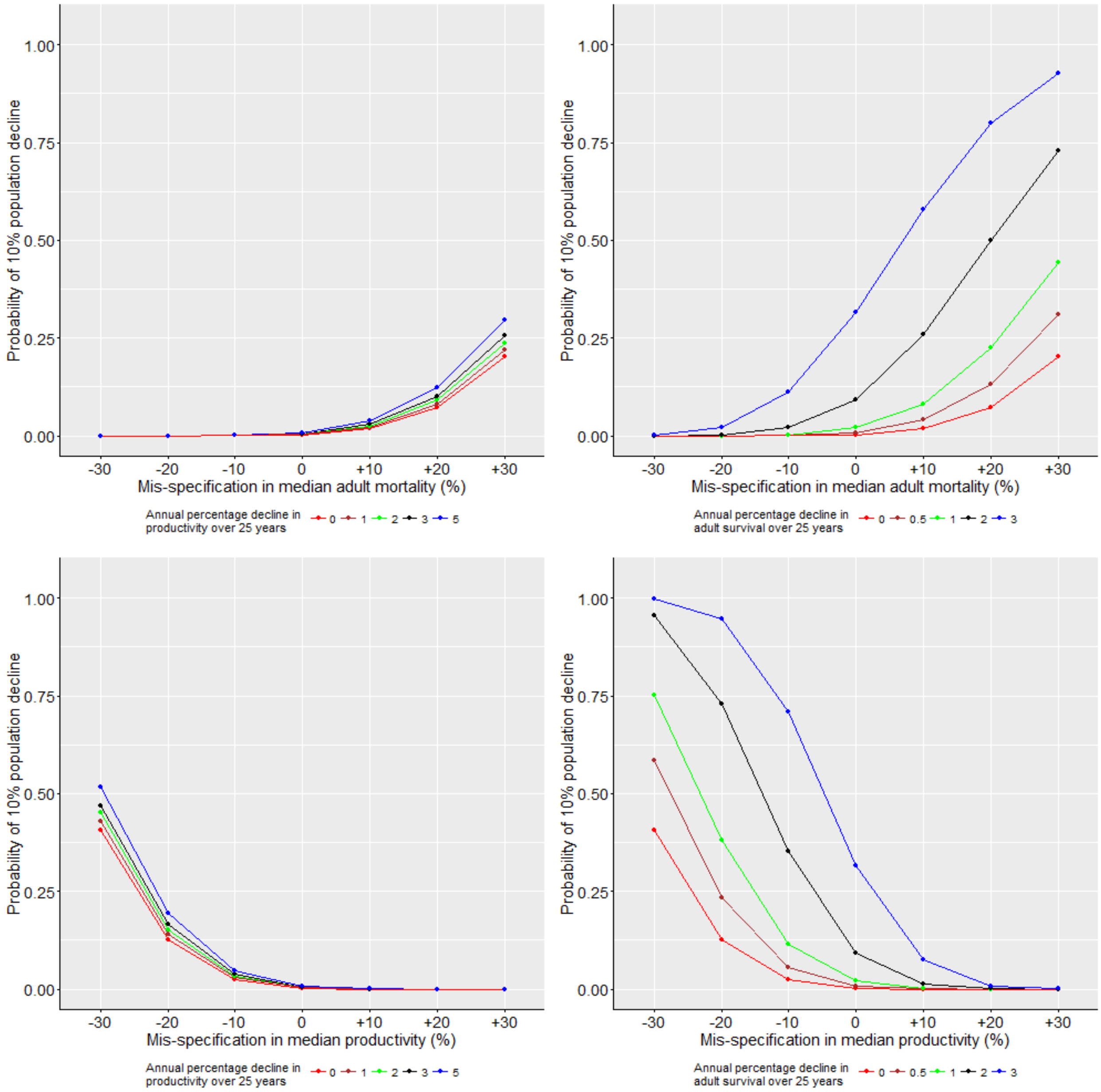


Figure A2.6f. PVA Metric E2 for St Abb's Guillemots – probability of population decline greater than 25% from 2016-2041.

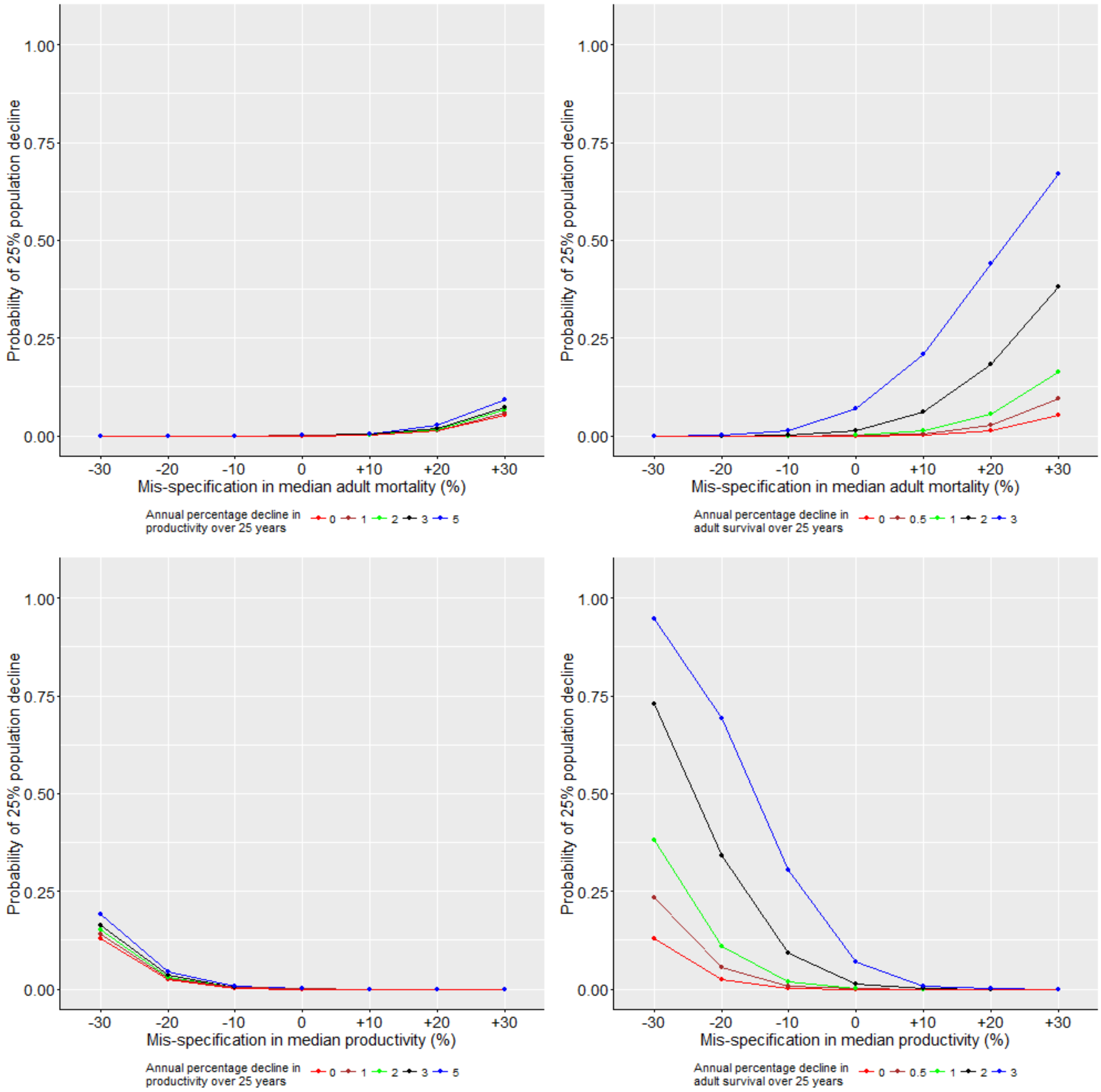


Figure A2.6g. PVA Metric E3 for St Abb's Guillemots – probability of population decline greater than 50% from 2016-2041.

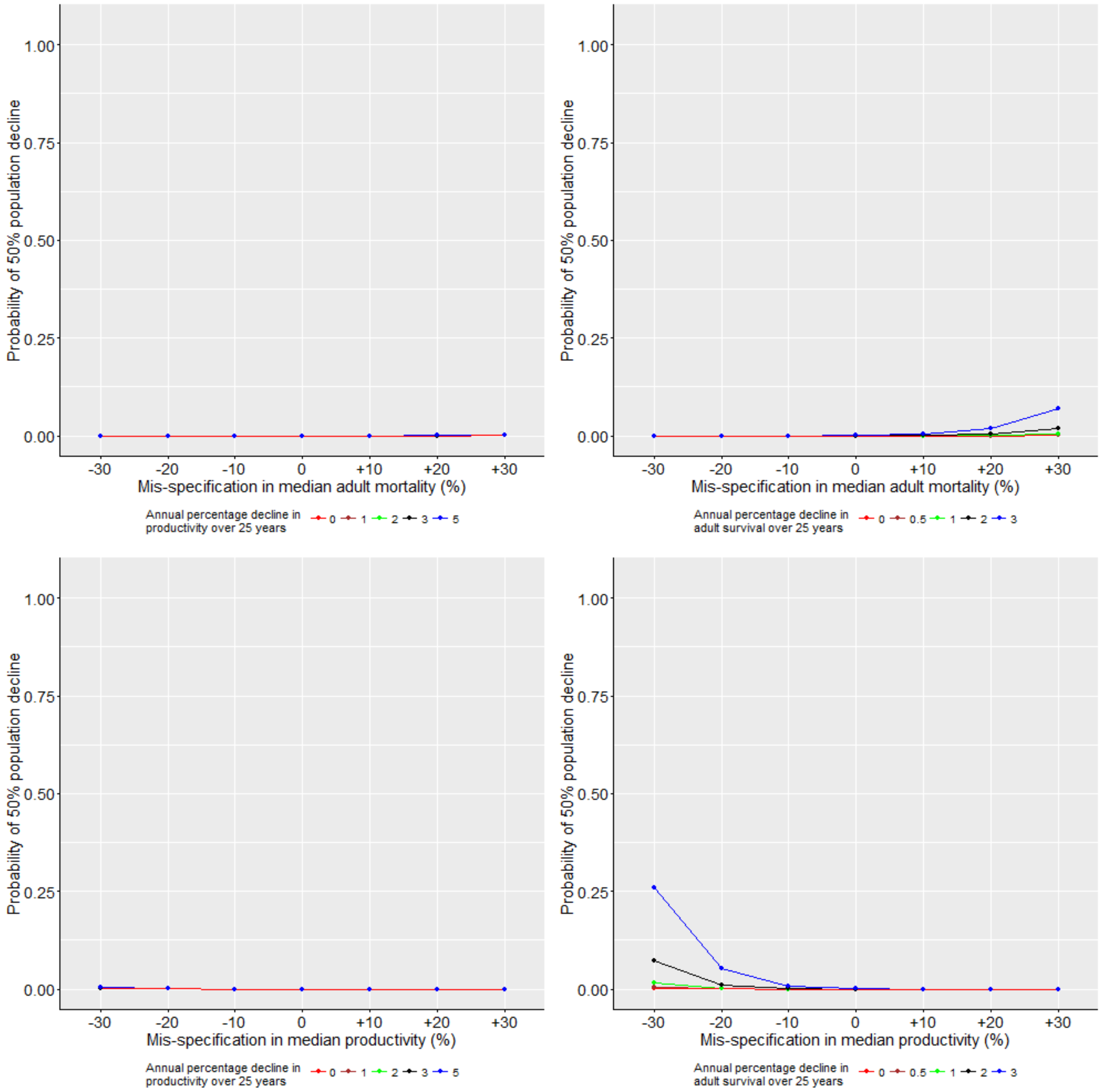
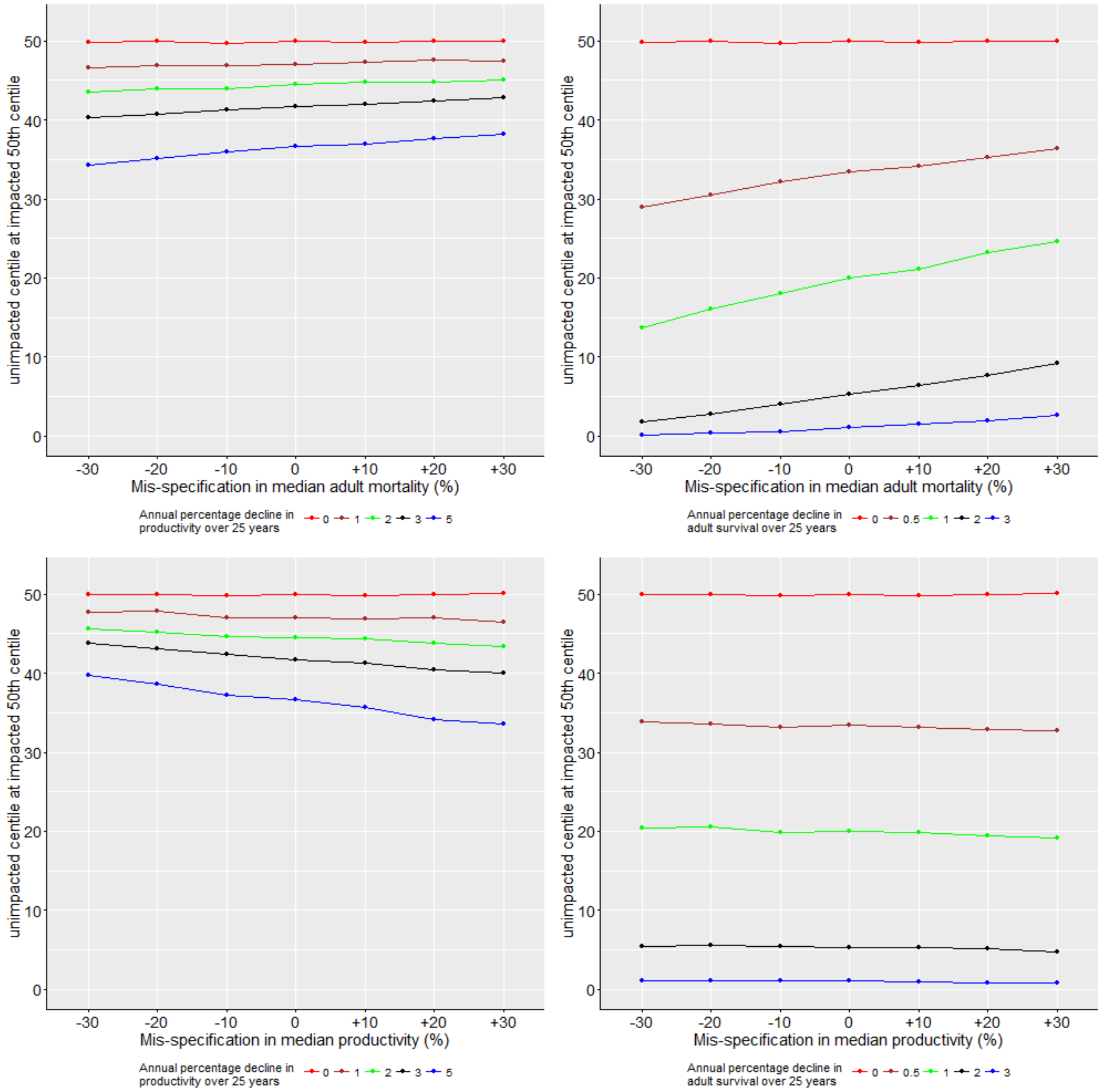


Figure A2.6h. PVA Metric F for St Abb's Guillemots – centile from un-impacted population size equal to the 50th centile of the impacted population size, at 2041.



7. **Guillemots at Fowlsheugh SPA**:

Figure A2.7a. PVA Metric A for Fowlsheugh Guillemots – ratio of population growth rate from 2016-2041, comparing impacted population vs. un-impacted population.

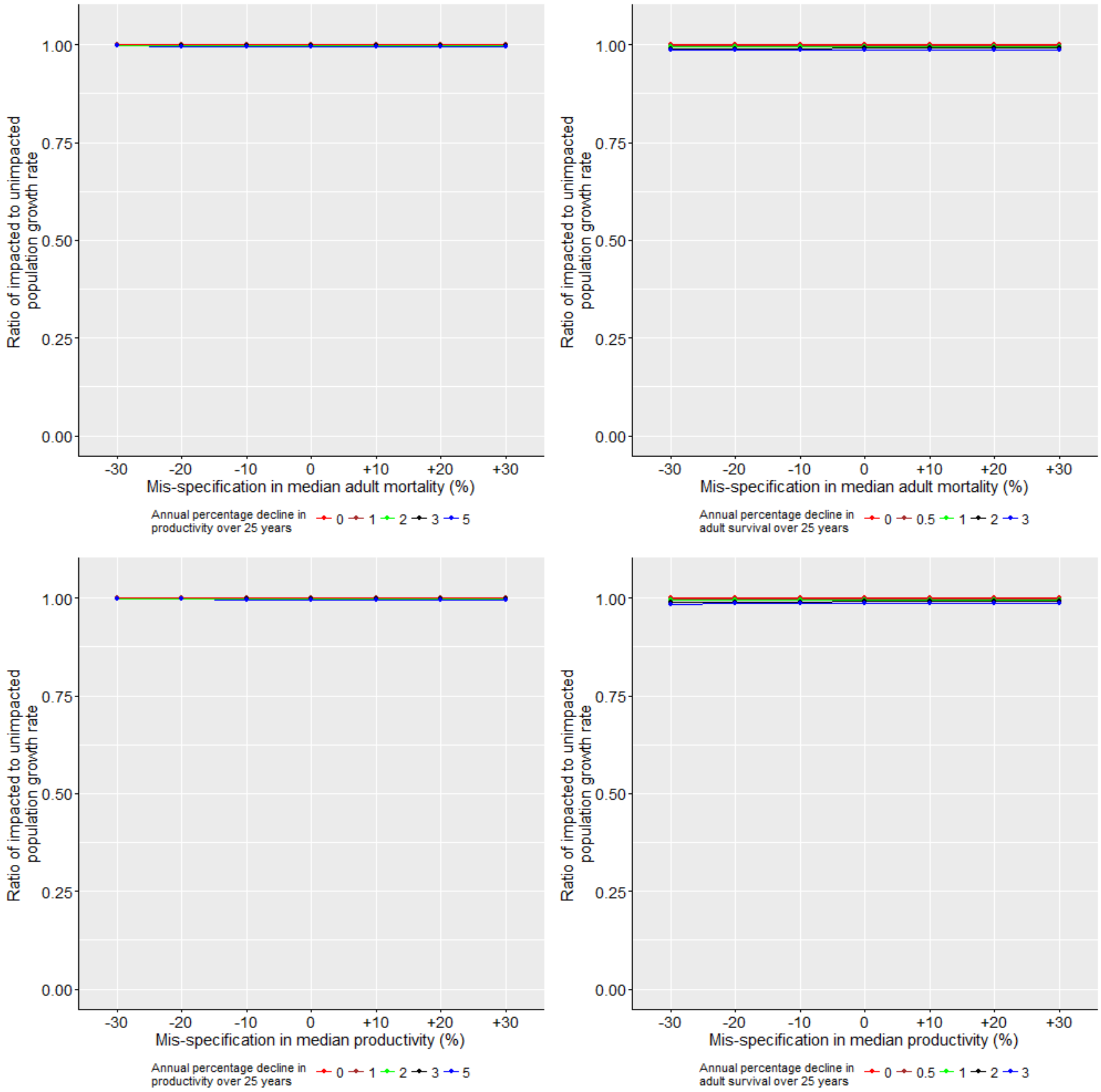


Figure A2.7b. PVA Metric B for Fowlsheugh Guillemots – ratio of population size at 2041, comparing impacted population vs. un-impacted population.

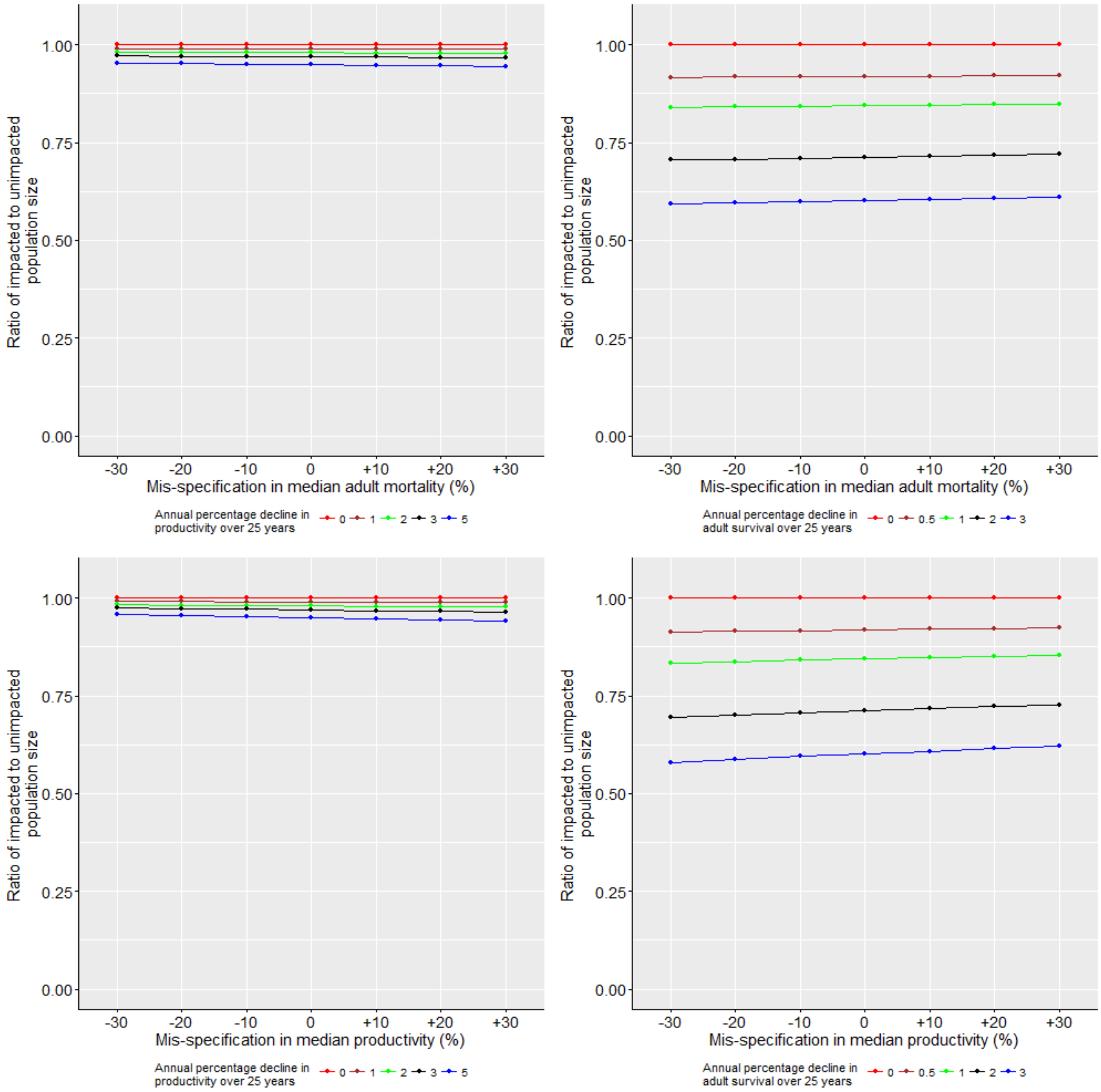


Figure A2.7c. PVA Metric C for Fowlsheugh Guillemots – difference in population growth rate from 2016-2041, comparing impacted population vs. un-impacted population.

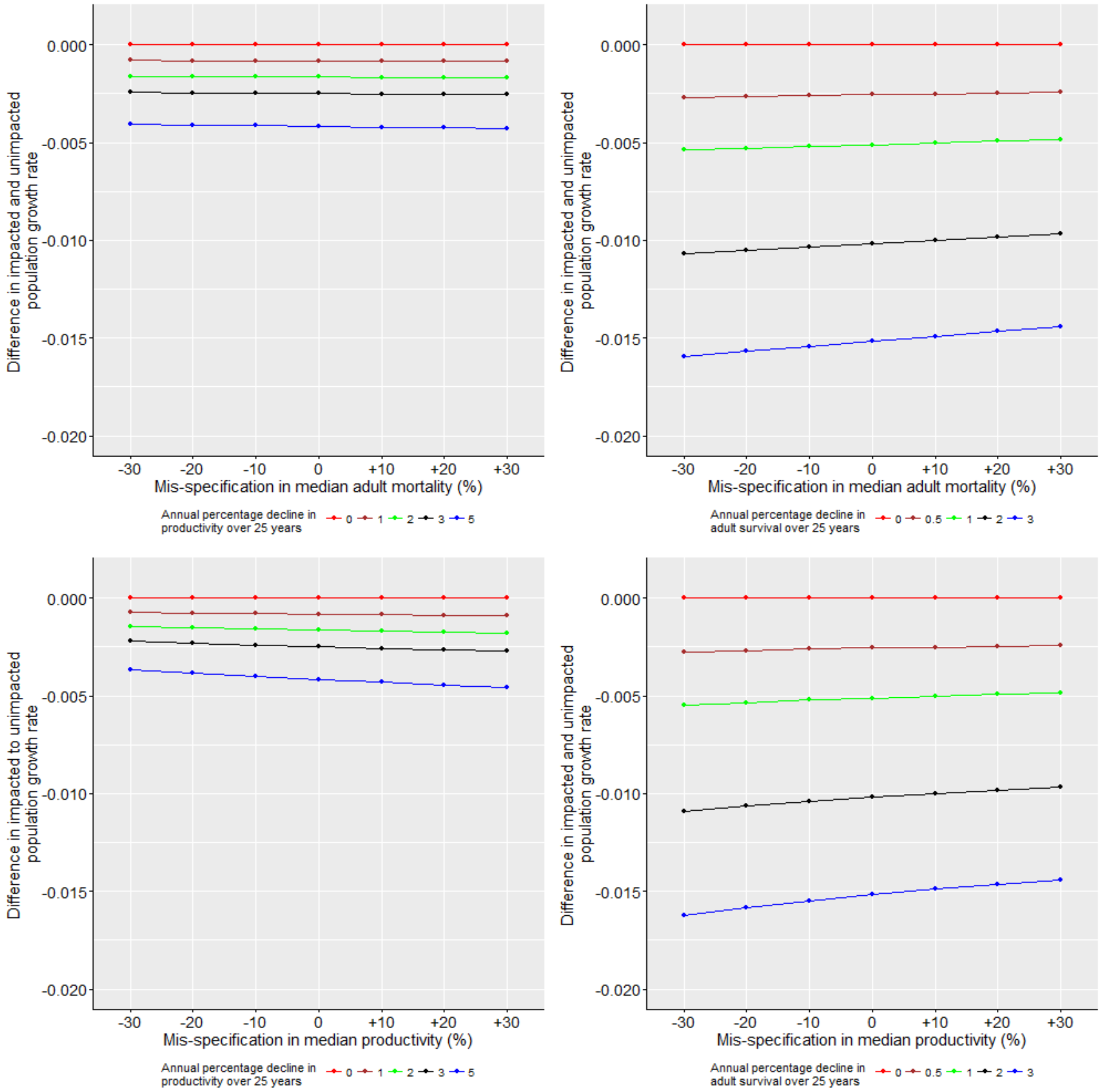


Figure A2.7d. PVA Metric D for Fowlsheugh Guillemots – difference in population size at 2041, comparing impacted population vs. un-impacted population.

