November 2023

PVA Analysis for key seabird populations in the MarPAMM area

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Executive Summary.

1. The EU Interreg VA funded Marine Protected Area Management and Monitoring (MarPAMM) project area, which comprises the seas between the west coast of Scotland and the island of Ireland, support internationally important populations of breeding seabirds. Some of these populations have undergone dramatic declines over the past few decades, linked to a range of human pressures including climate change, introduced predators and fisheries bycatch.
2. Through cross-border collaboration, the MarPAMM project aims to develop a suite of regional management plans in order to preserve and enhance important seabird populations. Using project partners and stakeholder engagement we identify vulnerable species, key threats, and possible management interventions for key seabird colonies within the MarPAMM area.
3. For populations within Special Protection Areas (SPAs) across the four MarPAMM management regions, we ran Population Viability Analyses (PVA) to investigate the regional population-level impacts of alterations in two key demographic rates – breeding success and adult survival.
4. To do this we use the Natural England PVA Tool and associated *nepva* R Package. In each management region, PVAs were run across a suite of those SPAs for which a species is a designated feature.
5. Baseline models were run to simulate current conditions. Using the original baseline model framework, matched comparative scenario simulations were then run, incorporating percentage point (p.p.) increases in either breeding success (+5 p.p., +10 p.p., +20 p.p.) or adult survival (+1 p.p., +2 p.p., +3 p.p.), as proxies for reductions in key pressures through management interventions.
6. By calculating two key comparative metrics, the ratio of impacted (managed following management intervention) to unimpacted (i.e., without any additional management intervention) population size (RPS) and growth rate (PGR), we explore how regional seabird populations may respond in the presence and absence or reduction (through management interventions) of key pressures/ threats.

7. On average, based on model outputs, species which lay more than one egg benefited most from improvements in breeding success.
8. Over the range of values modelled in this study, the greatest increases in RPS/PGR resulted from improvements in breeding success (+5 p.p., +10 p.p., +20 p.p.) compared to increases in adult survival (+1 p.p., +2 p.p., +3 p.p.). Crucially, the relative effect of improvements in breeding success and adult survival was related to species-specific life-histories, in particular maximum brood size.
9. The greatest improvements resulting from increases in breeding success were achieved in species which lay more than one egg (gulls, terns, and Cormorants; maximum brood size > 1), with this appearing a more effective target for management interventions in these species. For changes to adult survival, on average population benefits were smaller compared to the modelled effects of improved breeding success. However, improvements in adult survival resulted in greater increases in RPS/RGR in some species, in particular those which lay a single egg (auk and petrel species).
10. Based on the results of the PVA analysis we provide some discussion of potential benefits from a number of potential management interventions which could be used to inform regional management planning as part of the MarPAMM project and beyond.

Introduction.

The seas between Northern Ireland, the border regions of Ireland and Western Scotland (the area contained within the EU Interreg VA funded area, i.e., the Marine Protected Area Management and Monitoring (MarPAMM) project area), support internationally important populations of breeding seabirds. These include: the isle of Rum which hosts ~25% of the global Manx shearwater *Puffinus puffinus* breeding population; St Kilda which supports the second largest Northern gannet *Morus bassanus* (hereafter Gannet) colony in the world (59,622 pairs) (Murray, Harris and Wanless, 2015); and Rathlin Island at which ~40.5% (20,960 pairs) of the UK and Ireland Razorbill *Alca torda* population breed (Mitchell *et al.*, 2004). Under the EU Birds Directive (Directive 2009/147/EC), a network of Special Protection Areas (SPAs) is designated for these breeding seabirds throughout the Interreg VA area, to protect nationally/internationally important colonies, or large seabird assemblages (20,000+ individuals).

To achieve Conservation Objectives and Favourable Conservation Status for designated SPA features at protected sites, management actions are generally implemented at the site-level. However, many individual species occur at multiple SPAs and may be impacted by common threats and pressures either at their breeding locations or in foraging areas at sea. The suite of colonies within each region may also be subject to meta-population processes forming a network of interconnected sites which may be better managed through combined actions, although this can be challenging. Thus, management interventions implemented at a regional, rather than site level scale may benefit multiple species across a suite of SPAs within a given area (Oppel *et al.*, 2018). As such, conservation outcomes and management plans may be more successful and cost-effective for seabird populations through regional actions across these SPA networks. Through cross-border collaboration, the MarPAMM project aimed to develop four regional management plans: two in Scotland (Argyll and Outer Hebrides) and two in Northern Ireland/Republic of Ireland (Co Down – Co Louth region, North Coast – North Channel). In order to identify appropriate options for seabird conservation within these regional management plan areas, it is first necessary to identify the species for which actions may be most effective, an understanding of which is currently lacking, helping to ensure resources and conservation funding are most effectively targeted.

Several seabird populations within the Interreg VA area have undergone dramatic declines over the past few decades. These include Northern fulmar *Fulmarus glacialis* (hereafter Fulmar) and black-legged kittiwake *Rissa tridactyla* (hereafter Kittiwake). Others have increased, such as the Gannet and Great skua (also known as 'bonxie') *Stercorarius skua* (JNCC*,* 2021; NatureScot, 2021). These changes are potentially linked to a range of human pressures including introduced predators, fisheries activities, marine industries and climate change (Croxall *et al.*, 2012; Dias *et al.*, 2019). Incidental bycatch in fisheries has recently been identified as a threat to seabirds in the region with potential for population-level impacts (Northridge, Kingston and Coram, 2020). Invasive mammalian predators are also present at several seabird colonies, including the globally important Manx shearwater colony on Rum (Lambert, Carlisle and Cain, 2015; Carsile, 2019). There are also marine renewables proposed within the region which may have potential impacts on seabird populations (Furness, Wade and Masden, 2013; Scottish Government, 2020; Searle *et al.*, 2020). Conversely, the population increases observed in some species are potentially linked to favourable environmental conditions in recent decades and immigration from other regions (Murray *et al.*, 2015; Phillips *et al.*, 1999). Further, targeted removal of invasive mammalian predatory mammals through programmes such as the Biosecurity for Life (https://biosecurityforlife.org.uk/) has led to recolonization of several islands by some seabirds (i.e. the Shiant Isles), along with increased breeding success at some colonies (Luxmoore, Swann and Bell, 2019).

The regional management plans developed as part of the MarPAMM project may provide an opportunity to reduce and mitigate the impacts of these pressures across a suite of SPAs, helping to sustain important seabird populations. For instance, eradication of invasive predators across breeding colonies may result in improved breeding success, while reduced seabird bycatch within the region may increase survival rates. Moreover, addressing these threats may increase resilience to climate change impacts which are a key current and future pressure on seabirds in the region (Cleasby, L. and Davies, 2021; Davies, Humphreys and Pearce-Higgins, 2021; Johnston *et al.*, 2021; Pearce-Higgins, Davies and Humphreys, 2021). To achieve any mitigation of current and future pressures, it is first necessary to explore how improvements in demographic rates may impact population trajectories, which can then be used to prioritise between management intervention options, selecting those most likely to be effective at enhancing regional populations. As such, this paper details how we modelled the population level impacts of improvements in breeding success and adult survival as proxies of potential management interventions, providing a framework to prioritise regional management planning.

To explore the future population consequences of management interventions, Population Viability Analysis (PVA) can be used to simulate different scenarios in the presence /absence of different pressures following potential management interventions (Akçakaya and Sjögren-Gulve, 2000; Hostetler *et al.*, 2013). PVAs utilise species specific life-history values (age at first breeding, brood size) and demographic rates (breeding success and survival) to parameterise a mathematical population model to forecast future population sizes (Boyce, 1992). By alternating the input parameters, it is then possible to compare population trajectories under different scenarios. PVAs have been used for a wide range of purposes and can be used to predict future population sizes, extinction probabilities and population growth rates (Coulson *et al.*, 2001; Miller *et al.*, 2019). However, ratio metrics are considered more accurate than specific modelled population sizes/extinction probabilities which are considered to have relatively high uncertainty (Cook and Robinson, 2016; Green *et al.*, 2016; Jitlal *et al.*, 2017). By modelling the populations under both baseline (current) and scenario conditions (through improvements in breeding success/survival), the outputs of these models can then be used to examine the influence of potential management interventions on future regional seabird populations (Maclean, Frederiksen and Rehfisch, 2007; Freeman *et al.*, 2014).

PVAs were run for a suite of seabird species that are named designated SPA features (excluding seabird assemblages) within each regional management plan region. For each species within a region, a suite of individual closed sub-population models was run for each SPA at which the species is a designated feature. These were then combined into a single, regional PVA. For the two Scottish regional management plans (Argyll and Outer Hebrides), all SPAs for which seabirds are a designated feature were included. Within the two NI/RoI management regions, a refined list was agreed with project partners. Three individual sites were also included to the 4 management regions at the request of stakeholders: Canna and Sanday, Rum, and Monach Isles. Canna and Sanday SPA has achieved successful invasive mammal eradication (Luxmoore, Swann and Bell, 2019). Rum SPA hosts a globally important Manx shearwater colony (Mitchell *et al.*, 2004). Finally, anecdotal evidence suggesting that the black guillemot (*Cepphus grylle*) population is in decline at the Monach Isles Nature Conservation Marine Protected Area (NC MPA; (Pers. Comm. Glen Tyler)) has led to the recommendation of its inclusion into the PVA.

For each species-region/site combination, a baseline population model was run to represent current conditions and associate impacts on demographic rates. Scenario simulations were then run incorporating increases in breeding success and adult survival across a range of values, as a proxy for potential management interventions. The resultant outputs were then compared to address the following specific aims:

1. Compare modelled future population metrics between baseline and scenario conditions (i.e., improved demographic rates).
2. Identify the species for which management interventions which target breeding success and adult survival may be most effective.
3. Provide recommendations as to which management actions may be utilised to enhance seabird populations across the MarPAMM area.

Methods.

Site selection.

Information on protected sites and species was provided in a list format by staff under the MarPAMM Management Plan Work Package (Technical Work Package T5). This list, along with input from regional stakeholders, was used to identify key species and SPAs of interest to the MarPAMM project. The resultant list comprised 22 SPAs and 20 designated species across the four management regions, with 9 populations from seven species across the additional three sites (**Table 1**).

Population size and demographic data.

The latest population size estimates[1] were calculated for each designated species within an SPA using data from the Seabird Monitoring Programme (SMP) database (JNCC: Daisy Burnell/Ilka Win), or from other available databases where more recent counts were available (Bird Watch Ireland/National Parks and Wildlife Service: Kendrew Colhoun/David Tierney; RSPB: Mark Bolton). For the SMP data, each SPA ("Master Site" in the SMP database) comprises multiple "Site" sections which are surveyed periodically. For each designated species within an SPA, the total Master Site population count was calculated by summing the most recent population count across individual sites. These data comprised counts of Apparently Occupied Burrows (AOB: 30), counts of Apparently Occupied Nests (AON: 96), counts of Apparently Occupied Sites (AOS: 48), counts of Apparently Occupied Territories (AOT: 4) and Individuals (IND: 67; Appendix 1: Count Units). Where site counts were of individuals (Guillemot: 38; Razorbill: 22; Black guillemot: 7), these were converted to breeding pairs using a multiplication factor of 0.67 (Harris *et al.*, 2015). For comparability and to fit within the same modelling framework, all species were modelled as (and hereafter referred to) breeding pairs. Where site counts were from more than one year, the median

[1] At the time of writing this report, the Seabirds Count census, including the MarPAMM commissioned seabird counts were still ongoing and so contemporary counts for many of the sites were unavailable. Data were provided by the Seabird Monitoring Programme, a Scheme funded jointly by the British Trust for Ornithology and Joint Nature Conservation Committee, in association with the Royal Society for the Protection of Birds, with fieldwork conducted by both non-professional and professional surveyors.

year was used as the start year at a given SPA in the PVA (number: 18 SPAs; median year: 2015; year range: 1998-2018). We considered this the most pragmatic approach for this indicative exercise, such that we ensured the most recent counts were used rather than those which were often >20 years old (i.e., from the Seabird 2000 census).

For Leach's storm petrel (*Hydrobates leucorhous*), a recent population estimate for St Kilda SPA was provided by Mark Bolton (9918 AOB: 2019), while the most recent population estimate for North Rona and Sula Sgeir SPA (805 AOB: 2015) was extracted from Murray *et al.* (2016). Since no comparable recent population estimates were available for European storm petrels (*Hydrobates pelagicus*) at these SPAs, current (2019 and 2015, respectively) estimates were calculated based on the observed declines in Leach's storm petrel at these colonies (St Kilda SPA: 1121 AOB in 1999 to 791 AOB in 2019 based on a 70.6% reduction pers comm. Mark Bolton; North Rona and Sula Sgeir SPA: 760 AOB in 2001 to 334 AOB in 2015 based on the 0.44% reduction reported in Murray, Harris and Wanless, 2015). The results of a recent European storm petrel census on the Treshnish Isles SPA were extracted from the JNCC website (Committee, 2021).

It was considered that any populations with very low population sizes would result in unreliable outputs when input to the population models. As such, we omitted species for which the summed regional population comprised of less than 20 pairs across all SPAs combined. This resulted in the removal of 3 species from 2 regions (County Down to County Lough: Roseate tern (*Sterna dougallii*) – 1 pair; North Coast to North Channel: Arctic tern (*Sterna paradisaea*) – 0 pairs; Outer Hebrides: Little tern (*Sternula albifrons*) – 13 pairs).

For the majority of species, life-history/demographic rates (age at first breeding, recruitment age, breeding success and adult survival) were extracted from Horswill & Robinson (2015; **Table A2**). Where these were not available, rates were extracted from the literature.

For the majority of species, a single UK and Ireland level mean (± Standard Deviation - SD) breeding success value was utilised (Appendix 2: Demographic Rates). However, regional

(Western Scotland and Northern Ireland; (Cook and Robinson, 2010): European shag (*Gulosus aristotelis*; hereafter Shag); Gannet; Common guillemot) or SPA-level (St Kilda SPA: fulmar, Leach's storm-petrel, Atlantic puffin (*Fratercula arctica*; hereafter Puffin); Rum SPA: Manx shearwater; Canna and Sanday SPA: Shag) values were used where available. For adult survival, a single UK and Ireland level mean (±SD) value was utilised. Where values were absent for a given species, some were "borrowed" from a closely related species with the most similar life history (breeding success SD: European storm petrel value used for Leach's storm petrel; Adult Survival SD: Leach's storm petrel value used for European storm petrel; Lesser black-backed gull (*Larus fuscus*) value used for Great black-backed gull (*Larus marinus*), as recommended by Horswill and Robinson (2015).

As survival estimates for all immature age-classes were unavailable for the majority of species, we considered adult survival only in our modelling and thus made the necessary simplifying (though unrealistic) assumption that immature survival is the same as adult survival for the purpose of the population models.

Population Viability Analysis.

To assess the potential impacts of different management interventions on regional seabird populations we undertook a PVA approach. All analyses were conducted using R programming software (R Development Core Team, 2016), with figures created using the 'ggplot2' package (Wickham, 2009). PVAs were run using the "simple.scenarios" function from the 'nepva' package (Searle *et al.*, 2019) (for details of model options/inputs Appendix 3: Model Inputs). Leslie Matrix models are relatively simple to build and interpret, allowing different survival and breeding success rates to be modelled (Caswell, 2006). However, there is limited information on demographic rates at individual sites throughout the MarPAMM region. Uncertainty in demographic rates used to parameterise PVA models could lead to uncertainty in the predicted magnitude of the impact from a management intervention. To manage this uncertainty, we used stochastic models which are considered to be more precautionary than deterministic models (Lande, Engen and Saether, 2003). However, since confidence intervals generated by stochastic models only incorporate known/quantified uncertainty they are likely to be an underestimate of overall uncertainty around model outputs, and thus only qualitative comparison of outputs is appropriate. Both demographic and environmental stochasticity were included in population simulations to permit both chance population (demographic) and large-scale (environmental) effects to be simulated. Breeding success and adult survival rates were drawn from gamma and beta distributions respectively, ensuring rates were constrained to lie within biologically reasonable bounds.

PVA assumptions.

Several simplifying assumptions underpin the models:

- The relative impact of management measures remained constant over time, in proportion to the population size.
- Current (baseline) demographic parameters include the effects from all current pressures, in addition to natural (i.e., non-anthropogenic) processes operating on breeding success and adult survival. It is acknowledged that many of these rates are historical and may not consider many of the current pressures which seabirds face in the region. However, these are the only and best available data.
- These demographic parameters were assumed to remain constant during the period of the model, thus do not take account of environmental change or changes in anthropogenic pressures not considered in the models.
- The population affected by management measures for a given species comprises solely of individuals contained within SPAs for which the species is a feature in each region (i.e., they do not include neighbouring non-SPA colonies).
- Although metapopulation structures are known to occur between seabird populations, reliable connectivity estimates are currently unknown. As such, populations were modelled as closed populations, with no immigration and emigration accounted for between SPAs included in the analysis or other colonies at which the species is present, but not a designated feature. While this is unrealistic, it was considered a pragmatic and precautionary approach since immigration may buffer impacts on the focal population.

- Reproductive rates are independent of age - once an adult has reached 'age at first breeding', reproductive rates are constant across individuals of breeding age.
- Although the wider population may have more of one sex, we assumed an equal sex ratio when modelling (breeding pairs).
- Although there is some evidence of density dependence in some seabird populations, there is considerable uncertainty on the form and strength of these relationships. Density dependence was therefore not included in these analyses.

Scenarios.

Leslie Matrix models (Leslie, 1945) were used to compare the future population abundance generated by two scenarios - 1) simulated under current conditions (subject to the assumptions stated above) and associated values of breeding success and adult survival (i.e. baseline), and 2) simulated with the assumption of an unmeasured pressure being alleviated or mitigated through management interventions. To investigate the potential benefits of management interventions, scenarios were modelled to manipulate improvements in two key demographic rates that may be affected by these interventions - breeding success and adult survival. As long-lived species, with delayed maturity and low fecundity, seabird populations are generally considered more sensitive to changes in annual survival than changes in reproductive output, with the latter generally more variable between years in most species (Ricklefs, 1990; Schreiber and Burger, 2002). Further, for some species, greater relative increases in breeding success may be easier to achieve using certain management interventions (e.g., through rat eradication). As such, percentage point increases in breeding success were modelled over a greater range than survival (breeding success: +5 p.p.; +10 p.p.; +20 p.p.; survival: +1 p.p.; +2 p.p.; +3 p.p.). These range of values were agreed with key stakeholders and are in line with estimated or perceived impacts of current pressures on seabird populations (Jones *et al.*, 2008; Jitlal *et al.*, 2017; Northridge, Kingston and Coram, 2020), although these may be under/over-estimates of potential management benefits.

We implemented regional PVAs, incorporating each SPA where the species is a designated feature as a sub-population. Projections were run for each sub-population (SMP Master Site), which were combined to produce regional level population predictions. This approach may be useful when there are multiple count units over different years and calculating an initial total population is problematic. Where the count year differed between sub-populations, each sub-population was run from a different starting year, with the PVA combined after the latest year of initial population size. PVAs were run for 30 years (2020-2050), and impacts initiated in 2025 to align with likely timeframes of any management interventions resulting from MarPAMM to be initiated and take effect.

For Shag, population explosions occurred at +20 p.p. breeding success. As such, PVAs for this species were restricted to 5 p.p. and 10 p.p. only. It was not possible to model cormorant due to population explosions and associated modelling errors following any increases in breeding success and adult survival, and so this species was omitted from the analysis.

To test the impact of including different immature survival rates on the model outputs we ran a supplementary PVA for shag, for which complete immature survival rates were available (Juvenile (0-1): 0.51 ± 0.25 SD; Immature (1-2 year): 0.74 ± 0.18 SD; Adult survival (>3 year): 0.86 ± 0.19 SD). While the main analysis appears to overestimate modelled impacts, results were comparable between main and supplementary analysis. We therefore considered the use of a single adult survival rate across the age classes as appropriate for this indicative modelling exercise (Appendix 5: Supplementary analysis including immature Shag).

Metrics.

Where there is significant uncertainty in the life history and population parameters used to parameterise PVA models, it has been demonstrated that these can lead to unreliable results (Beissinger and Westphal, 1998; Chaudhary and Oli, 2020). This is particularly true for seabirds (Cook and Robinson, 2010; Masden *et al.*, 2015; Green *et al.*, 2016). Thus, Leslie Matrix models are best used to investigate relative population changes under different management scenarios, rather than attempting to generate quantitative predictions about future population sizes (Jitlal *et al.*, 2017). As such, we calculated two commonly used output metrics - 1) the ratio of impacted versus un-impacted population size (RPS) and 2) the ratio of impacted versus un-impacted population growth rate (RGR). For both, a ratio of

1 would equate to no change, whereas a ratio of 1.5 would mean a relative increase of 50% compared to the baseline scenario. In this context ‘impacted’ refers to populations for which potential management interventions have been implemented. RPS and RGR are reported as the predicted ratio in the final year of modelling (2050), due to the potential of sustained increases in mean population breeding success and adult survival following management interventions.

Results.

Argyll.

Of the four species modelled within the Argyll management region, the Common tern (*Sterna hirundo*) displayed the greatest increase in the ratio of impacted to un-impacted population size (RPS) and growth rate (RGR) following improvements in breeding success (**Figure 2**, **Figure 4**, Appendix 4: Modelled Values). In both Common tern and Kittiwake, improvements in breeding success were more effective than adult survival. For European storm petrel and Guillemot, improvements in adult survival were more effective.

Outer Hebrides.

Of the 11 species modelled within this region, the Shag displayed the most pronounced increase in RPS/RGR over the range of breeding success values modelled (+5 to +10 p.p.) (**Figure 2**, **Figure 4**, Appendix 4: Modelled Values), followed by two gull species, Kittiwake and the Great black-backed gull. The remaining species, including the petrels (Fulmar, Manx shearwater, European storm petrel, Leach's storm petrel), gannet and auks (Guillemot, Razorbill and Puffin) benefitted more from improvements in adult survival, which had a greater influence on RPS/RGR than improvements in breeding success over the range of values modelled.

North Coast - North Channel.

Of the seven species modelled in this region, the Shag displayed the most substantial increase in RPS/RGR following improvements in breeding success (+5 to +10 p.p.; **Figure 2**, **Figure 4**, Appendix 4: Modelled Values). The three gull species also displayed relatively greater increases in these two metrics following improvements in breeding success compared to adult survival, with the greatest increases predicted in the Common gull (*Larus canus*) followed by Kittiwake and Herring gull (*Larus argentatus*). Smaller effects of increased breeding success were observed in Fulmar, Guillemot and Razorbill, with improvements in adult survival more effective in these species.

County Down - County Louth.

Of the four species modelled within this region, the three tern species displayed substantial increases in both RGR and RPS following improved breeding success (**Figure 2**, **Figure 4**, Appendix 4: Modelled Values). The greatest increase was in the Arctic tern followed by the Common tern and Sandwich tern (*Thalasseus sandvicensis*). For Manx shearwater, improvements in breeding success only resulted in a modest increase in RPG/RGR. The four species displayed relatively consistent improvements in RGR/RPS following increases in survival.

Individual Sites.

Kittiwake benefited most from improvements in breeding success at Canna and Sanday SPA. This site also showed that Manx shearwater and Guillemot had the greatest increases in RPS/RGR from improvements in adult survival. Within the Monach Isles NC MPA, Black guillemot benefitted most from improvements in adult survival compared to breeding success, although the difference was marginal. On Rum SPA, Kittiwake benefited most from improvements in breeding success overall compared to any improvements in adult survival. Considering all 3 individual sites together, Guillemot benefitted the least from improvements in breeding success while for this species and Manx shearwater, improvements in adult survival were more effective than breeding success (**Figure 3**, **Figure 5**, Appendix 4: Modelled Values).

Discussion.

This analysis identifies a suite of species for which improvements in breeding success or adult survival may provide an appropriate target for management interventions within each of the MarPAMM regions. However, while the focus of this analysis is to inform potential management interventions within the different regions, for each species the results were largely similar across the four regions. This was because due to the lack of local demographic data, the majority of models used UK and Ireland level rates. As such, the only consistent difference between regions was the regional population sizes. Thus, here we provide a general overview of the patterns observed and discuss a suite of potential management interventions that could be considered which may be used to guide management planning.

General patterns.