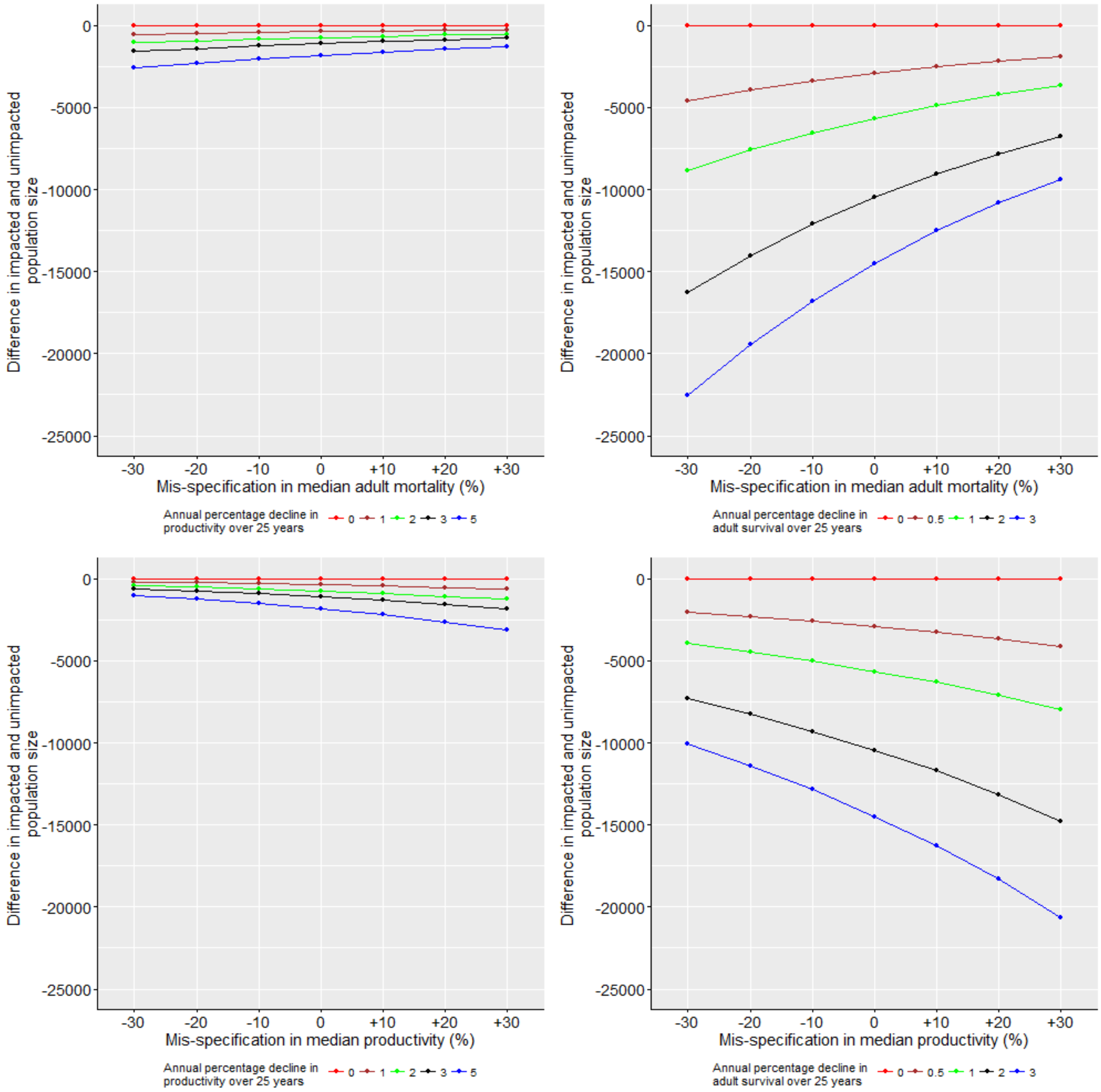


Figure A2.7e. PVA Metric E1 for Fowlsheugh Guillemots – probability of population decline greater than 10% from 2016-2041.

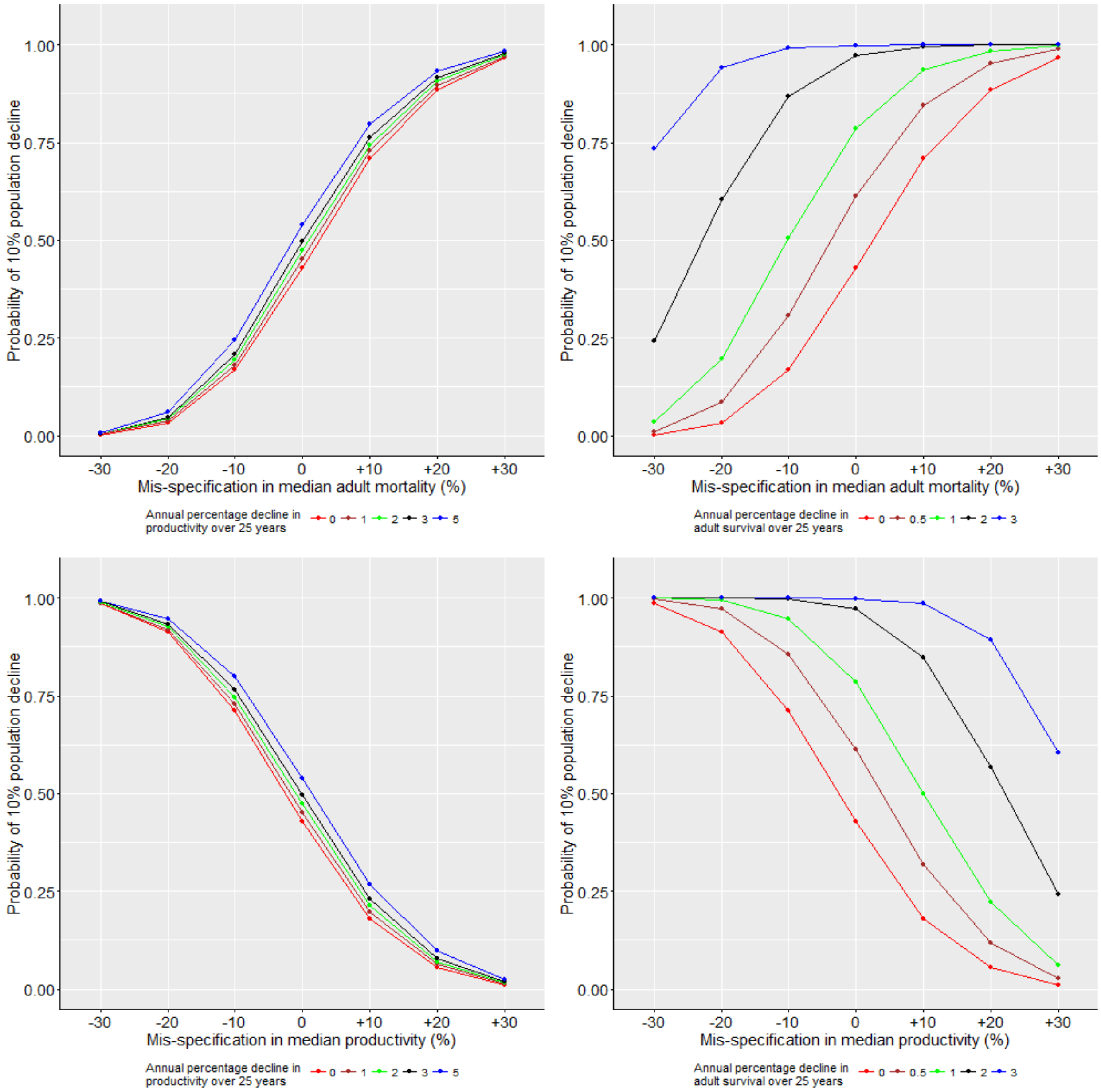


Figure A2.7f. PVA Metric E2 for Fowlsheugh Guillemots – probability of population decline greater than 25% from 2016-2041.

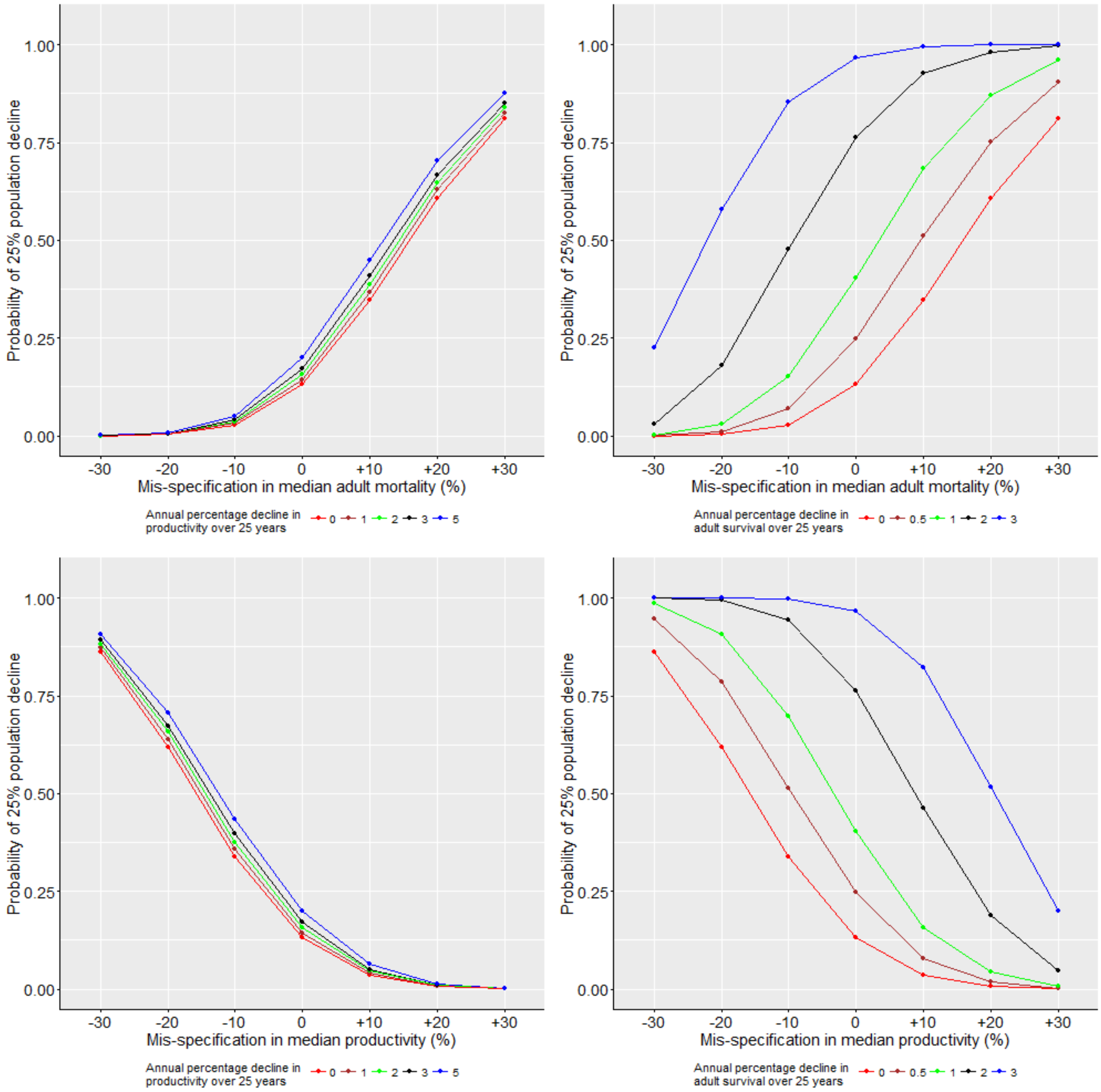


Figure A2.7g. PVA Metric E3 for Fowlsheugh Guillemots – probability of population decline greater than 50% from 2016-2041.

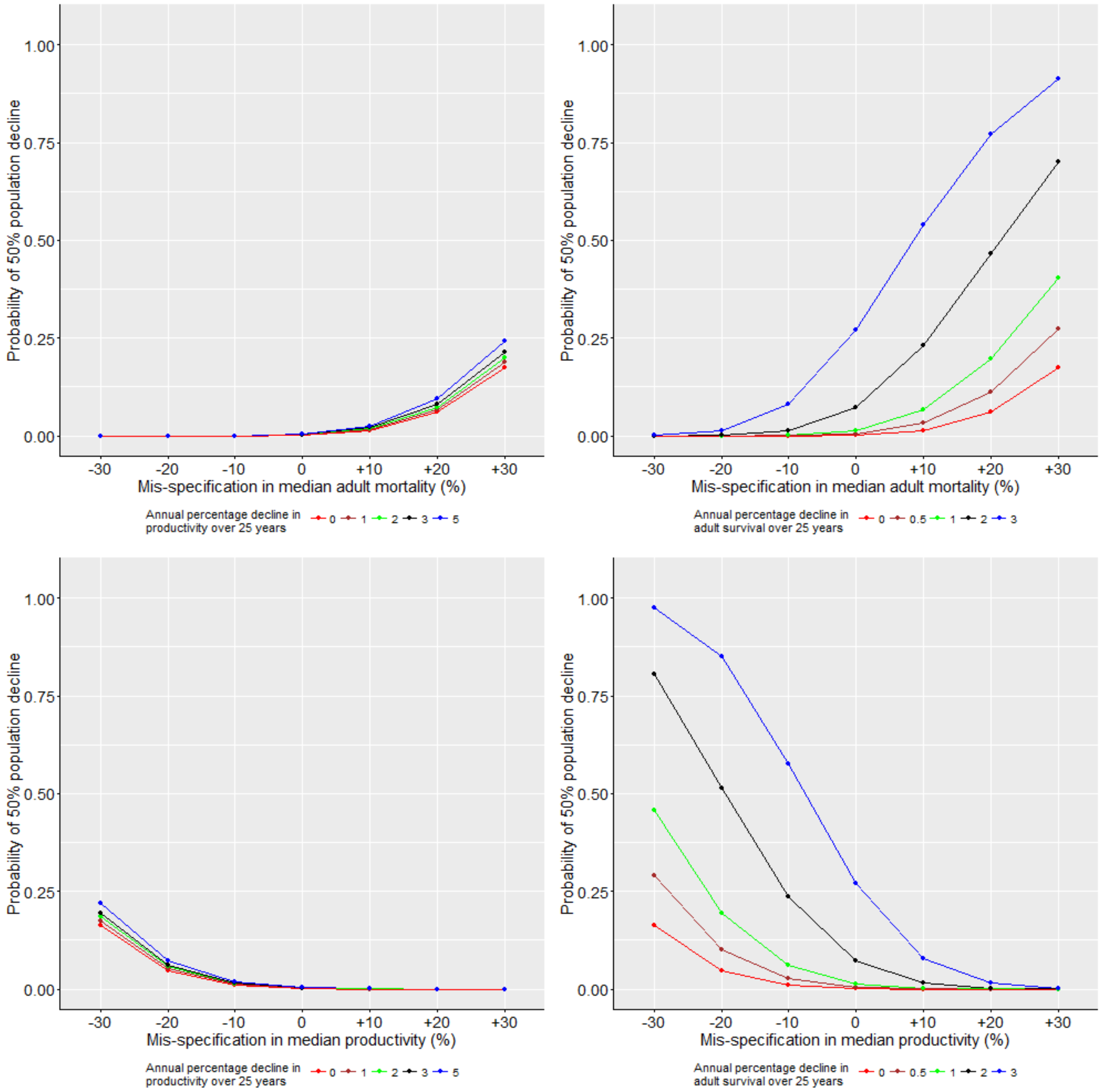
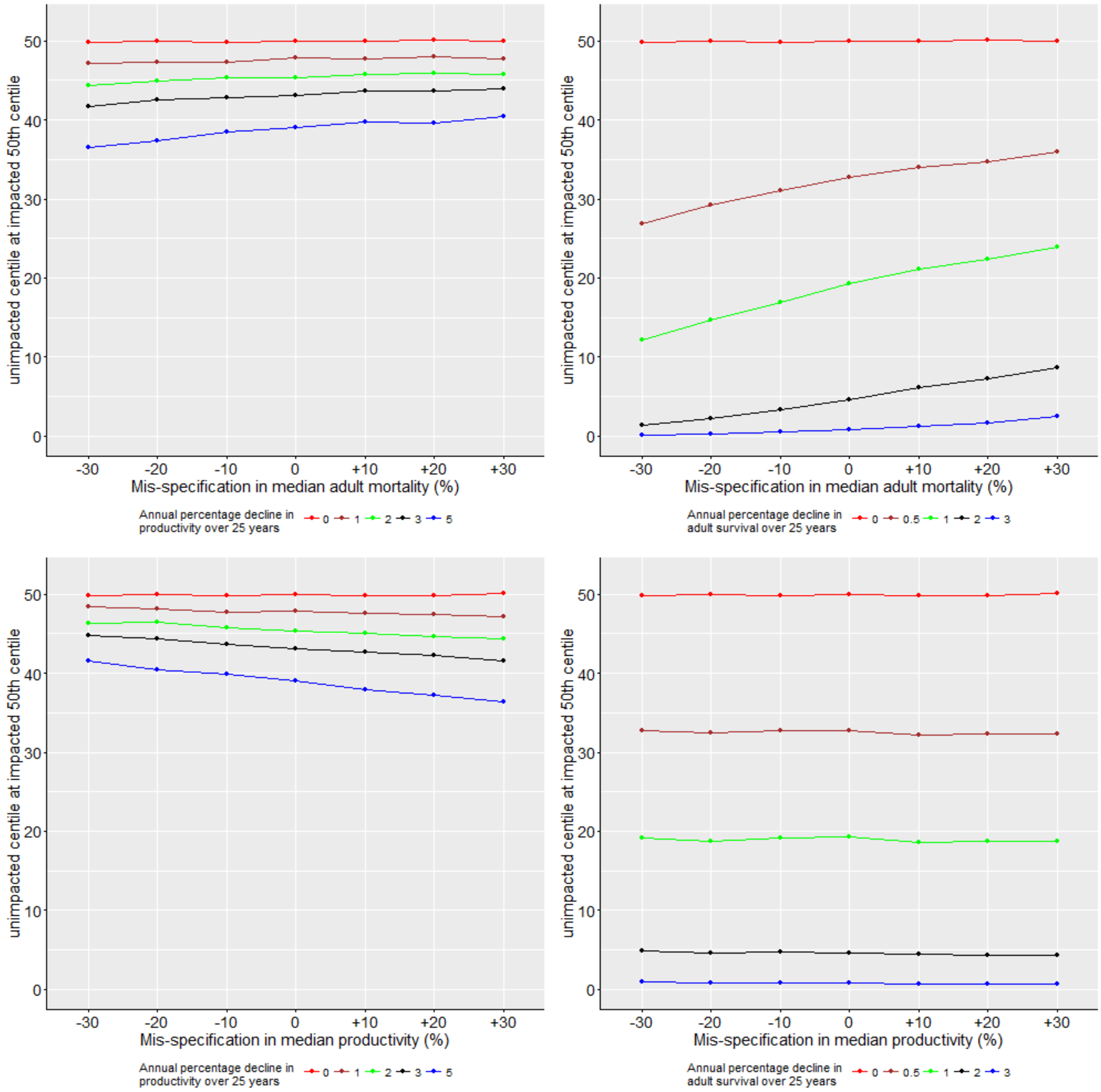


Figure A2.7h. PVA Metric F for Fowlsheugh Guillemots – centile from un-impacted population size equal to the 50th centile of the impacted population size, at 2041.



8. **Guillemots at Buchan Ness to Collieston Coast SPA**:

Figure A2.8a. PVA Metric A for Buchan Ness Guillemots – ratio of population growth rate from 2016-2041, comparing impacted population vs. un-impacted population.

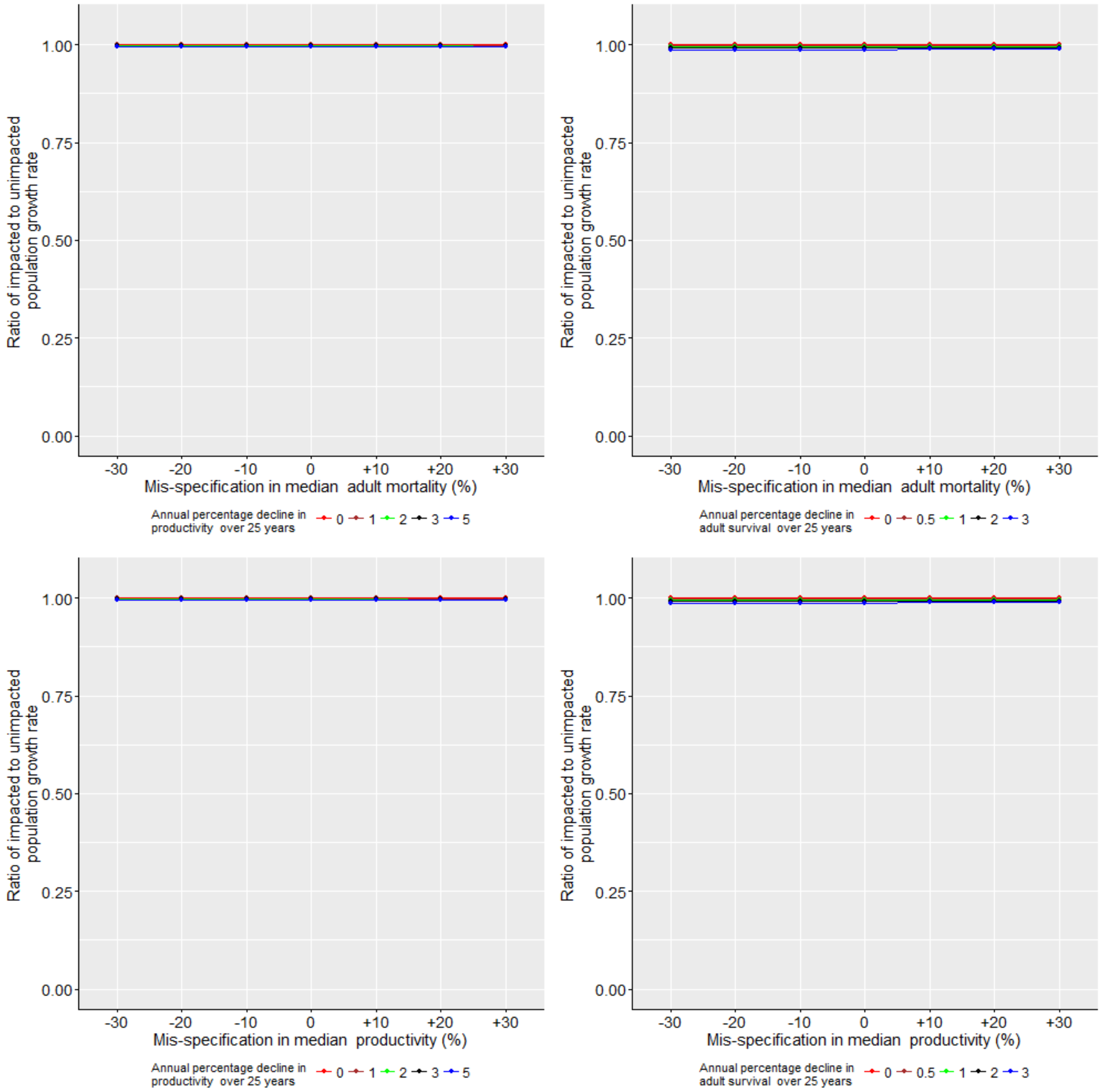


Figure A2.8b. PVA Metric B for Buchan Ness Guillemots – ratio of population size at 2041, comparing impacted population vs. un-impacted population.

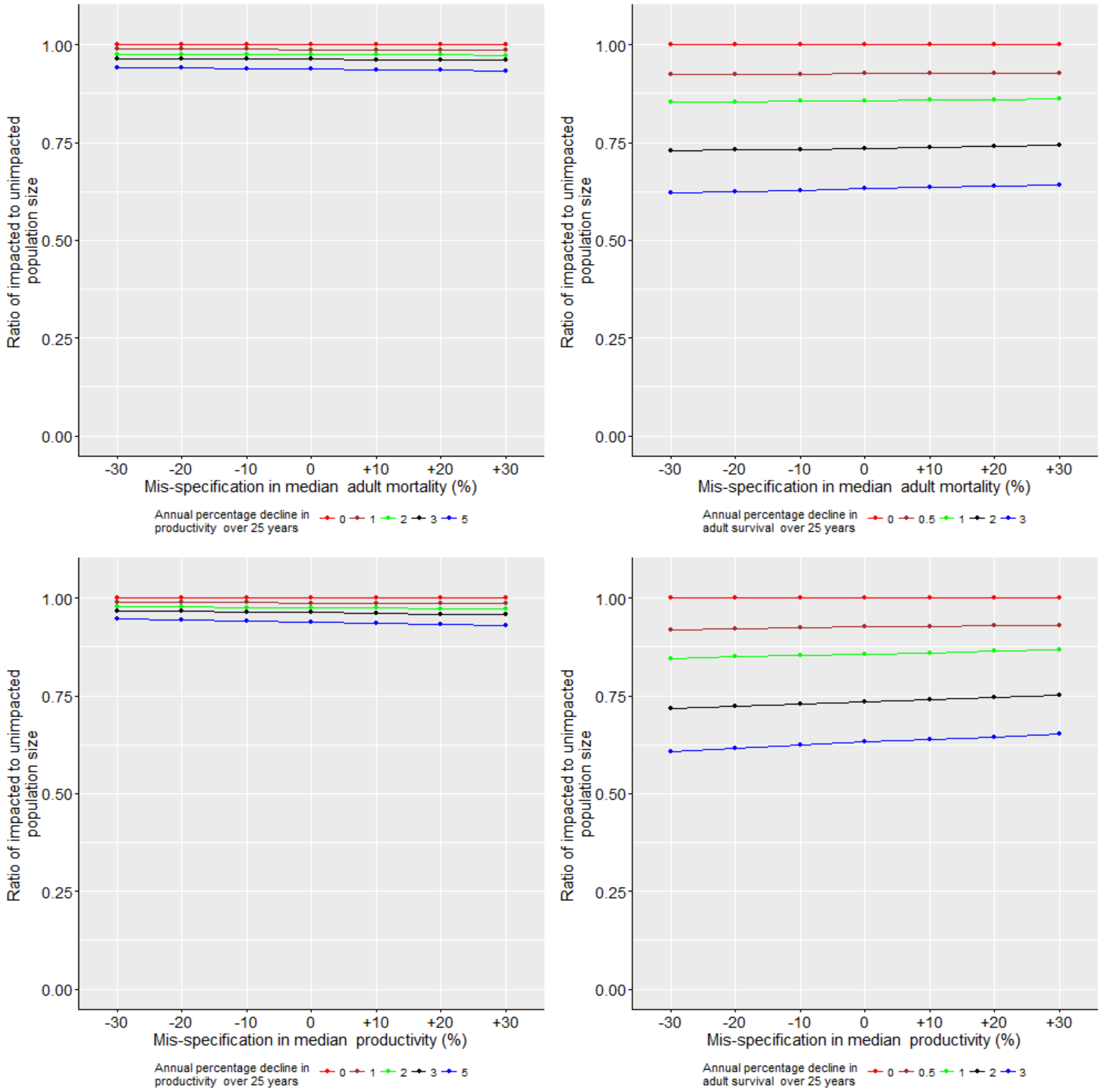


Figure A2.8c. PVA Metric C for Buchan Ness Guillemots – difference in population growth rate from 2016-2041, comparing impacted population vs. un-impacted population.

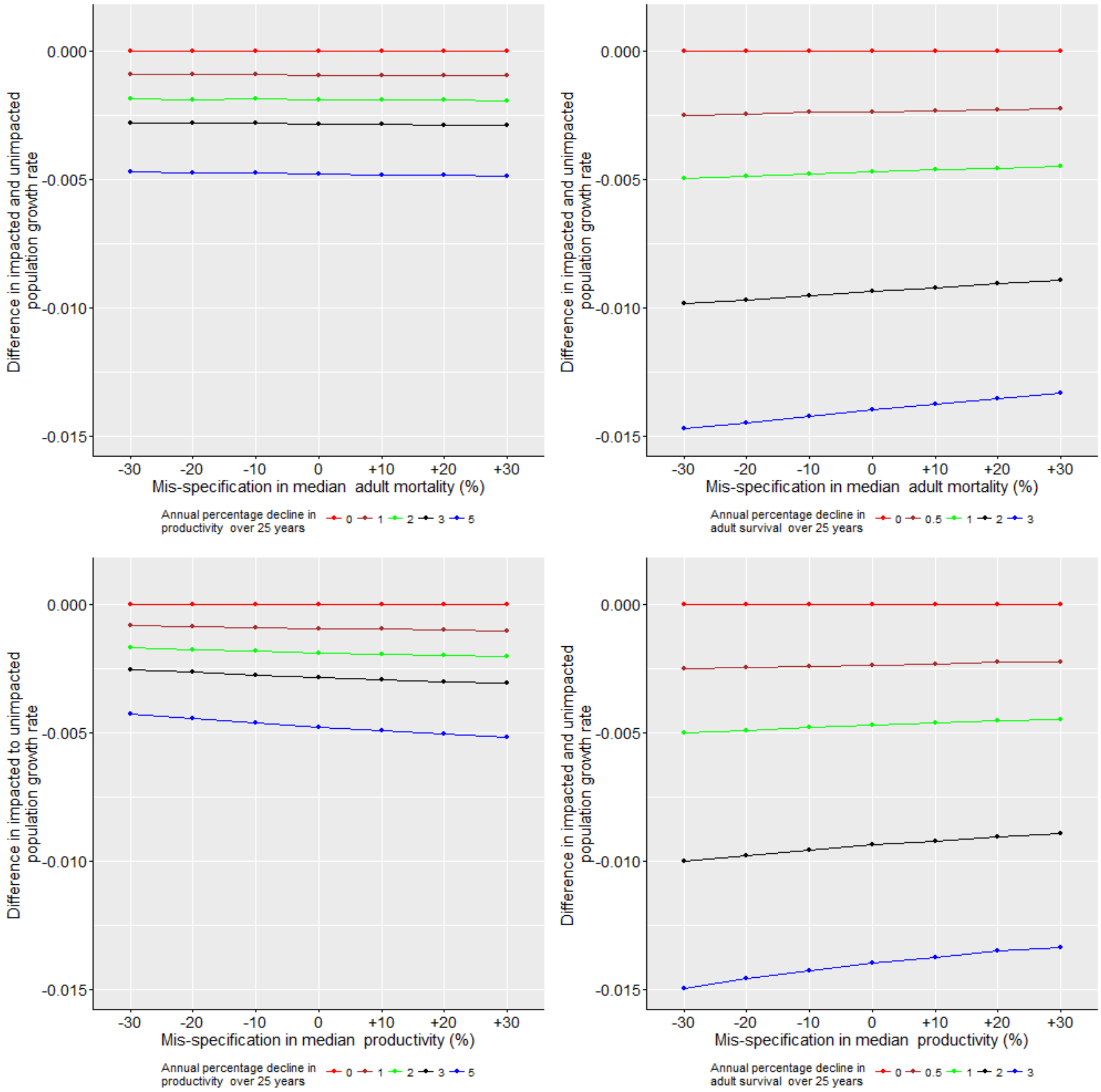


Figure A2.8d. PVA Metric D for Buchan Ness Guillemots – difference in population size at 2041, comparing impacted population vs. un-impacted population.

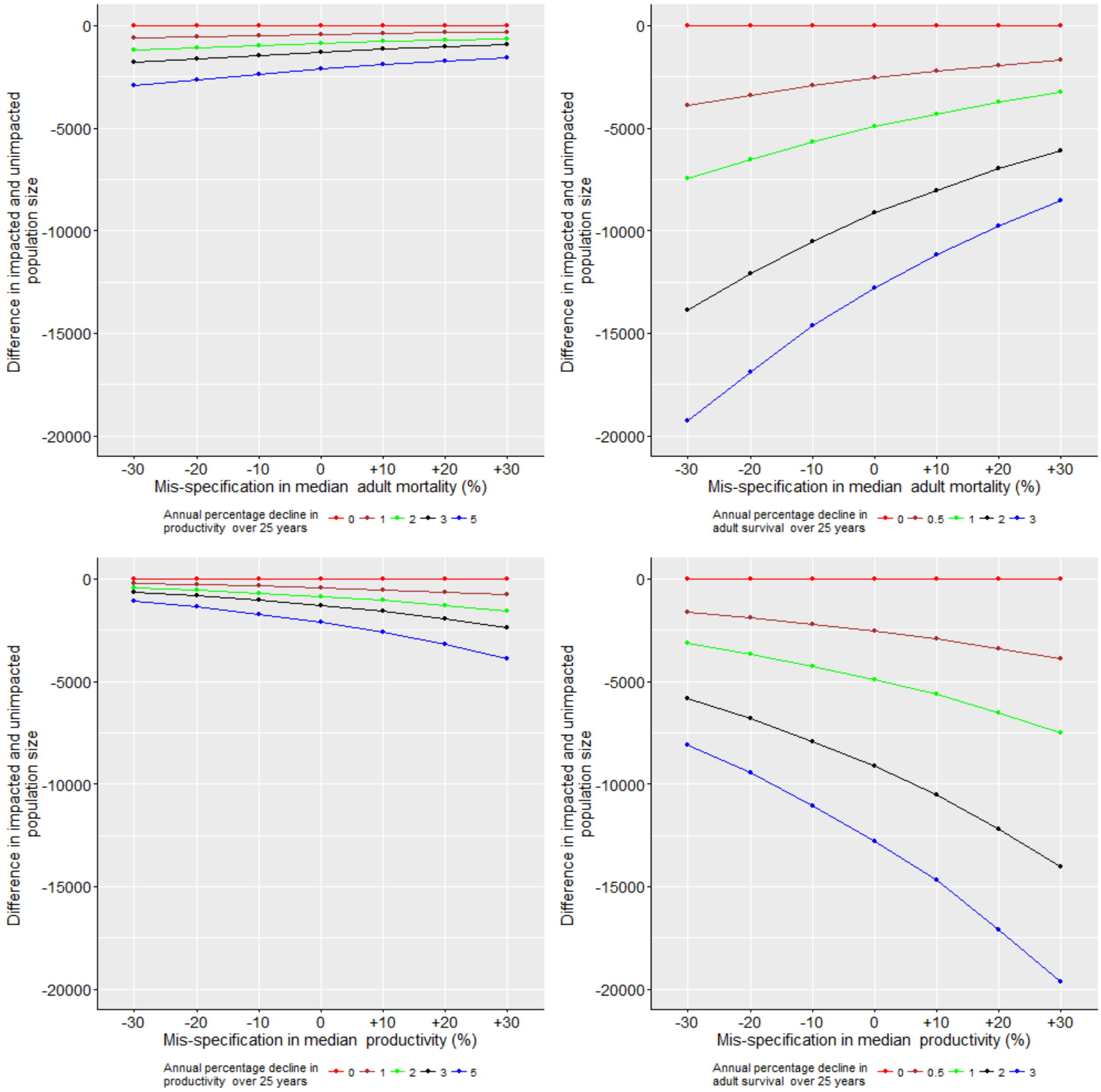


Figure A2.8e. PVA Metric E1 for Buchan Ness Guillemots – probability of population decline greater than 10% from 2016-2041.

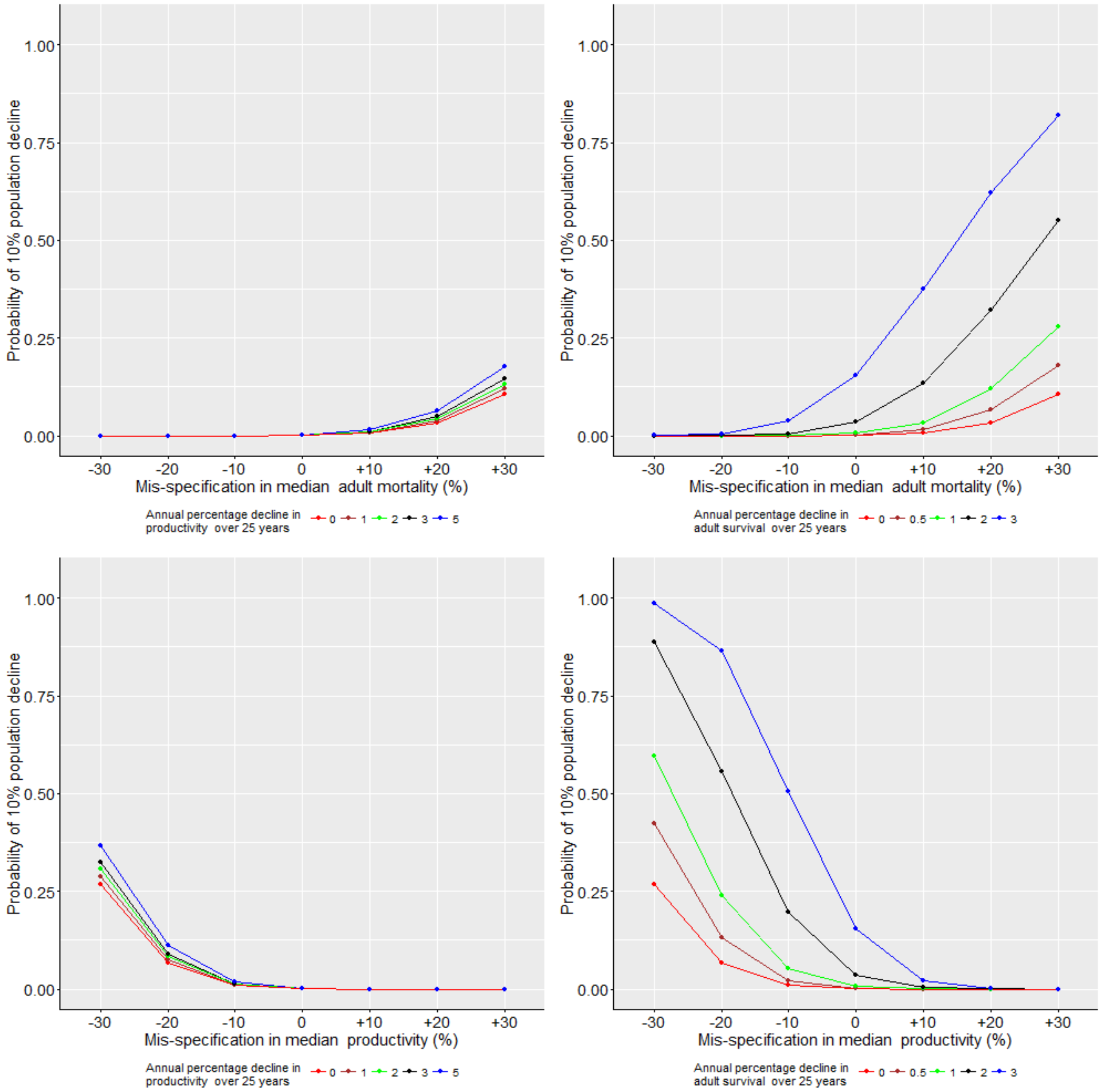


Figure A2.8f. PVA Metric E2 for Buchan Ness Guillemots – probability of population decline greater than 25% from 2016-2041.

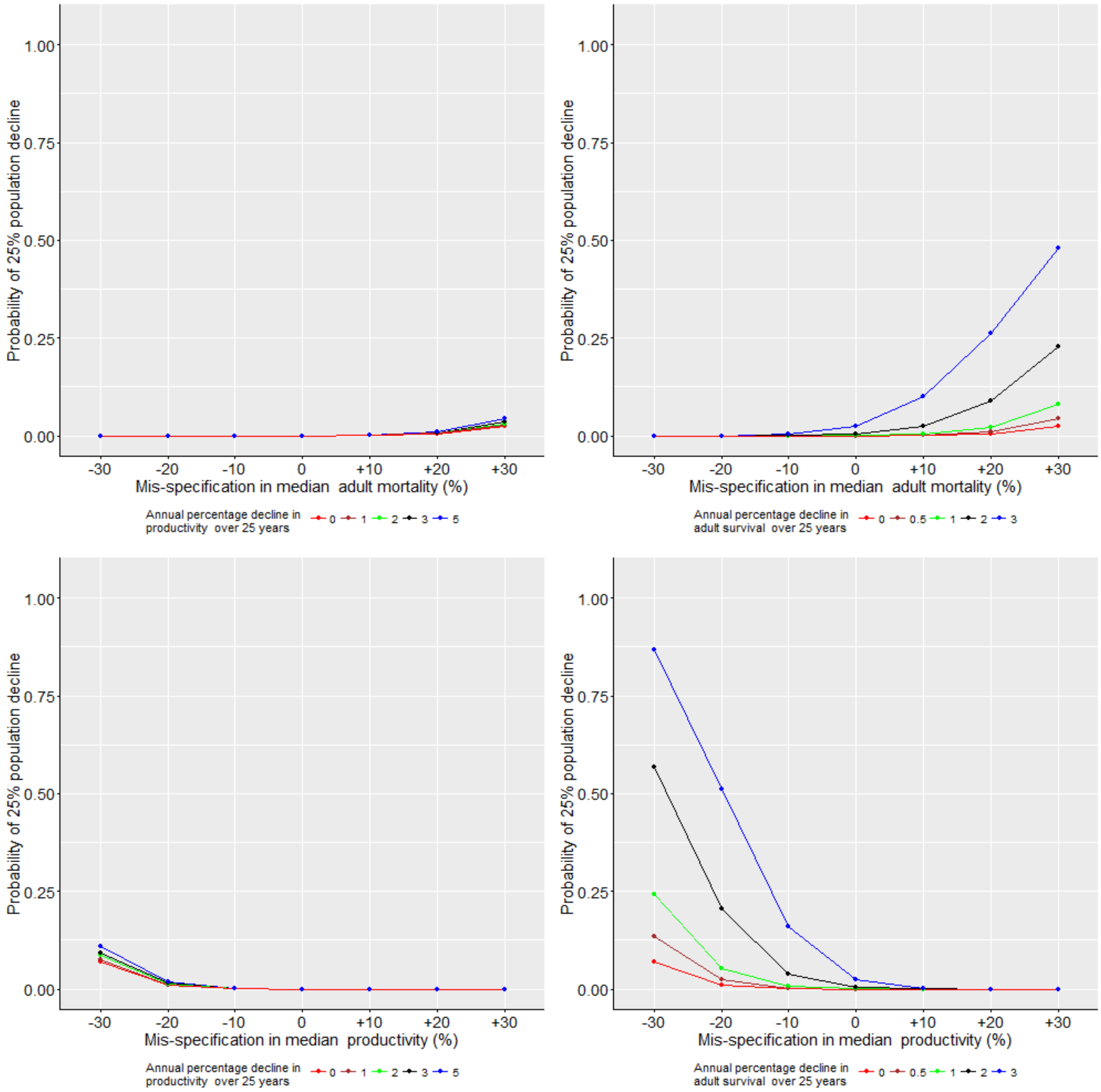


Figure A2.8g. PVA Metric E3 for Buchan Ness Guillemots – probability of population decline greater than 50% from 2016-2041.

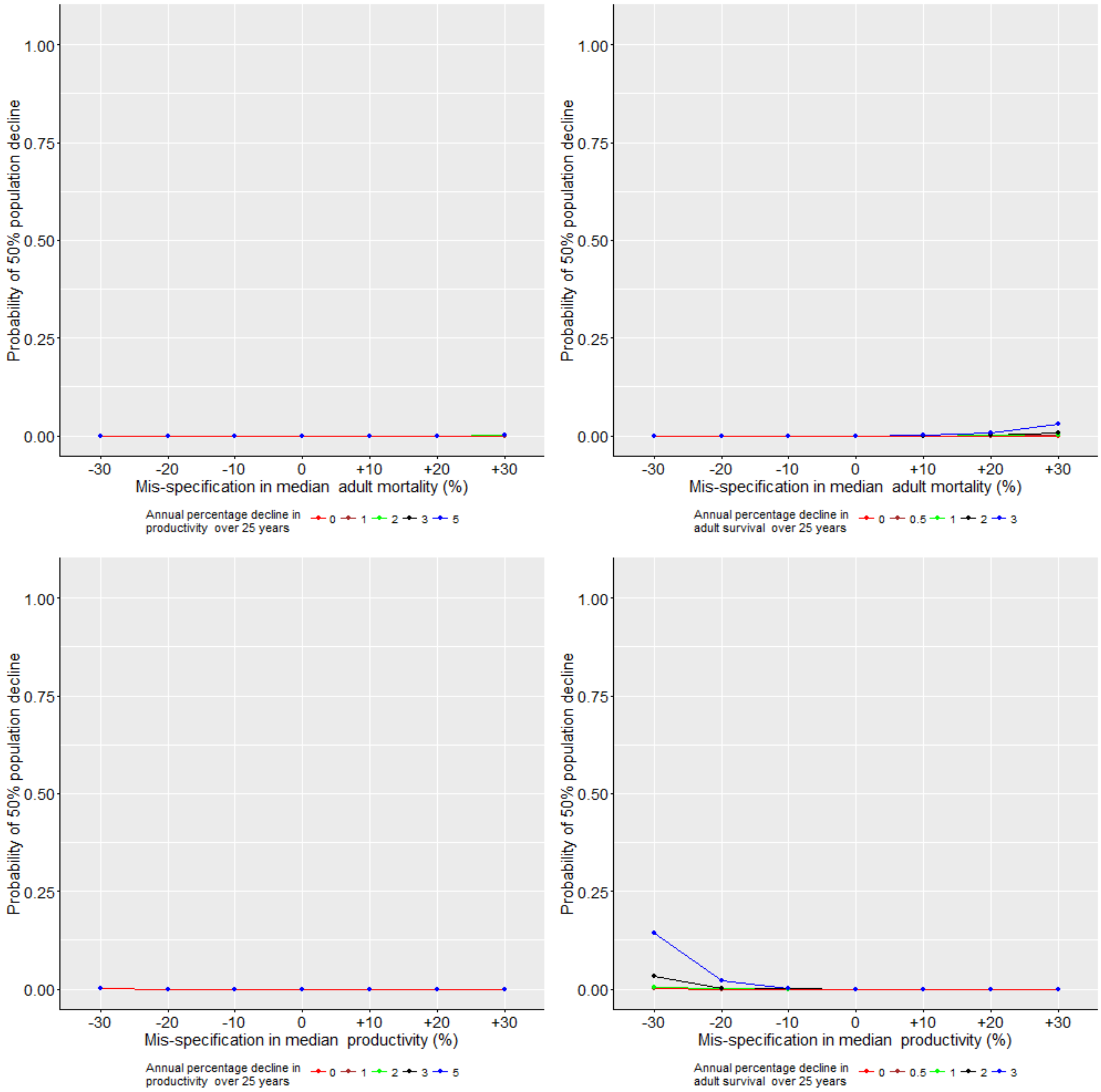
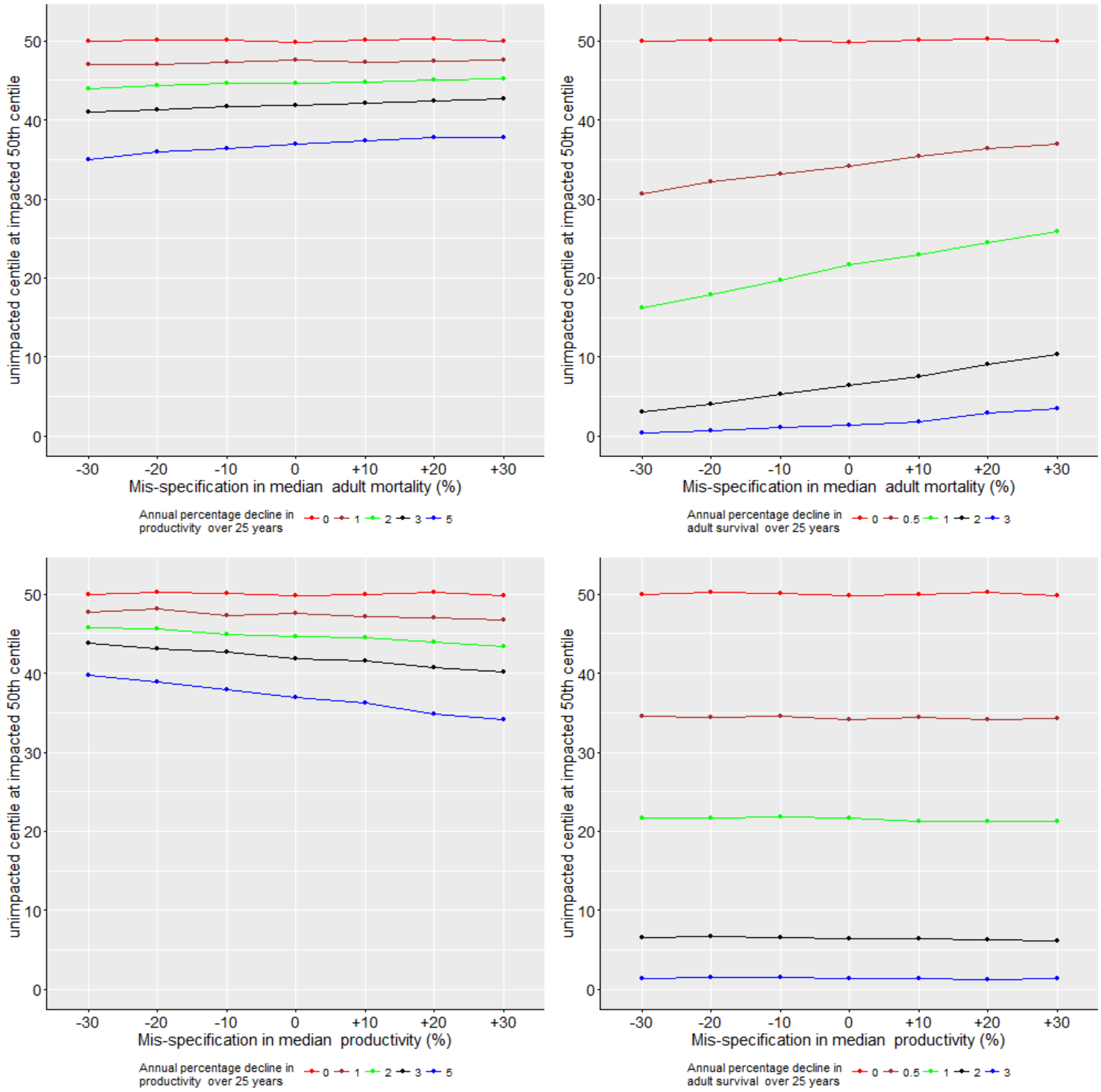


Figure A2.8h. PVA Metric F for Buchan Ness Guillemots – centile from un-impacted population size equal to the 50th centile of the impacted population size, at 2041.



9. **Razorbills at Forth Islands SPA**:

Figure A2.9a. PVA Metric A for Forth Razorbills – ratio of population growth rate from 2016-2041, comparing impacted population vs. un-impacted population.

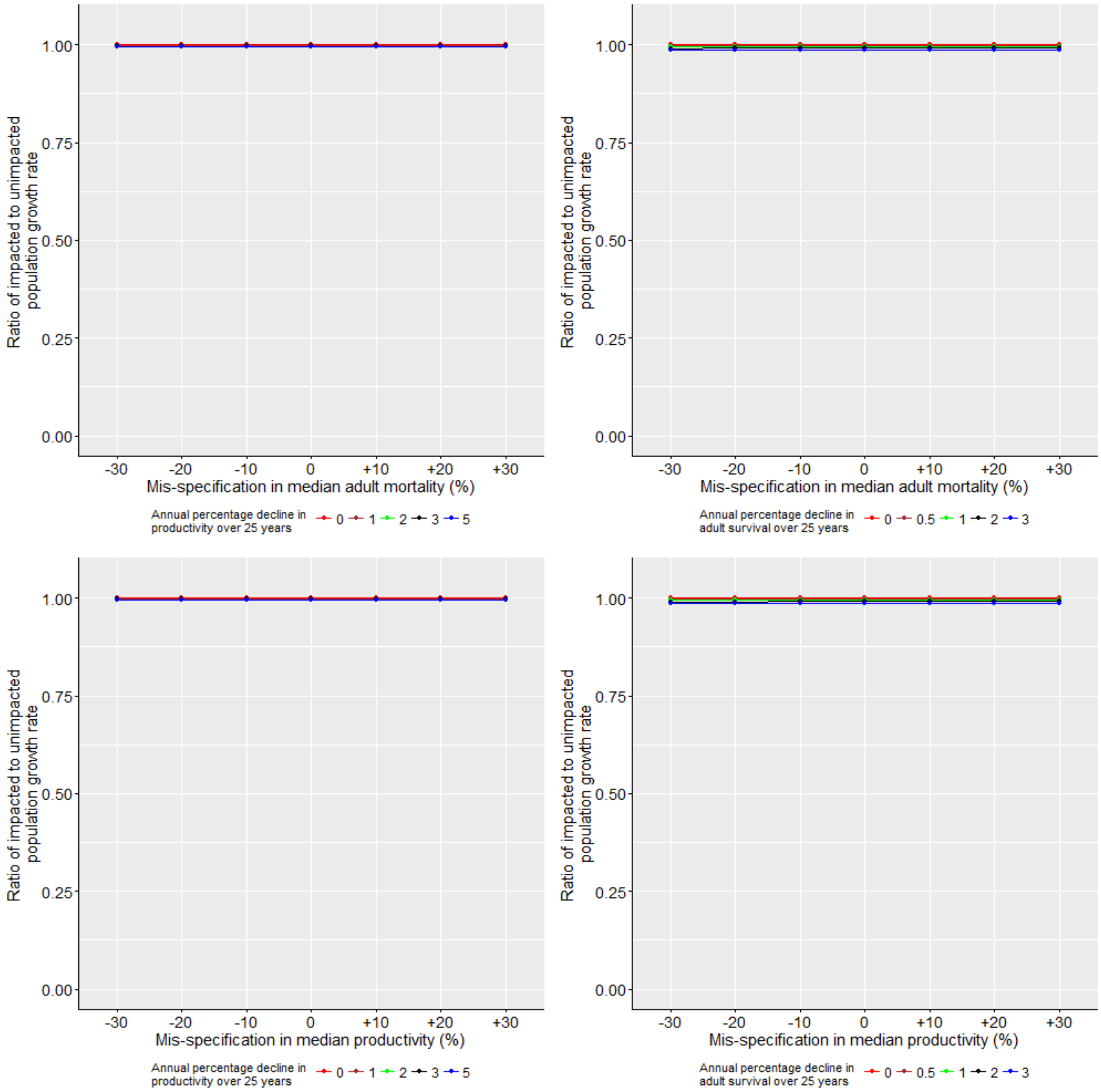


Figure A2.9b. PVA Metric B for Forth Razorbills – ratio of population size at 2041, comparing impacted population vs. un-impacted population.

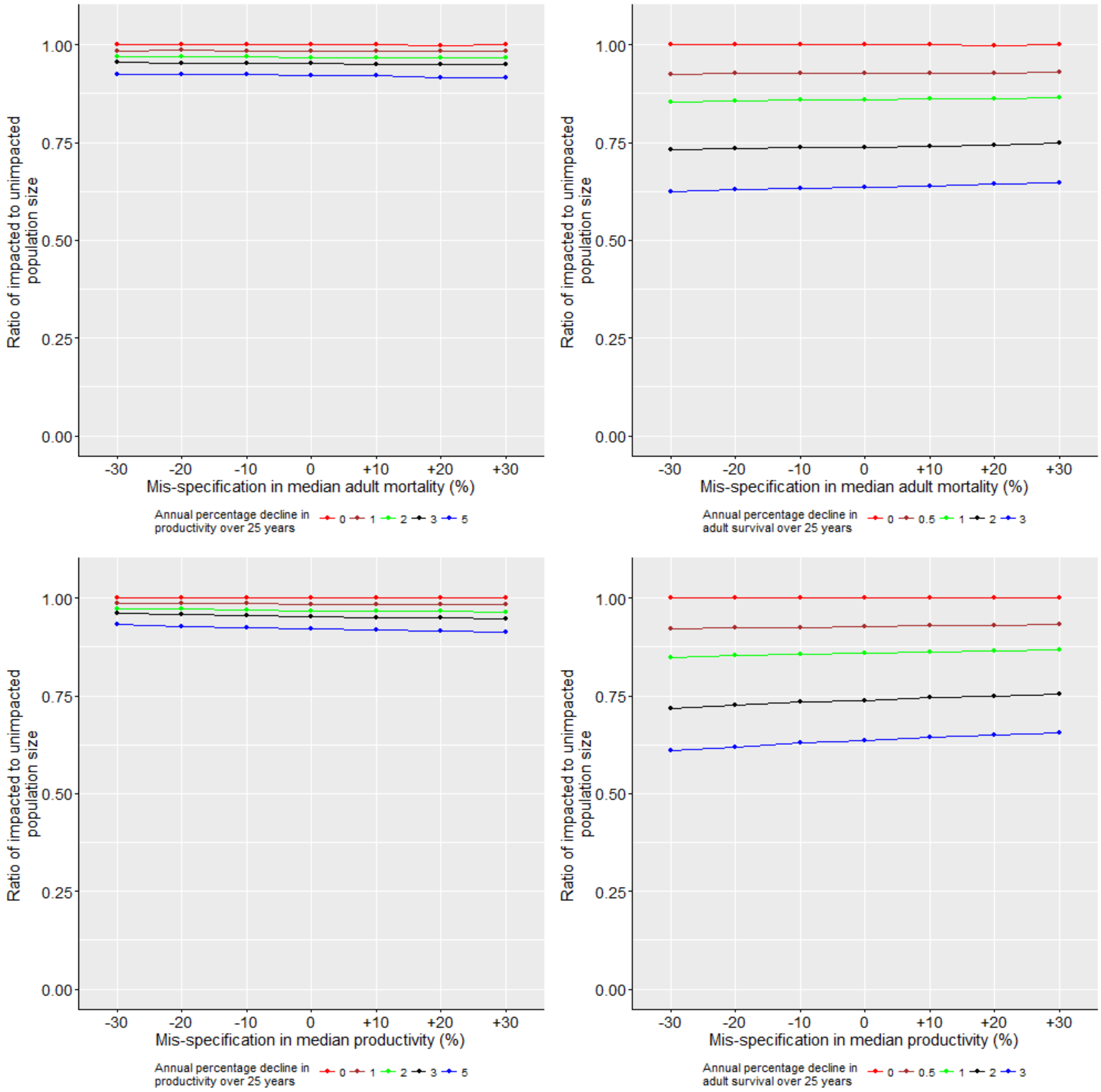


Figure A2.9c. PVA Metric C for Forth Razorbills – difference in population growth rate from 2016-2041, comparing impacted population vs. un-impacted population.