Over the range of values modelled in this study, the greatest increases in RPS/RGR resulted from improvements in breeding success (+5 p.p., +10 p.p., +20 p.p.) compared to increases in adult survival (+1 p.p., +2 p.p., +3 p.p.). Crucially, the relative effect of improvements in breeding success and adult survival was related to species-specific life histories, in particular maximum brood size. For example, the greatest improvements resulting from improved breeding success was achieved in species which lay more than one egg (i.e., gulls, terns, and Cormorants; maximum brood size > 1). In Shags, the effects were so substantial that the highest increase (+20 p.p.) could not be modelled due to an ecologically implausible population explosion and associated modelling errors. Cormorant was not modelled for the same reason. However, such rapid population growth ties in with other studies on these species, which demonstrate “boom and bust” population trends (Frederiksen *et al.*, 2008). Although on average the improvements in adult survival were smaller when compared to some of the modelled effects of improved breeding success, improvements in adult survival resulted in greater increases in RPS/RGR in some species, in particular those which lay a single egg.

When interpreting these results, it is important to consider the relative effect of increasing demographic rates between species. For example, a 20 p.p increase in breeding success results in a relatively larger increase in the number of chicks fledged in species that raise more than one chick (such as Shags) compared to species that lay a single egg. Similarly, in species which have relatively higher survival rates, the ability to achieve improvements over the range of values modelled here may be more challenging than in a species with a lower rate. These results highlight the importance of considering seabird life history and specific demographic rates when considering management interventions.

Management interventions.

We have highlighted below some of the potential management interventions which may be utilised within and across the four management regions (Argyll, Outer Hebrides, North Coast – North Channel, County Down – County Louth) to improve breeding success and adult survival across the species included in the analysis. While this does not provide an exhaustive list, we have used a combination of expert opinion and literature to inform our suggestions, which combined with the PVA results may be used to inform regional management plans[2].

Sustainable management of prey populations.

Prey availability is a key determinant of seabird breeding success (Cury *et al.*, 2011; Saraux *et al.*, 2021), operating either directly through impacts on the quantity/quality of food delivered to chicks, or through indirect effects on parental foraging behaviour, with consequences on provisioning/attendance rates and predation risk. Thus, a key focus of regional management plans should be to ensure that key prey populations are managed through an ecosystem approach, with consideration to the food requirements of marine predators, including seabirds, and important feeding locations protected. This requires an understanding of where and when birds forage (which is lacking for many colonies), an understanding of which is being collated through seabird tagging activities undertaken as

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[2] Note these were collated separately to Pearce-Higgins et al. 2021. Species and habitat climate change adaptation options for seabirds within the INTERREG VA area (https://www.mpa-management.eu/?p=1252). However, we advise that readers also consider that report.

part the wider MarPAMM project. While managing prey populations will likely benefit the majority of seabirds in the region, such measures may be particularly beneficial for surface feeding species, such as terns and Kittiwake, which may have limited dietary flexibility compared to some other species. The extent to which prey populations are limited across the MarPAMM region by fisheries was not assessed during this project, so it is not clear what specific potential there is for seabird populations to benefit within the region.

Reduce seabird bycatch.

Incidental bycatch in fisheries has been identified as a threat to seabirds in the MarPAMM region, with the potential for population-level impacts for some species, notably fulmar for the region (Miles, Parsons and O'Brian, 2020; Northridge, Kingston and Coram, 2020). Thus, there is increasing interest in bycatch mitigation options in the region to reduce seabird mortality (e.g., the UK Marine wildlife bycatch mitigation initiative)[3]. Effective development and implementation of such measures requires consideration of fishing industries and communities, to avoid unintended consequences on these communities (Komoroske and Lewison, 2015). Thus, a key focus of the regional management plans may be to work with key stakeholders to further understand the extent and factors associated with a seabird bycatch in the region, to help identify solutions. While risk factors associated with seabird bycatch vary between fishery activities and seabird species ecology, surface feeding species, (such as Fulmar) may be particularly vulnerable to longlines (Bradbury *et al.*, 2017), and thus may provide targets for management interventions in some of the MarPAMM regions.

Invasive predator removal/eradication.

Non-native predators, including mink, ferrets and rats are a key threat to seabirds in the MarPAMM region and are present at several of the SPAs included in the PVA analysis, such as the globally important Manx shearwater colony on Rum (Lambert, Carlisle and Cain, 2015; Carsile, 2019). While requiring substantial resources, successful eradication of mammalian predators has been achieved at several seabird colonies worldwide, resulting in population recovery (Jones *et al.*, 2016; Williams *et al.*, 2020). Eight out of 25 priority sites

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[3] DEFRA (2022). Marine wildlife bycatch mitigation initiative. https://www.gov.uk/government/publications/marine-wildlife-bycatch-mitigation-initiative/marine-wildlife-bycatch-mitigation-initiative

identified for invasive mammal eradication in UK and Ireland in an earlier study are within the MarPAMM area (Stanbury *et al.*, 2017). As such, regional management plans may provide a mechanism through which to coordinate effective biosecurity, monitoring and eradication practices. This would require considerable stakeholder engagement but could build upon the success of the Biosecurity for Life project (due to end in 2022) and feed into the recently established LIFE Raft Project focussed on Rathlin Island. Removal of invasive species may benefit all species but is likely to be particularly effective for burrow nesting species such as petrels and Puffin, and ground-nesting species such as gulls and terns.

Exclude native predators.

Native predators including mammals, such as otters and foxes, and avian predators such as crows and Sea eagles can also impact seabird populations, by reducing breeding success. Management of this pressure may be achieved through protective fencing to exclude ground predators (Babcock and Booth, 2020; Williams *et al.*, 2020), the use of canes to deter aerial predators (Boothby, Redfern and Schroeder, 2019), or licensed removal of eggs/culling is also an option in specific circumstances. Although such interventions relating to natural or invasive predators would need to be done at site level, a regional joined up approach to the design and deployment of such activities would ensure that the most cost-effective use of resources is achieved and utilised. Such interventions may be particularly effective at improving breeding success of ground-nesting species such as gulls and terns. Such measures must be very carefully considered weighing up the costs and benefits.

Create nesting habitat.

While the focus of regional management plans is likely to be mitigating existing pressures and maintaining current seabird colonies, nesting habitat creation is also a potential option for some species (Furness *et al.*, 2013; Williams *et al.*, 2020). However, uptake of such structures is variable and so careful consideration must be given to both the location and species-specific behaviours. Habitat creation could involve the construction of nesting platforms or islands for species such as gulls and terns, artificial nesting platforms for Kittiwakes, and nesting boxes or colonies for burrow nesting species, such as petrels.

Careful planning for renewable developments.

To date, there has been limited Offshore Wind Farm (OWF) development in the MarPAMM region. However, both the Scottish and Irish Governments have ambitious targets for marine renewable energy generation over the next decade (Scottish Government 2020, Government of Ireland 2021). As OWFs have the potential to impact seabird populations in the region, this expansion requires careful spatial management. The two main mechanisms through which OWFs may affect seabird populations is via collision and displacement/barrier effects (Searle *et al.*, 2019). Further, although there is currently a limited understanding of underwater collision risk (Grecian *et al.*, 2012), diving seabirds, such as auks and Shag could also potentially collide with tidal turbines (Furness *et al.*, 2012).

Reduce disturbance.

Seabirds are vulnerable to disturbance from human activities, both on land during breeding season and at sea when foraging/resting. Disturbance from recreation can cause flushing of adults, leaving eggs/chicks vulnerable to the elements/predators, with impacts on breeding success (Watson, Bolton and Monaghan, 2014). At sea, vessel traffic, associated with shipping, construction, or recreation can displace birds from key foraging areas (Fliessbach *et al.*, 2019), which may be particularly important when adults are provisioning for young or when disturbance occurs during the energetically challenging winter months, with impacts on individual condition and survival. To manage and reduce recreational pressures on land, clear signage and fencing could be installed at key colonies, which has been shown to reduce human nuisance behaviours (Allbrook and Quinn, 2020). At sea, vessel wildlife codes could be developed for vessel operators while shipping guidance may help reduce disturbance to birds engaging in foraging/maintenance behaviours at sea. Cormorants, Shags and auks are particularly vulnerable to disturbance at the colony level, while auks, particularly during the flightless period, are strongly affected by disturbance at sea.

Marine Litter.

Marine litter is an increasing threat to seabirds, via both entanglement at sea, ingestion during feeding activity and during the breeding season with population-level impacts yet to be fully understood (Provencher *et al.*, 2020). While sources of marine litter are varied, engagement with fisheries may be one option to reduce marine litter alongside a coordinated monitoring regime to help identify litter hotspots and sources (O'Hanlon *et al.*, 2019; Thompson *et al.*, 2020). Surface feeding species, such as Fulmar, may be particularly vulnerable to ingestion, while nest building species, including Gannet, gulls and Shag may be vulnerable to entanglement.

Limitations.

While this analysis utilises the most up to date data for seabird populations within the MarPAMM region, the demographic rates used are unlikely to be true representations of current conditions. This is because the demographic rates used are based on historical data, largely collated over the latter half of the 20th century, where pressures may have varied compared to those currently affecting populations. For example, fisheries practices may have changed, such as the species/stocks targeted, areas used and technologies/equipment deployed, with impacts on discards and bycatch rates.

While count units were generally consistent within individual SPAs, there were some differences between sites included within the same regional SPA (Appendix 1: Count Units). As confirmed nests or occupied burrows provide a more robust measure of a breeding pairs than AONs or the number of individuals. As we are not able to account for the error associated with these different count units, this may reduce certainty in our results. To improve the representativeness of this analysis, more up to date information would be needed for demographic rates and population sizes.

Another key limitation was that while populations were modelled at a regional scale, incorporating multiple colonies in a single PVA, these were closed populations, with no immigration or emigration. Despite this being a biologically unrealistic assumption (Miller *et al.*, 2019), we had no data to model metapopulational dynamics. Thus, future analyses should aim to incorporate immigration/emigration throughout the regional (and wider) meta-population, to provide a better approximation of reality. Similarly, by neither modelling different adult and immature rates nor adding different impacts to these age classes, we may be under/over estimating effects. For example, juveniles may be particularly vulnerable to predation or display increased bycatch rates, and so management interventions targeted at this age class may be more effective (Genovart, Oro and Tenan, 2018). Indeed, the supplementary analysis of Shag modelled to include juvenile age classes with lower survival rates, indicates that our main results overestimate the potential benefits of management interventions.

Finally, while substantial climate change impacts are predicted to occur on seabirds in within the MarPAMM area (Cleasby, L. and Davies, 2021; Davies, Humphreys and Pearce-Higgins, 2021), this report does not account for such effects in the analysis or cover specific management interventions to mitigate this pressure (but see Pearce-Higgins *et al*., 2021). However, the suite of potential management interventions outlined may increase resilience in seabird populations within the MarPAMM area, helping to buffer the overall negative impacts associated with climate change. When developing management interventions, it will be important to consider both these predicted climate-mediated alterations changes in seabird populations and distributions, in addition to any potential changes in environmental conditions (e.g., sea level rise/prey populations).

Conclusions.

This analysis explores those seabird species within the MarPAMM region which may benefit from management interventions targeted at improving either breeding success or adult survival. The results of the analysis and the potential management interventions identified may be used as a guide for the development of regional management plans now and in the future, which through cross-border collaboration may allow for more effective seabird conservation to be achieved. In addition to the MarPAMM project, a number of national seabird conservation strategies are currently under development, which may provide a further mechanism through which some of these potential management interventions may be considered. While all of the measures outlined here are intended to result in positive effect on seabird demography overall, it is also important to consider unintended consequences of management interventions. For example, removal of invasive predators may lead to accidental poisoning of non-target species or alterations to key prey populations (Travers *et al.*, 2021), while fishery closures can lead to increased bycatch in non-target species (Abbott and Haynie, 2012). Thus, where possible, to ensure maximum benefit is achieved from management interventions, they should be designed and implemented such that data is collected or experiments incorporated so new learning can be gained to inform future action (Ockendon *et al.*, 2021). Further, for management interventions to be developed and deployed effectively and result in long-lasting, and meaningful change, it is essential that local communities and stakeholders are involved with these processes (Lewison *et al.*, 2012), which is a key focus of MarPAMM. Finally, while the Seabirds Count census has recently been completed, the results are not yet collated, summarised and published. Given the unprecedented threats that seabirds in the UK and Ireland face this report, combined with these updated counts, will provide key information upon which to prioritise management and conservation of the internationally important seabird populations within the MarPAMM area and beyond.

Tables.