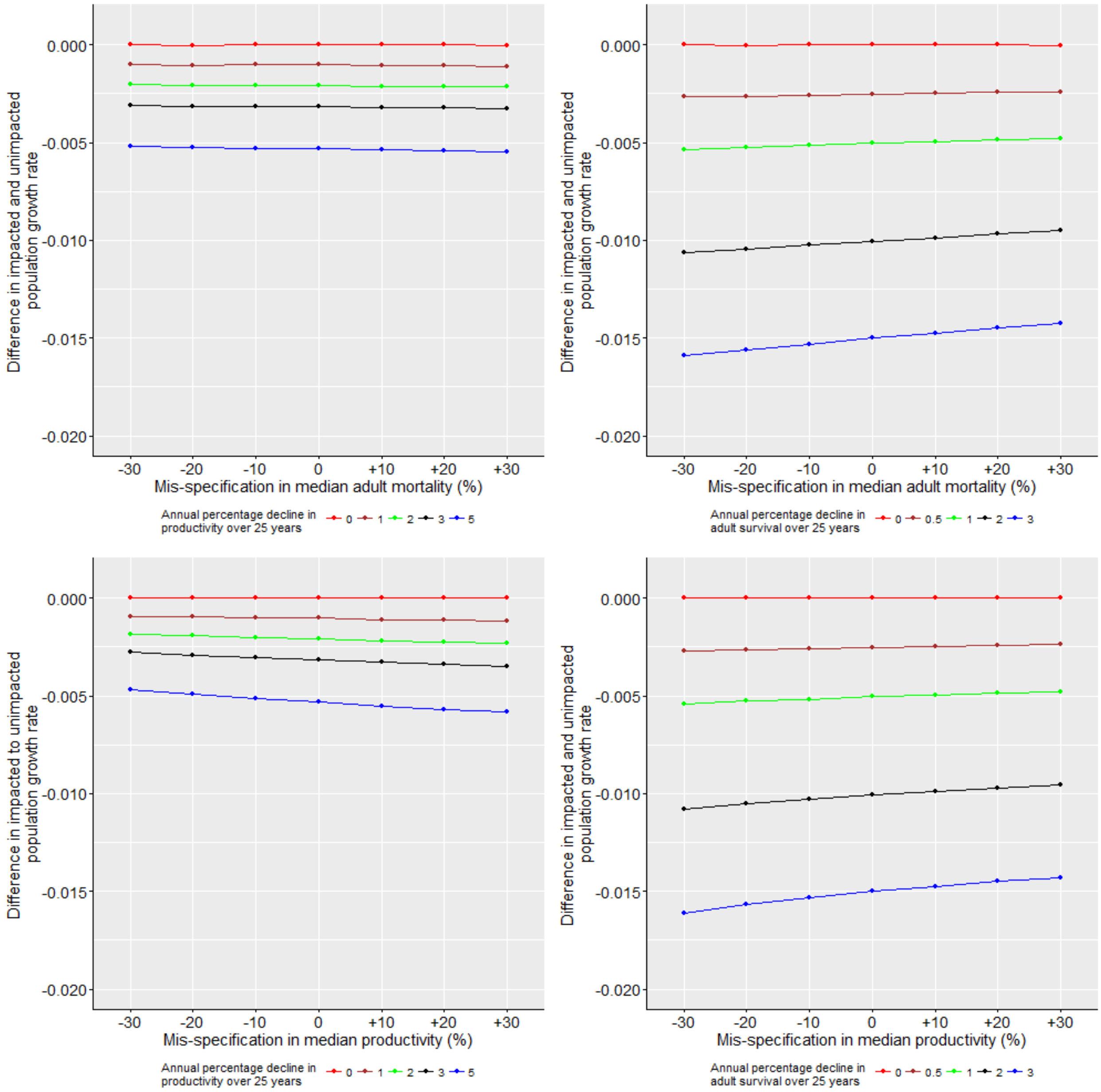


Figure A2.9d. PVA Metric D for Forth Razorbills – difference in population size at 2041, comparing impacted population vs. un-impacted population.

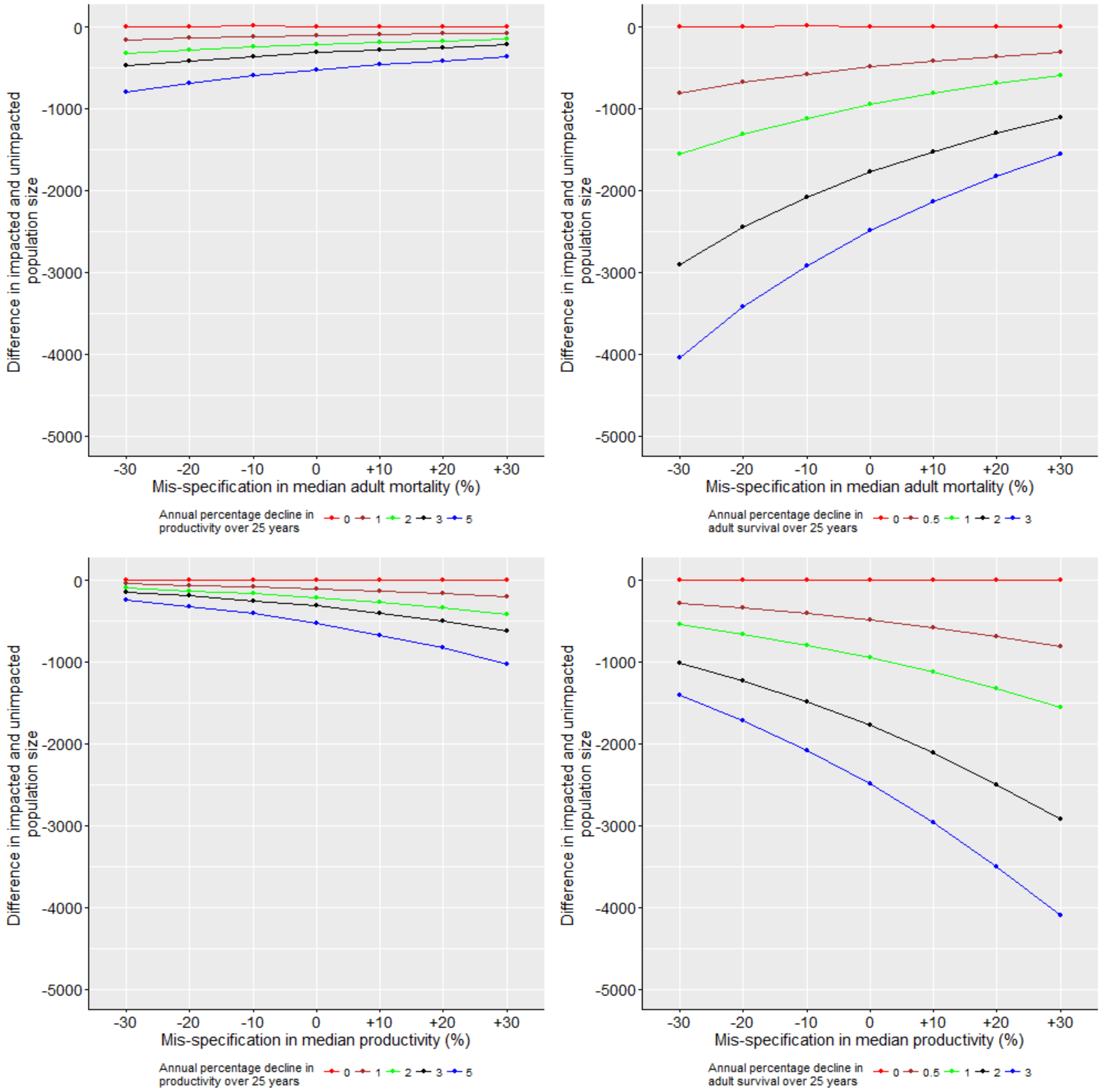


Figure A2.9e. PVA Metric E1 for Forth Razorbills – probability of population decline greater than 10% from 2016-2041.

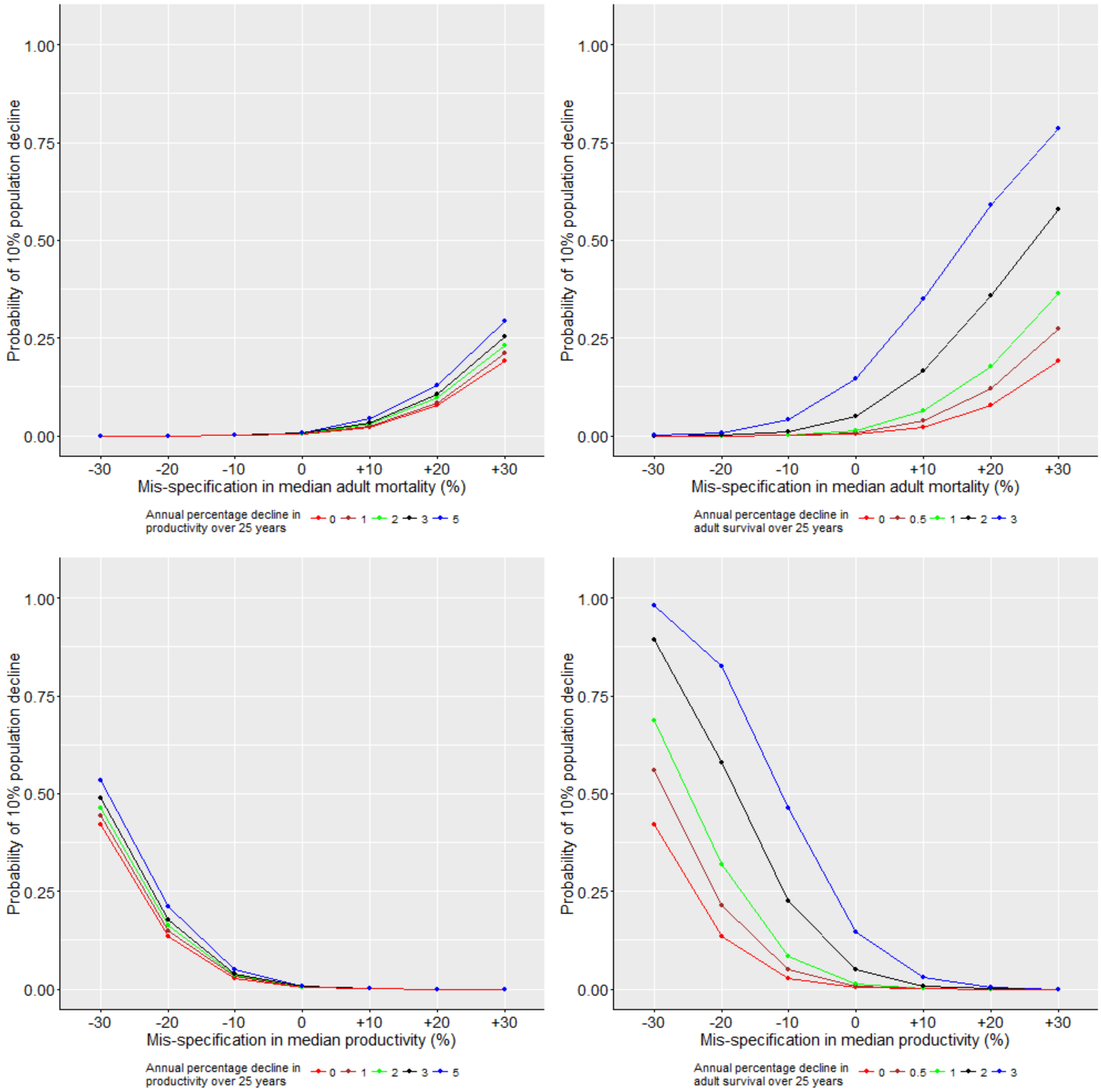


Figure A2.9f. PVA Metric E2 for Forth Razorbills – probability of population decline greater than 25% from 2016-2041.

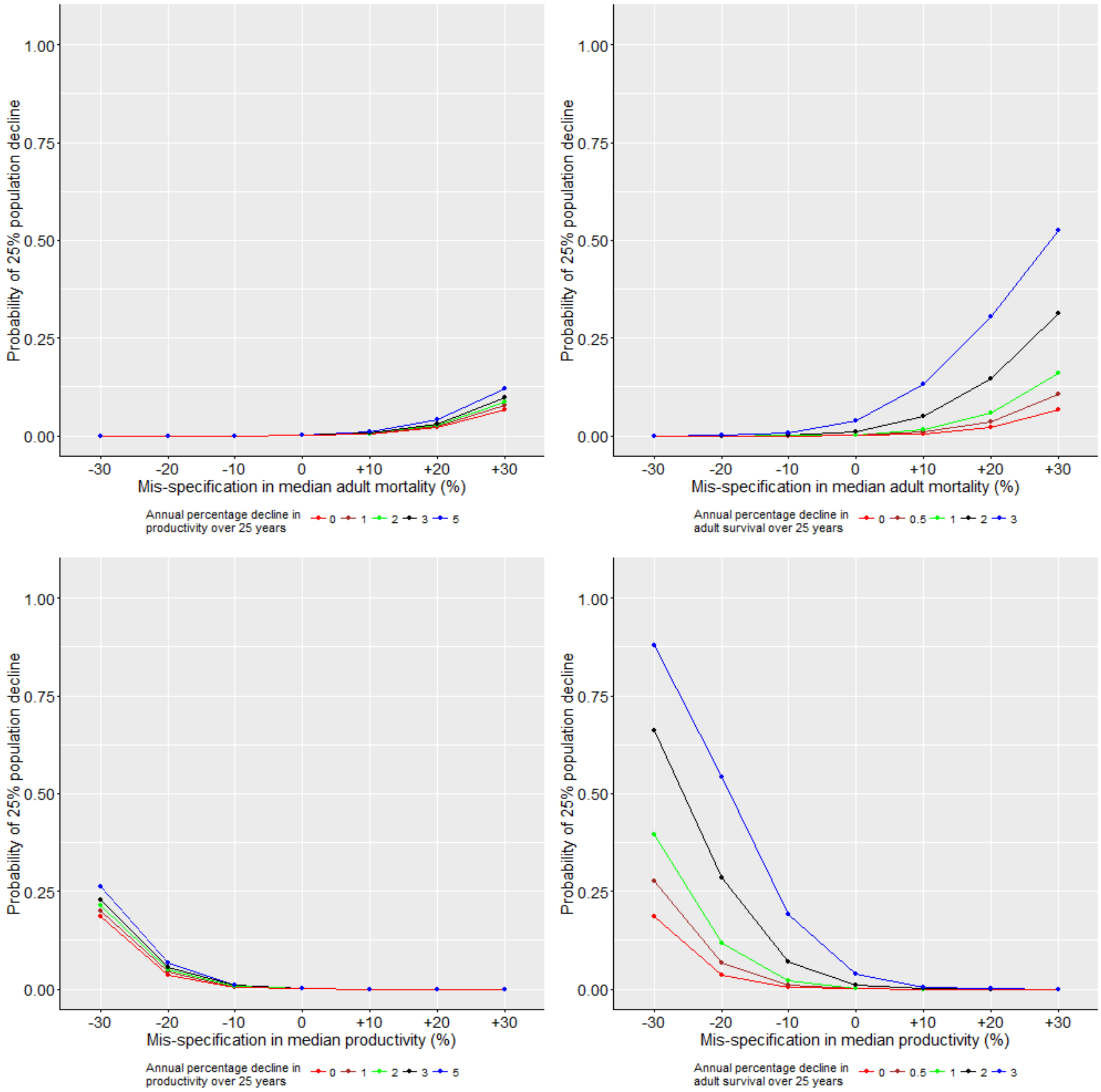


Figure A2.9g. PVA Metric E3 for Forth Razorbills – probability of population decline greater than 50% from 2016-2041.

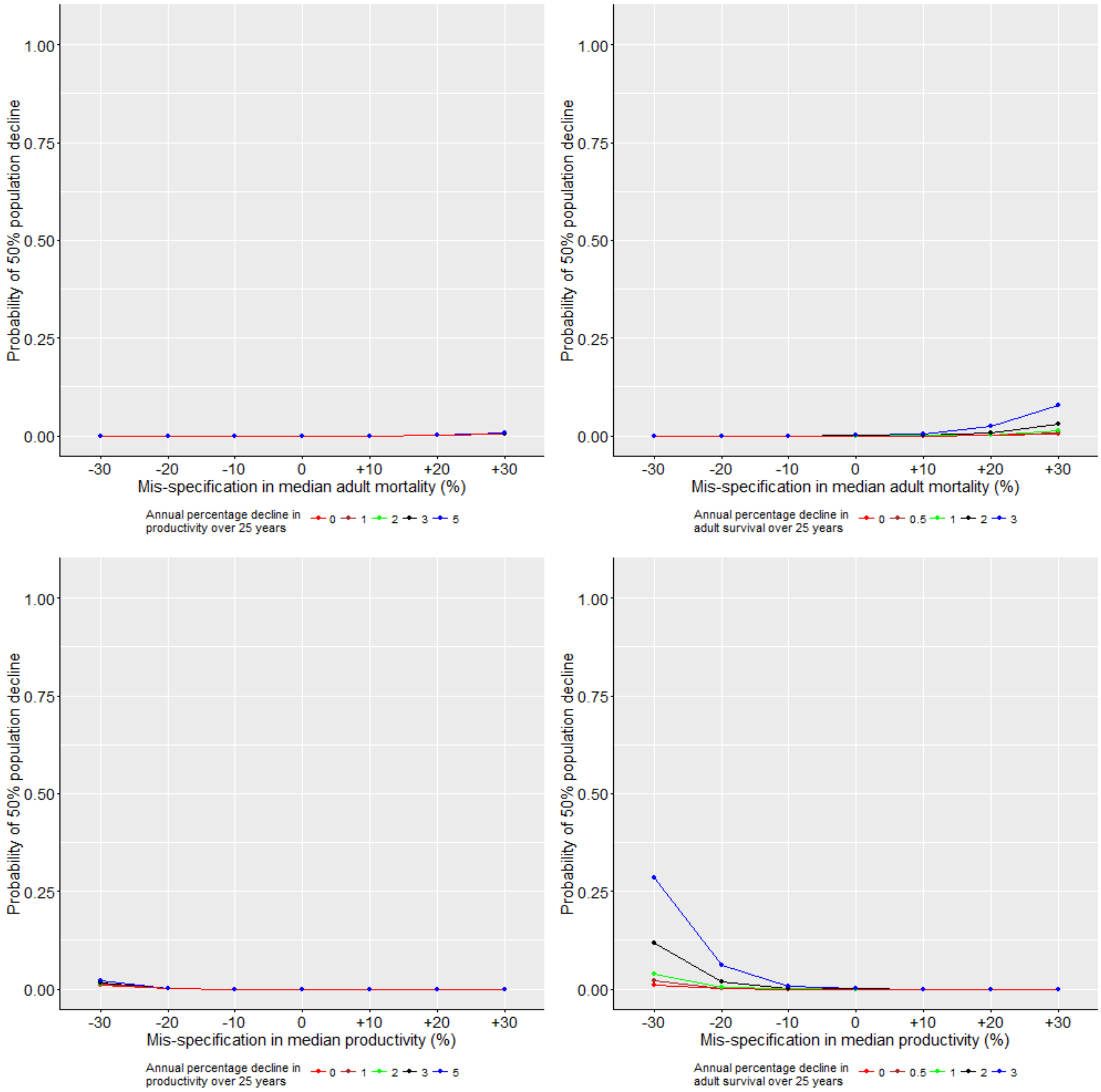
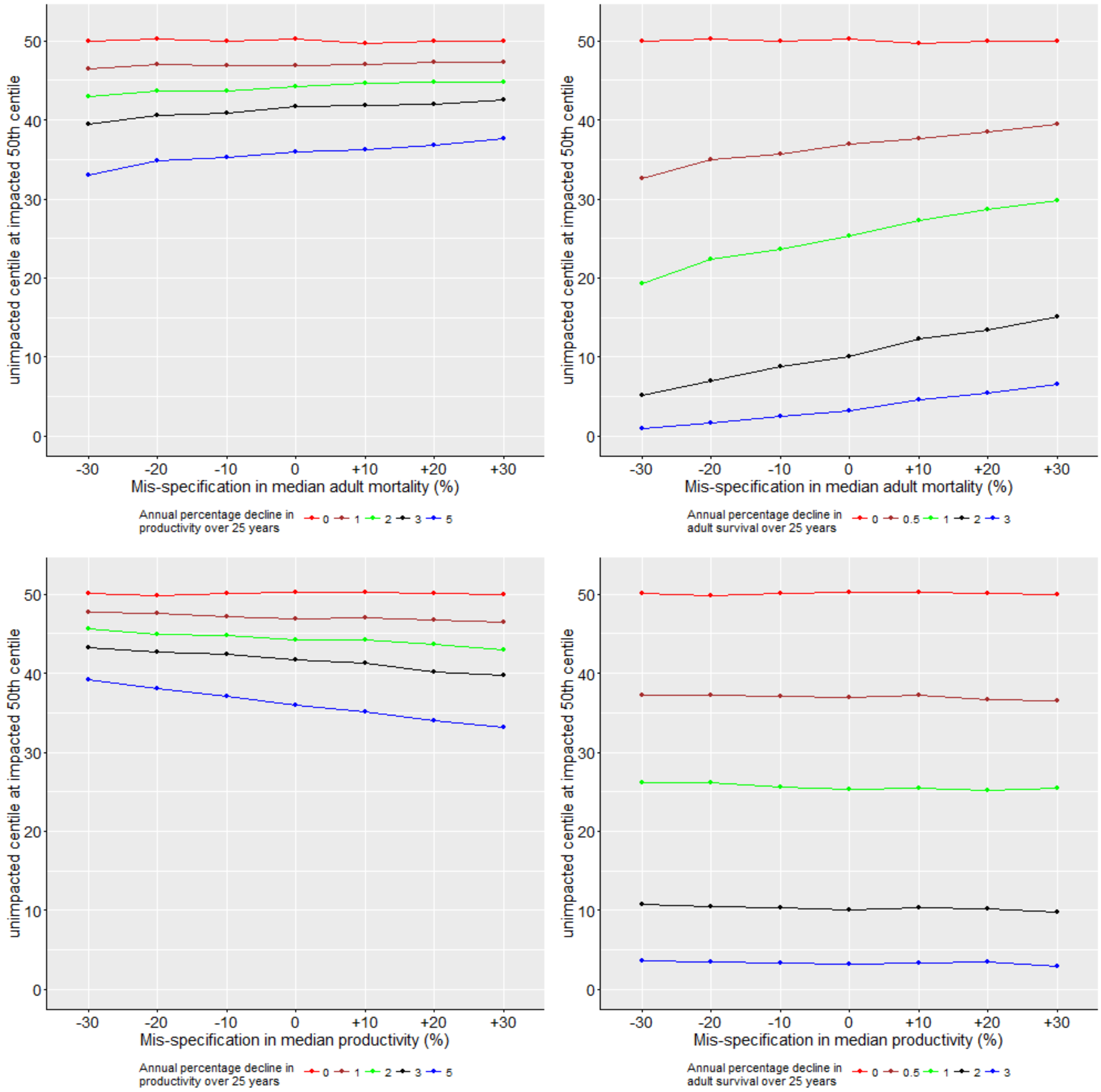


Figure A2.9h. PVA Metric F for Forth Razorbills – centile from un-impacted population size equal to the 50th centile of the impacted population size, at 2041.



10. **Razorbills at St Abb's Head SPA**:

Figure A2.10a. PVA Metric A for St Abb's Razorbills – ratio of population growth rate from 2016-2041, comparing impacted population vs. un-impacted population.

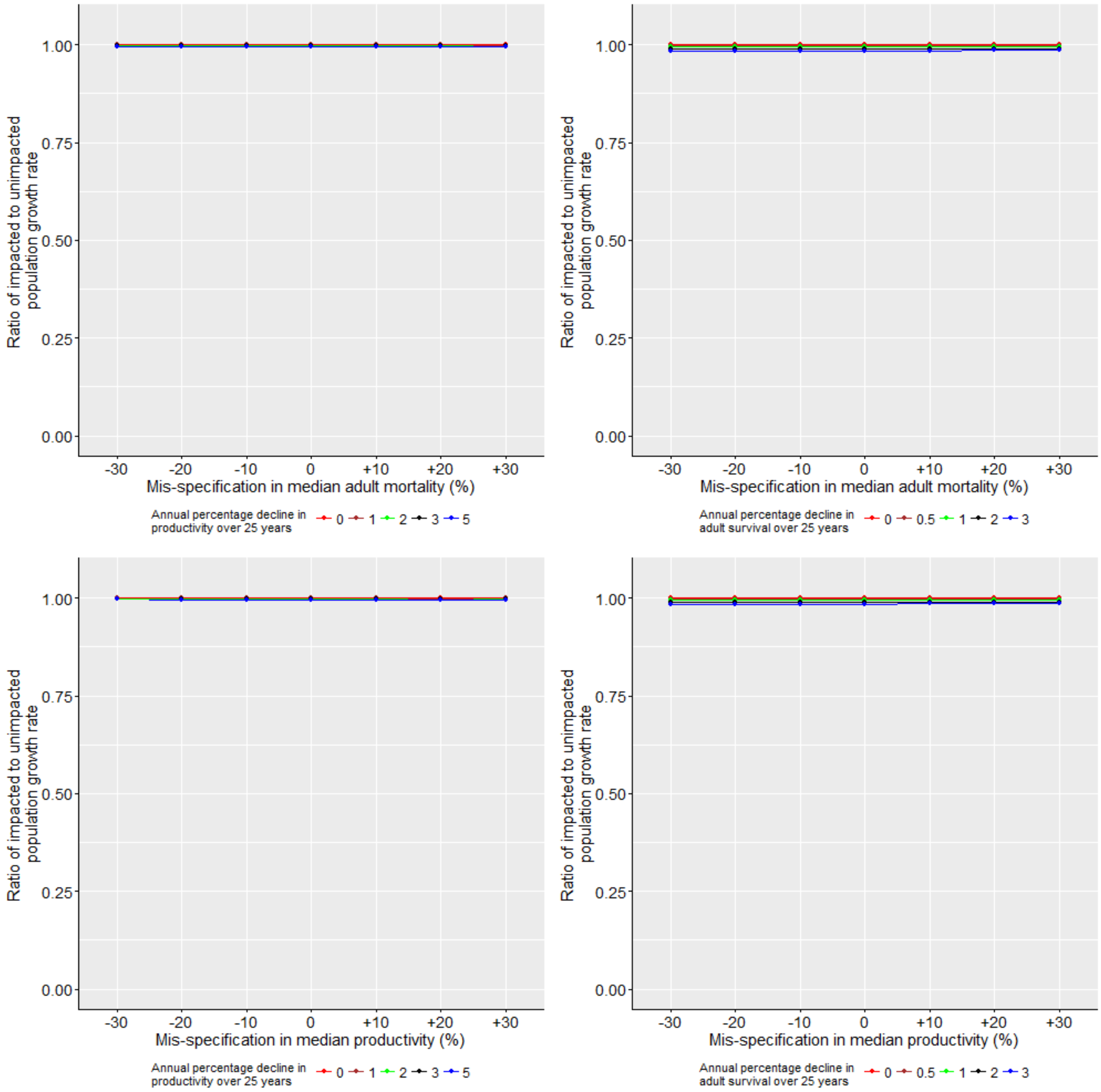


Figure A2.10b. PVA Metric B for St Abb's Razorbills – ratio of population size at 2041, comparing impacted population vs. un-impacted population.

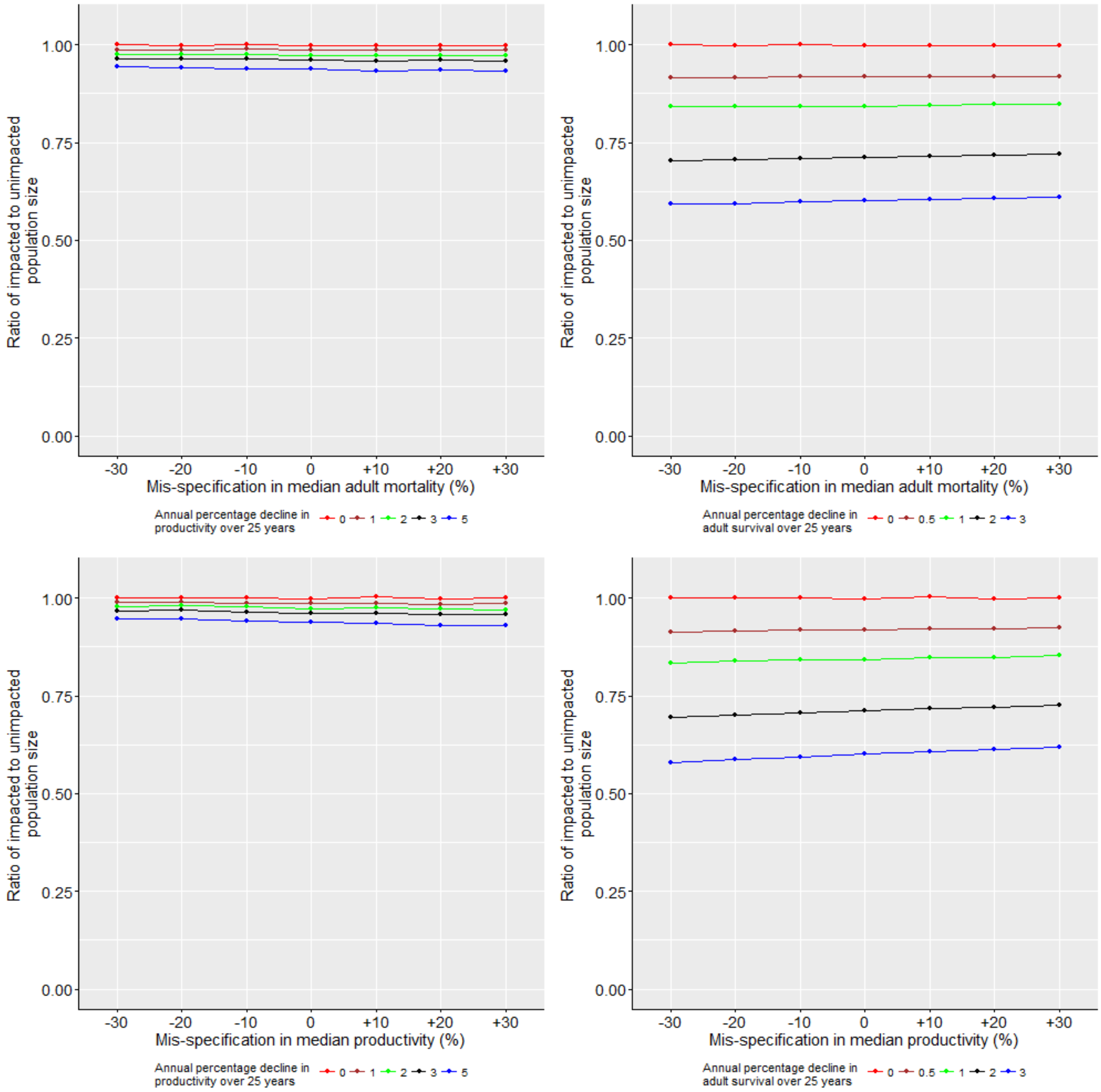


Figure A2.10c. PVA Metric C for St Abb's Razorbills – difference in population growth rate from 2016-2041, comparing impacted population vs. un-impacted population.