Table 1. *SPA site list and species which are designated features across the four MarPAMM management regions. Ticks (✓) indicate species designation at an SPA, with the total number of SPAs in which a species is a designated feature provided in* ***bold****. NC MPA: Nature Conservation Marine Protected Area. * were not modelled due to low regional population sizes (<20 pairs).*

	Fulmar	**Manx shearwater**	**European storm petrel**	**Leach's storm petrel**	**Gannet**	**Cormorant**	**Shag**	**Kittiwake**	**Common Gull**	**Herring Gull**	**Great black-backed gull**	**Little tern**	**Sandwich tern**	**Common tern**	**Roseate tern**	**Arctic tern**	**Guillemot**	**Razorbill**	**Black guillemot**	**Puffin**	**Total Number of Species**
Argyll			**1**					**1**						**1***			**1**				**4**
Glas Eileanan SPA														✓							1
North Colonsay and Western Cliffs SPA								✓									✓				2
Treshnish Isles SPA			✓																		1
County Down - County Louth		**1**											**3**	**4**	**1***	**4**					**13**
Belfast Lough SPA														✓		✓					2
Carlingford Lough SPA													✓	✓							2
Copeland Islands SPA		✓														✓					2
Larne Lough SPA													✓	✓	✓						3
Outer Ards SPA																✓					1
Strangford Lough SPA													✓	✓		✓					3
North Coast - North Channel	**2**					**2**	**5**	**3**	**2**	**3**						**1**	**2**	**3**			**23**
Horn Head to Fanad Head SPA	✓					✓	✓	✓									✓	✓			6
Inishmurray SPA							✓			✓						✓					3
Inishtrahull SPA							✓		✓												2

Rathlin Island SPA								✓									✓	✓			3
West Donegal Coast SPA	✓					✓	✓	✓		✓								✓			6
West Donegal Islands SPA							✓		✓	✓											3
Outer Hebrides	**5**	**1**	**2**	**3**	**2**		**2**	**5**			**1**	**2***					**5**	**5**		**5**	**38**
Flannan Isles SPA	✓			✓				✓									✓	✓		✓	6
Mingulay and Berneray SPA	✓						✓	✓									✓	✓		✓	6
Monach Isles SPA												✓									1
North Rona and Sula Sgeir SPA	✓		✓	✓	✓			✓			✓						✓	✓		✓	9
Shiant Isles SPA	✓						✓	✓									✓	✓		✓	6
South Uist Machair and Lochs SPA												✓									1
St Kilda SPA	✓	✓	✓	✓	✓			✓									✓	✓		✓	9
Individual Sites		**1**					**1**	**2**		**1**							**2**		**1**	**1**	**9**
Canna and Sanday SPA							✓	✓		✓							✓			✓	5
Monach Isles NC MPA																			✓		1
Rum SPA		✓						✓									✓				3
Total Number of Sites	7	3	3	3	2	2	8	11	2	4	1	2	3	5	1	5	10	8	1	6	87

Figures.

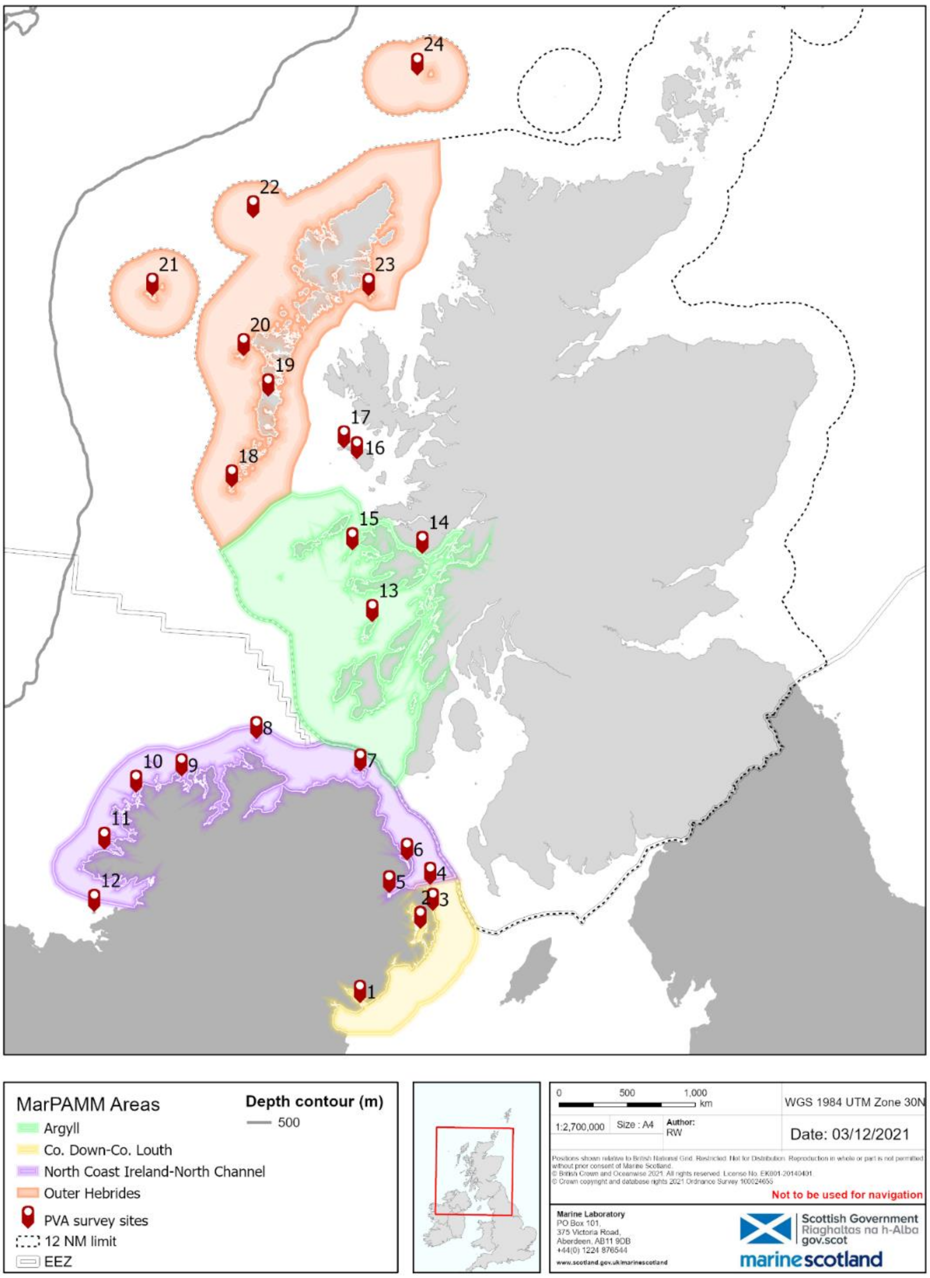

Figure 1. *Map of seabird breeding sites included in the analysis.*

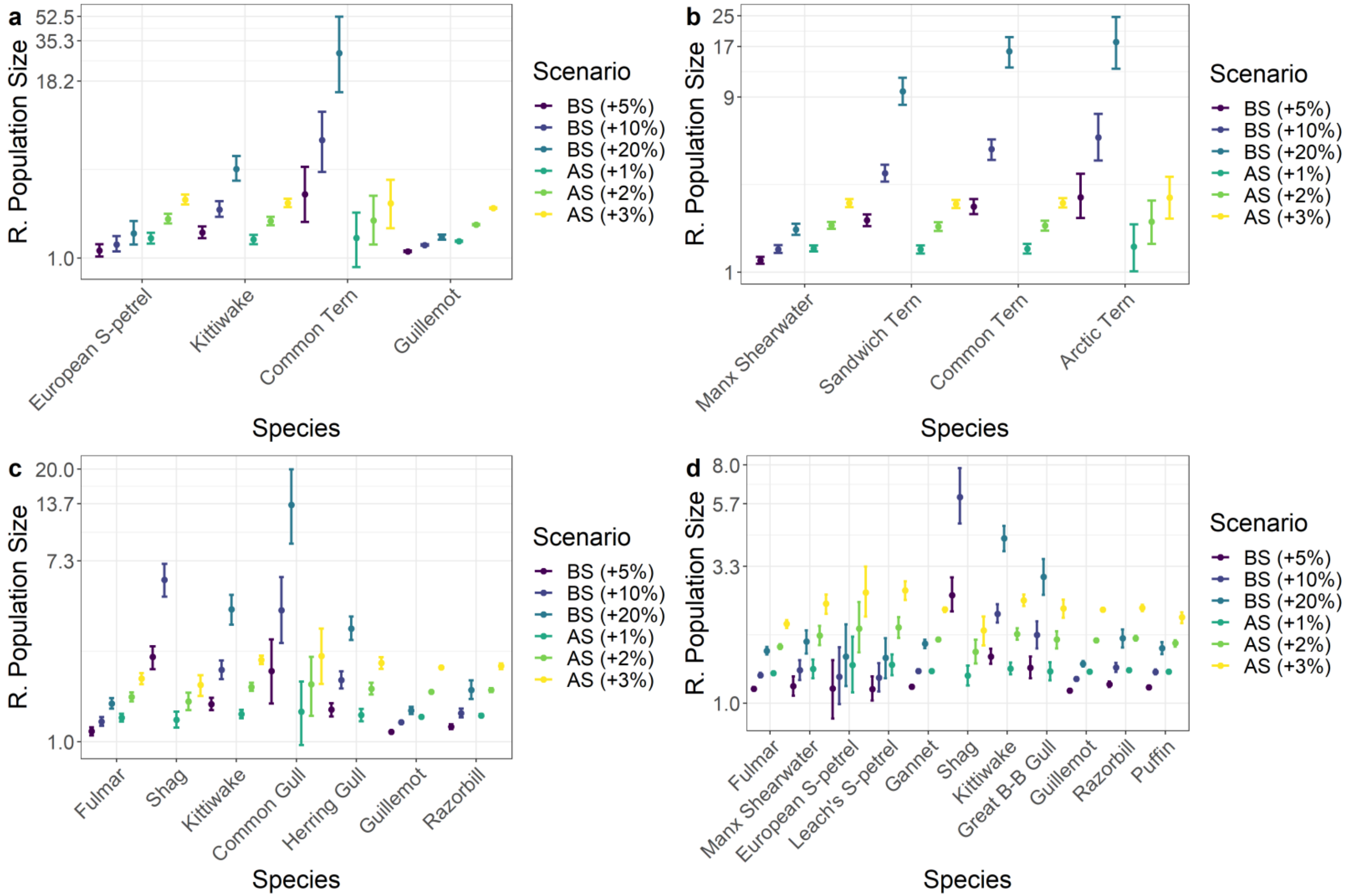

Figure 2. *Ratio of impacted to un-impacted Population Size for all species occurring within the a) Argyll, b) County Down - County Louth, c) North Coast - North Channel, and d) Outer Hebrides management regions by 2050, following increases in BS = Breeding Success and AS = Adult Survival.*

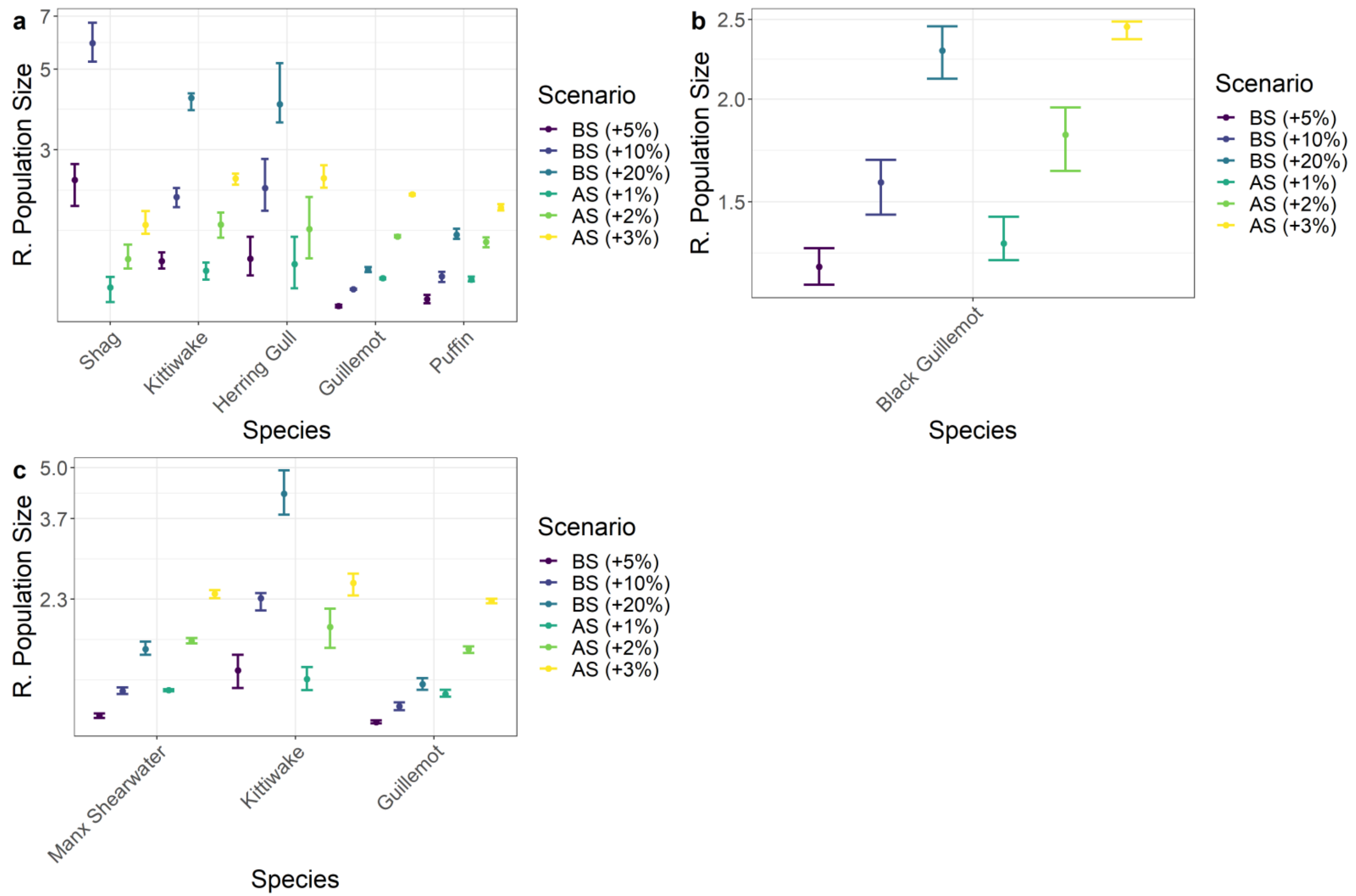

Figure 3. *Ratio of impacted to un-impacted Population Size for all species occurring within a) Canna and Sanday SPA, b) Monach Isles NC MPA, and c) Rum SPA by 2050, following increases in BS = Breeding Success and AS = Adult Survival.*

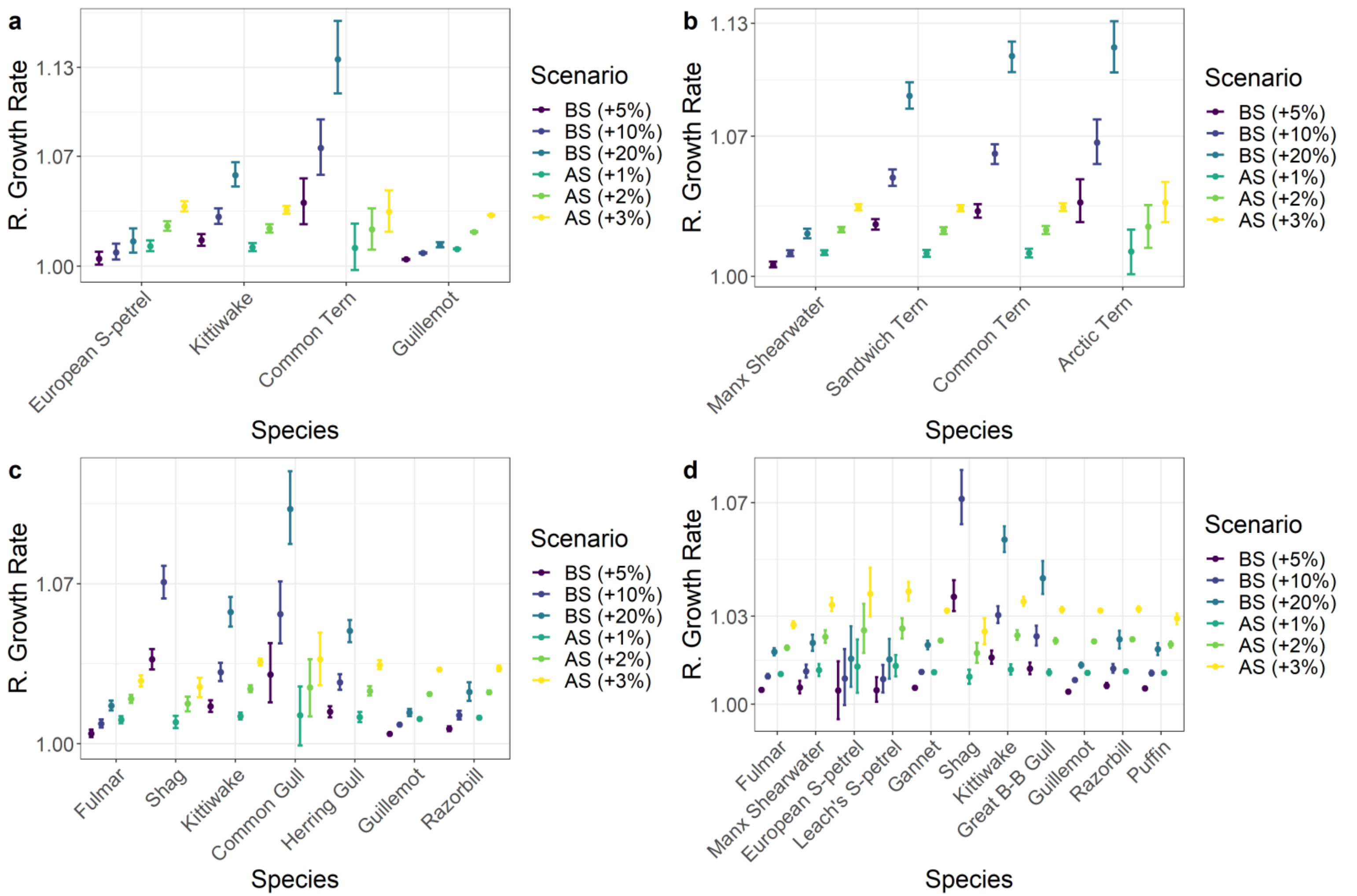

Figure 4. *Ratio of impacted to un-impacted Growth Rate for all species occurring within the a) Argyll, b) County Down - County Louth, c) North Coast - North Channel, and d) Outer Hebrides management regions by 2050, following increases in BS = Breeding Success and AS = Adult Survival.*

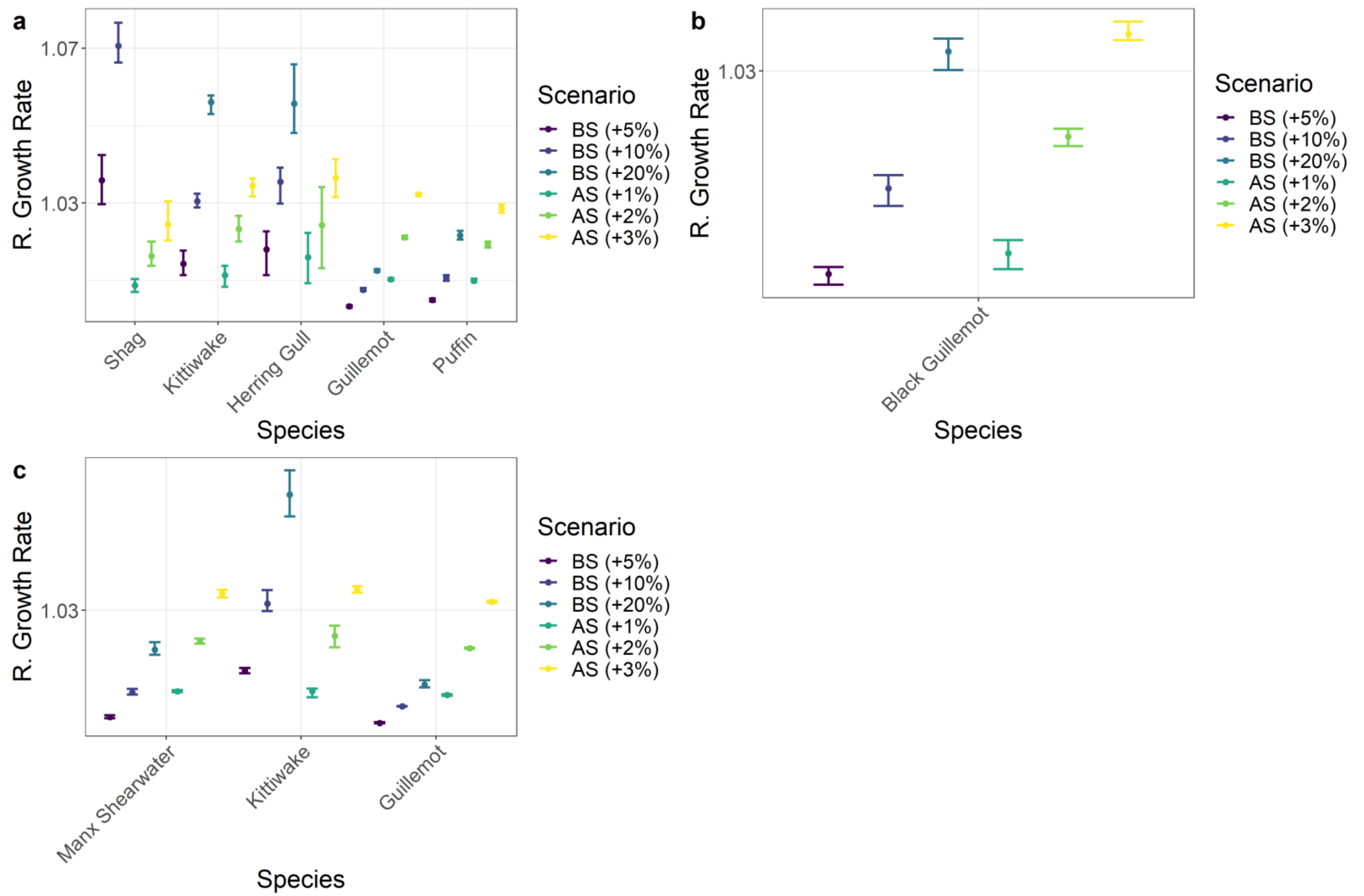

Figure 5. *Ratio of impacted to un-impacted Population Growth Rate for all species occurring within a) Canna and Sanday SPA , b) Monach Isles NC MPA, and c) Rum SPA by 2050, following increases in BS = Breeding Success and AS = Adult Survival.*

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Appendix.

Appendix 1: Count Units.

Table A1. *Count units used to calculate total population sizes for each of the SPAs included in the PVA analysis. AOB: Apparently Occupied Burrows, AON: Apparently Occupied Nests; AOS: Apparently Occupied Sites, IND: Individuals.*

Common Name	**AOB**	**AON**	**AOS**	**AOT**	**IND**
Fulmar	0	1	21	0	0
Manx shearwater	2	0	8	0	0
European storm petrel	1	0	9	0	0
Leach’s storm petrel	0	0	9	0	0
Gannet	0	2	0	0	0
Cormorant	0	2	0	0	0
Shag	0	13	1	0	0
Kittiwake	0	41	0	0	0
Common Gull	0	2	0	0	0
Herring gull	0	3	0	1	0
Great black-backed gull	0	2	0	0	0
Little tern	0	4	0	0	0
Sandwich tern	0	5	0	0	0
Common tern	0	8	0	1	0
Roseate tern	0	1	0	0	0
Arctic tern	0	10	0	2	0
Guillemot	0	1	0	0	38
Razorbill	0	1	0	0	22
Black guillemot	0	0	0	0	7
Puffin	27	0	0	0	0
Total	30	96	48	4	67

Appendix 2: Demographic Rates.