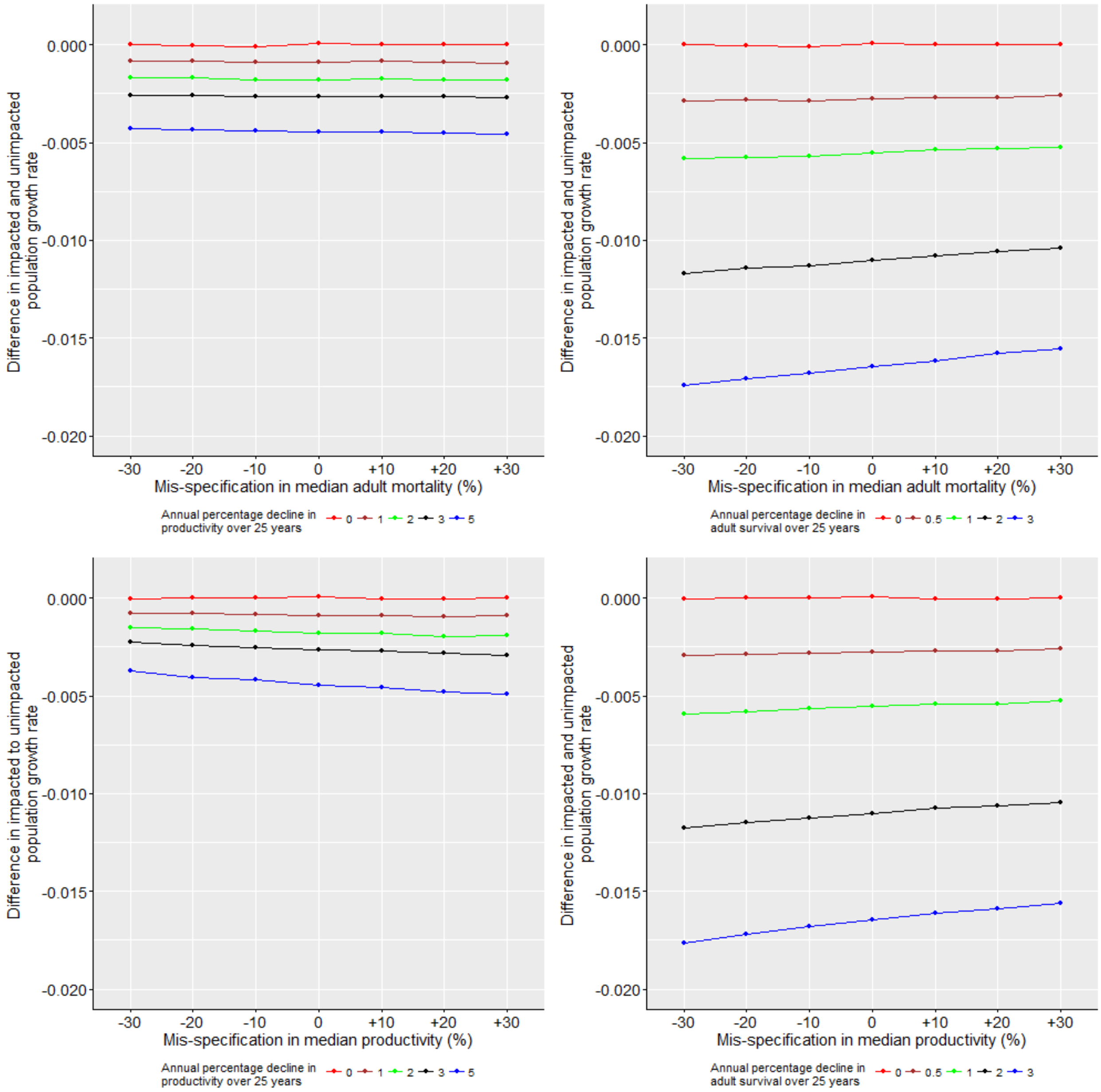


Figure A2.10d. PVA Metric D for St Abb's Razorbills – difference in population size at 2041, comparing impacted population vs. un-impacted population.

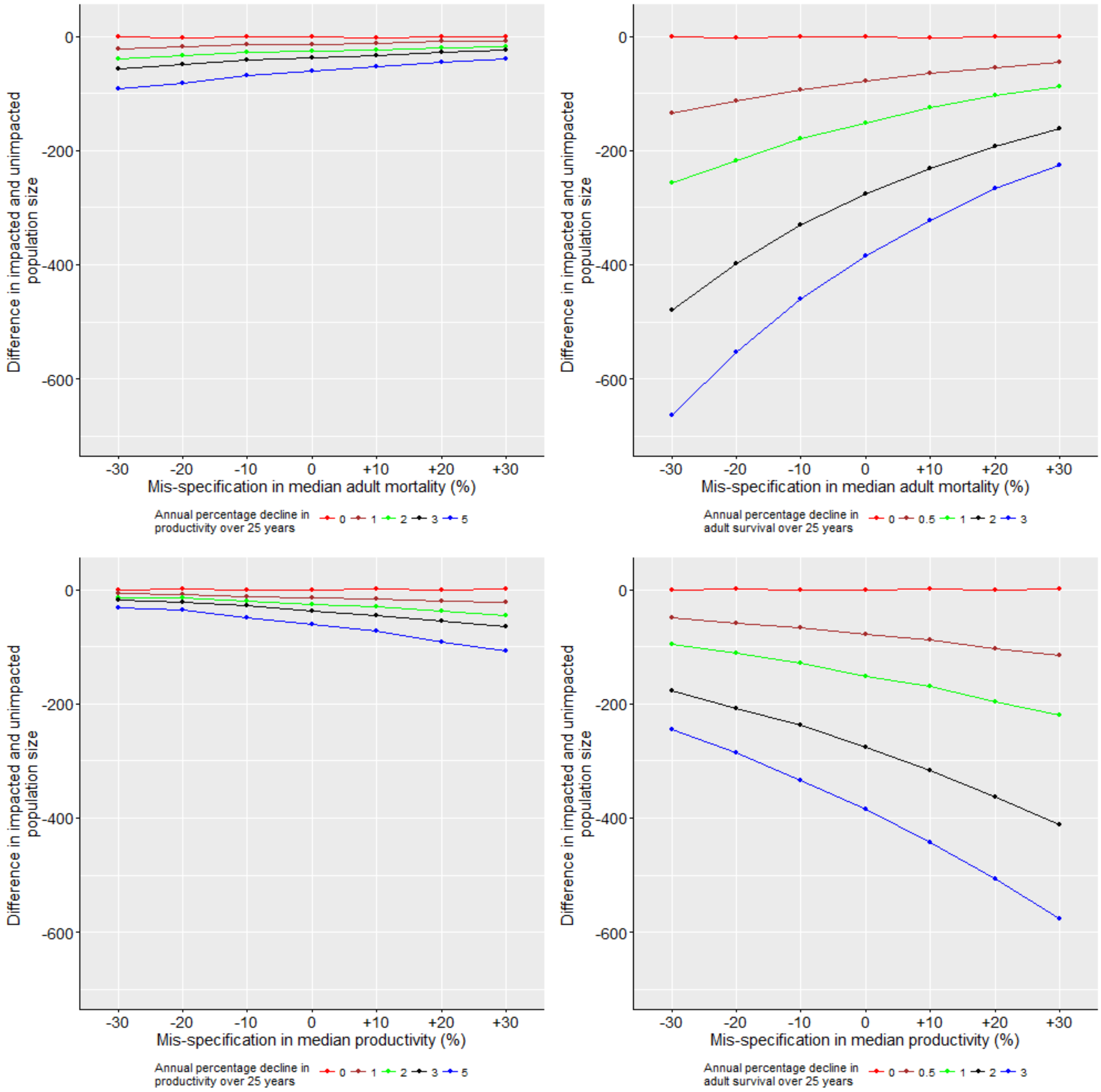


Figure A2.10e. PVA Metric E1 for St Abb's Razorbills – probability of population decline greater than 10% from 2016-2041.

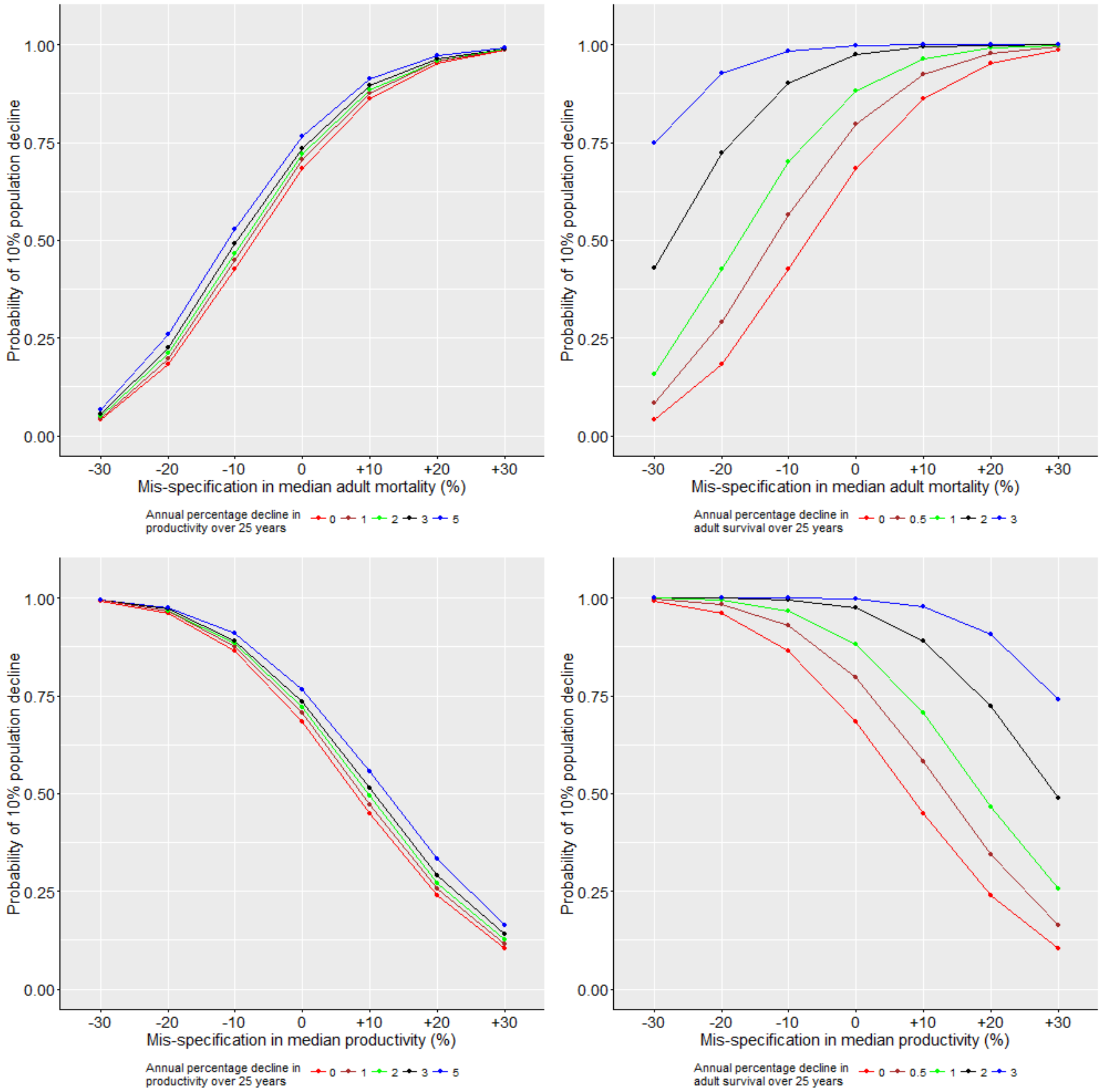


Figure A2.10f. PVA Metric E2 for St Abb's Razorbills – probability of population decline greater than 25% from 2016-2041.

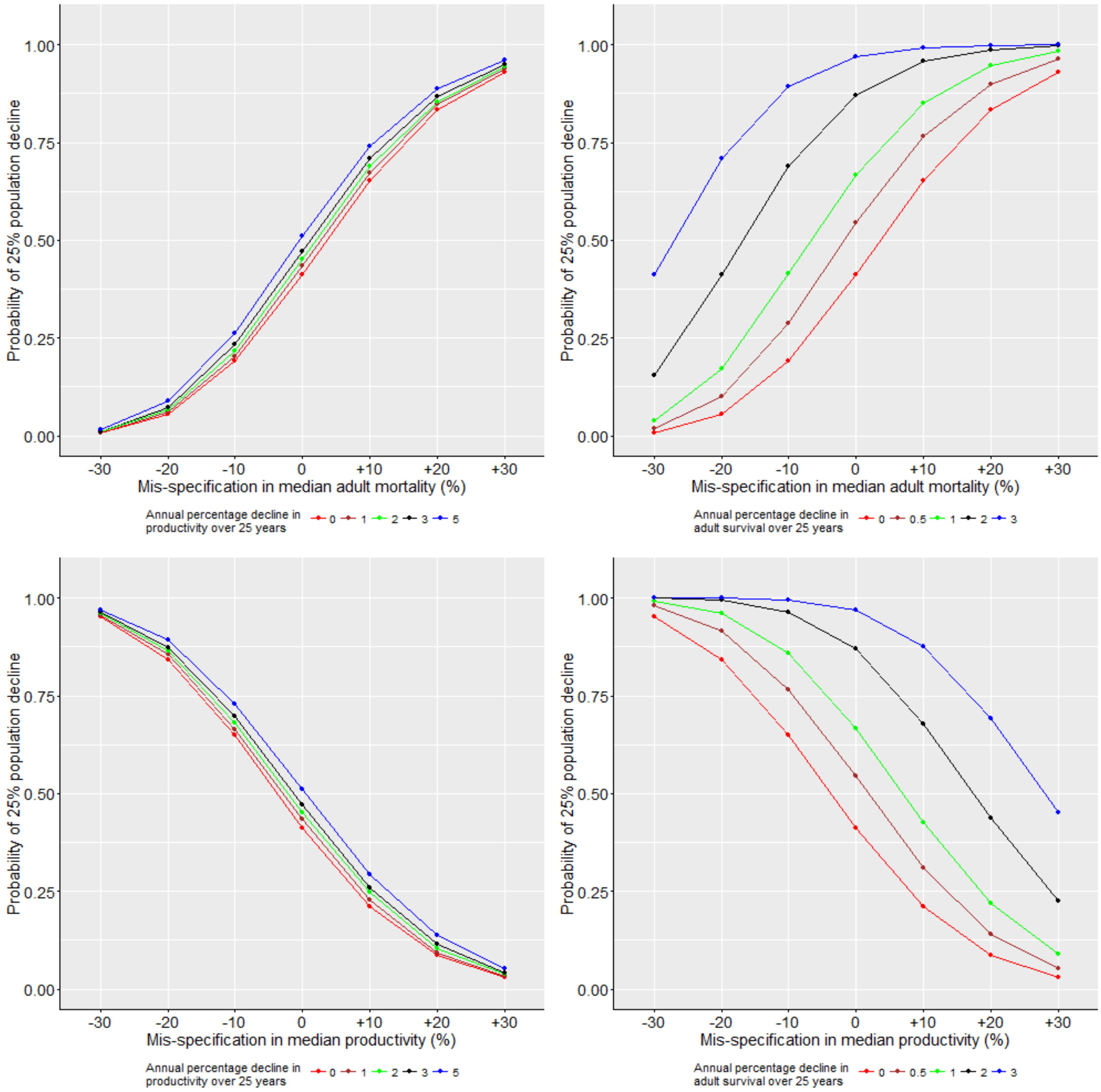


Figure A2.10g. PVA Metric E3 for St Abb's Razorbills – probability of population decline greater than 50% from 2016-2041.

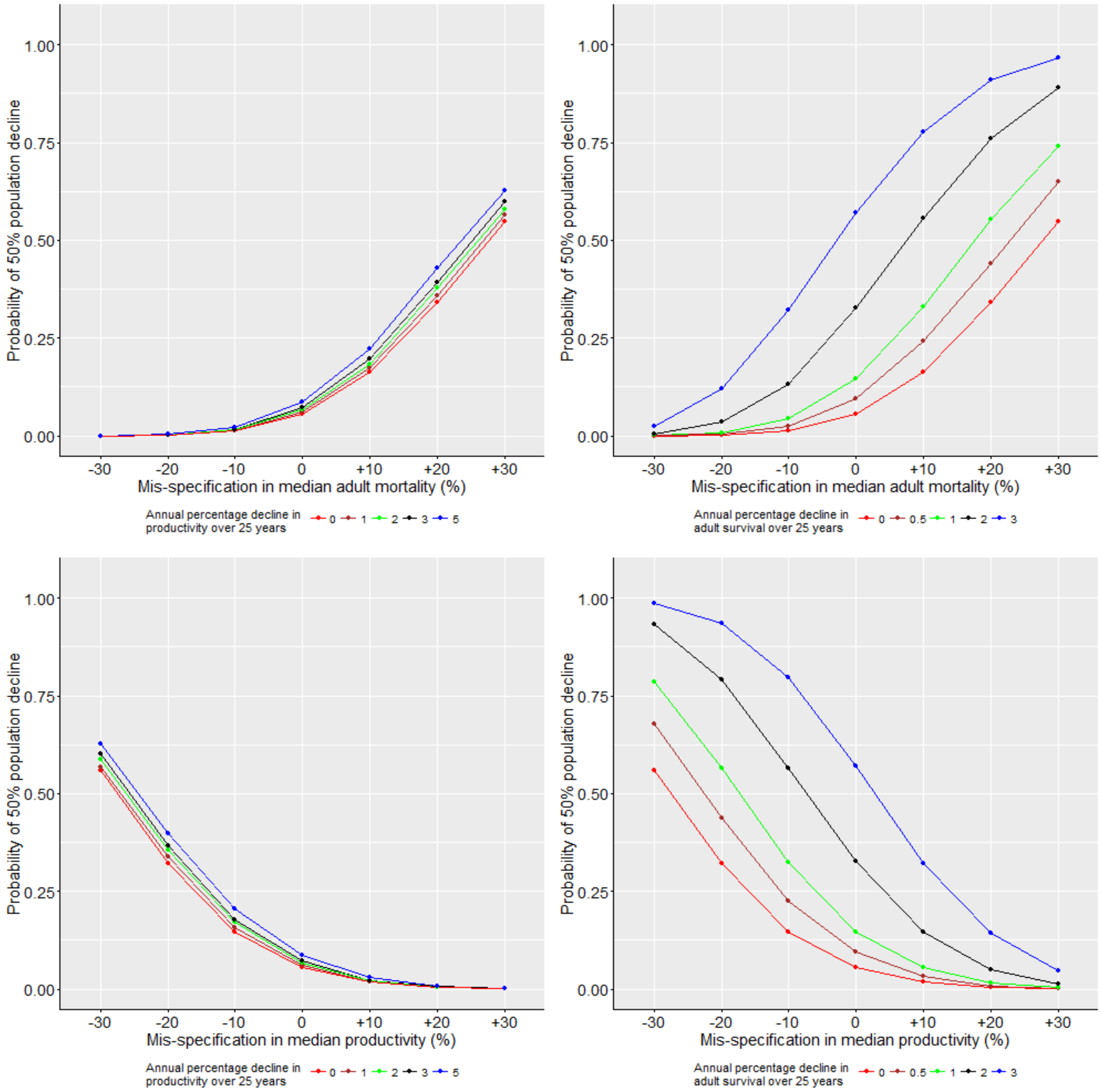
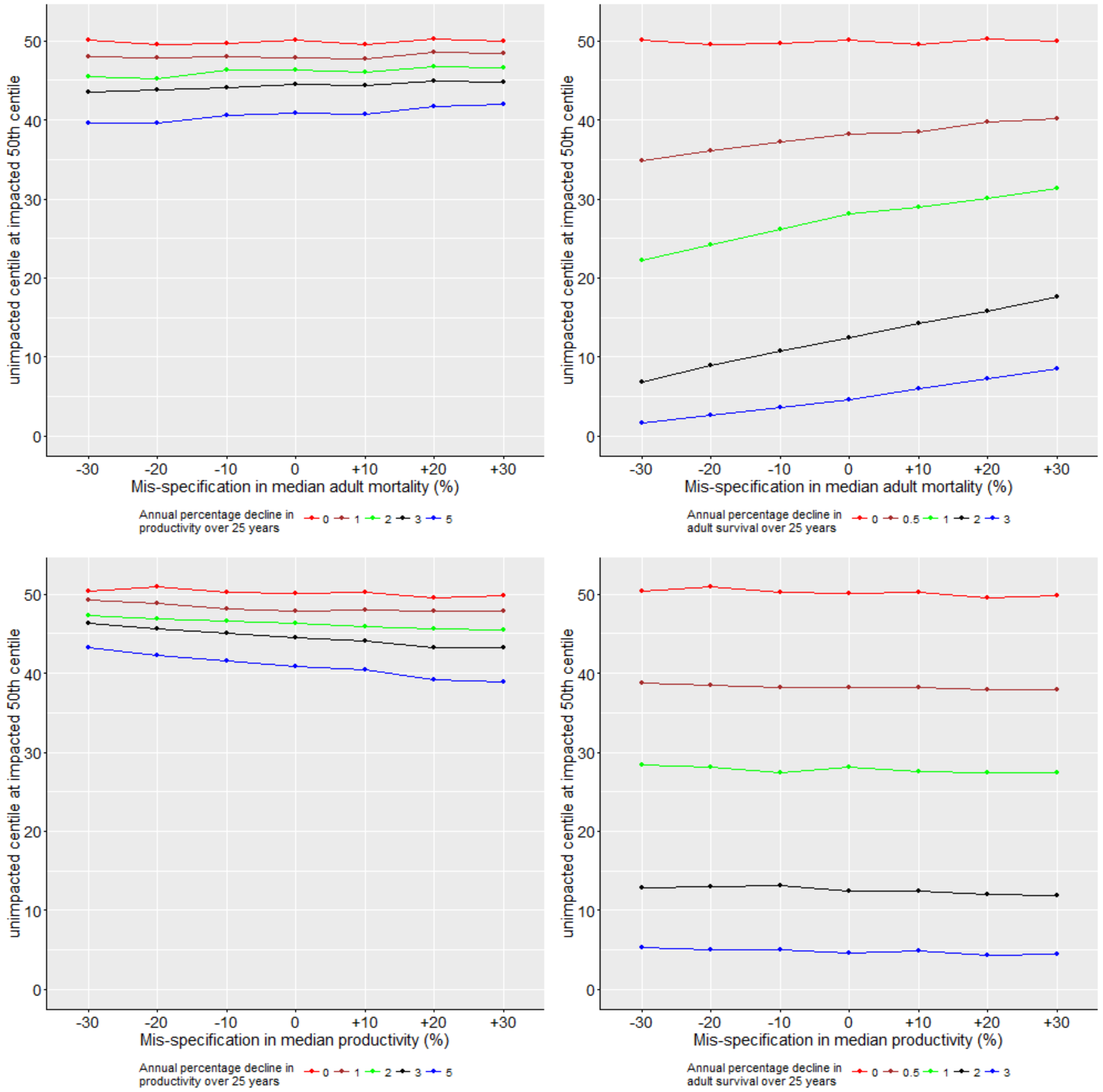


Figure A2.10h. PVA Metric F for St Abb's Razorbills – centile from un-impacted population size equal to the 50th centile of the impacted population size, at 2041.



11. **Razorbills at Fowlsheugh SPA**:

Figure A2.11a. PVA Metric A for Fowlsheugh Razorbills – ratio of population growth rate from 2016-2041, comparing impacted population vs. un-impacted population.

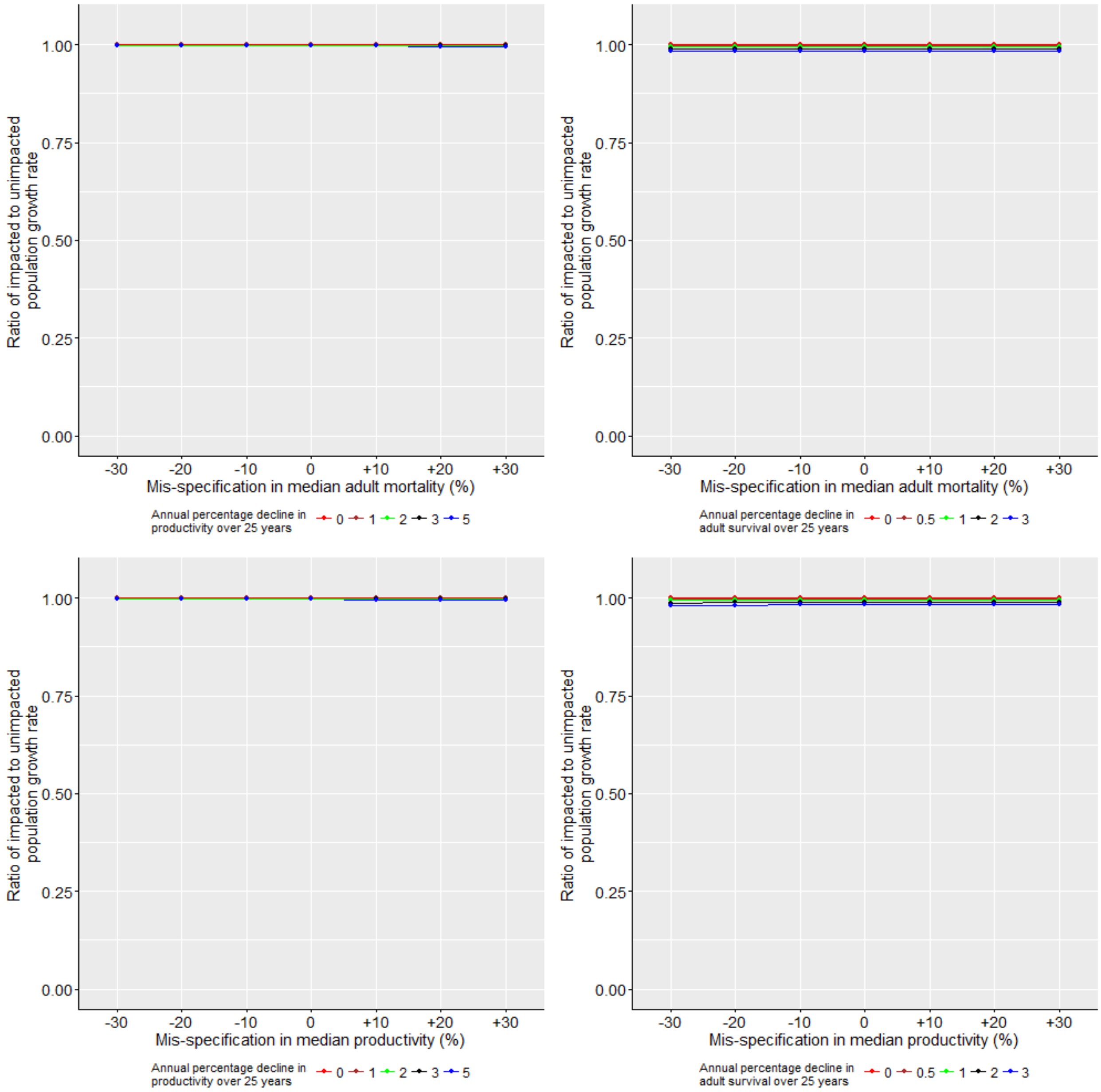


Figure A2.11b. PVA Metric B for Fowlsheugh Razorbills – ratio of population size at 2041, comparing impacted population vs. un-impacted population.