Table A2. *Demographic Rates used in the PVA analysis. Maximum Brood size = MBS, Age at First Breeding = AFB. For the majority of species demographic rates were taken from Horswill and Robinson (2015). For European storm petrel and Leach's storm petrel, these values were taken from the literature with these indicated with footnotes. Where values were absent for a given species similar values were borrowed from a closely related species (breeding success SD: Leach's storm petrel uses European storm petrel; Adult Survival S: European storm petrel uses Leach's storm petrel; Great black-backed gull uses Lesser black-backed gull).*

Common Name	MBS	AFB	Extent	Breeding Success		Adult Survival	
				Mean	SD	Mean	SD
Fulmar	1	9	UK and Ireland	0.42	0.13	0.94	0.06
	1	9	St Kilda	0.28	0.07		
Manx shearwater	1	5	National	0.70	0.10	0.87	0.08
	1	5	Rum	0.68	0.14		
European storm petrel	1	5	UK and Ireland	0.60[4]	0.12*	0.80[5]	0.04^
Leach's storm petrel	1	5	St Kilda	0.64[6]	0.12*	0.78[7]	0.04^
Gannet	1	5	UK and Ireland	0.70	0.08	0.92	0.04
	1	5	West	0.71	0.11		
Cormorant	4	3	UK and Ireland	1.99	0.66	0.87	0.06
Shag	4	2	UK and Ireland	1.30	0.48	0.86	0.19
	4	2	West	1.13	0.51		
	4	2	Canna and Sanday	1.60	0.51		
Kittiwake	2	4	National	0.53	0.33	0.85	0.05
Common gull	3	3	National	0.54	0.39	0.83	0.05
Herring gull	3	5	National	0.92	0.48	0.89	0.02
	3	5	Canna and Sanday	0.74	0.64		
Lesser black-backed gull	3	5	National	0.53	0.33	0.89	0.02!
Great black-backed gull	3	5	National	1.14	0.53	0.93	0.02!
	3	5	Canna and Sanday	0.81	0.62		
Little tern	1	2	National	0.52	0.52	0.80	0.03
Sandwich tern	3	3	National	0.70	0.37	0.90	0.03
Common tern	3	3	National	0.76	0.47	0.88	0.01
	3	3	Glas Eileanan SPA	0.53	0.53		
Arctic tern	3	4	National	0.38	0.33	0.84	0.04
Guillemot	1	6	National	0.67	0.15	0.94	0.02
	1	6	West	0.82	0.06		

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Razorbill	1	5	National	0.57	0.25	0.90	0.07
Black guillemot	2	4	National	1.30	0.32	0.87	0.03
Puffin	1	5	National	0.62	0.15	0.91	0.08
	1	5	St Kilda SPA	0.79	0.08		

Appendix 3: Model Inputs.

Table A3. *Model options used with the NE PVA Tool.*

Model component	nevpa option	Justification
Environmental stochasticity	model.envstoch = “betagamma”	Included to account for large-scale environmental effects.
Demographic stochasticity	Model.demostoch = TRUE	Included to account for population effects.
Density dependence	model.dd = “nodd”	Density Dependence not included as insufficient evidence for these populations.
Scenarios	nscen = 6, include.baseline = TRUE	Used to specify the number of scenarios (6) in which a demographic parameter was improved. Baseline conditions also modelled.
Simulations	sim.n = 1000	1000 simulations run for each scenario run.
Years modelled	output.year.start = 2020, output.year.end = 2050	Range of years over wghich models are run (2020-2050)
Rates	demobase.splitpops = TRUE	Allows lowest level of demographic rates to be used
Productivity	model.prodmax = TRUE	Productivity rates constrained at Maximum Brood Size to ensure biologically valid ranges.
Demographic rates	demobase.splitpops = TRUE	Demographic rates were specified for each sub population at the lowest level possible: UK and Ireland < Western Region < SPA.
Immature rates	demobase.splitimmat = FALSE	Due to limited information on immature demographic rates, adult rates were used across all age classes.
Starting population sizes	inipop.inputformat = breeding.pairs	Initial population sizes were calculated as breeding pairs from nests/sites or transformed individual counts.
Management interventions start/end	impacts.year.start = 2025, impacts.year.end = 2050	Range of years over which management interventions are included (2020-2050)
Management interventions rates	impacts.prod.mean = c(0.05, 0.10, 0.20, 0, 0, 0), impacts.survadult.mean = c(0, 0, 0, 0.01, 0.02, 0.03)	Range of percentage point changes in breeding success (5 p.p., 10, p.p, 20 p.p.) and adult survival (1 p.p., 2, p.p, 3 p.p.)

Appendix 4: Modelled Values.

Table A4a. *Modelled Ratio of impacted to un-impacted Population Size by 2050, for each species within the four MarPAMM management regions and Individual Sites, following percentage point (p.p.) increases in breeding success (BS) and adult survival (AS).*

Region/ Individual Site	Common name	+5 p.p. BS	+10 p.p. BS	+20 p.p. BS	+1 p.p. AS	+2 p.p. AS	+3 p.p. AS
Argyll	European storm-petrel	1.13 (1.03-1.26)	1.25 (1.11-1.43)	1.5 (1.25-1.83)	1.38 (1.26-1.51)	1.9 (1.76-2.06)	2.61 (2.41-2.83)
	Kittiwake	1.52 (1.38-1.68)	2.21 (1.96-2.54)	4.31 (3.55-5.31)	1.36 (1.25-1.46)	1.83 (1.71-1.96)	2.46 (2.3-2.64)
	Common tern	2.84 (1.81-4.45)	6.91 (4.1-11.01)	28.8 (15.15-52.28)	1.39 (0.86-2.11)	1.85 (1.25-2.77)	2.45 (1.63-3.61)
	Guillemot	1.12 (1.11-1.13)	1.24 (1.22-1.25)	1.41 (1.35-1.47)	1.32 (1.3-1.33)	1.73 (1.71-1.75)	2.26 (2.24-2.28)
Co. Down to Co. Louth	Manx shearwater	1.16 (1.11-1.21)	1.33 (1.27-1.4)	1.71 (1.6-1.84)	1.35 (1.29-1.4)	1.8 (1.72-1.88)	2.38 (2.26-2.5)
	Sandwich tern	1.92 (1.78-2.07)	3.47 (3.12-3.86)	9.69 (8.17-11.48)	1.33 (1.26-1.4)	1.78 (1.68-1.87)	2.35 (2.24-2.47)
	Common tern	2.28 (2.07-2.51)	4.68 (4.08-5.29)	15.99 (13.02-19.11)	1.34 (1.26-1.43)	1.79 (1.68-1.91)	2.38 (2.25-2.53)
	Arctic tern	2.56 (1.97-3.44)	5.43 (4.06-7.28)	17.96 (12.86-24.64)	1.38 (1.01-1.82)	1.88 (1.43-2.46)	2.55 (1.96-3.31)
N. Coast to N. Channel	Fulmar	1.12 (1.07-1.17)	1.25 (1.19-1.31)	1.52 (1.44-1.61)	1.3 (1.25-1.36)	1.64 (1.56-1.72)	2 (1.88-2.12)
	Shag	2.54 (2.22-2.85)	5.93 (4.92-7.07)	-	1.27 (1.17-1.39)	1.56 (1.41-1.71)	1.87 (1.65-2.08)
	Kittiwake	1.51 (1.41-1.62)	2.2 (1.99-2.44)	4.28 (3.61-5.01)	1.35 (1.29-1.42)	1.83 (1.75-1.91)	2.46 (2.35-2.57)
	Common gull	2.17 (1.52-3.08)	4.24 (2.96-6.11)	13.52 (8.8-19.94)	1.39 (0.96-1.93)	1.88 (1.33-2.54)	2.56 (1.89-3.47)

	Herring gull	1.42 (1.32-1.53)	1.97 (1.79-2.17)	3.47 (3.06-3.95)	1.34 (1.25-1.43)	1.79 (1.68-1.91)	2.38 (2.22-2.53)
	Guillemot	1.12 (1.11-1.12)	1.24 (1.23-1.25)	1.41 (1.35-1.47)	1.32 (1.31-1.32)	1.73 (1.72-1.74)	2.26 (2.25-2.27)
	Razorbill	1.18 (1.15-1.21)	1.37 (1.3-1.44)	1.77 (1.6-1.96)	1.33 (1.31-1.35)	1.76 (1.72-1.8)	2.3 (2.21-2.36)
Outer Hebrides	Fulmar	1.13 (1.12-1.15)	1.28 (1.25-1.3)	1.58 (1.53-1.63)	1.3 (1.29-1.32)	1.64 (1.6-1.67)	2 (1.93-2.06)
	Manx shearwater	1.16 (1.07-1.26)	1.34 (1.22-1.46)	1.71 (1.54-1.89)	1.35 (1.24-1.46)	1.8 (1.66-1.96)	2.38 (2.18-2.58)
	European storm-petrel	1.14 (0.88-1.45)	1.26 (0.99-1.63)	1.5 (1.16-1.99)	1.4 (1.1-1.79)	1.92 (1.56-2.41)	2.63 (2.13-3.29)
	Leach's storm-petrel	1.13 (1.02-1.26)	1.25 (1.11-1.42)	1.49 (1.24-1.77)	1.4 (1.28-1.53)	1.94 (1.77-2.12)	2.68 (2.46-2.9)
	Gannet	1.16 (1.15-1.17)	1.33 (1.31-1.35)	1.68 (1.62-1.74)	1.32 (1.32-1.33)	1.74 (1.72-1.76)	2.27 (2.21-2.31)
	Shag	2.56 (2.23-3)	6.05 (4.8-7.78)	-	1.27 (1.17-1.39)	1.57 (1.41-1.74)	1.89 (1.65-2.13)
	Kittiwake	1.5 (1.41-1.61)	2.18 (2.02-2.37)	4.22 (3.77-4.7)	1.35 (1.29-1.43)	1.83 (1.74-1.92)	2.45 (2.34-2.58)
	Great black-backed gull	1.36 (1.24-1.5)	1.81 (1.61-2.05)	3.01 (2.57-3.52)	1.32 (1.22-1.43)	1.74 (1.61-1.87)	2.29 (2.11-2.47)
	Guillemot	1.12 (1.11-1.12)	1.24 (1.23-1.25)	1.41 (1.38-1.44)	1.32 (1.31-1.32)	1.73 (1.72-1.74)	2.26 (2.25-2.28)
	Razorbill	1.18 (1.15-1.21)	1.37 (1.31-1.42)	1.76 (1.63-1.9)	1.33 (1.31-1.36)	1.76 (1.73-1.8)	2.3 (2.23-2.36)
	Atlantic puffin	1.15 (1.14-1.16)	1.31 (1.29-1.34)	1.62 (1.54-1.7)	1.32 (1.3-1.33)	1.69 (1.64-1.73)	2.12 (2.01-2.21)
Canna and Sanday SPA	Shag	2.62 (2.13-3.24)	6.31 (4.6-9.18)	-	1.29 (1.13-1.47)	1.59 (1.38-1.86)	1.94 (1.64-2.34)
	Kittiwake	1.52 (1.33-1.74)	2.22 (1.91-2.65)	4.32 (3.48-5.45)	1.36 (1.19-1.52)	1.83 (1.61-2.06)	2.46 (2.19-2.77)

	Herring gull	1.55 (1.14-2.11)	2.26 (1.64-3.09)	4.3 (2.92-6.46)	1.36 (1.01-1.85)	1.81 (1.38-2.4)	2.41 (1.83-3.17)
	Guillemot	1.12 (1.09-1.14)	1.24 (1.21-1.27)	1.41 (1.35-1.47)	1.32 (1.29-1.35)	1.73 (1.69-1.77)	2.26 (2.22-2.32)
	Atlantic puffin	1.17 (1.12-1.23)	1.36 (1.3-1.44)	1.78 (1.65-1.94)	1.32 (1.26-1.38)	1.69 (1.6-1.78)	2.12 (1.96-2.27)
Monach Isles NC MPA	Black guillemot	1.27 (1.15-1.4)	1.58 (1.42-1.74)	2.3 (2.01-2.6)	1.35 (1.23-1.48)	1.81 (1.65-1.99)	2.42 (2.2-2.66)
Rum SPA	Manx shearwater	1.16 (1.14-1.18)	1.34 (1.3-1.37)	1.73 (1.64-1.82)	1.35 (1.33-1.36)	1.8 (1.77-1.83)	2.38 (2.3-2.45)
	Kittiwake	1.52 (1.23-1.85)	2.22 (1.79-2.74)	4.32 (3.34-5.55)	1.36 (1.13-1.65)	1.84 (1.53-2.2)	2.46 (2.03-2.95)
	Guillemot	1.12 (1.09-1.14)	1.24 (1.2-1.27)	1.41 (1.34-1.48)	1.32 (1.28-1.35)	1.73 (1.68-1.77)	2.26 (2.2-2.32)