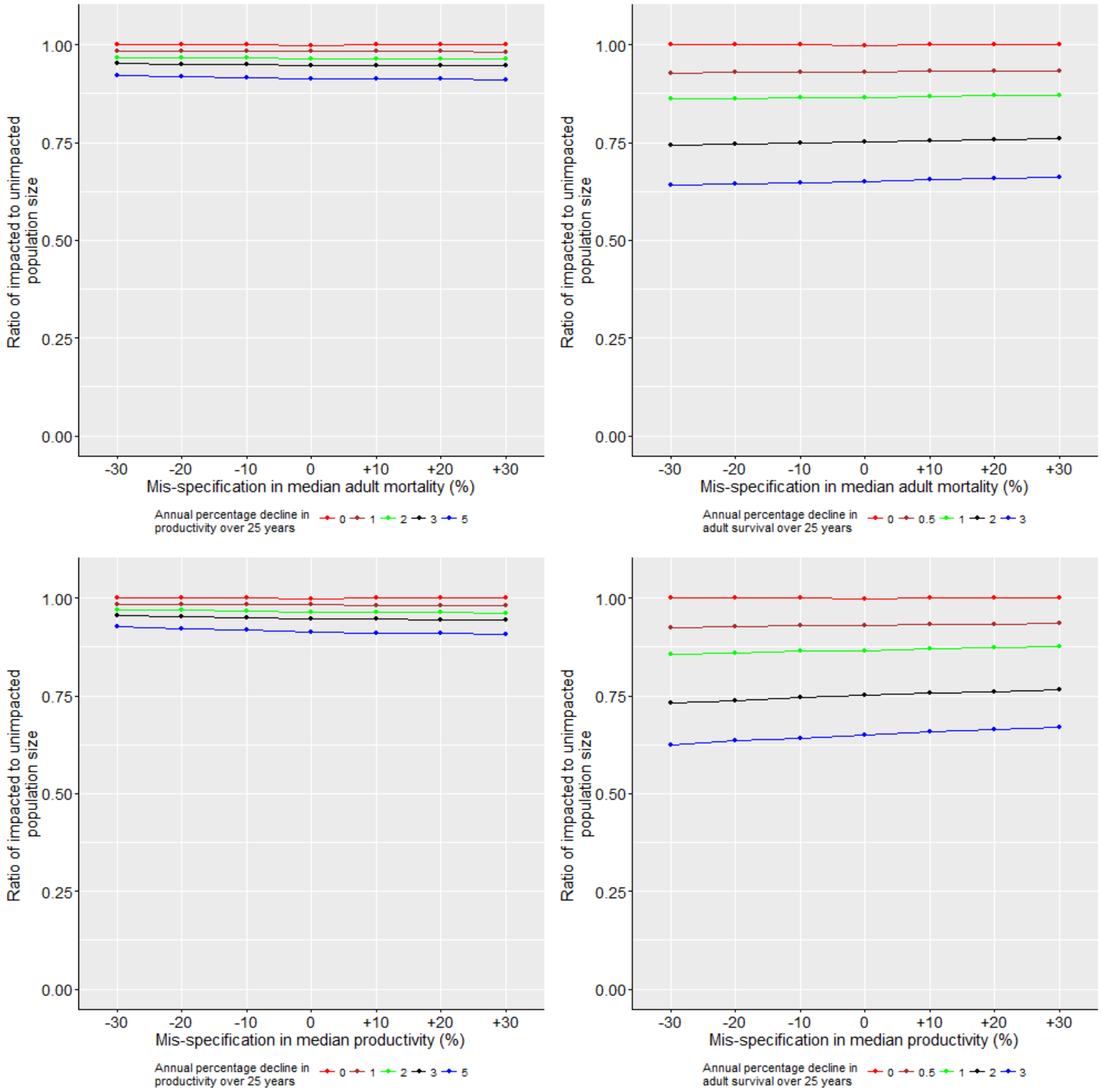


Figure A2.11c. PVA Metric C for Fowlsheugh Razorbills – difference in population growth rate from 2016-2041, comparing impacted population vs. un-impacted population.

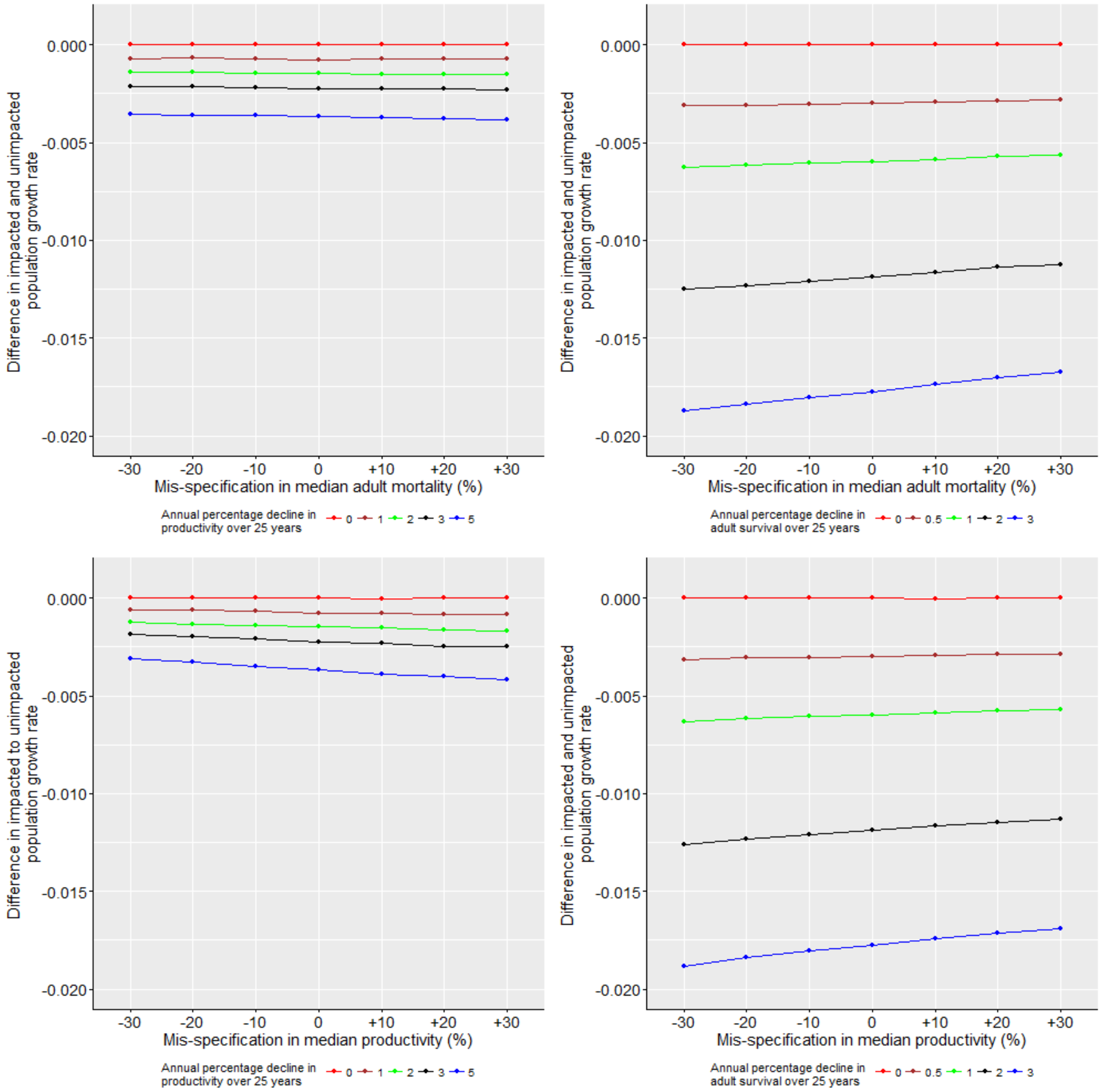


Figure A2.11d. PVA Metric D for Fowlsheugh Razorbills – difference in population size at 2041, comparing impacted population vs. un-impacted population.

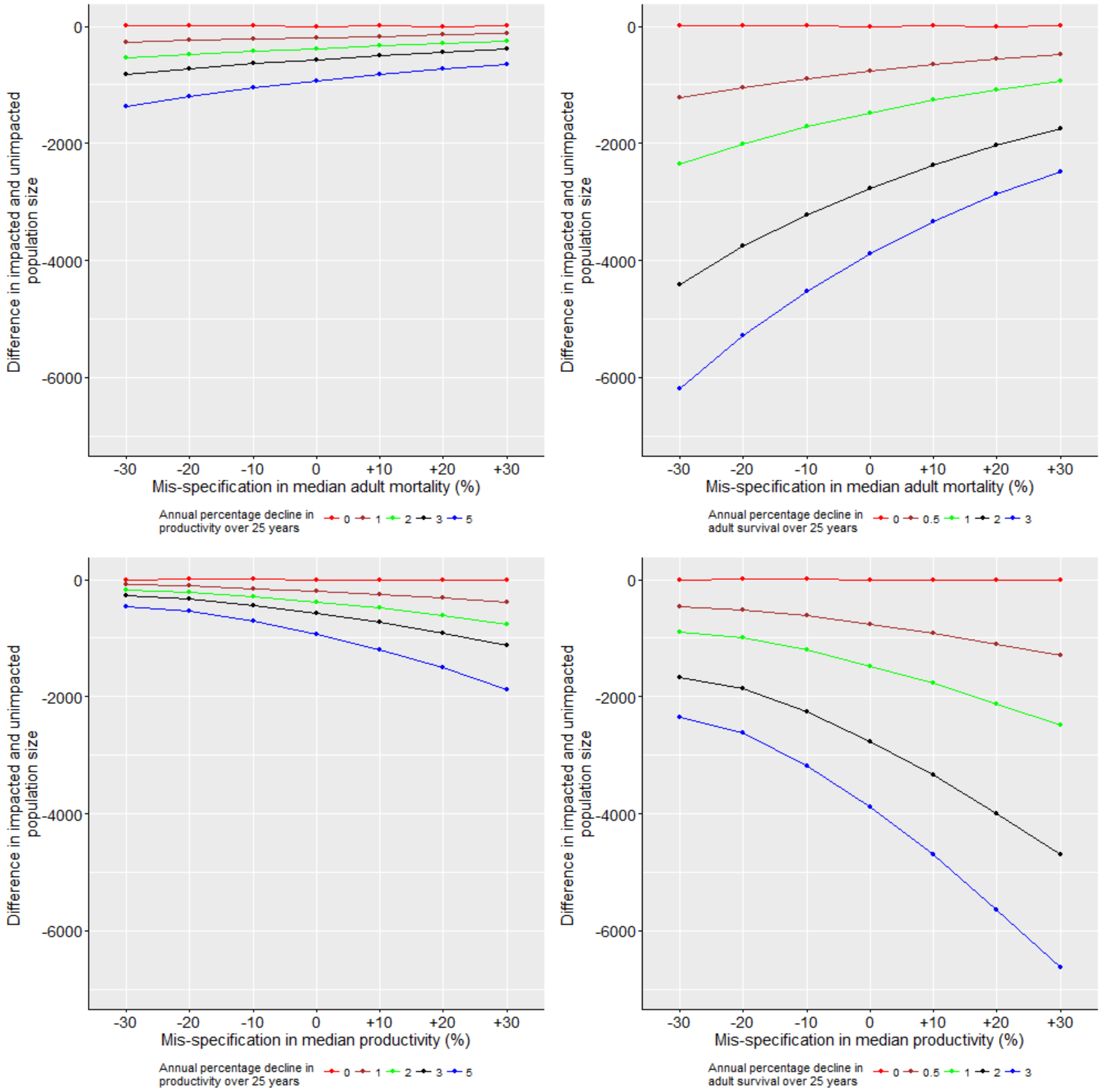


Figure A2.11e. PVA Metric E1 for Fowlsheugh Razorbills – probability of population decline greater than 10% from 2016-2041.

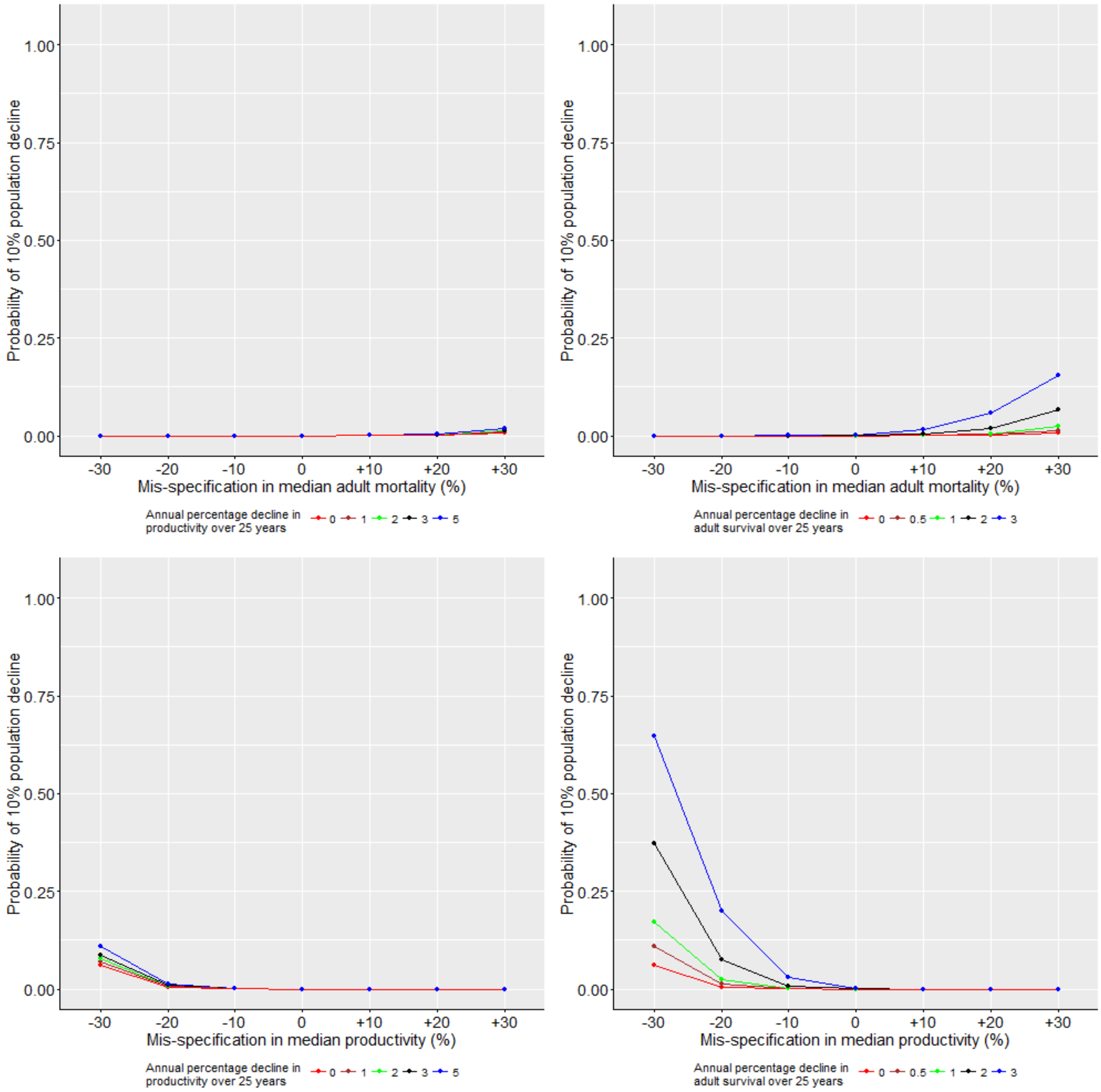


Figure A2.11f. PVA Metric E2 for Fowlsheugh Razorbills – probability of population decline greater than 25% from 2016-2041.

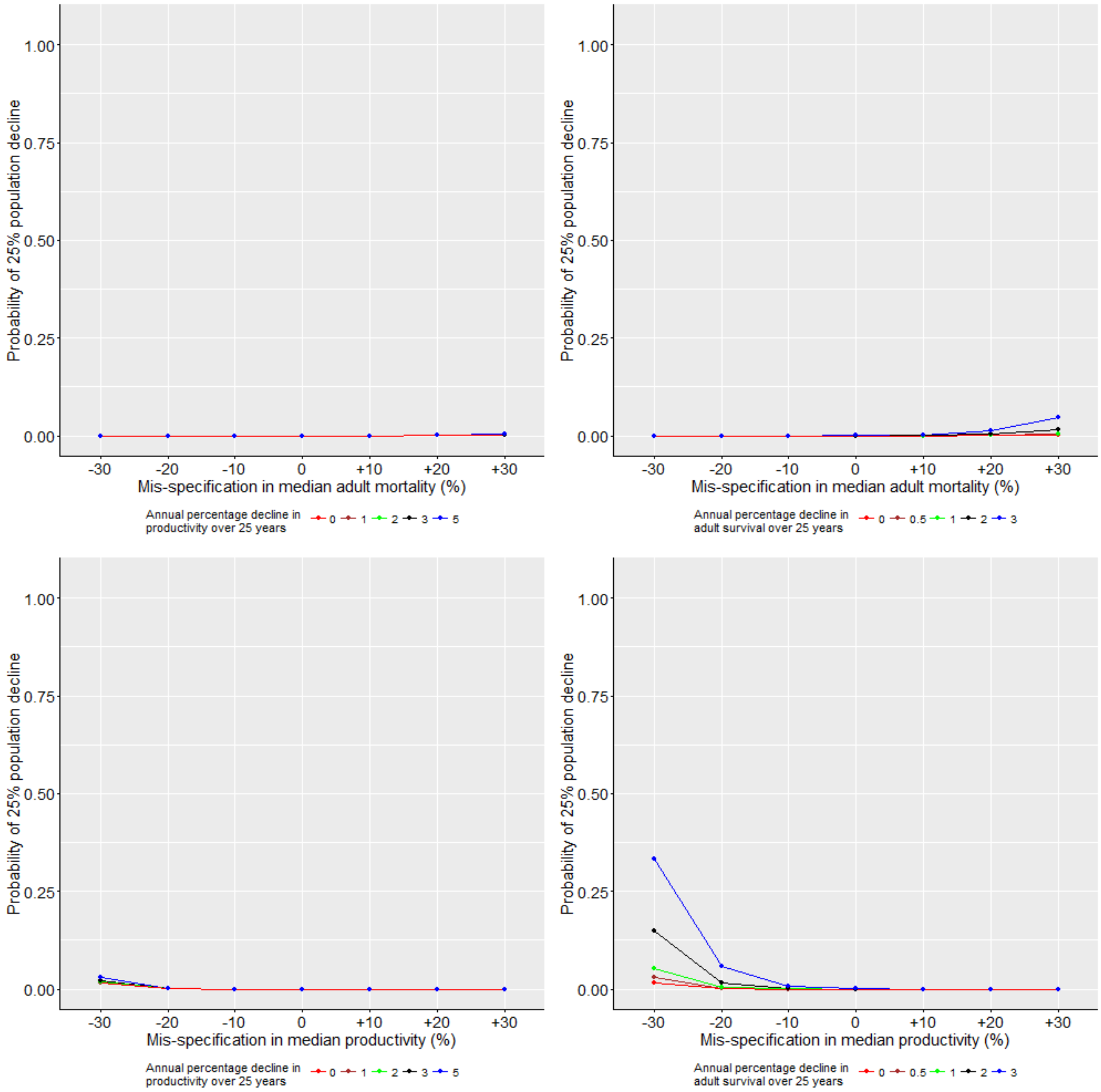


Figure A2.11g. PVA Metric E3 for Fowlsheugh Razorbills – probability of population decline greater than 50% from 2016-2041.

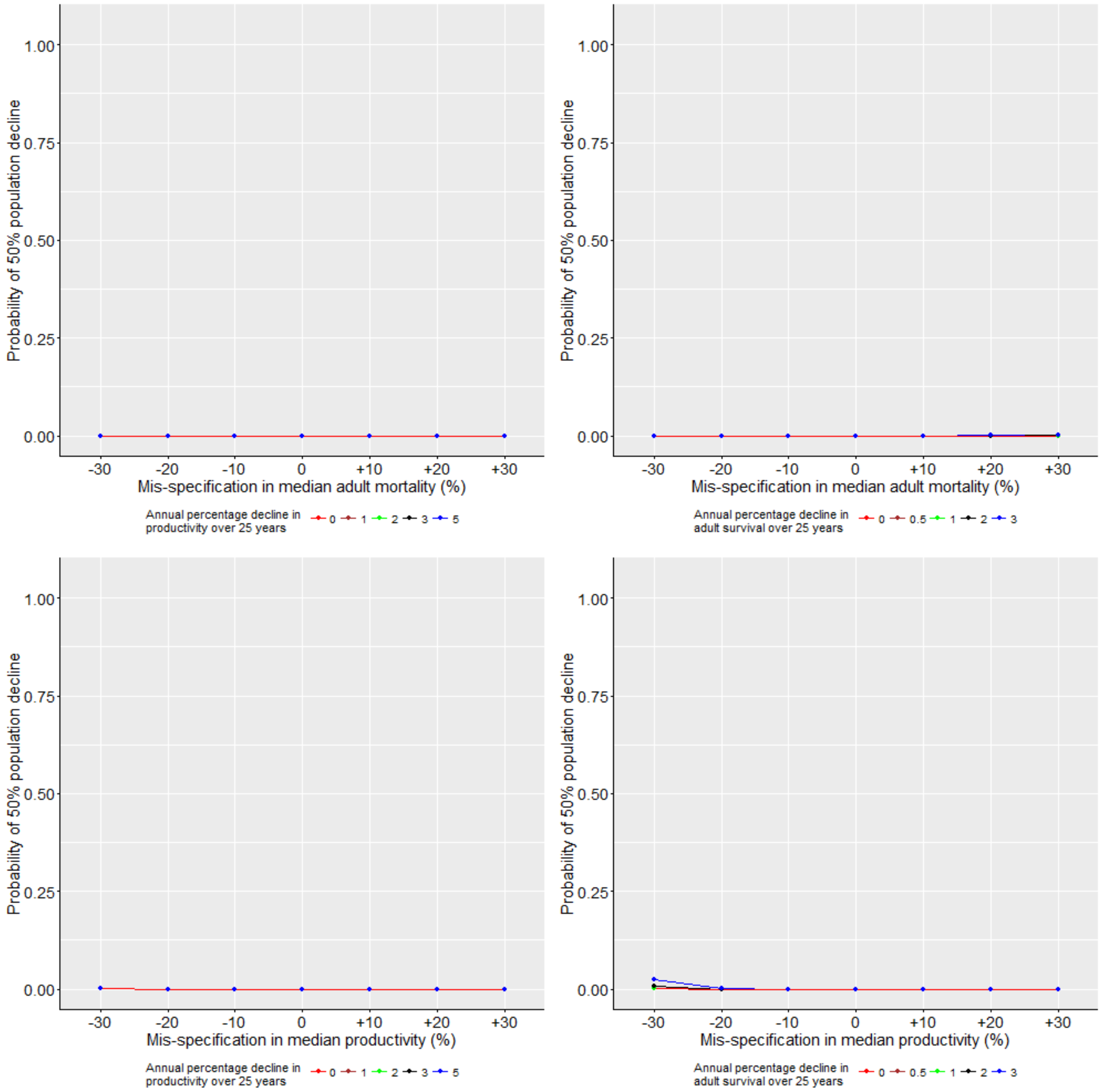
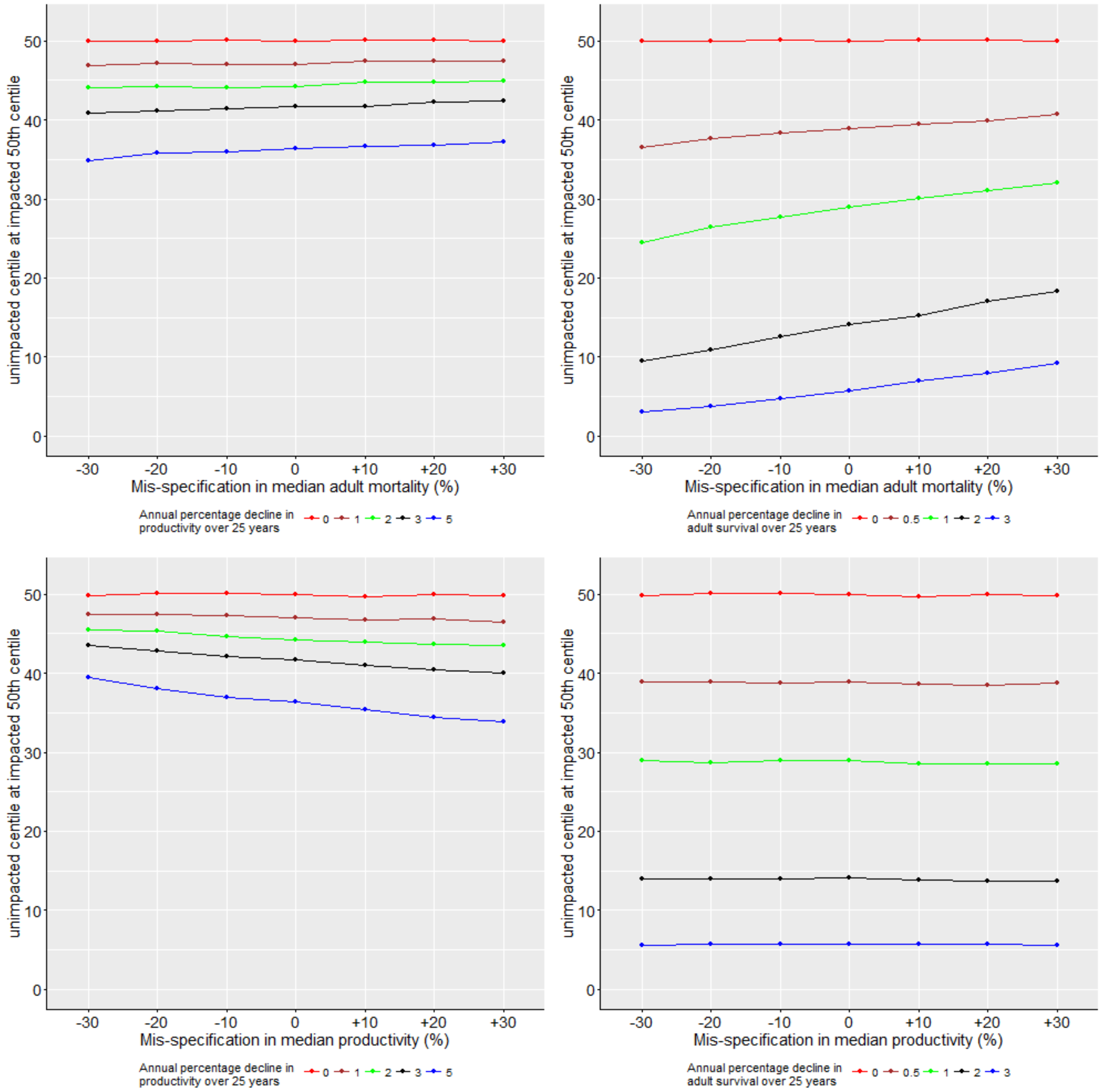


Figure A2.11h. PVA Metric F for Fowlsheugh Razorbills – centile from un-impacted population size equal to the 50th centile of the impacted population size, at 2041.



12. **Shags at Forth Islands SPA**:

Figure A2.12a. PVA Metric A for Forth Shags – ratio of population growth rate from 2016-2041, comparing impacted population vs. un-impacted population.

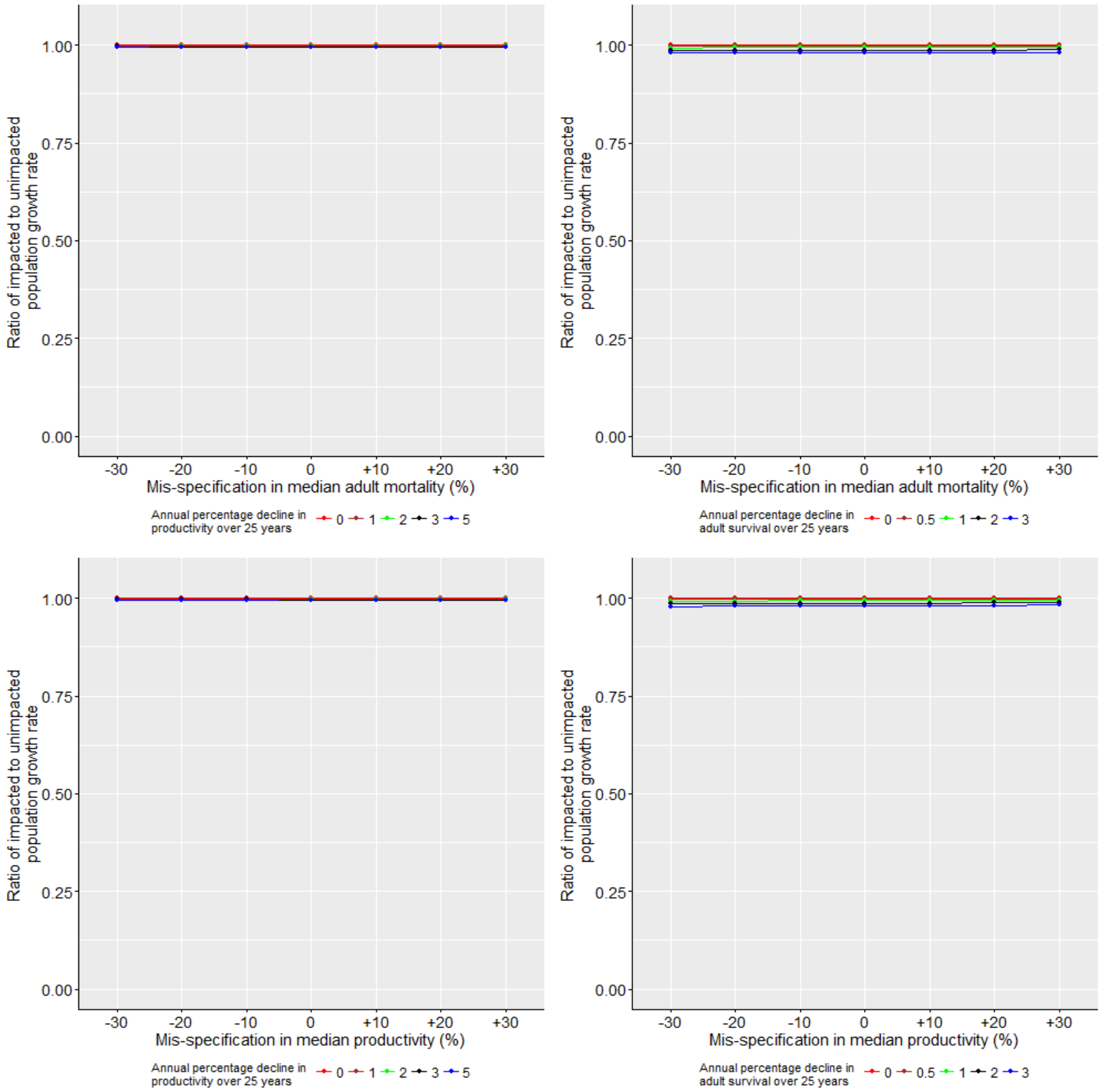


Figure A2.12b. PVA Metric B for Forth Shags – ratio of population size at 2041, comparing impacted population vs. un-impacted population.

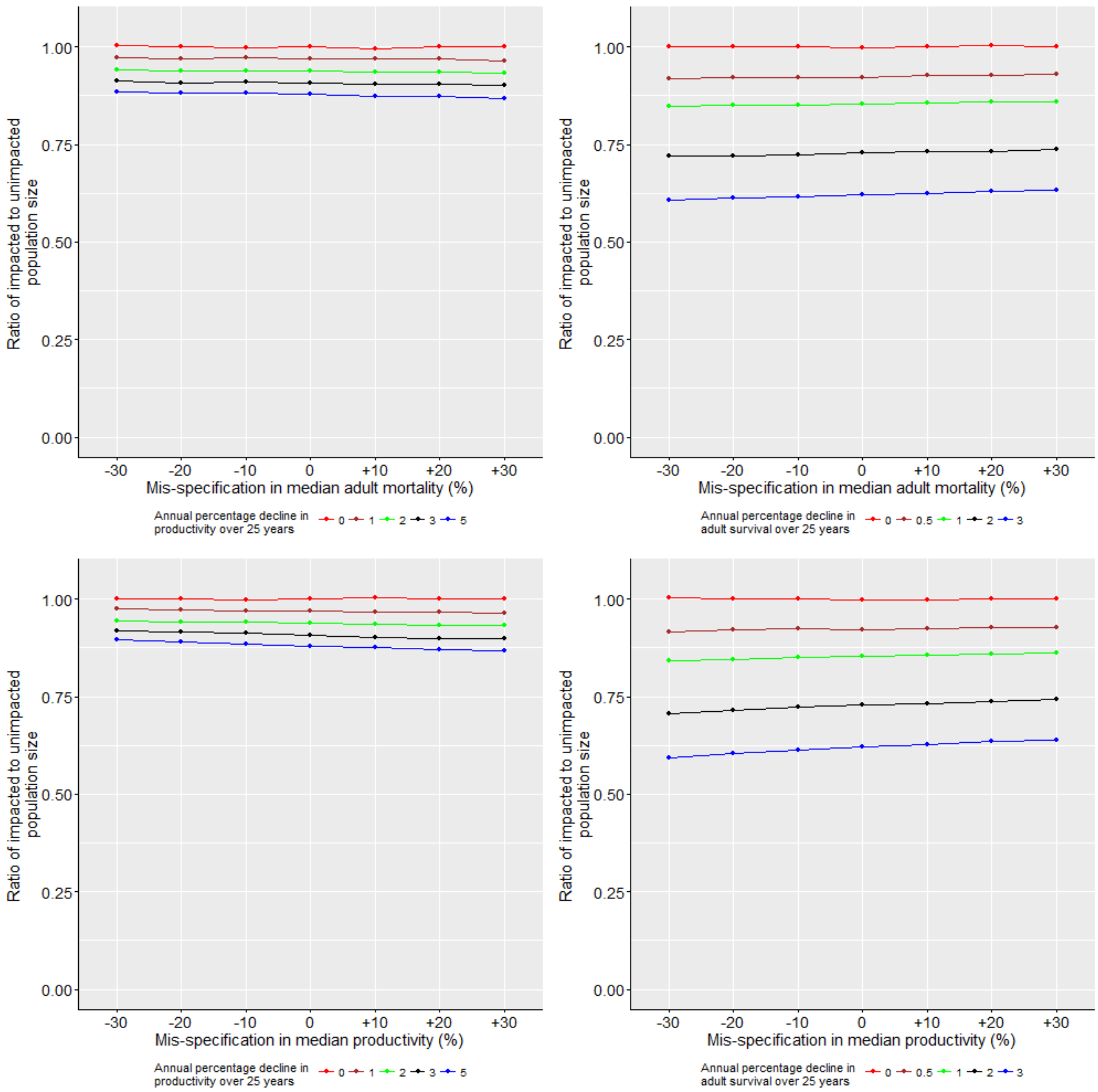


Figure A2.12c. PVA Metric C for Forth Shags – difference in population growth rate from 2016-2041, comparing impacted population vs. un-impacted population.

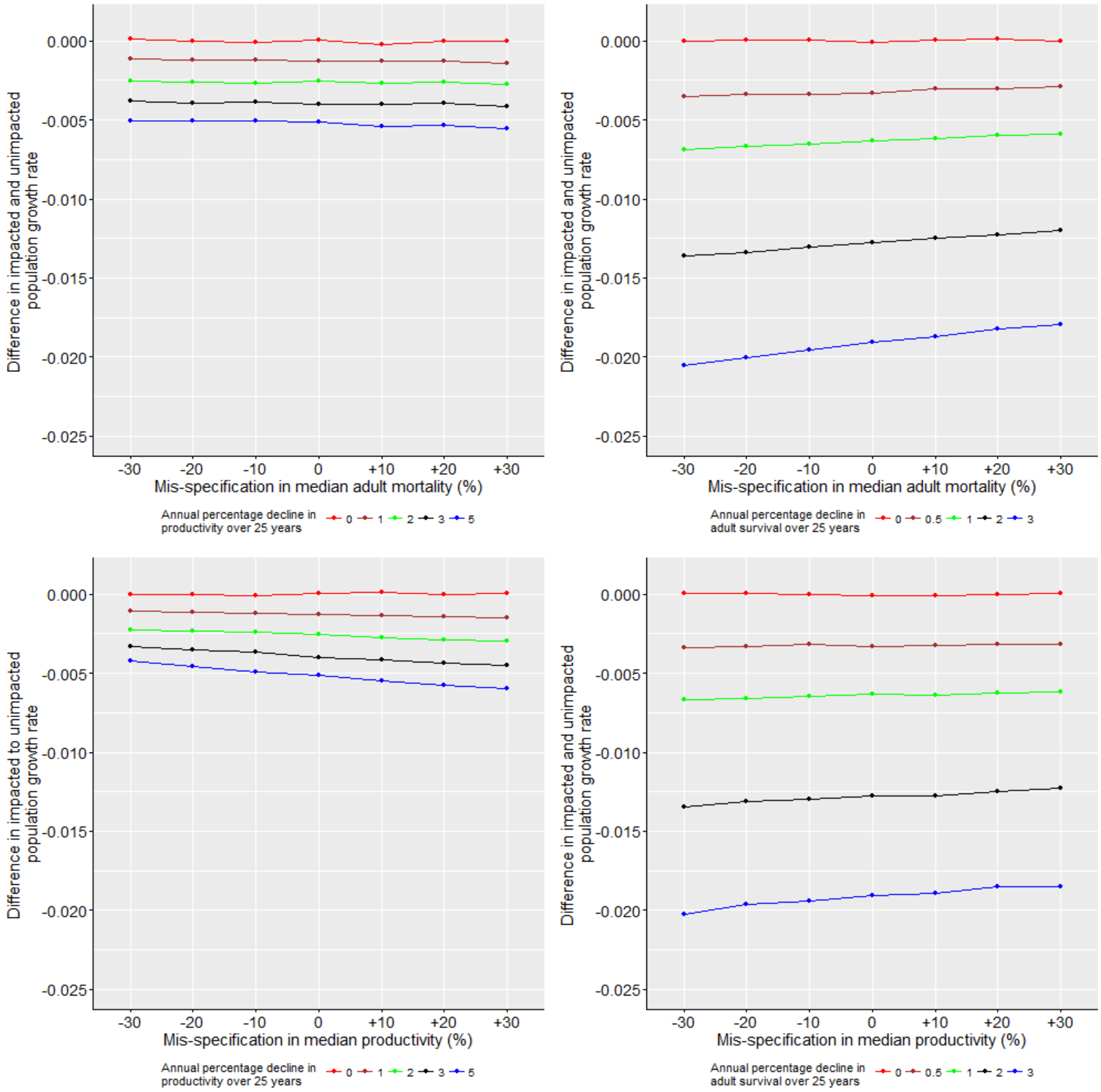


Figure A2.12d. PVA Metric D for Forth Shags – difference in population size at 2041, comparing impacted population vs. un-impacted population.

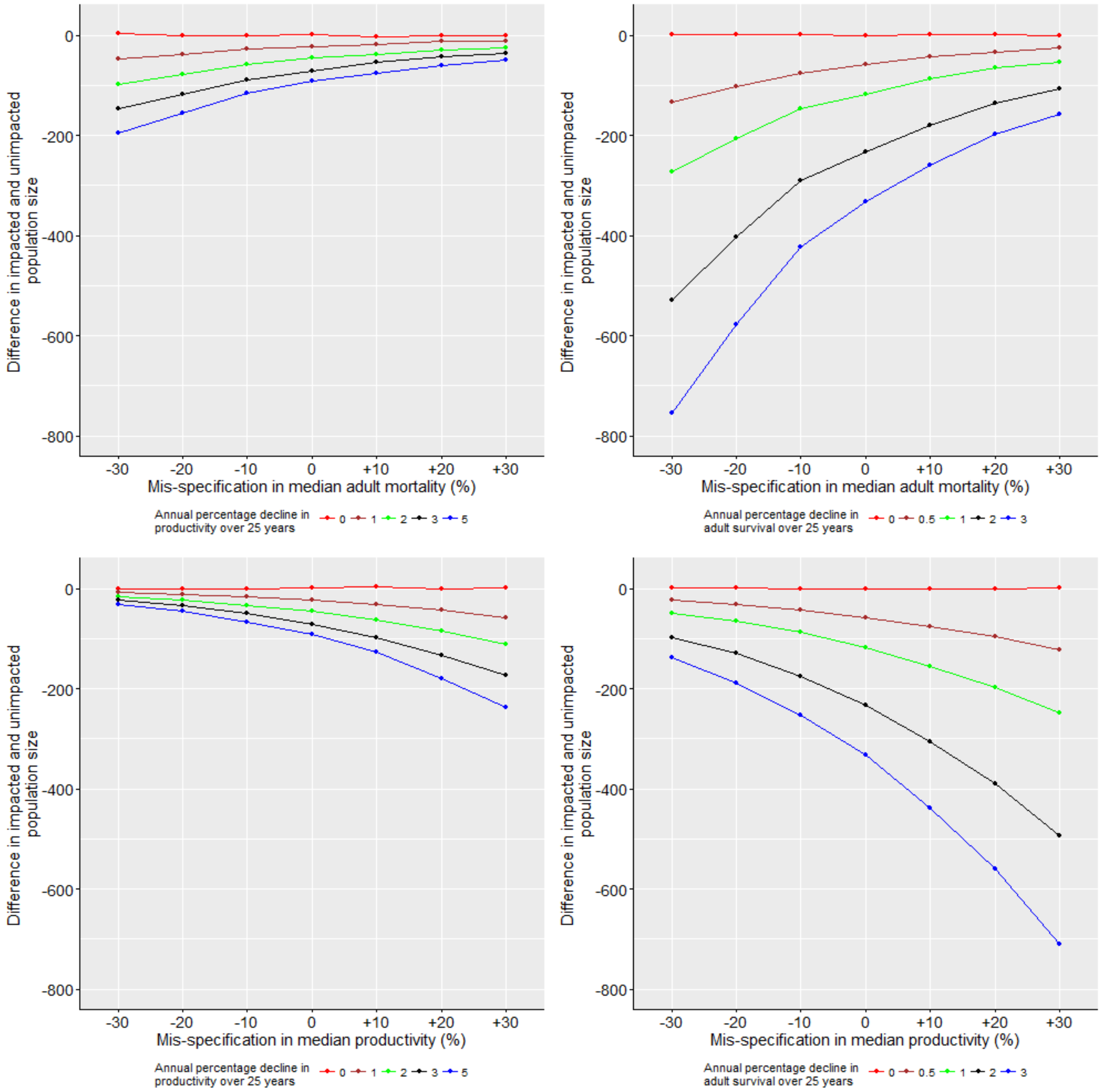


Figure A2.12e. PVA Metric E1 for Forth Shags – probability of population decline greater than 10% from 2016-2041.

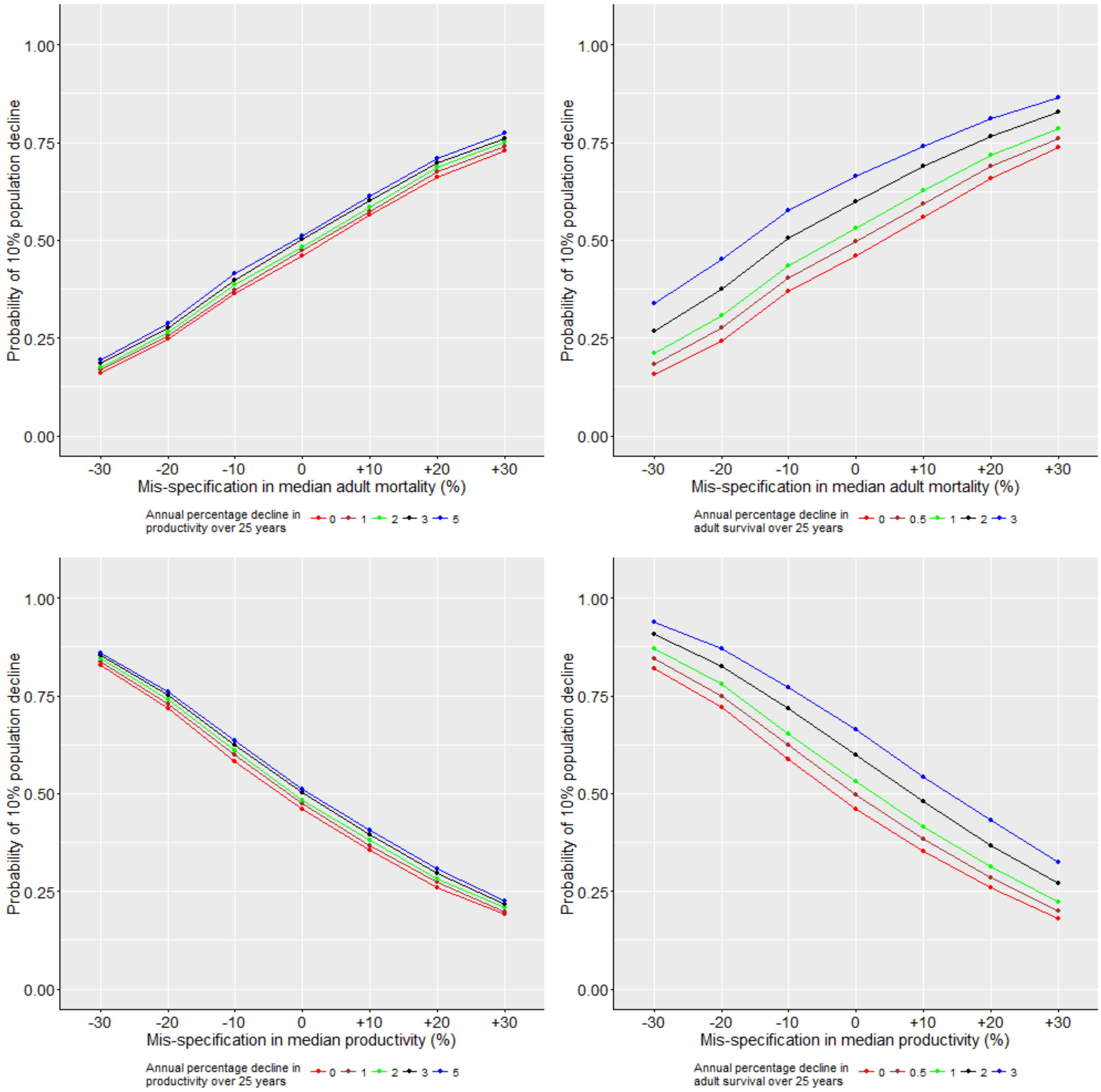


Figure A2.12f. PVA Metric E2 for Forth Shags – probability of population decline greater than 25% from 2016-2041.

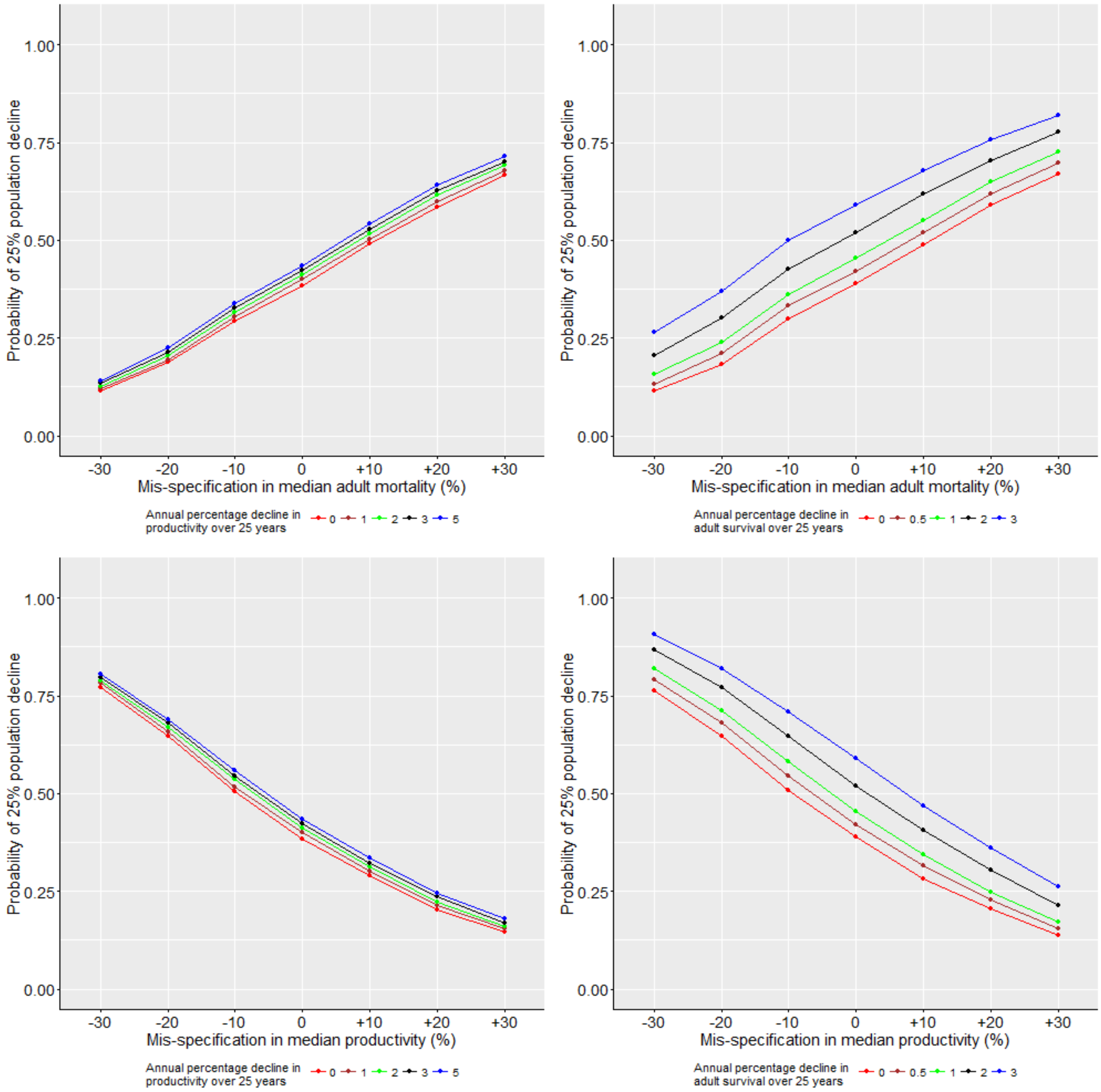


Figure A2.12g. PVA Metric E3 for Forth Shags – probability of population decline greater than 50% from 2016-2041.

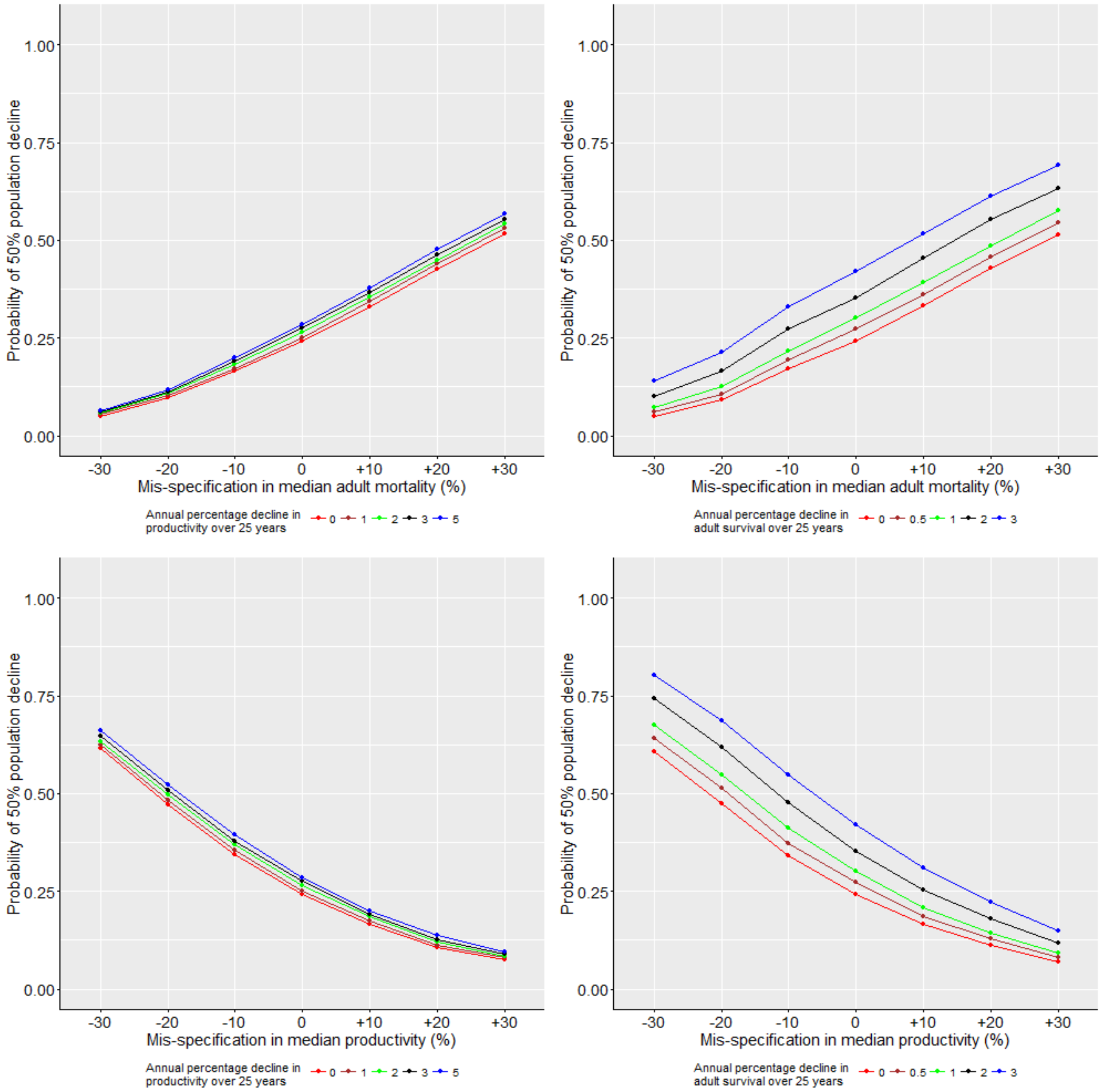
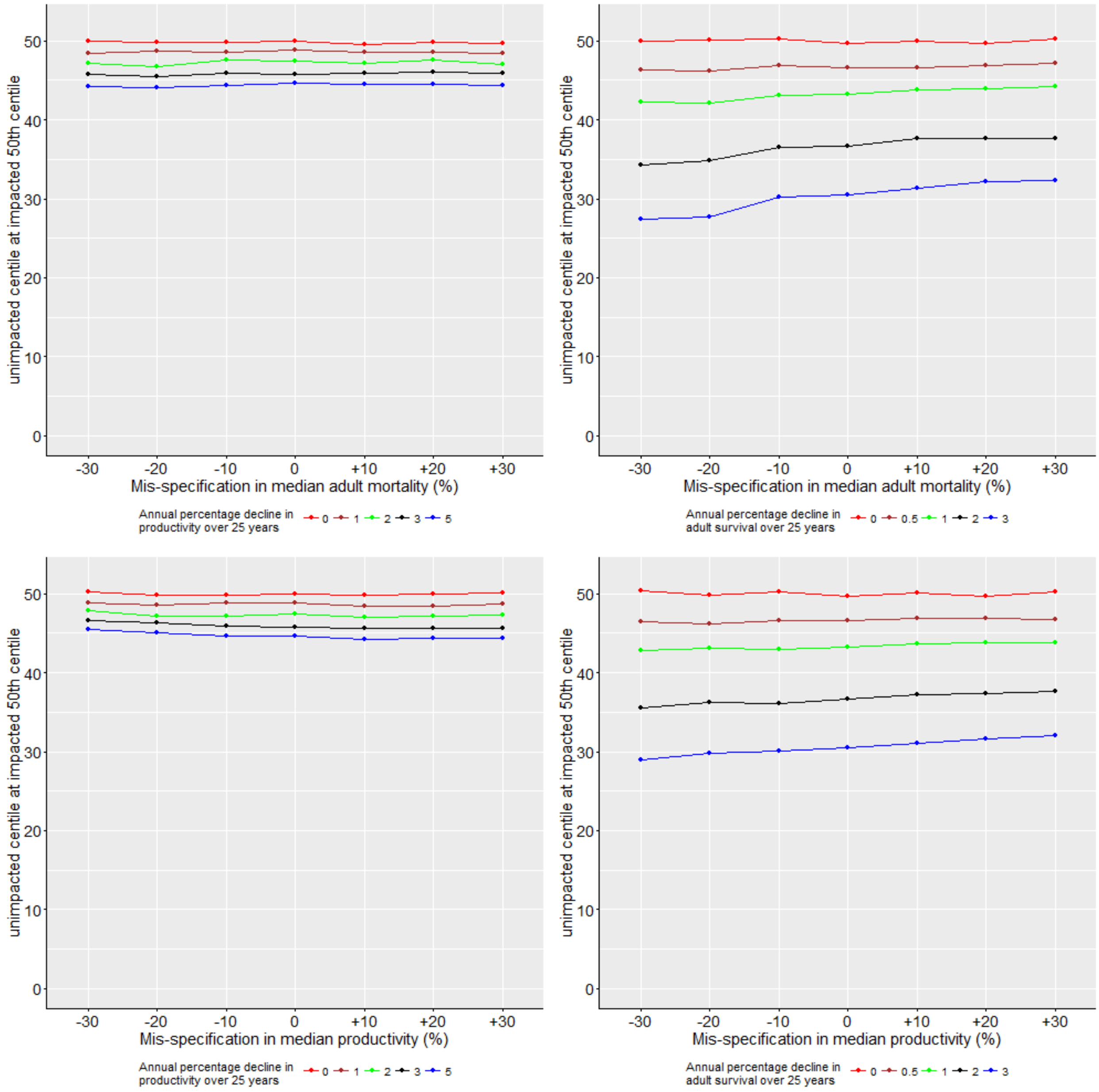


Figure A2.12h. PVA Metric F for Forth Shags – centile from un-impacted population size equal to the 50th centile of the impacted population size, at 2041.



13. **Shags at St Abb's Head SPA**:

Figure A2.13a. PVA Metric A for St Abb's Shags – ratio of population growth rate from 2016-2041, comparing impacted population vs. un-impacted population.