Table A4b. *Modelled Ratio of Ratio of impacted to un-impacted Population Growth Rate by 2050, for each species within the four MarPAMM management regions and individual sites, following percentage point (p.p.) increases in breeding success (BS) and adult survival (AS).*

Region	Common name	+5 p.p. BS	+10 p.p. BS	+20 p.p. BS	+1 p.p. AS	+2 p.p. AS	+3 p.p. AS
Argyll	European storm-petrel	1 (1-1.01)	1.01 (1-1.01)	1.02 (1.01-1.02)	1.01 (1.01-1.02)	1.02 (1.02-1.03)	1.04 (1.03-1.04)
	Kittiwake	1.02 (1.01-1.02)	1.03 (1.03-1.04)	1.06 (1.05-1.07)	1.01 (1.01-1.01)	1.02 (1.02-1.03)	1.04 (1.03-1.04)
	Common tern	1.04 (1.03-1.06)	1.08 (1.06-1.09)	1.14 (1.11-1.16)	1.01 (1-1.03)	1.02 (1.01-1.04)	1.03 (1.02-1.05)
	Guillemot	1 (1-1)	1.01 (1.01-1.01)	1.01 (1.01-1.01)	1.01 (1.01-1.01)	1.02 (1.02-1.02)	1.03 (1.03-1.03)
Co. Down to Co. Louth	Manx shearwater	1.01 (1-1.01)	1.01 (1.01-1.01)	1.02 (1.02-1.02)	1.01 (1.01-1.01)	1.02 (1.02-1.02)	1.03 (1.03-1.04)
	Sandwich tern	1.03 (1.02-1.03)	1.05 (1.04-1.05)	1.09 (1.08-1.1)	1.01 (1.01-1.01)	1.02 (1.02-1.02)	1.03 (1.03-1.04)
	Common tern	1.03 (1.03-1.04)	1.06 (1.06-1.07)	1.11 (1.1-1.12)	1.01 (1.01-1.01)	1.02 (1.02-1.02)	1.03 (1.03-1.04)
	Arctic tern	1.04 (1.03-1.05)	1.07 (1.06-1.08)	1.12 (1.1-1.13)	1.01 (1-1.02)	1.02 (1.01-1.04)	1.04 (1.03-1.05)
N. Coast to N. Channel	Fulmar	1 (1-1.01)	1.01 (1.01-1.01)	1.02 (1.01-1.02)	1.01 (1.01-1.01)	1.02 (1.02-1.02)	1.03 (1.02-1.03)
	Shag	1.04 (1.03-1.04)	1.07 (1.06-1.08)	-	1.01 (1.01-1.01)	1.02 (1.01-1.02)	1.02 (1.02-1.03)
	Kittiwake	1.02 (1.01-1.02)	1.03 (1.03-1.03)	1.06 (1.05-1.06)	1.01 (1.01-1.01)	1.02 (1.02-1.03)	1.04 (1.03-1.04)
	Common gull	1.03 (1.02-1.04)	1.06 (1.04-1.07)	1.1 (1.09-1.12)	1.01 (1-1.02)	1.02 (1.01-1.04)	1.04 (1.03-1.05)
	Herring gull	1.01 (1.01-1.02)	1.03 (1.02-1.03)	1.05 (1.04-1.05)	1.01 (1.01-1.01)	1.02 (1.02-1.02)	1.03 (1.03-1.04)
	Guillemot	1 (1-1)	1.01 (1.01-1.01)	1.01 (1.01-1.01)	1.01 (1.01-1.01)	1.02 (1.02-1.02)	1.03 (1.03-1.03)

	Razorbill	1.01 (1.01-1.01)	1.01 (1.01-1.01)	1.02 (1.02-1.03)	1.01 (1.01-1.01)	1.02 (1.02-1.02)	1.03 (1.03-1.03)
Outer Hebrides	Fulmar	1 (1-1.01)	1.01 (1.01-1.01)	1.02 (1.02-1.02)	1.01 (1.01-1.01)	1.02 (1.02-1.02)	1.03 (1.03-1.03)
	Manx shearwater	1.01 (1-1.01)	1.01 (1.01-1.01)	1.02 (1.02-1.02)	1.01 (1.01-1.01)	1.02 (1.02-1.03)	1.03 (1.03-1.04)
	European storm-petrel	1 (1-1.01)	1.01 (1-1.02)	1.02 (1.01-1.03)	1.01 (1-1.02)	1.03 (1.02-1.03)	1.04 (1.03-1.05)
	Leach’s storm-petrel	1 (1-1.01)	1.01 (1-1.01)	1.02 (1.01-1.02)	1.01 (1.01-1.02)	1.03 (1.02-1.03)	1.04 (1.04-1.04)
	Gannet	1.01 (1.01-1.01)	1.01 (1.01-1.01)	1.02 (1.02-1.02)	1.01 (1.01-1.01)	1.02 (1.02-1.02)	1.03 (1.03-1.03)
	Shag	1.04 (1.03-1.04)	1.07 (1.06-1.08)	-	1.01 (1.01-1.01)	1.02 (1.01-1.02)	1.02 (1.02-1.03)
	Kittiwake	1.02 (1.01-1.02)	1.03 (1.03-1.03)	1.06 (1.05-1.06)	1.01 (1.01-1.01)	1.02 (1.02-1.03)	1.04 (1.03-1.04)
	Great black-backed gull	1.01 (1.01-1.01)	1.02 (1.02-1.03)	1.04 (1.04-1.05)	1.01 (1.01-1.01)	1.02 (1.02-1.02)	1.03 (1.03-1.03)
	Guillemot	1 (1-1)	1.01 (1.01-1.01)	1.01 (1.01-1.01)	1.01 (1.01-1.01)	1.02 (1.02-1.02)	1.03 (1.03-1.03)
	Razorbill	1.01 (1.01-1.01)	1.01 (1.01-1.01)	1.02 (1.02-1.02)	1.01 (1.01-1.01)	1.02 (1.02-1.02)	1.03 (1.03-1.03)
	Atlantic puffin	1.01 (1.01-1.01)	1.01 (1.01-1.01)	1.02 (1.02-1.02)	1.01 (1.01-1.01)	1.02 (1.02-1.02)	1.03 (1.03-1.03)
Canna and Sanday SPA	Shag	1.04 (1.03-1.05)	1.07 (1.06-1.09)	-	1.01 (1.01-1.01)	1.02 (1.01-1.02)	1.03 (1.02-1.03)
	Kittiwake	1.02 (1.01-1.02)	1.03 (1.03-1.04)	1.06 (1.05-1.07)	1.01 (1.01-1.02)	1.02 (1.02-1.03)	1.04 (1.03-1.04)
	Herring gull	1.02 (1.01-1.03)	1.03 (1.02-1.04)	1.06 (1.04-1.07)	1.01 (1-1.02)	1.02 (1.01-1.03)	1.03 (1.02-1.04)
	Guillemot	1 (1-1)	1.01 (1.01-1.01)	1.01 (1.01-1.01)	1.01 (1.01-1.01)	1.02 (1.02-1.02)	1.03 (1.03-1.03)

	Atlantic puffin	1.01 (1-1.01)	1.01 (1.01-1.01)	1.02 (1.02-1.03)	1.01 (1.01-1.01)	1.02 (1.02-1.02)	1.03 (1.03-1.03)
Monach Isles NC MPA	Black guillemot	1.01 (1.01-1.01)	1.02 (1.01-1.02)	1.03 (1.03-1.04)	1.01 (1.01-1.01)	1.02 (1.02-1.03)	1.03 (1.03-1.04)
Rum SPA	Manx shearwater	1.01 (1.01-1.01)	1.01 (1.01-1.01)	1.02 (1.02-1.02)	1.01 (1.01-1.01)	1.02 (1.02-1.02)	1.03 (1.03-1.04)
	Kittiwake	1.02 (1.01-1.02)	1.03 (1.02-1.04)	1.06 (1.05-1.07)	1.01 (1.01-1.02)	1.02 (1.02-1.03)	1.04 (1.03-1.04)
	Guillemot	1 (1-1)	1.01 (1.01-1.01)	1.01 (1.01-1.01)	1.01 (1.01-1.01)	1.02 (1.02-1.02)	1.03 (1.03-1.03)

Appendix 5: Supplementary analysis including immature Shag survival rates.

Table A5a. *Modelled Ratio of Ratio of impacted to un-impacted Population Size by 2050 for Shag following percentage point (p.p.) increases in breeding success (BS) and adult survival (AS), when including a single (adult) survival rate across age classes (0.86 ± 0.19) or different immature survival rates (Juvenile (0-1): 0.51 ± 0.25; Immature (1-2 year): 0.74 ± 0.18; Adult survival (>3 year): 0.86 ± 0.19).*

Region	**Analysis**	**+5 p.p. BS**	**+10 p.p. BS**	**+20 p.p. BS**	**+1 p.p. AS**	**+2 p.p. AS**	**+3 p.p. AS**
N. Coast to N. Channel	Adult Rate	2.54 (2.22-2.85)	5.93 (4.92-7.07)	-	1.27 (1.17-1.39)	1.56 (1.41-1.71)	1.87 (1.65-2.08)
	Immature Rates	1.88 (1.68-2.08)	3.54 (2.77-4.08)	-	1.3 (1.16-1.47)	1.77 (1.51-1.97)	2.29 (1.92-2.74)
Outer Hebrides	Adult Rate	2.56 (2.23-3)	6.05 (4.8-7.78)	-	1.27 (1.17-1.39)	1.57 (1.41-1.74)	1.89 (1.65-2.13)
	Immature Rates	2.1 (1.7-2.43)	4.08 (3.28-5.2)	-	1.37 (1.28-1.54)	1.82 (1.64-2.1)	2.44 (2.01-2.72)
Canna and Sanday SPA	Adult Rate	2.62 (2.13-3.24)	6.31 (4.6-9.18)	-	1.29 (1.13-1.47)	1.59 (1.38-1.86)	1.94 (1.64-2.34)
	Immature Rates	1.97 (1.73-2.49)	3.66 (3.13-4.59)	-	1.36 (1.21-1.52)	1.81 (1.58-2.09)	2.4 (1.9-3.28)

Table A5b. *Modelled Ratio of Ratio of impacted to un-impacted Population Growth Rate by 2050 for Shag following percentage point (p.p.) increases in breeding success (BS) and adult survival (AS), when including a single (adult) survival rate across age classes (0.86 ± 0.19) or different immature survival rates (Juvenile (0-1): 0.51 ± 0.25; Immature (1-2 year): 0.74 ± 0.18; Adult survival (>3 year): 0.86 ± 0.19).*

Region	Analysis	+5 p.p. BS	+10 p.p. BS	+20 p.p. BS	+1 p.p. AS	+2 p.p. AS	+3 p.p. AS
N. Coast to N. Channel	Adult Rate	1.04 (1.03-1.04)	1.07 (1.06-1.08)	-	1.01 (1.01-1.01)	1.02 (1.01-1.02)	1.02 (1.02-1.03)
	Immature Rates	1.03 (1.02-1.03)	1.05 (1.04-1.05)	-	1.01 (1.01-1.01)	1.02 (1.02-1.02)	1.03 (1.03-1.04)
Outer Hebrides	Adult Rate	1.04 (1.03-1.04)	1.07 (1.06-1.08)	-	1.01 (1.01-1.01)	1.02 (1.01-1.02)	1.02 (1.02-1.03)
	Immature Rates	1.03 (1.02-1.04)	1.06 (1.05-1.07)	-	1.01 (1.01-1.01)	1.02 (1.02-1.03)	1.03 (1.03-1.04)
Canna and Sanday SPA	Adult Rate	1.04 (1.03-1.05)	1.07 (1.06-1.09)	-	1.01 (1.01-1.01)	1.02 (1.01-1.02)	1.03 (1.02-1.03)
	Immature Rates	1.03 (1.02-1.03)	1.05 (1.04-1.06)	-	1.01 (1.01-1.02)	1.02 (1.02-1.03)	1.03 (1.03-1.04)

Appendix 6: Modelled Population Sizes.

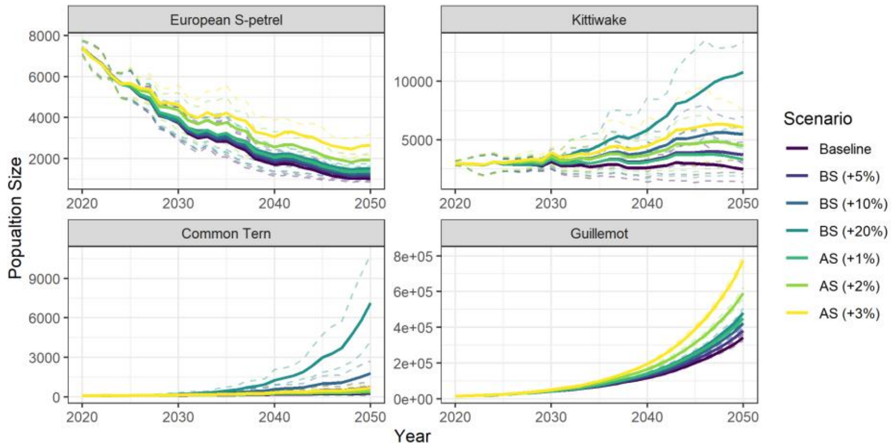

Figure A6a. *Population Size modelled between 2020 and 2050 for species occurring within the Argyll Management Region following improvments in Breeding Success (BS: 5 p.p., 10 p.p., 20 p.p.) and Adult Survival (AS: 1 p.p., 2 p.p., 3 p.p.).*

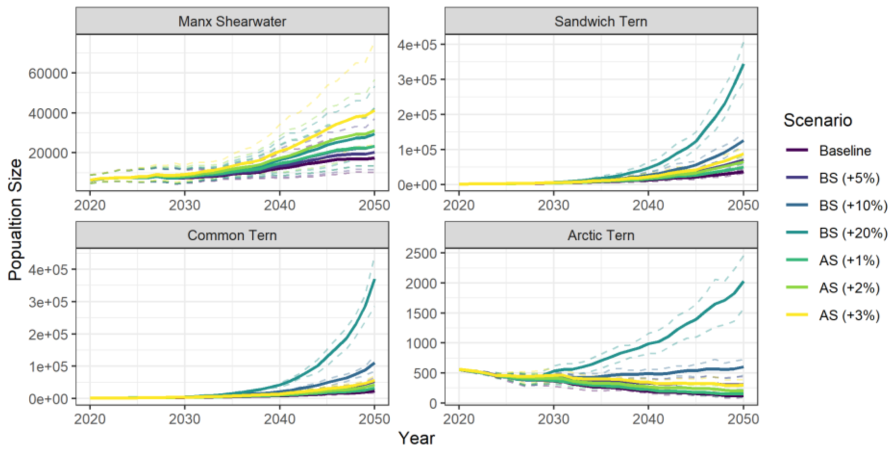

Figure A6b. *Population Size modelled between 2020 and 2050 for species occurring within the County Down – County Louth Management Region following improvments in Breeding Success (BS: 5 p.p., 10 p.p., 20 p.p.) and Adult Survival (AS: 1 p.p., 2 p.p., 3 p.p.).*

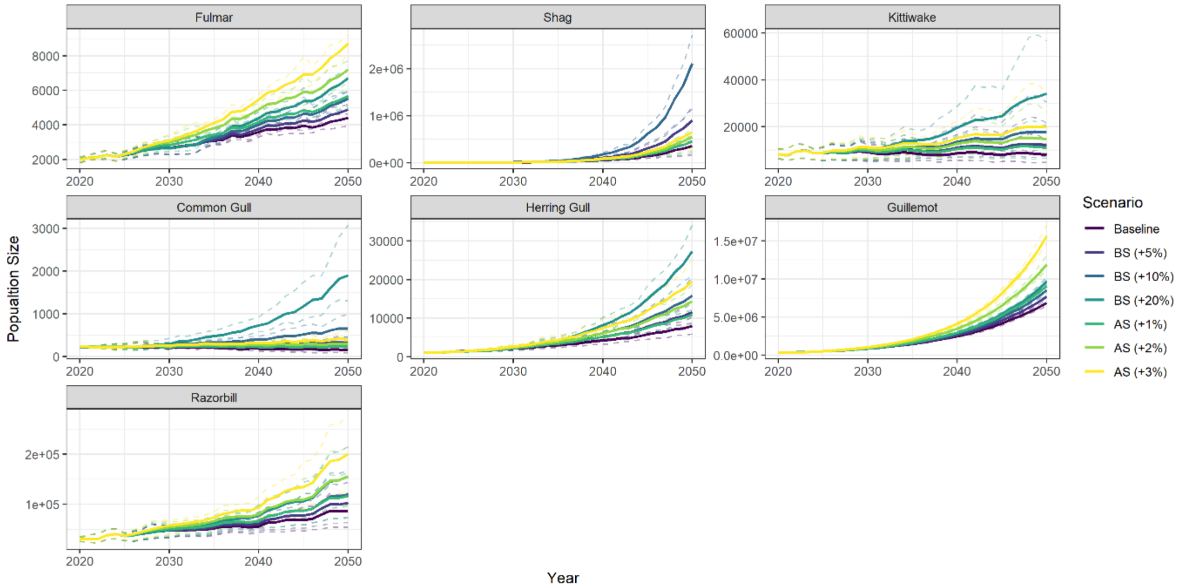

Figure A6c. *Population Size modelled between 2020 and 2050 for species occurring within the North Coast-North Channel Management Region following improvments in Breeding Success (BS: 5 p.p., 10 p.p., 20 p.p.) and Adult Survival (AS: 1 p.p., 2 p.p., 3 p.p.).*

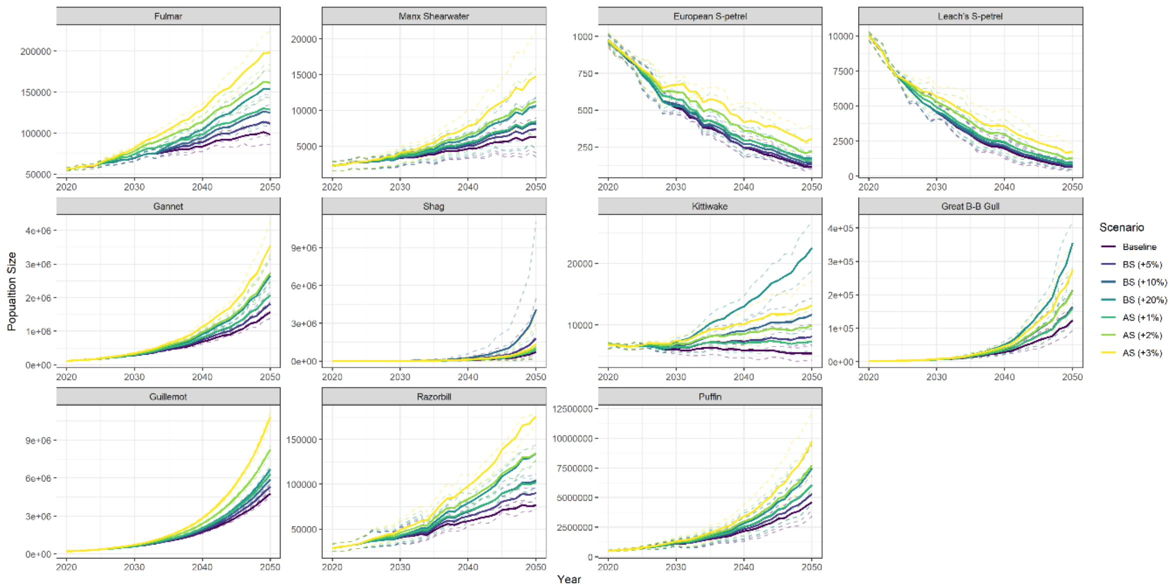

Figure A6d. *Population Size modelled between 2020 and 2050 for species occurring within the Outer Hebrides Management Region following improvments in Breeding Success (BS: 5 p.p., 10 p.p., 20 p.p.) and Adult Survival (AS: 1 p.p., 2 p.p., 3 p.p.).*

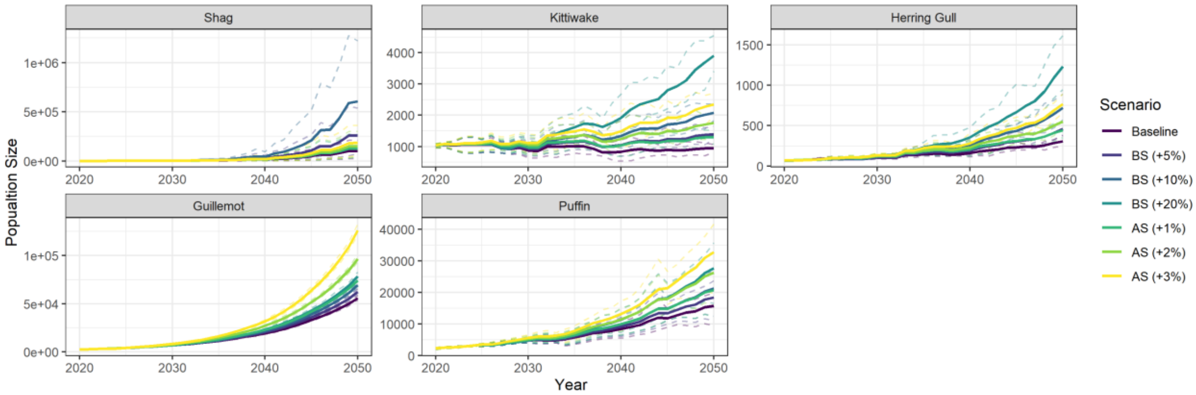

Figure A6e. *Population Size modelled between 2020 and 2050 for species occurring within Canna and Sanday SPA following improvments in Breeding Success (BS: 5 p.p., 10 p.p., 20 p.p.) and Adult Survival (AS: 1 p.p., 2 p.p., 3 p.p.).*

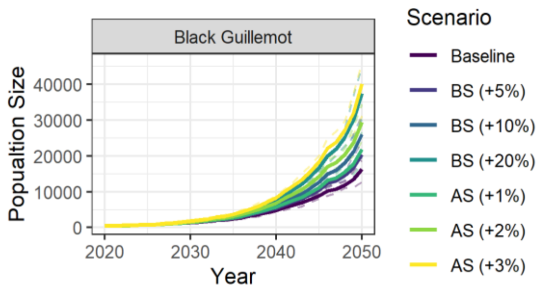

Figure A6f. *Population Size modelled between 2020 and 2050 for species occurring within Monach Isles NC MPA SPA following improvments in Breeding Success (BS: 5 p.p., 10 p.p., 20 p.p.) and Adult Survival (AS: 1 p.p., 2 p.p., 3 p.p.).*

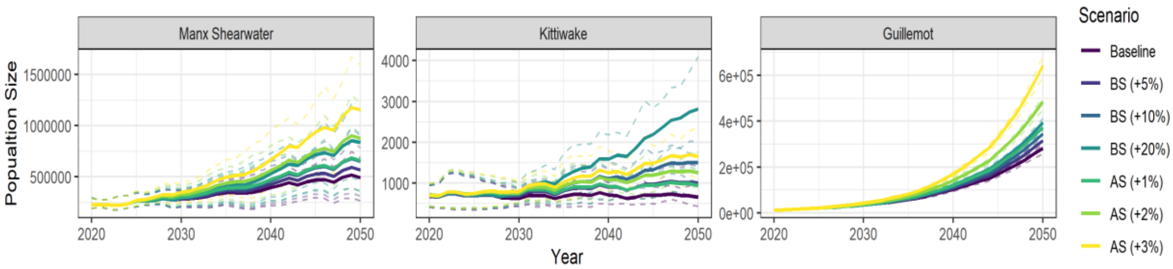

Figure A6g. *Population Size modelled between 2020 and 2050 for species occurring within Rum SPA following improvments in Breeding Success (BS: 5 p.p., 10 p.p., 20 p.p.) and Adult Survival (AS: 1 p.p., 2 p.p., 3 p.p.).*

Project partners

Citation

Rich Howells, Chelsea Ward, & Tom Evans (2023). PVA Analysis for key seabird populations in the MarPAMM area

Cover photographs

This work was produced as part of the Marine Protected Area Management and Monitoring (MarPAMM) project. MarPAMM has been supported by the European Union's INTERREG VA Programme, managed by the Special EU Programmes Body.

For more information on the MarPAMM project please visit the project website:

www.mpa-management.eu