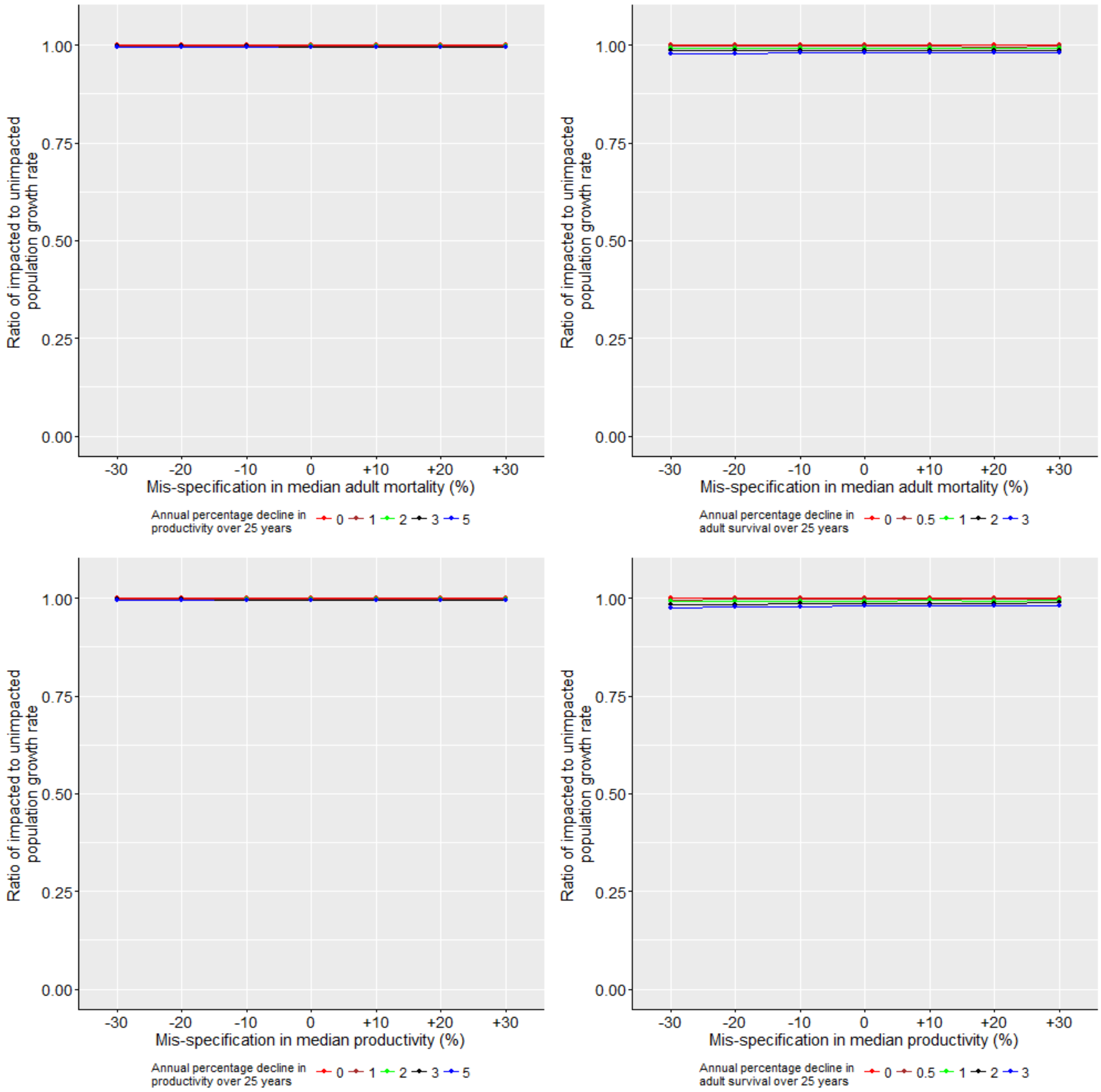


Figure A2.13b. PVA Metric B for St Abb's Shags – ratio of population size at 2041, comparing impacted population vs. un-impacted population.

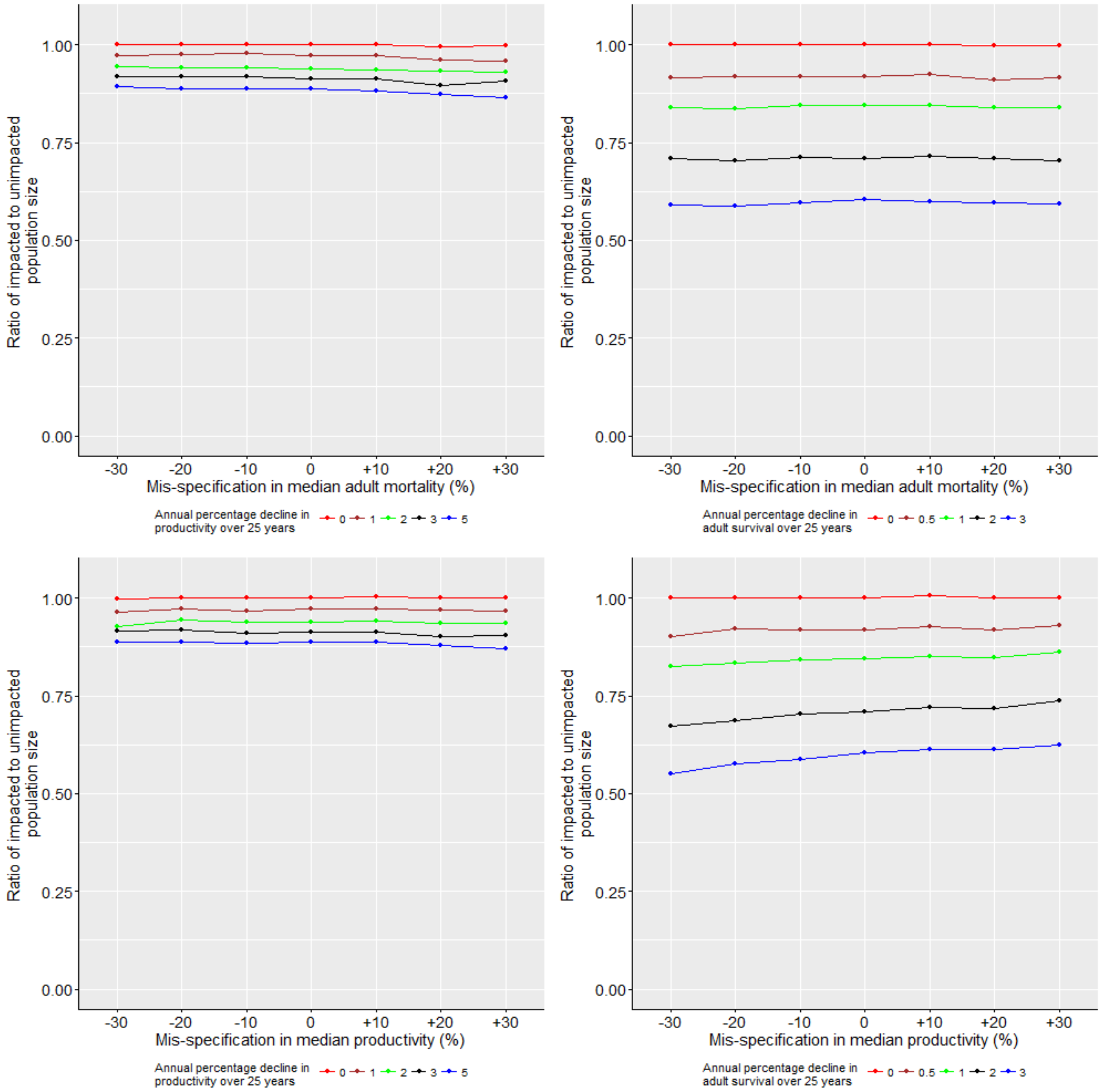


Figure A2.13c. PVA Metric C for St Abb's Shags – difference in population growth rate from 2016-2041, comparing impacted population vs. un-impacted population.

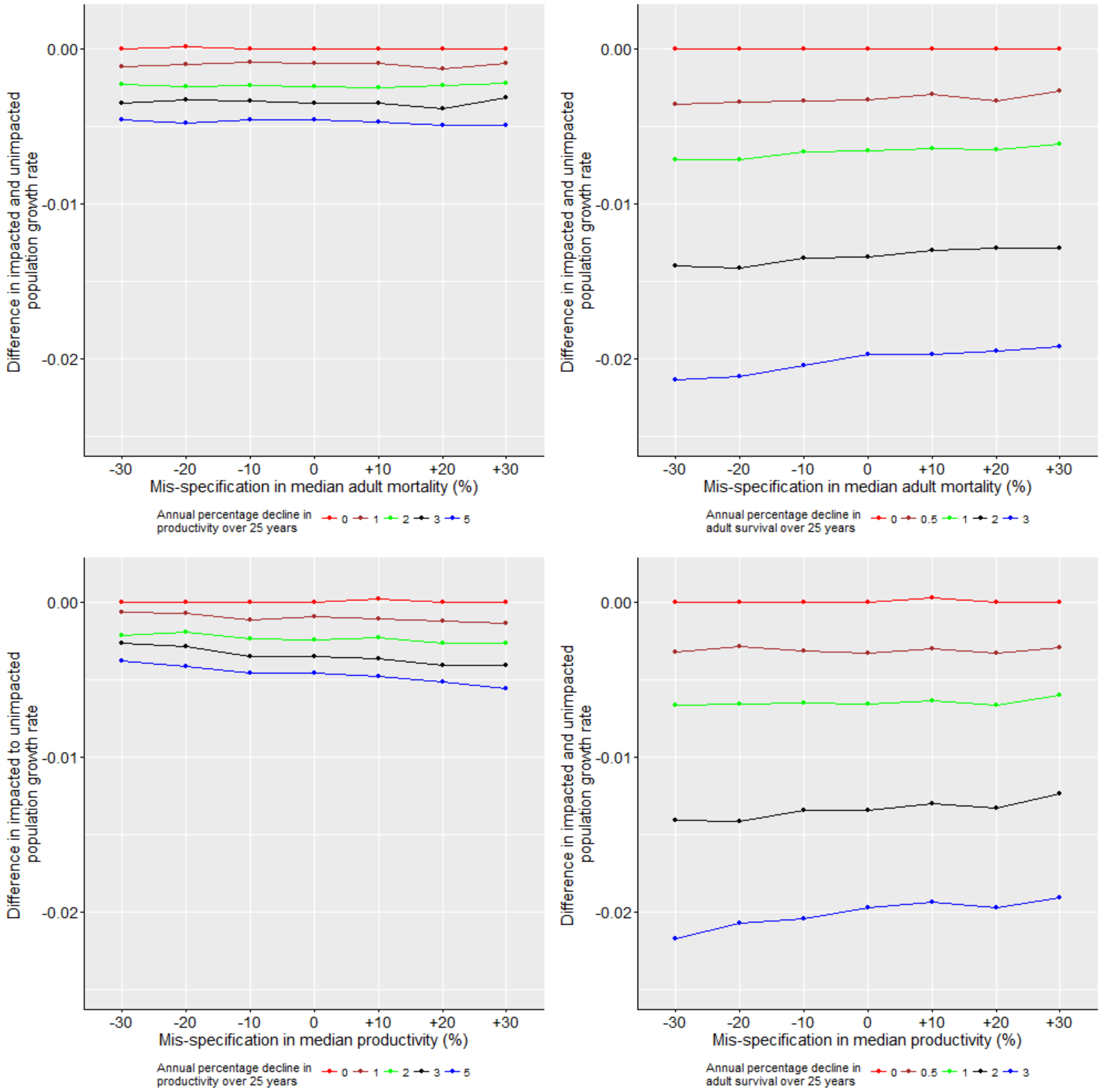


Figure A2.13d. PVA Metric D for St Abb's Shags – difference in population size at 2041, comparing impacted population vs. un-impacted population.

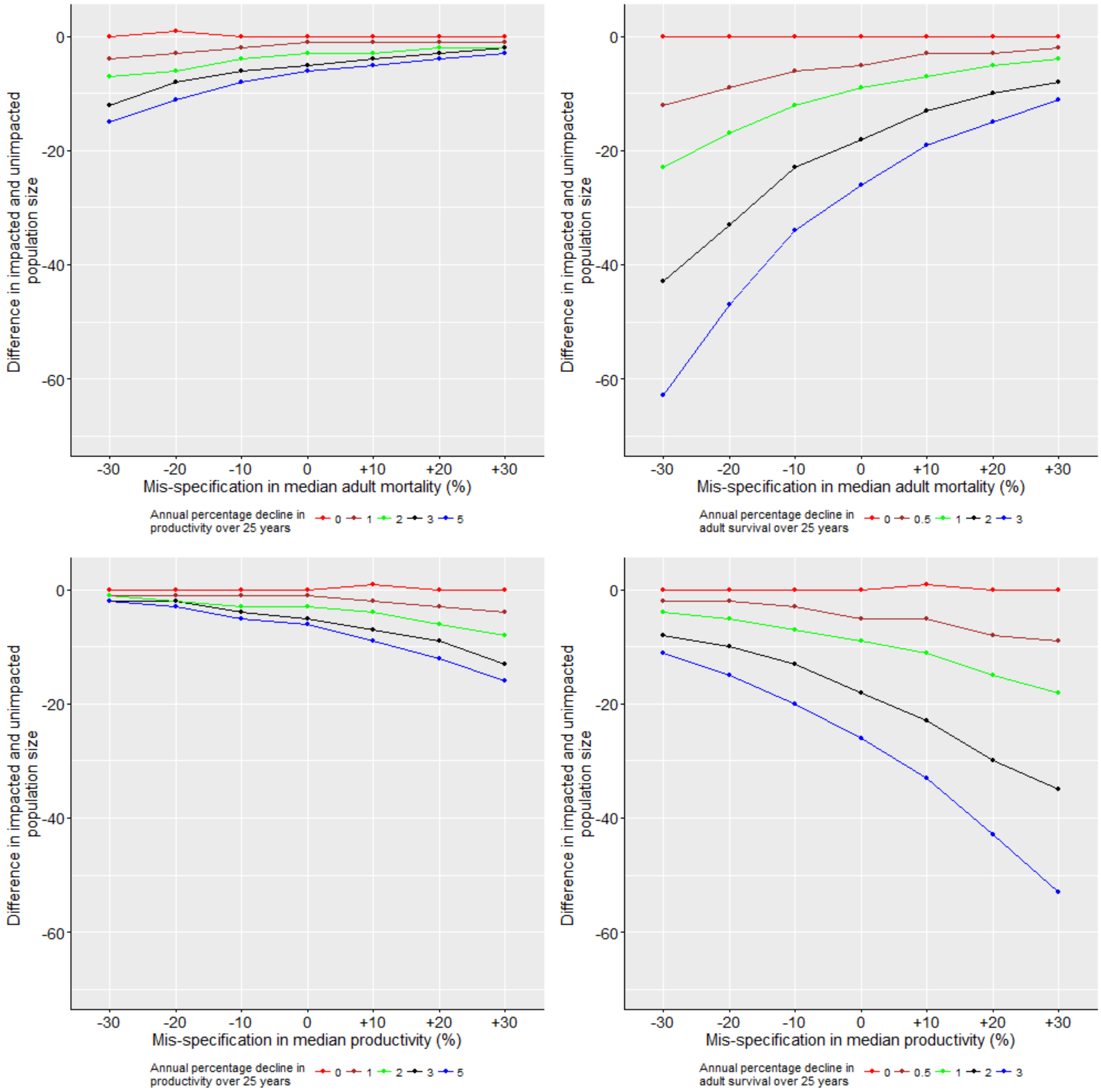


Figure A2.13e. PVA Metric E1 for St Abb's Shags – probability of population decline greater than 10% from 2016-2041.

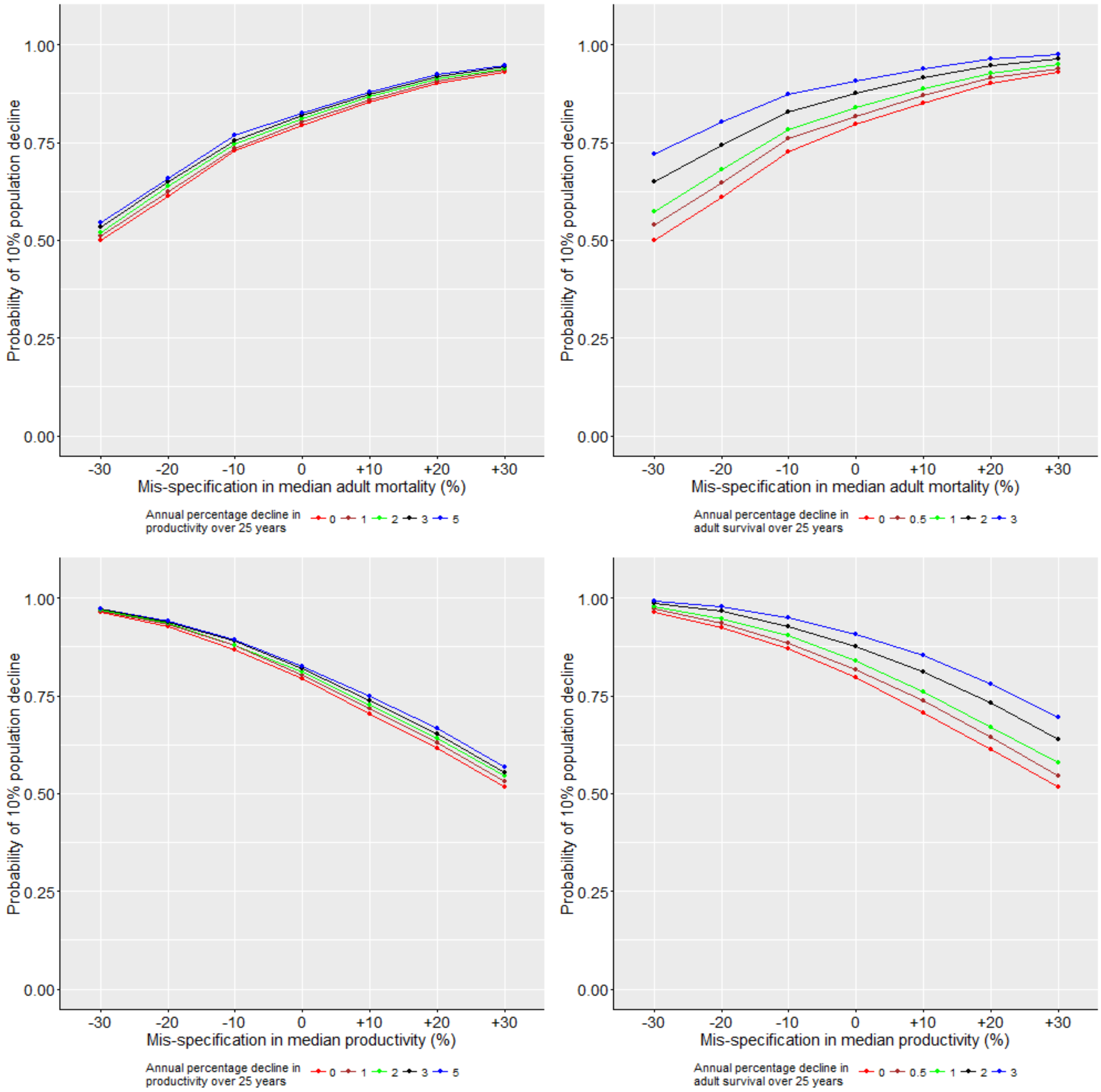


Figure A2.13f. PVA Metric E2 for St Abb’s Shags – probability of population decline greater than 25% from 2016-2041.

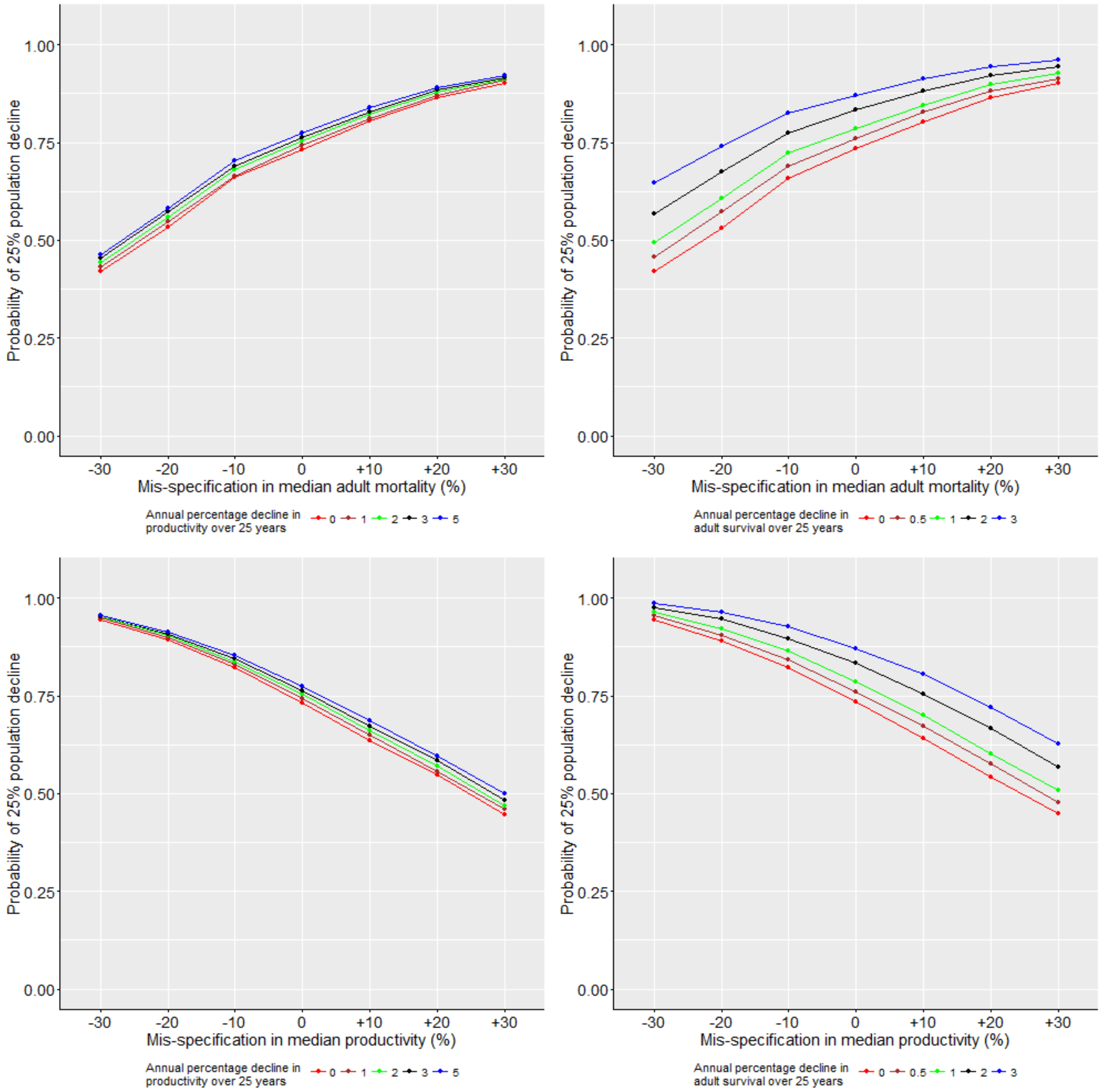


Figure A2.13g. PVA Metric E3 for St Abb's Shags – probability of population decline greater than 50% from 2016-2041.

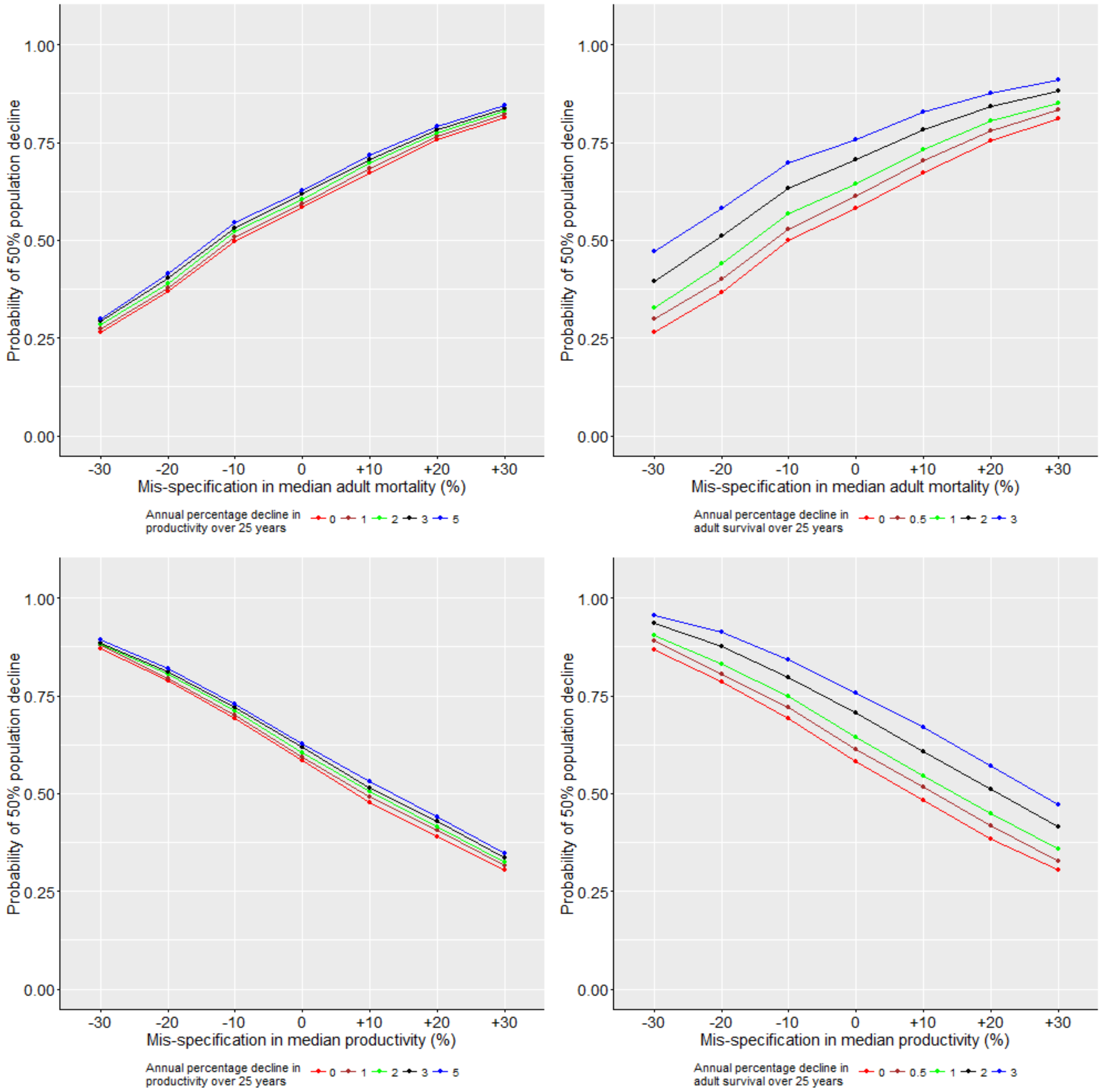
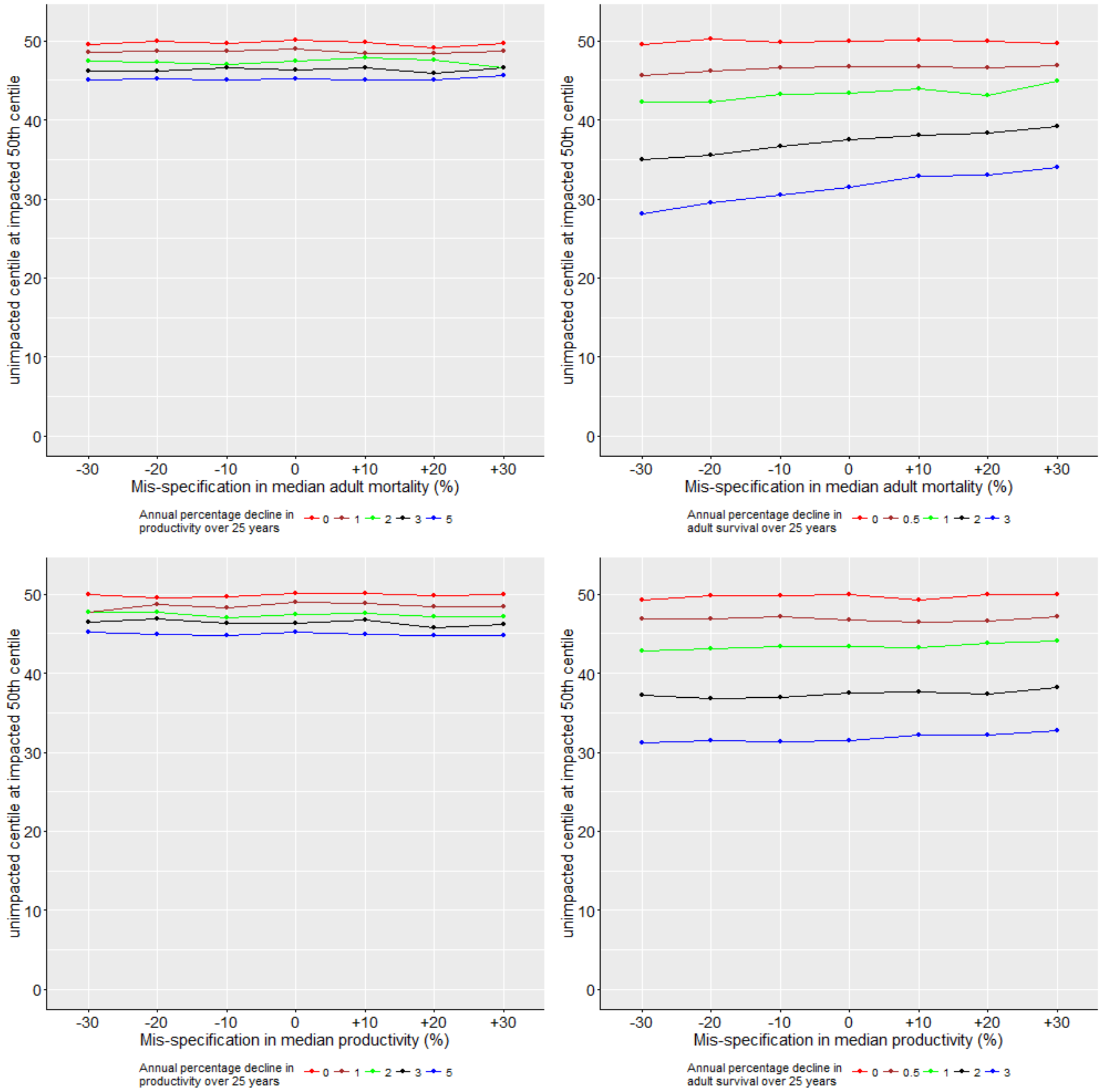


Figure A2.13h. PVA Metric F for St Abb's Shags – centile from un-impacted population size equal to the 50th centile of the impacted population size, at 2041.



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